



US Army Corps  
of Engineers

Waterways Experiment  
Station

# *Environmental Effects of Dredging*

*Section 04 - Regulatory  
Technical Notes  
EEDP-04-1 through EEDP-04-17*

Compiled by  
Dredging Operations Technical  
Support Program

## **Section 04—Regulatory**

### **EEDP-04-1 through EEDP-04-17**

- EEDP-04-1 Interim Guidance for Predicting Quality of Effluent Discharged from Confined Dredged Material Disposal Areas—General (June 1985)
- EEDP-04-2 Interim Guidance for Predicting Quality of Effluent Discharged from Confined Dredged Material Disposal Areas—Test Procedures (June 1985)
- EEDP-04-3 Interim Guidance for Predicting Quality of Effluent Discharged from Confined Dredged Material Disposal Areas—Data Analysis (June 1985)
- EEDP-04-4 Interim Guidance for Predicting Quality of Effluent Discharged from Confined Dredged Material Disposal Areas—Application (June 1985)
- EEDP-04-5 Interim Procedures for Estimating Mixing Zones for Effluent from Dredged Material Disposal Sites (Single-Point Discharge) (September 1987)
- EEDP-04-6 Regulatory Identification of Hydrocarbon Contaminants in Dredged Material (March 1987)
- EEDP-04-7 Routine and Atypical Wetland Determination According to CE Wetlands Delineation Manual (January 1988)
- EEDP-04-8 Corps of Engineers' Procedures and Policies on Dredging and Dredged Material Disposal (The Federal Standard) (August 1988)
- EEDP-04-9 General Guidelines for Monitoring Effluent Quality from Confined Dredged Material Disposal Areas (November 1988)
- EEDP-04-10 Procedures for Examining the Relationship Between Sediment Geochemistry and Biological Impacts of Contaminants (December 1989)
- EEDP-04-11 A Plant Bioassay for Assessing Plant Uptake of Heavy Metals from Contaminated Freshwater Dredged Material (December 1989)
- EEDP-04-12 A Computerized Procedure for Predicting Plant Uptake of Heavy Metals from Contaminated Freshwater Dredged Material (March 1990)
- EEDP-04-13 Environmental Interpretation and Evaluation of Hydrocarbon Contaminants in Dredged Material (August 1990)
- EEDP-04-14 Residue Effects Data Base on the Relationship Between Dioxin and Biological Effects in Aquatic Animals (February 1992)
- EEDP-04-15 New Technique for Sediment/Organism Equilibrium Partitioning Studies (December 1992)
- EEDP-04-16 The  $K_{oc}$  of Nonpolar Organic Compounds in Sediment (December 1992)

EEDP-04-17 Critical Body Residue (CBR) Approach for Interpreting the Consequences of Bioaccumulation of Neutral Organic Contaminants (December 1992)



# *Environmental Effects of Dredging Technical Notes*



## INTERIM GUIDANCE FOR PREDICTING QUALITY OF EFFLUENT DISCHARGED FROM CONFINED DREDGED MATERIAL DISPOSAL AREAS--GENERAL

**PURPOSE:** The following series of technical notes describe the functions necessary for predicting the quality of effluent discharged from confined dredged material disposal areas during disposal operations.\*

EEDP-04-1	General
EEDP-04-2	Test Procedures
EEDP-04-3	Data Analysis
EEDP-04-4	Application

The guidance was developed as a part of on-going research conducted under the Long-Term Effects of Dredging Operations (LEDO) Program. Procedures for such predictions are being refined and verified under LEDO through comparative evaluations of predictions and field measurement of effluent water quality.

**BACKGROUND:** Confined dredged material disposal has increased because of constraints on open-water disposal. The quality of water discharged from confined disposal areas (effluent) is a major environmental concern associated with such disposal.

A schematic of a typical active confined disposal area is illustrated in Figure 1. Dredged material placed in a disposal area undergoes sedimentation that results in a thickened deposit of material overlaid by clarified water (supernatant), which is discharged as effluent from the site during disposal operations. The concentrations of suspended solids in the effluent can be determined by column settling tests.

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\* The modified elutriate test does not account for long-term geochemical changes that may occur following disposal and subsequent drying of the dredged material and therefore should not be used to evaluate quality of surface runoff from the disposal site.

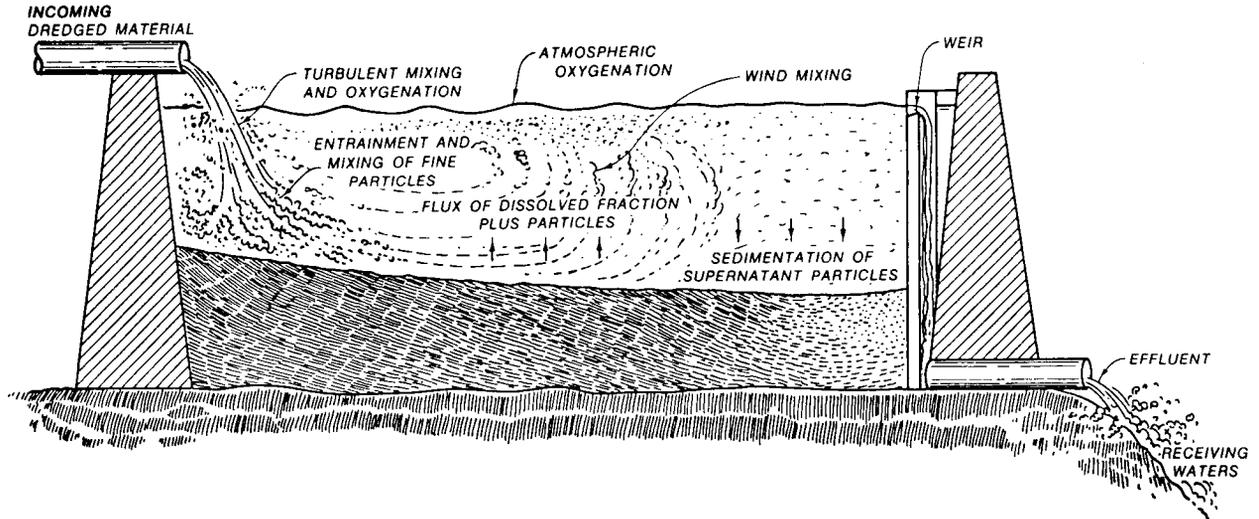


Figure 1. Schematic of factors affecting quality of effluent from confined disposal areas

The effluent may contain both dissolved and particle-associated contaminants. A large portion of the total contaminant level is particle associated. Results of the standard elutriate test do not reflect the conditions in confined disposal sites that influence contaminant release. A modified elutriate test procedure was therefore developed for use in predicting both the dissolved and particle-associated concentrations of contaminants in the effluent from confined disposal areas. The modified test simulates contaminant release under confined disposal area conditions and reflects the sedimentation behavior of dredged material, retention time of the disposal area, and chemical environment in ponded water during disposal.

**REGULATORY ASPECTS:** Guidelines have been published to reflect the 1977 Amendments of Section 404 of the Clean Water Act (EPA 1980a). Proposed testing requirements define dredged material according to the four categories shown in Figure 2 (EPA 1980b). Category 3 includes potentially contaminated material proposed for confined disposal that has "potential for contamination of the receiving water column only." The proposed testing requirements call for evaluation of short-term water column impacts of disposal area effluents. Predicted contaminant levels based on results of modified elutriate and column settling tests along with operational considerations can be used with appropriate water-quality standards to determine the mixing zone required to dilute the effluent to an acceptable level (Environmental Effects Laboratory 1976, EPA/CE 1977).

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

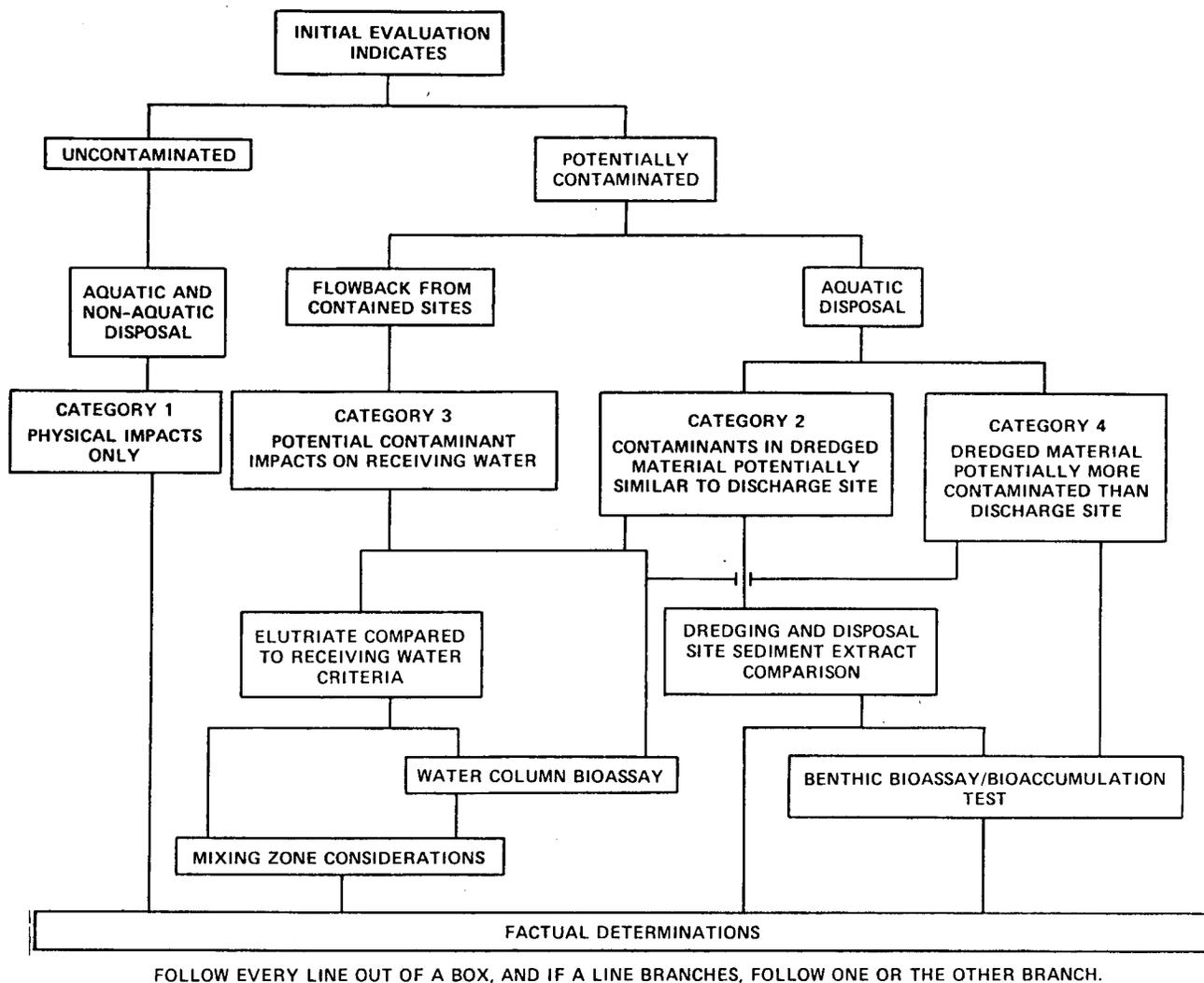


Figure 2. Proposed dredged material regulatory testing flow chart (EPA 1980b)

### Predictive Technique

The prediction of the quality of effluent from confined dredged material disposal areas must account for both the dissolved concentrations of contaminants and that fraction in the total suspended solids. A modified elutriate test procedure, developed for this purpose, defines dissolved concentrations of contaminants and contaminant fractions in the total suspended solids under quiescent settling conditions and accounts for the geochemical changes occurring in the disposal area during active disposal operations. Column settling test procedures (Montgomery 1978; Palermo, Montgomery, Poindexter 1978) were refined and extended to define the concentration of suspended solids in the effluent for given operational conditions

(i.e., surface area, ponding depth, inflow rate, and hydraulic efficiency).

Using results from both of these tests, a prediction of the total concentration of contaminants in the effluent can be made. A flow chart illustrating the technique is shown in Figure 3. The procedures for conducting both tests are given in Technical Note EEDP-04-2.

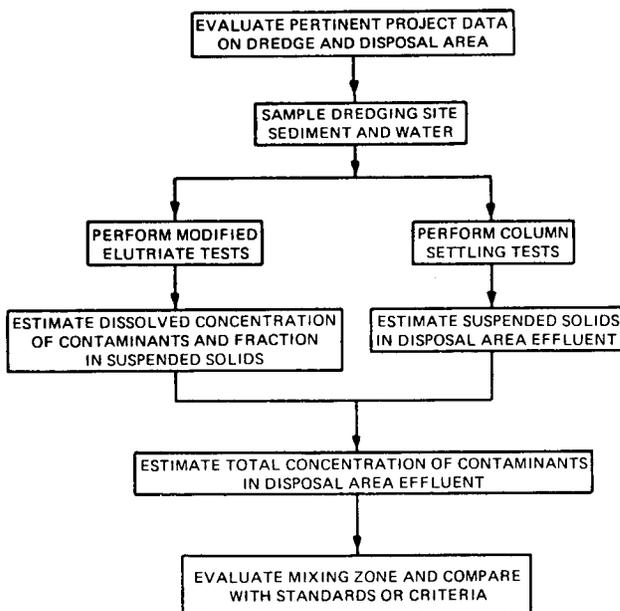


Figure 3. Steps for predicting effluent water quality.

in dredging and disposal activities for the project under consideration or for similar projects. Data relating to the characteristics of the sediment must be determined from samples of the sediment to be dredged and the dredging site water column.

A summary of the data requirements for effluent quality predictions is given in Table 1. Some of the data can be determined from the design or from evaluation of the site using procedures described by Montgomery (1978) and Palermo, Montgomery, and Poindexter (1978). The remaining data must be developed using the procedures described in Technical Note EEDP-04-2.

### Data Requirements

Data requirements for prediction of effluent quality include those pertaining to operational considerations (i.e., disposal site characteristics and dredge characteristics) and those pertaining to the properties of the sediment to be dredged (i.e., contaminant-release characteristics and sedimentation characteristics). Data relating to operational considerations are usually determined from the disposal area design and by past experience

### Sampling Requirements

Samples of sediment and water from a proposed dredging site are required for characterizing the sediment to be dredged and for conducting modified elutriate tests and column settling tests. The level of effort, including the

Table 1  
Data Requirements for Predicting the Quality of Effluent  
 from Confined Dredged Material Disposal Area\*

Data Required	Symbol	Source of data
Dredge inflow rate	$Q_i$	Project information, site design
Dredge inflow solids concentration	$C_i$	Project information, site design
Ponded area in disposal site	$A_p$	Project information, site design
Average ponding depth in disposal site and at the weir	$D_p, D_{pw}$	Project information, site design
Hydraulic efficiency factor	HEF	Dye tracer or theoretical determination
Effluent total suspended solids concentration	$SS_{eff}$	Column settling tests
Dissolved concentration of contaminant in effluent	$C_{diss}$	Modified elutriate tests
Fraction of contaminant in the total suspended solids in effluent	$F_{SS}$	Modified elutriate tests

\* This summary includes only those data required for effluent quality prediction. It was assumed that the disposal area under consideration was designed for effective sedimentation and storage capacity. Data requirements for design or evaluation of a disposal area are found in Palermo, Montgomery, and Poindexter (1978).

number of sampling stations, quantity of material, and any scheme used for compositing samples, is highly project specific. If at all possible, the sampling operations required for sediment characterization (both physical and chemical), for design or evaluation of the disposal site, and for modified elutriate and column settling tests should be conducted simultaneously to avoid duplication of effort and to ensure sample similarity.

Normally effluent quality will be of concern for maintenance dredged material. Representative samples of sediments proposed for maintenance dredging are satisfactory for obtaining the quantities needed for all testing requirements. General guidance on sampling for chemical characterization purposes is found in Plumb (1981). This reference should be used for guidance in obtaining samples for use in the modified elutriate testing.

## Application

The technique for predicting the quality of effluent discharged from confined dredged material disposal areas is described in Technical Note EEDP-04-3. The technique can be applied to predict the performance of existing sites or to design new sites.

For existing sites, the technique can be used to predict effluent quality for a given set of anticipated operational conditions (known flow and ponding conditions). In a similar manner, the procedure can be used to determine the operational conditions (size, geometry, maximum allowable dredge size, etc.) for a proposed site to meet a given effluent quality requirement. Examples of both of these cases are presented in Technical Note EEDP-04-4.

References

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- Plumb, R. H. 1981. "Procedures for Handling and Chemical Analysis of Sediment and Water Samples," EPA/CE Technical Committee on Criteria for Dredged and Fill Material, US Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

## Notations

The notations used in Technical Notes EEDP-04-1 through 4 are defined as follows.

$A_p$	Area ponded, acres
$C_{diss}$	Dissolved concentration of constituent, milligrams per liter
$C_i$	Inflow solids concentration, grams per liter
$C_{slurry}$	Solids concentration of slurry, grams per liter (dry weight basis)
$C_{sediment}$	Solids concentration of sediment, grams per liter (dry weight basis)
$C_{total}$	Total concentration of constituent, milligrams per liter
$F_{SS}$	Fraction of constituent in total suspended solids, milligrams per kilogram
$D_p$	Depth of ponding in disposal site, ft
$D_{pw}$	Desired ponding depth or ponding depth at weir, ft
HEF	Hydraulic efficiency factor
$Q_i$	Inflow rate, cubic feet per second
P	Percent of suspended solids remaining at test interval
R	Percent of solids removed from suspension at test interval
RF	Resuspension factor
SS	Total suspended solids concentration, milligrams per liter
$SS_{col}$	Suspended solids concentration determined by column test, milligrams per liter
$SS_{eff}$	Suspended solids concentration of effluent considering anticipated resuspension, milligrams per liter of water
T	Theoretical detention time, hours
$T_d$	Field mean detention time, hours
t	Sampling time, hr
$V_{sediment}$	Volume of sediment, liters
$V_p$	Volume ponded, acre-feet
$V_{water}$	Volume of water, liters
z	Sample depth, feet
$\phi$	Percent of initial suspended solids concentration (beginning of column settling test used as 100 percent)



# *Environmental Effects of Dredging Technical Notes*



## INTERIM GUIDANCE FOR PREDICTING QUALITY OF EFFLUENT DISCHARGED FROM CONFINED DREDGED MATERIAL DISPOSAL AREAS--TEST PROCEDURES

**PURPOSE:** The following series of technical notes describe the functions necessary for predicting the quality of effluent discharged from confined dredged material disposal areas during dredging operations.\*

EEDP-04-1	General
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The guidance was developed as a part of on-going research conducted under the Long-Term Effects of Dredging Operations (LEDO) Program. Procedures for such predictions are being refined and verified under LEDO through comparative evaluations of predictions and field measurement of effluent water quality.

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Dredged material placed in a disposal area undergoes sedimentation that results in a thickened deposit of material overlaid by clarified water (called supernatant), which is discharged as effluent from the site during disposal operations. The concentrations of suspended solids in the effluent can be determined by column settling tests.

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\* The modified elutriate test does not account for long-term geochemical changes that may occur following disposal and subsequent drying of the dredged material and therefore should not be used to evaluate quality of surface runoff from the disposal site.

The effluent may contain both dissolved and particle-associated contaminants. A large portion of the total contaminant content is particle associated. The modified elutriate test was developed for use in predicting both the dissolved and particle-associated concentrations of contaminants in the effluent from confined disposal areas.

REGULATORY ASPECTS: Guidelines have been published to reflect the 1977 Amendments of Section 404 of the Clean Water Act (EPA 1980b). Proposed testing requirements define dredged material according to four categories. Category 3 includes potentially contaminated material proposed for confined disposal that has "potential for contamination of the receiving water column only." The proposed testing requirements call for evaluation of the short-term water-column impacts of disposal area effluents. Predicted contaminants levels based on results of modified elutriate and column settling tests along with operational considerations can be used with appropriate water-quality standards to determine the mixing zone required to dilute the effluent to an acceptable level (Environmental Effects Laboratory 1976, EPA/CE 1977).

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### Initial Screening

An initial screening for contamination must be performed as outlined in the testing requirements for Section 404 of the Clean Water Act (EPA 1980b). The evaluation is designed to determine if there is reason to believe that the sediment contains any contaminant at a significant concentration (above background levels) and to identify the contaminants of concern that should be considered for analysis in the modified elutriate test. Considerations include but are not limited to:

- a. Potential routes by which contaminants could reasonably have been introduced to the sediment.
- b. Data from previous tests of the sediment or other similar sediment in the vicinity, provided comparison would still be appropriate.
- c. Probability of contamination from surface runoff.
- d. Spills of contaminants in the area to be dredged.
- e. Industrial and municipal waste discharges.

### Modified Elutriate Test

The modified elutriate test should be conducted and appropriate chemical analyses should be performed as soon as possible after sample collection. The volume of elutriate sample needed for chemical analyses will depend on the

number and types of analyses to be conducted (Plumb 1981). Both dissolved and total concentrations of contaminants must be determined. The volume required for each analysis, the number of parameters measured, and the desired analytical replication will influence the total elutriate sample volume required. A 4-l cylinder is normally used for the test, and the supernatant volume available for sample extraction will vary from approximately 500 to 1,000 ml, depending on the sediment properties, settling times, and initial concentration of the slurry. It may be necessary to composite several extracted samples or use large-diameter cylinders to obtain the total required volume.

#### Apparatus

The following items are required:

- a. Laboratory mixer, preferably with Teflon shaft and blades.
- b. Several 4-l graduated cylinders. Larger cylinders may be used if large sample volumes are required. Nalgene cylinders are acceptable for testing involving analysis of metals and nutrients. Glass cylinders are required for testing involving analysis of organics.
- c. Assorted glassware for sample extraction and handling.
- d. Compressed air source with deionized water trap and tubing for bubble aeration of slurry.
- e. Vacuum or pressure filtration equipment, including vacuum pump or compressed air source and appropriate filter holder capable of accommodating 47-, 105-, or 155-mm-diam filters.
- f. Presoaked filters with 0.45- $\mu$ m pore-size diameter.
- g. Plastic sample bottles, 500-ml capacity for storage of water and liquid phase samples for metal and nutrient analyses.
- h. Wide-mouth 1-gal-capacity glass jars with Teflon-lined screw-type lids for sample mixing. These jars should also be used as sample containers when samples are to be analyzed for pesticide materials.

Prior to use, all glassware, filtration equipment, and filters should be thoroughly cleaned. Wash all glassware with detergent; rinse five times with tap water; place in a clean 10-percent (or stronger) HCl acid bath for a minimum of 4 hr; rinse five times with tap water; and then rinse five times with distilled or deionized water. Soak filters for a minimum of 2 hr in a 5-M HCl bath and then rinse 10 times with distilled water. It is also a good practice to discard the first 50 ml of water or liquid phase filtered. Wash all glassware to be used in preparation and analysis of pesticide residues using the eight-step procedure given EPA (1980a).

## Test procedure

The step-by-step procedure for conducting a modified elutriate test, as shown in Figure 1, is given in the following paragraphs.

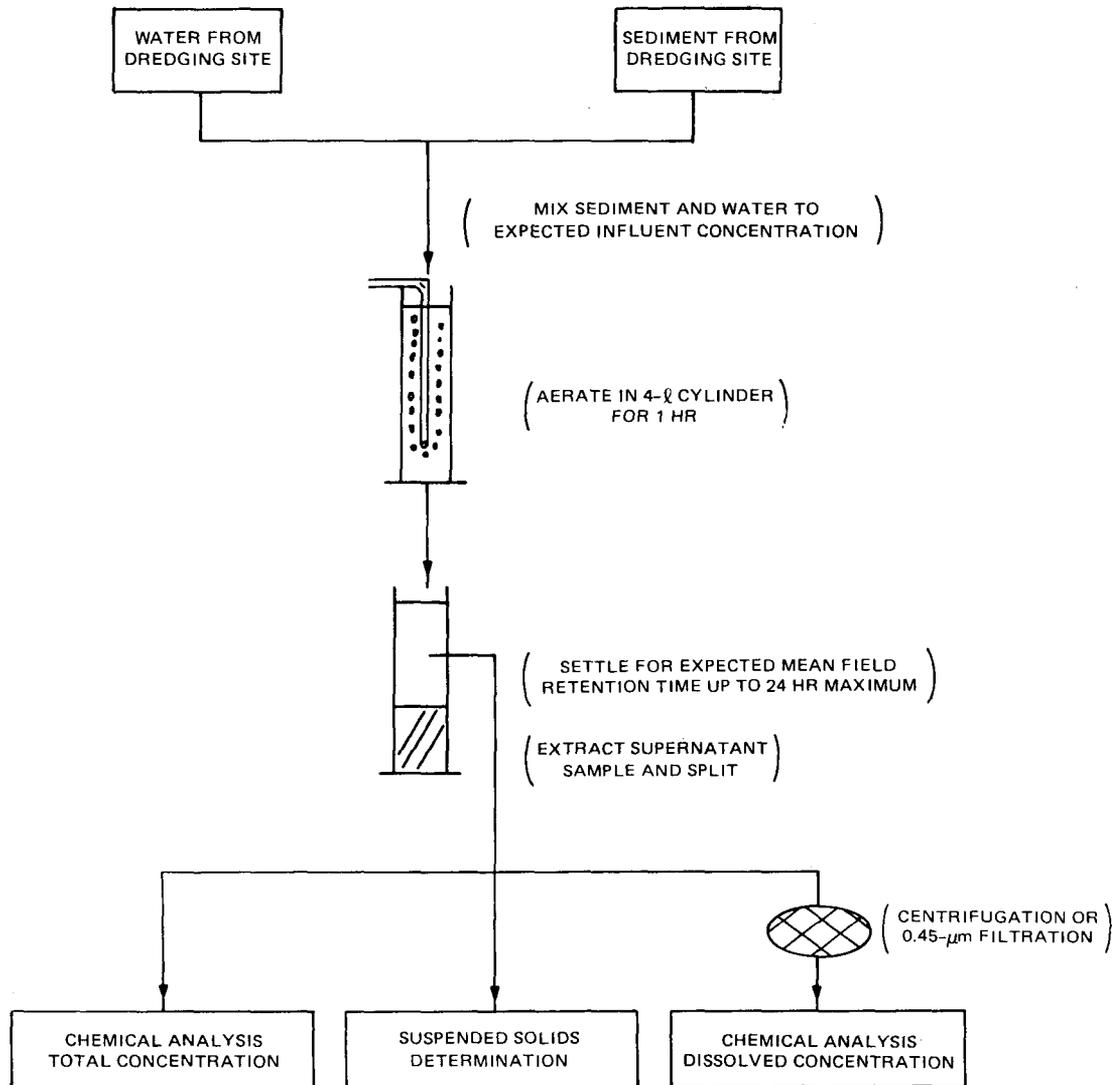


Figure 1. Modified elutriate test procedure

Step 1 - Slurry preparation. The sediment and dredging site water should be mixed to approximately equal the expected average field inflow concentration. If estimates of the average field inflow concentration cannot be made based on past data, a slurry concentration of 150 g/ℓ (dry-weight basis) should be used. Predetermine the concentration of the well-mixed sediment in grams per liter (dry-weight basis) by oven drying a small subsample of known volume. Each 4-ℓ cylinder to be filled will require a mixed slurry volume of 3-3/4 ℓ. The volumes of sediment and dredging site water to be mixed for a 3-3/4-ℓ slurry volume can be calculated using the following expressions:

and 
$$V_{\text{sediment}} = 3.75 \frac{C_{\text{slurry}}}{C_{\text{sediment}}} \quad (1)$$

$$V_{\text{water}} = 3.75 - V_{\text{sediment}} \quad (2)$$

where

$V_{\text{sediment}}$  = volume of sediment, liters

3.75 = volume of slurry for 4-l cylinder, liters

$C_{\text{slurry}}$  = desired concentration of slurry, grams per liter  
(dry-weight basis)

$C_{\text{sediment}}$  = predetermined concentration of sediment, grams per liter  
(dry-weight basis)

$V_{\text{water}}$  = volume of dredging site water, liters

Step 2 - Mixing. Mix the 3-3/4 l of slurry by placing appropriate volumes of sediment and dredging site water in 1-gal glass jars and mixing for 5 min with a laboratory mixer. The slurry should be mixed to a uniform consistency with no unmixed agglomerations of sediment.

Step 3 - Aeration. Bubble aeration is used to ensure oxidizing conditions in the supernatant water during the subsequent settling phase. Pour the mixed slurry into a 4-l graduated cylinder. Attach glass tubing to the aeration source and insert tubing to the bottom of the cylinder. The tubing can be held in place by insertion through a predrilled No. 4 stopper placed in the top of the cylinder. Compressed air should be passed through a deionized water trap, through the tubing, and bubbled through the slurry. The flow rate should be adjusted to agitate the mixture vigorously, and bubbling should be continued for 1 hr.

Step 4 - Settling. Remove the tubing and allow the aerated slurry to undergo quiescent settling for a time period equal to the anticipated field mean retention time up to a maximum of 24 hr. If the field mean retention time is not known, allow settling for 24 hr. Guidance for estimating the field mean retention is given in Technical Note EEDP-04-3.

Step 5 - Sample extraction. After the settling period, an interface will usually be evident between the supernatant water with low concentration of suspended solids and the more concentrated settled material. Samples of the supernatant water should be extracted from the cylinder at a point midway between the water surface and the interface using syringe and tubing. Care should be taken not to resuspend settled material.

Step 6 - Sample preservation and analysis. The sample should be analyzed as soon as possible after extraction to determine the total suspended solids and the dissolved and total concentrations of selected constituents. The fraction of a constituent in the total suspended solids can then be calculated. Filtration using 0.45- $\mu\text{m}$  filters should be used to obtain subsamples for analysis of dissolved concentrations. Samples to be analyzed for dissolved pesticides or PCB must be free of particles but should not be filtered, due to the tendency for these materials to adsorb on the filter. However, particulate matter can be removed before analysis by high-speed centrifugation at 10,000 times gravity using Teflon, glass, or aluminum centrifuge tubes (Fulk et al. 1975). The total suspended solids concentration can also be determined by filtration (0.45  $\mu\text{m}$ ). The fraction of a constituent in the total suspended solids is calculated as follows:

$$F_{SS} = (1 \times 10^6) \frac{C_{\text{total}} - C_{\text{diss.}}}{SS} \quad (3)$$

where

- $F_{SS}$  = fraction of constituent in the total suspended solids, milligrams per kilogram of suspended solids
- $C_{\text{total}}$  = total concentration of constituent, milligram per liter of sample
- $C_{\text{diss.}}$  = dissolved concentration of constituent, milligrams per liter of sample
- $SS$  = total suspended solids concentration, milligrams per liter of sample

Subsamples for analyses of total concentrations should undergo appropriate digestion prior to analysis. All digestion and chemical analyses should be performed using accepted procedures (American Public Health Association 1985; EPA 1980a; and EPA 1979).

Samples to be analyzed for pesticides or PCB should immediately undergo solvent extraction. The extract may then be held in clean uncontaminating containers for periods up to three or four weeks at -15 to -20° C before further analyses are performed.

Samples for metals analysis should be preserved immediately by lowering the pH to <2 with 3 to 5 ml of concentrated nitric acid per liter (EPA 1979). High purity acid, either purchased commercially or prepared in a subboiling unit, must be used.

Nutrient analyses should be conducted as soon as possible. Acidification with  $H_2SO_4$  to  $pH < 2$  and storage at  $4^\circ C$  will allow the sample to be held for maximum of 24 hr for ammonia nitrogen, Kjeldahl nitrogen, and nitrate nitrogen analyses (EPA 1979). Storage at  $4^\circ C$  will allow holding of samples to be analyzed for dissolved orthophosphate and total dissolved phosphorus for up to 24 hr. Subsamples to be analyzed for cyanide should be preserved with 2 ml of 10 N sodium hydroxide per liter of sample ( $pH > 12$ ) (EPA 1979).

### Column Settling Test

Sedimentation tests, performed in 8-in.-diam ported columns as shown in Figure 2, are necessary to provide data for design or evaluation of disposal areas for retention of suspended solids. These tests were originally designed to define the settling behavior of a particular sediment and to provide information concerning the volumes occupied by newly placed layers of dredged material. The test procedures were modified to obtain data for use in predicting the concentration of suspended solids in the effluent.

Sedimentation of freshwater slurries of solids concentration less than 100 g/l can generally be characterized by flocculent settling properties. As solids concentrations exceed 100 g/l, the sedimentation process may be characterized by zone settling properties in which a clearly defined interface is formed between the clarified supernatant water and the more concentrated settled material. Zone settling properties also govern when the sediment/water salinity is greater than 3 ppt. Recent studies have shown that flocculent settling governs behavior of suspended solids in the clarified supernatant water above the sediment/water interface for slurries exhibiting an interface.

#### Apparatus

A settling column such as shown in Figure 2 is used. The test column depth should approximate the effective settling depth of the proposed disposal area. A practical limit on the depth of test is 6 ft. The column should be at least 8 in. in diameter with interchangeable sections and with sample port at 1-ft or closer intervals in the lower 3 ft and at 1/2-ft intervals in the upper 3 ft. The column should have provisions to bubble air from the bottom to keep the slurry mixed during the column filling period. Shop drawings for construction of the test columns are available from the Waterways Experiment Station.\*

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\* Address request for the shop drawings to the attention of WESEP-E.

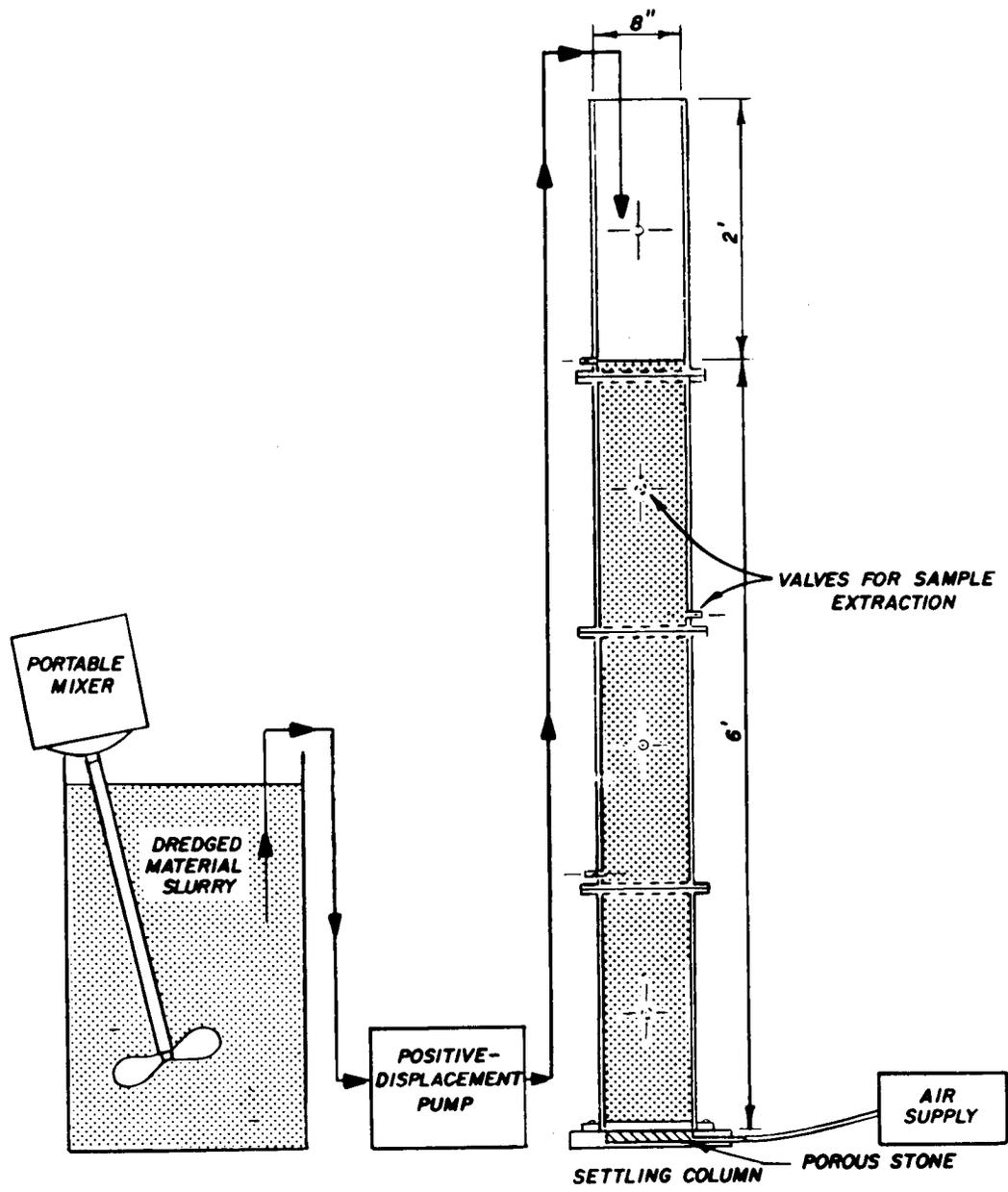


Figure 2. Schematic of apparatus for column settling tests (Montgomery 1978)

Flocculent settling test

Test data required to design or evaluate a disposal area in which flocculent settling governs and to predict the concentration of suspended solids in the effluent can be obtained using procedures described by Montgomery (1978) and Palermo, Montgomery, and Poindexter (1978). The flocculent settling test consists of withdrawing samples from each sample port at regular time intervals to determine the concentration of suspended solids at various depths.

### Zone settling test

Information required to design or evaluate a disposal area in which zone settling governs can be obtained by conducting a series of zone settling tests (Montgomery 1978 and Palermo, Montgomery, and Poindexter 1978). One of the tests should be performed on sediment slurries at a concentration equal to the expected mean field inflow concentration. This test should be continued for a period of at least 15 days to provide data for estimating volume requirements and to obtain data for prediction of effluent suspended solids concentrations.

The procedures described below include those modifications of the procedures described by Montgomery (1978) and Palermo, Montgomery, and Poindexter (1978) required to define the flocculent process governing the sedimentation of suspended solids above the interface. The flocculent settling test as modified consists of measuring the concentration of suspended solids at various depths and time intervals by withdrawing samples from the settling column and timing the fall of liquid/solids interface.

Step 1 - Slurry preparation and loading. Mix the sediment slurry to the desired suspended solids concentration in a container with sufficient volume to fill the test column. The test should be performed at the concentration  $C_i$  selected to represent the anticipated concentration of the dredged material influent. Field studies indicate that for maintenance dredging in fine-grained material, the disposal concentration will average about 150 g/l. This value may be used for  $C_i$  if no better data are available.

Step 2 - Settling and sampling. For sediments exhibiting zone settling behavior, an interface will form between the more concentrated settled material and the clarified supernatant water. The first sample should be extracted immediately after the interface has fallen sufficiently below the uppermost port to allow extraction. This sample can usually be extracted within a few hours after initiation of the test, depending on the initial slurry concentration and the spacing of ports.

As the interface continues to fall, extract samples from all ports above the interface at regular time intervals. Substantial reductions of suspended solids will occur during the early part of the test, but reductions will lessen at longer retention times. Therefore, the intervals can be extended as the test progresses. A suggested sequence of intervals would be 2, 4, 8, 12, 24, 48, 96 hr, etc. Continue to take samples throughout the 15-day test or until the suspended solids concentration of the extracted samples shows no decrease. Record the time of extraction and the port height for each port sample taken (Figure 3).



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- Palermo, M. R., Montgomery, R. L., and Poindexter, M. 1978. "Guidelines for Designing, Operating, and Managing Dredged Material Containment Areas," Technical Report DS-78-10, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
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# *Environmental Effects of Dredging Technical Notes*



## INTERIM GUIDANCE FOR PREDICTING QUALITY OF EFFLUENT DISCHARGED FROM CONFINED DREDGED MATERIAL DISPOSAL AREAS--DATA ANALYSIS

**PURPOSE:** The following series of technical notes described the functions necessary for predicting the quality of effluent discharged from confined dredged material disposal areas during dredging operations.\*

EEDP-04-1	General
EEDP-04-2	Test Procedures
EEDP-04-3	Data Analysis
EEDP-04-4	Application

The guidance was developed as a part of on-going research conducted under the Long-Term Effects of Dredging Operation (LEDO) Program. Procedures for such predictions are being refined and verified under LEDO through comparative evaluation of predictions and field measurement of effluent water quality.

**BACKGROUND:** Confined dredged material disposal has increased because of constraints on open-water disposal. The quality of water discharged from confined disposal areas during disposal operations (effluent) is a major environmental concern associated with such disposal.

Dredged material placed in a disposal area undergoes sedimentation that results in a thickened deposit of material overlaid by clarified water (called supernatant), which is discharged as effluent from the site during disposal operations. The concentrations of suspended solids in the effluent can be determined by column settling tests.

The effluent may contain both dissolved and particle-associated contaminants. A large portion of the total contaminant content is particle associated. The modified elutriate tests was developed for use in predicting

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\* The modified elutriate test does not account for long-term geochemical changes that may occur following disposal and subsequent drying of the dredged material and therefore should not be used to evaluate quality of surface runoff from the disposal sites.

both the dissolved and particle-associated concentrations of contaminants in the effluent from confined disposal areas.

REGULATORY ASPECTS: Guidelines have been published to reflect the 1977 Amendments of Section 404 of the Clean Water Act (EPA 1980). Proposed testing requirements define dredged material according to four categories. Category 3 includes potentially contaminated material proposed for confined disposal that has "potential for contamination of the receiving water column only." The proposed testing requirements call for evaluation of the short-term water column impacts of modified elutriate and column settling tests along with operational considerations can be used with appropriate water-quality standards to determine the mixing zone required to dilute the effluent to an acceptable level (Environmental Effects Laboratory 1976, EPA/CE 1977).

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

### Data Analysis

The results of the column settling tests are used to determine the concentrations of suspended solids in the effluent from a confined disposal site.

Sedimentation of freshwater slurries with solids concentrations of less than 100 g/l are generally characterized by flocculent settling properties. When solids concentrations exceed 100 g/l, the sedimentation process may be characterized by zone settling properties in which a clearly defined interface is formed between the clarified supernatant water and the more concentrated settled material. Zone settling properties also govern when the sediment/water salinity is greater than 3 ppt. Recent studies have shown that flocculent settling governs behavior of the suspended solids in the clarified supernatant water above the sediment/water interface for slurries exhibiting an interface.

For the flocculent case, the procedures for data analysis given in Montgomery (1978) and Palermo, Montgomery, and Poindexter (1978) may be used. For the zone settling case, flocculent settling behavior governs in the supernatant water above the interface. Therefore, a modified flocculent data analysis procedure as outlined in the following paragraphs is required. Example calculations are given in Technical Note EEDP-04-4.

Step 1. Compute values of  $z$ , the depth of sampling below the fluid surface as shown in Figure 1. In computing  $\phi$ , the fraction remaining, the highest concentration of the first port samples is considered the initial concentration  $SS_0$ .



Step 2. Plot the values of fractions remaining  $\phi$  and  $z$  using column settling data to form a concentration profile diagram (Figure 2). Concentration profiles should be plotted for each time of sample extraction.

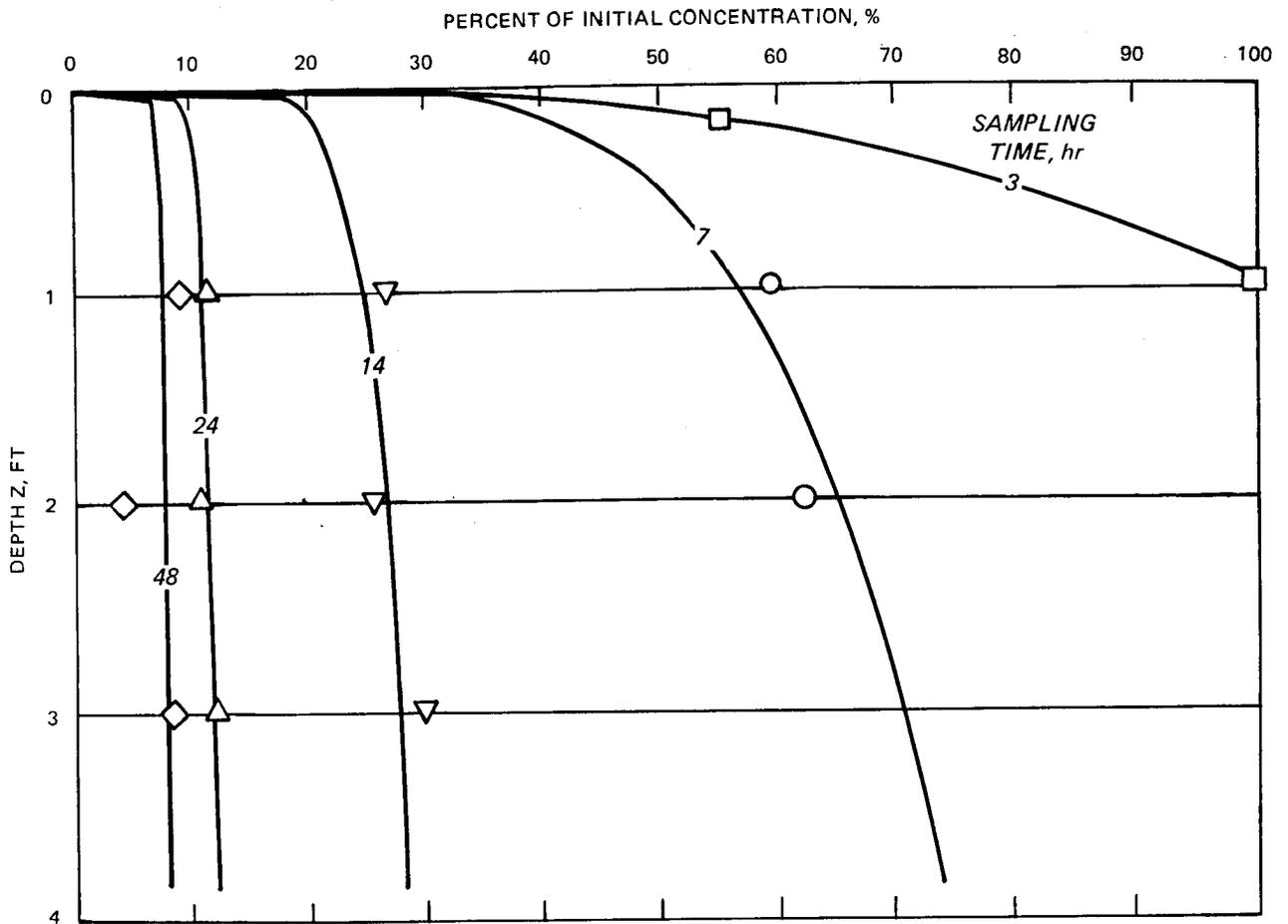


Figure 2. Concentration profile diagram

Step 3. Use the concentration profile diagram to graphically determine  $R$ , the percentages of solids removed for the various time intervals for any desired ponding depth  $D_{pw}$ . This is done by determining the area to the right of each concentration profile and its ratio to the total area above the depth  $D_{pw}$ . The removal percentage  $R$  is calculated as follows:

$$R = \frac{\text{Area Right of Profile}}{\text{Total Area}} 100 \quad (1)$$

Step 4. Compute  $P$ , the percentage of suspended solids remaining in suspension, as simply 100 minus the percentage removed as follows:

$$P = 100 - R \quad (2)$$

Step 5. Compute values for suspended solids for each time of extraction as follows:

$$SS = P \times SS_0 \quad (3)$$

Tabulate  $R$ , and  $P$ , and  $SS$  for each sampling time.

Step 6. Plot a relationship for suspended solids concentration versus time using the value for each sampling time (Figure 3). An exponential or power curve fitted through the data points is recommended.

By repeating steps 4 through 6 for each of several values of  $D_{pw}$ , a family of curves showing suspended solids versus retention time for each of several ponding depths can be developed as shown in Figure 3. These curves can be used for prediction of effluent suspended solids concentrations under quiescent settling conditions for any estimated ponding depth and field mean retention time. Simply enter a curve with the estimated field mean retention time  $T_D$  and select the value of suspended solids as estimated from the column test  $SS_{col}$ . Guidance for adjusting the value derived from the column test for anticipated resuspension and for estimated field mean retention time is given in the following paragraphs.

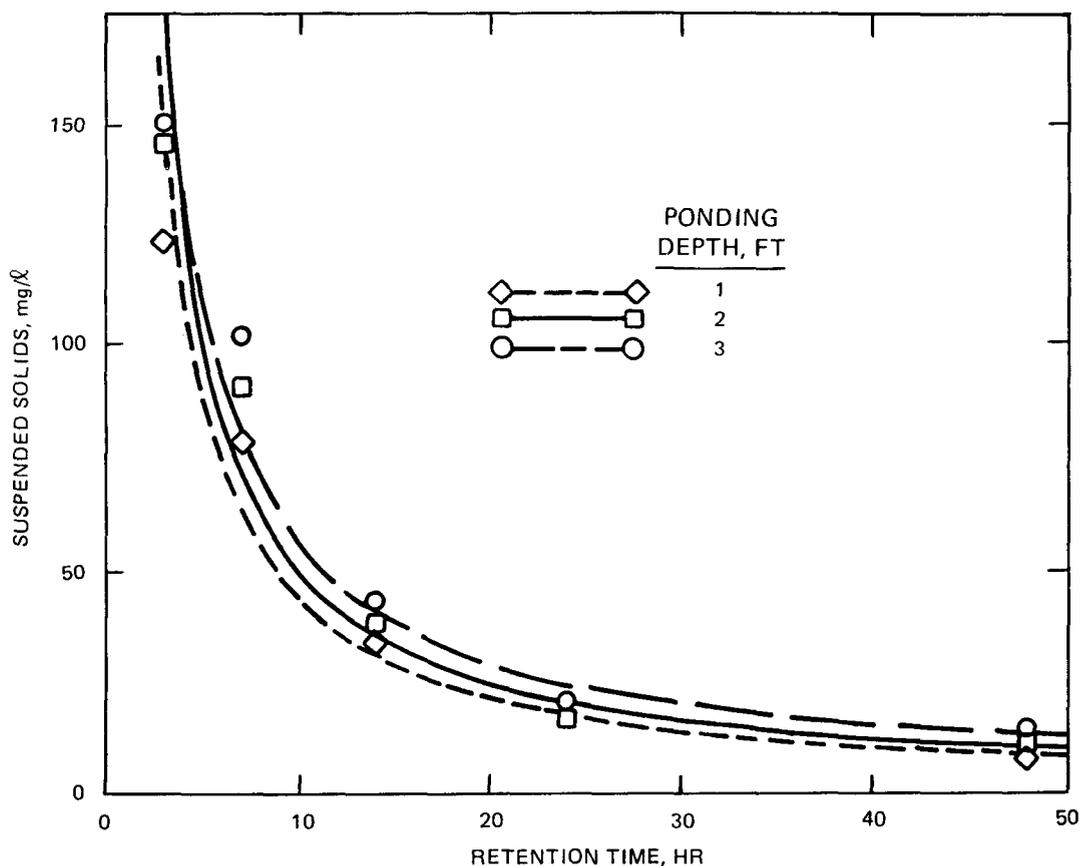


Figure 3. Supernatant suspended solids concentration versus time from column settling test

## Effluent Suspended Solids Concentration

A prediction of the concentration of total suspended solids in the effluent must consider the anticipated retention time in the disposal area and must account for the possible resuspension of settled material because of wind effects. The relationship of supernatant suspended solids versus time developed from the column settling test is based on quiescent settling conditions found in the laboratory. The anticipated retention time in the disposal area under consideration can be used to determine a predicted suspended solids concentration from the relationship. This predicted value can be considered a minimum value that could be achieved in the field assuming little or no resuspension of settled material.

For dredged material exhibiting flocculent settling behavior, the concentration of particles in the ponded water is on the order of 1 g/l or higher. The resuspension resulting from normal wind conditions will not significantly increase this concentration; therefore, an adjustment for resuspension is not required for the flocculent settling case.

However, an adjustment for anticipated resuspension is appropriate for dredged material exhibiting zone settling. The minimum expected value and the value adjusted for resuspension provide a range of anticipated suspended solids concentrations for use in predicting the total concentrations of contaminants in the effluent.

The following tabulation summarizes recommended resuspension factors (RF) based on comparisons of suspended solids concentrations as predicted from column settling tests and field data from a number of sites with various site conditions.

<u>Anticipated Ponded Area acres</u>	<u>Resuspension Factor- Average Ponded Depth</u>	
	<u>Less than 2 ft</u>	<u>2 ft or Greater</u>
Less than 100	2.0	1.5
Greater than 100	2.5	2.0

The value of  $SS_{eff}$ , suspended solids concentration of the effluent considering anticipated resuspension, is calculated using equation 4.

$$SS_{eff} = SS_{col} \times RF \quad (4)$$

where

$SS_{eff}$  = suspended solids concentration of effluent considering anticipated resuspension, milligrams per liter of water

$SS_{col}$  = suspended solids concentration of effluent estimated from column settling tests, milligrams per liter of water

RF = resuspension factor

### Field Mean Retention Time

Estimates of the field mean retention time for expected operational conditions are required for selecting appropriate settling times in the modified elutriate test and for determination of suspended solids concentrations in the effluent. Estimates of the retention time must consider the hydraulic efficiency of the disposal area, defined as the ratio of mean retention time to theoretical retention time. Field mean retention time  $T_d$  can be estimated for given flowrate and ponding conditions by applying a hydraulic efficiency factor to the theoretical detention time  $T$  as follows:

$$T_d = \frac{T}{HEF} \quad (5)$$

where

$T_d$  = mean detention time, hr

$T$  = theoretical detention time, hr

HEF = hydraulic efficiency factor (HEF >1.0) defined as the inverse of the hydraulic efficiency

The theoretical detention time is calculated as follows:

$$T = \frac{V_p}{Q_i} \quad (12.1) = \frac{A_p D_p}{Q_i} \quad (12.1) \quad (6)$$

where

$T$  = theoretical detention time, hr

$V_p$  = volume ponded, acre-ft

$Q_i$  = average inflow rate, cfs

$A_p$  = area ponded, acres

$D_p$  = average depth on ponding, ft

12.1 = conversion factor acre-ft/cfs to hr

The hydraulic efficiency factor HEF can be estimated by several methods. The most accurate estimate for existing sites is made from field dye-tracer data previously obtained at the site under operational conditions similar to those for the operation under consideration. Guidance for conducting such field tests is presented by Schroeder et al. (in preparation).

Hydraulic flow models can also be used to evaluate the efficiency factor. Koussis, Saenz, and Thackston\* recommended steady-state two-dimensional models for such evaluations. Development of such techniques is still under study (Schroeder et al. in preparation).

In absence of dye-tracer data or values obtained from other theoretical approaches, the HEF can be assumed based on values obtained by dye-tracer studies at similar sites and under similar conditions. Montgomery (1978) recommended a value for HEF of 2.25 based on field studies conducted at several sites.

#### Total Concentrations of Contaminants

For each contaminant of interest, the modified elutriate test procedure defines the dissolved concentration and the fraction of the particle-associated contaminant in the total suspended solids under quiescent settling conditions and accounts for geochemical changes occurring in the disposal area during active disposal operations. Using these test results in conjunction with those from column settling tests, the total concentration of the contaminant in the effluent can be determined based on the estimated sedimentation condition as follows:

$$C_{\text{total}} = C_{\text{diss.}} + \frac{F_{\text{SS}} \times \text{SS}_{\text{eff.}}}{1 \times 10^6} \quad (7)$$

where

- $C_{\text{total}}$  = estimated total concentration in effluent, milligrams per liter of water
- $C_{\text{diss.}}$  = dissolved concentration as determined by modified elutriate tests, milligrams per liter of water

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\* A. D. Koussis, M. A. Saenze, and E. L. Thackston. 1982. "Evaluation of Hydraulic Models for Dredged Material Containment Areas," report prepared under contract for the US Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

- $F_{SS}$  = fraction of contaminant in the total suspended solids as calculated from modified elutriate results, milligrams per kilogram of suspended solids
- $SS_{eff.}$  = suspended solids concentration of effluent as estimated from evaluation of sedimentation performance, milligrams per liter of water
- $1 \times 10^6$  = conversion of milligrams per milligram to milligrams per kilogram

The acceptability of the proposed confined disposal operation can then be evaluated by comparing the predicted total contaminant concentrations with applicable water quality standards, considering an appropriate mixing zone. (Environmental Effects Laboratory 1976, EPA/CE 1977).

#### References

- Environmental Effects Laboratory. 1976. "Ecological Evaluation of Proposed Discharge of Dredged Material into Navigable Waters," Miscellaneous Paper D-76-17, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
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# *Environmental Effects of Dredging Technical Notes*



## INTERIM GUIDANCE FOR PREDICTING QUALITY OF EFFLUENT DISCHARGED FROM CONFINED DREDGED MATERIAL DISPOSAL AREAS--APPLICATION

**PURPOSE:** The following series of technical notes describe the functions necessary for predicting the quality of effluent discharged from confined dredged material disposal areas during disposal operations.\*

EEDP-04-1	General
EEDP-04-2	Test Procedures
EEDP-04-3	Data Analysis
EEDP-04-4	Application

The guidance was developed as a part of on-going research conducted under the Long-Term Effects of Dredging Operation (LEDO) Program. Procedures for such predictions are being refined and verified under LEDO through comparative evaluation of predictions and field measurement of effluent water quality.

**BACKGROUND:** Confined dredged material disposal has increased because of constraints on open-water disposal. The quality of water discharged from confined disposal areas during disposal operations (effluent) is a major environmental concern associated with such disposal.

Dredged material placed in a disposal area undergoes sedimentation that results in a thickened deposit of material overlaid by clarified water (called supernatant), which is discharged as effluent from the site during disposal operations. The concentrations of suspended solids in the effluent can be determined by column settling tests.

The effluent may contain both dissolved and particle-associated contaminants. A large portion of the total contaminant content is particle

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\* The modified elutriate test does not account for long-term geochemical changes that may occur following disposal and subsequent drying of the dredged material and therefore should not be used to evaluate quality of surface runoff from the disposal sites.

associated. The modified elutriate test was developed for use in predicting both the dissolved and particle-associated concentrations of contaminants in the effluent from confined disposal areas.

REGULATORY ASPECTS: Guidelines have been published to reflect the 1977 Amendments of Section 404 of the Clean Water Act (EPA 1980). Proposed testing requirements define dredged material according to four categories. Category 3 includes potentially contaminated material proposed for confined disposal that has "potential for contamination of the receiving water column only." The proposed testing requirements call for evaluation of the short-term water column impacts of disposal area effluents. Predicted contaminant levels based on results of modified elutriate and column settling tests along with operational considerations can be used with appropriate water-quality standards to determine the mixing zone required to dilute the effluent to an acceptable level (Environmental Effects Laboratory 1976, EPA/CE 1977).

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

### Example 1: Evaluation of Effluent Water Quality for an Existing Disposal Area

#### Project information

Dredged material from a maintenance project will be placed in an existing disposal site. The site will be ponded over an area of approximately 35 acres. The design indicated that the surface area was adequate for effective sedimentation if a minimum ponding depth  $D_{pw}$  of 2 ft was maintained. The dredging equipment and anticipated pumping conditions will result in a flowrate of approximately 30 cfs. A field mean retention time of 20 hr was determined from a dye tracer test run during earlier disposal operations at this site under similar operational conditions. Previous sampling of inflow from the dredged pipe under similar conditions indicated an influent solids concentrations of approximately 150 g/l.

The quality of effluent must be predicted and compared to applicable water quality standards so that the acceptability of the proposed discharge can be evaluated. A mixing evaluation was conducted, and a dilution factor of 38 was determined for the allowable mixing zone. The water quality standard for copper at the perimeter of the mixing zone was set at 0.004 mg/l (whole water). The concentration of copper in the effluent at the point of discharge must, therefore, be less than 0.15 mg/l.

#### Modified elutriate test

Modified elutriate tests were conducted on samples of sediment and

water from three stations at the proposed dredging site. Modified elutriate tests were run at the anticipated influent solids concentration  $C_{\text{slurry}}$  of 150 g/l. Sediment samples from each sampling station were homogenized.

For one of the homogenized samples, a sediment solids concentration  $C_{\text{sediment}}$  of 450 g/l was determined by oven drying a sample of known volume. The volumes of sediment and water to be mixed to obtain 3-3/4 l of slurry with 150 g of solids per liter was determined as follows:

$$V_{\text{sediment}} = 3.75 \frac{C_{\text{slurry}}}{C_{\text{sediment}}} = 3.75 \frac{150}{450} = 1.25 \quad (1)$$

$$V_{\text{water}} = 3.75 - V_{\text{sediment}} = 3.75 - 1.25 = 2.50 \quad (2)$$

The modified elutriate tests were completed as described in Technical Note EEDP-04-2. A settling time of 20 hr was used since that was the estimated field retention for this case. Samples were extracted for the replicate tests and analyzed for total suspended solids and both dissolved and total concentration of contaminants of concern.

The total suspended solids concentration SS in one of the extracted samples was 40 mg/l. The dissolved concentration  $C_{\text{diss}}$  of copper in this sample was 0.06 mg/l, while the total concentration  $C_{\text{total}}$  of copper was 0.08 mg/l. The fraction of copper in the total suspended solids  $F_{\text{SS}}$  for this sample was determined as follows:

$$\begin{aligned} F_{\text{SS}} &= 1 \times 10^6 \left( \frac{C_{\text{total}} - C_{\text{diss}}}{\text{SS}} \right) \\ &= 1 \times 10^6 \left( \frac{0.08 - 0.06}{40} \right) \text{ or } 500 \text{ mg/kg SS} \end{aligned} \quad (3)$$

These calculations were repeated for other replicate tests, and the average dissolved and particulate copper concentrations were found to be 0.06 mg/l and 510 mg/kg SS, respectively.

#### Column settling test

Samples from all stations were homogenized into a composite for column settling tests. The test used for prediction of effluent suspended solids was run at a slurry concentration of 150 g/l, which was equal to the anticipated influent slurry concentration.

The interface was formed early in the test. Samples were extracted from

all ports above the interface at 3, 7, 14, 24, and 48 hr. The recorded observation and the subsequent computations are shown in Figure 1.

Since an interface formed in the test, the slurry mass was undergoing zone settling. Therefore, the initial supernatant solids concentration  $SS_0$  was assumed equal to the highest concentration of the first port samples taken, 169 mg/ℓ. In computing  $\phi$  and constructing the concentration profile diagram (Figure 2), 169 mg/ℓ was used as  $\phi = 100$  percent.

The concentration profile diagram (Figure 2) was used for graphical determination of  $R$ , the percentage of solids removed, for the various time intervals at  $z = 1, 2,$  and  $3$  ft, which was the range of anticipated depths of withdrawal influence at the weir. This was done by using a planimeter to measure the area to the right of each concentration profile (defined by circled numbers in the figure) and computing its ratio to the total area above 1, 2, and 3 ft.

An example calculation of removal percentage for the concentration profile at  $T = 14$  hr and a depth of influence of 2 ft is as follows:

$$R_{14} = \frac{\text{Area Right of Profile}}{\text{Total Area}} 100 = \frac{\text{Area 1-2-3-0}}{\text{Area 1-2-4-0}} 100 \text{ or } 78 \text{ percent} \quad (4)$$

The percentage of solids remaining at  $T = 14$  hr was found as follows:

$$P_{14} = 100 - R_{14} = 100 - 78 \text{ or } 22 \text{ percent} \quad (5)$$

The value for the suspended solids remaining at  $T = 14$  hr was determined as follows:

$$SS_{14} = \frac{P_{14}}{100} \times SS_0 = 0.22 \times 169 \text{ or } 37 \text{ mg/ℓ} \quad (6)$$

Values at other times were determined in a similar manner. The data for the 2-ft depth of influence were compiled as shown in the following tabulation.

Sample Extraction Time $t$ , hr	Removal Percentage $R_t$	Remaining Percentage $P_t$	Suspended Solids $SS_t$ , mg/ℓ
3	14	86	145
7	47	53	90
14	78	22	37
24	90	10	17
48	94	6	10



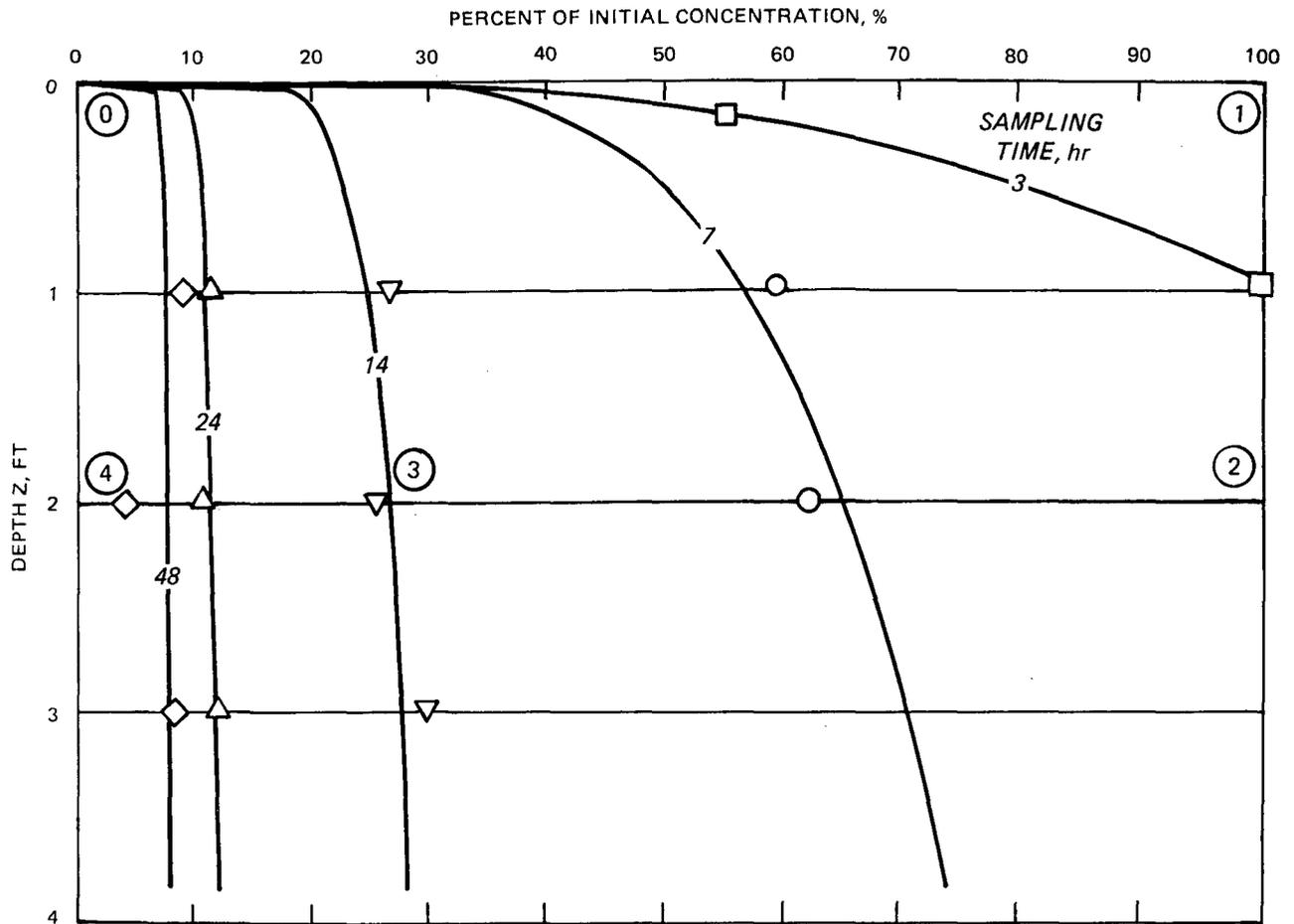


Figure 2. Concentration profile diagram

Similar calculations for other depths of influence were made. Curves were fitted to the total suspended solids versus retention time for depths of influence of 1, 2, and 3 ft, as shown in Figure 3.

Prediction of effluent suspended solids concentration

A value for effluent suspended solids can be determined for quiescent settling conditions using the column test relationships. In this case, the field mean retention time of 20 hr corresponds to a suspended solids concentration  $SS_{CO1}$  of 24 mg/l, as shown in Figure 3. This value should be adjusted for anticipated resuspension using the resuspension factors as given in Technical Note EEDP-04-3:

Anticipated Poned Area acres	Resuspension Factor- Average Poned Depth	
	Less than 2 ft	2 ft or Greater
Less than 100	2.0	1.5
Greater than 100	2.5	2.0

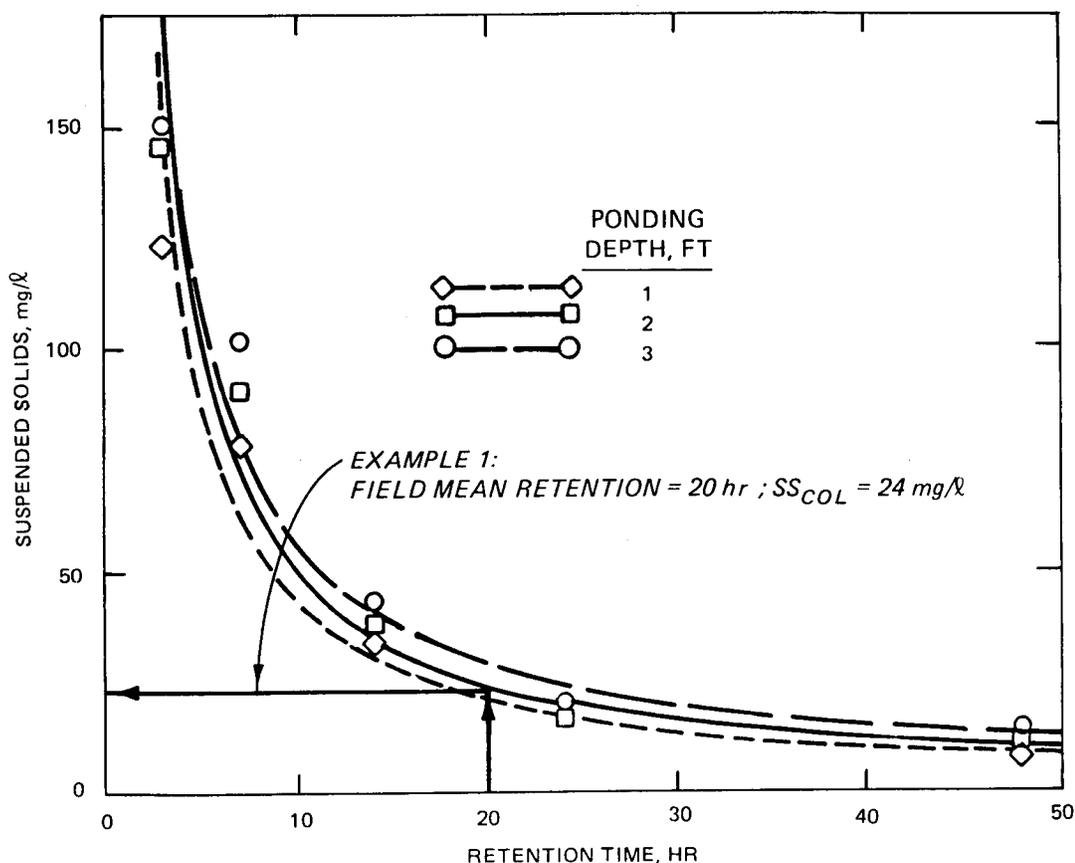


Figure 3. Suspended solids concentration estimated from column settling test

In this case, for a surface area less than 100 acres and average ponding depth of 2 ft, the resuspension factor  $RF$  is 1.5. The predicted total suspended solids concentration  $SS_{eff}$  in the effluent is calculated as follows:

$$SS_{eff} = SS_{col} \times RF = 24 \text{ mg/l} \times 1.5 \text{ or } 36 \text{ mg/l} \quad (7)$$

#### Prediction of contaminant concentrations

The modified elutriate test results indicated that the concentration of dissolved copper  $C_{diss}$  would be 0.06 mg/l and that the fraction of copper in the total suspended solids  $F_{SS}$  would be 510 mg/kg. The predicted total suspended solids concentration in the effluent  $SS_{eff}$  is 36 mg/l. The predicted concentration of total copper in the effluent  $C_{total}$  is calculated as follows:

$$C_{total} = C_{diss} + \frac{F_{SS} \times SS_{eff}}{1 \times 10^6} = 0.06 + \frac{510 \times 36}{1 \times 10^6} = 0.078 \text{ or } 0.08 \text{ mg/l} \quad (8)$$

The estimated concentrations of other contaminants in the disposal area

effluent can be determined in a similar manner. The acceptability of the proposed discharge can be evaluated by comparing the estimated effluent concentrations with applicable water-quality standards, considering an appropriate mixing zone. For total copper, the predicted concentration of 0.08 mg/l at the point of discharge is less than the maximum of 0.15 mg/l specified in the water-quality standards. The discharge would therefore be acceptable.

Example 2: Determination of Disposal Area Requirements  
to Meet a Given Effluent Quality Standard

Project information

A disposal area is planned for contaminated sediment from a small maintenance dredging project. Dredging plant traditionally used in the project area is capable of flowrates up to 15 cfs. Available real estate in the project vicinity is scarce with the maximum available area limited to 60 acres. The minimum disposal area requirements to meet applicable water-quality standards must be determined.

The design using procedures described by Montgomery (1978) and Palermo, Montgomery, and Poindexter (1978) indicated that a minimum ponded surface of 20 acres was required for effective sedimentation, assuming a flow rate of 15 cfs and a minimum ponding depth of 2 ft. A mixing evaluation was conducted and a dilution factor of 60 was determined for the allowable mixing zone. The water-quality standard for PCB at the perimeter of the mixing zone was set at 0.00003 mg/l. The concentrations of PCB in the effluent (at the point of discharge) must therefore be less than 0.0018 mg/l to meet the standards, considering an appropriate mixing zone.

Modified elutriate test

Modified elutriate tests were conducted and calculations made as described for Example 1. For this example, the mean field retention time for the proposed disposal area was not known, so the maximum laboratory retention of 24 hr was used for the tests. Since the inflow concentration was not known, the tests were run at a slurry concentration of 150 g/l. Results for replicate tests for this example were 0.001 mg/l for the concentration of dissolved PCB  $C_{diss}$  and 44 mg/kg for the fraction of PCB in the total suspended solids  $F_{SS}$ .

Column settling test

Column settling tests were run at a slurry concentration of 150 g/l ,

and the resulting concentration profile was developed as in Example 1 (Figure 2). For simplicity, the results of the column settling tests used in Example 1 will also be used for this example.

Determination of required effluent suspended solids concentration

Since this requires determination of disposal site characteristics to meet a given water-quality standard, the calculations proceeded in a manner similar to Example 1, but in a reverse sequence. The concentration of effluent suspended solids  $SS_{eff}$  required to meet water-quality standards must first be determined. For total PCB  $C_{total}$ , the standard at the point of discharge is 0.0018 mg/l. The suspended solids concentration required to meet this standard is calculated as follows:

$$C_{total} = C_{diss} + \frac{F_{SS} \times SS_{eff}}{1 \times 10^6} \quad (9)$$

or transposed,

$$\begin{aligned} SS_{eff} &= \frac{1 \times 10^6}{F_{SS}} (C_{total} - C_{diss}) \\ &= \frac{1 \times 10^6}{44.0} (0.0018 - 0.001) \text{ or } 18 \text{ mg/l} \end{aligned}$$

Based on this calculation, the effluent suspended solids concentration cannot exceed 18 mg/l without exceeding the standard for PCB. Similar determinations should be made for other contaminants being considered in order to define the limiting value for the required effluent suspended solids concentration. For this example, 18 mg/l was used as the limiting value.

Since the final site configuration is not known, a conservative resuspension factor  $RF$  should be selected from the tabulation given in Example 1. The minimum ponding depth of 2 ft required by the site design is used. A resuspension factor of 1.5 was selected corresponding to an area less than 100 acres and ponding depth of 2 ft.

The value of 18 mg/l suspended solids (including resuspended particles) must be met at the point of discharge. The corresponding value for total suspended solids concentration under quiescent settling condition is determined by transposing Equation 7 ( $SS_{eff} = SS_{col} \times RF$ ) as follows:

$$SS_{COL} = \frac{SS_{eff}}{RF} = \frac{18 \text{ mg/l}}{1.5} \text{ or } 12 \text{ mg/l}$$

The required configuration of the disposal area must correspond to a retention time that will allow the necessary sedimentation. The required retention time to achieve 12 mg/l under quiescent settling conditions can be determined from the laboratory column relationship for suspended solids versus retention time.

Using the concentration profile data and the assumed depth of ponding at the weir of 2 ft, the relationship for suspended solids versus field mean retention was developed as shown in Figure 4. Using Figure 4, 12 mg/l corresponds to a field mean retention time  $T_d$  of 36 hr. To determine the required disposal site geometry, the theoretical retention time  $T$  should be used. Since no other data were available, the hydraulic efficiency factor HEF was assumed as 2.25. The theoretical retention time  $T$  was calculated as follows:

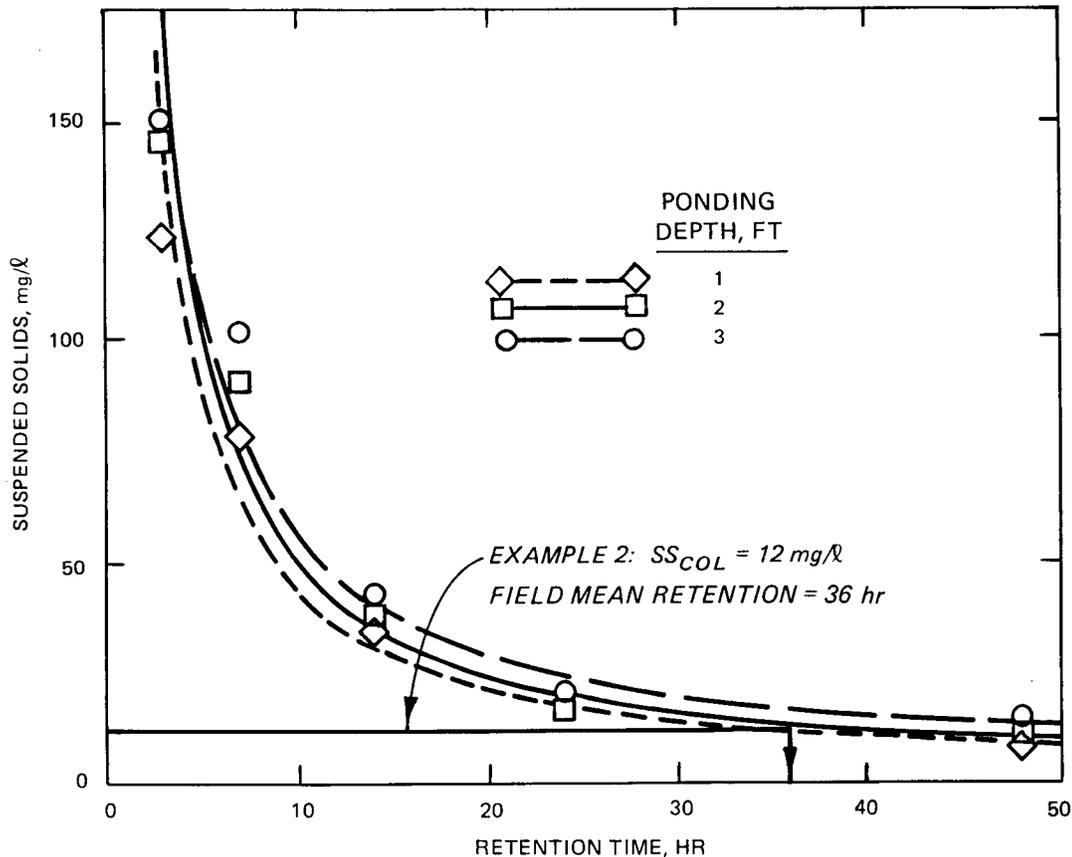


Figure 4. Field mean retention time estimated from column settling test

$$t_d = \frac{T}{(HEF)} \quad (10)$$

transposed to

$$T = T_d (HEF) = 36 (2.25) \text{ or } 81 \text{ hr}$$

Determination of disposal area configuration

The disposal area configuration can now be determined using data on anticipated flowrate and the required retention time. Since the dredging equipment available in the project area is capable of flowrates up to 15 cfs, the high value should be assumed.

The pond volume required is calculated as follows:

$$T = \frac{V_p}{Q_i} \quad (11)$$

transposed to

$$V_p = \frac{T Q_i}{12.1} = \frac{81 \text{ hr} \times 15 \text{ cfs}}{12.1} \text{ or } 100 \text{ acre-ft}$$

A ponding depth of 2 ft is the minimum required. This same depth should be maintained over the entire ponded surface area and at the weir. The disposal site should, therefore, encompass approximately 50 acres of ponded surface area if the dredge selected for the project has an effective flowrate not greater than 15 cfs. The surface area of 50 acres required to meet the water-quality standard controls over the design surface area of 20 acres required for effective sedimentation.

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Environmental Effects Laboratory. 1976. "Ecological Evaluation of Proposed Discharges of Dredged Material into Navigable Water," Miscellaneous Paper D-76-17, US Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

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# *Environmental Effects of Dredging Technical Notes*



## INTERIM PROCEDURES FOR ESTIMATING MIXING ZONES FOR EFFLUENT FROM DREDGED MATERIAL DISPOSAL SITES (Single-Point Discharge)

**PURPOSE AND SCOPE:** This technical note presents a simple analytical method for evaluating the size of mixing zones for effluents from confined dredged material disposal areas. The method involves a simplistic two-dimensional calculation based on dispersion principles. Discussions of the applicability and limitations of the technique and a stepwise procedure for performing the required calculations are presented. Recommendations for using computer-based approaches for more complex conditions are also discussed.

**BACKGROUND:** Contaminated dredged material that is unsuitable for disposal in open water is normally placed in confined upland disposal areas. The quality of effluent discharged from these sites is an environmental concern and is regulated as a discharge under Section 404 of the Clean Water Act. Further, water quality certification may be required from state regulatory agencies, and effluent standards may be set as a condition of the certification. This technical note addresses one aspect of the required estimation techniques: the evaluation of mixing-zone sizes in receiving waters for the effluent from confined disposal sites.

Whenever contaminant concentrations in an effluent at the point of discharge from a confined disposal site are above receiving water quality standards, there will be some limited initial mixing zone (or zone of dilution) in the vicinity of the discharge point where receiving water quality standards may be exceeded. The size of this mixing zone depends on a number of factors including the contaminant concentrations in the effluent, the applicable water quality standards, effluent density and flow rate, receiving water flow rate and turbulence, and the geometry of the discharge structure and the receiving water boundaries. Since the maximum allowable mixing zone specified by regulatory agencies is usually on the order of hundreds of meters, the evaluation of mixing-zone sizes must necessarily be based on calculation of near-field dilution and dispersion processes.

Mixing-zone sizes can be best determined from field studies at the proposed discharge site using dye plumes to measure dispersion. However, this method is time-consuming and expensive; therefore, a technique utilizing calculations based on theoretical and empirical studies of effluent dispersion is desirable. There are a variety of possible estimation techniques for most real mixing-zone problems, but any choice of a suitable technique involves some tradeoffs. The available techniques may be thought of as ranging from sophisticated computer models, which are sometimes capable of very accurate

predictions, to simple approximations that yield order-of-magnitude estimates. The most sophisticated models will not usually run on a microcomputer, and they may require a considerable amount of measured data and manpower for calibration of the model to a single site. By contrast, the simplest of approximations may be made on the basis of several simplifying assumptions and hand calculations.

An example of a simple approach to mixing-zone size estimation was proposed for discharges of dredged material into navigable waters (Environmental Effects Laboratory 1976). This report presented a simplified method to calculate mixing-zone sizes for offshore dumping of dredged material. The method involved the use of some characteristic plume shapes and an estimate of the amount of the dispersion based on a wide range of experimental studies. Unfortunately this method can only be applied in the case of open-water disposal well away from boundaries and flow restrictions. The method is not applicable to mixing zones for effluent from containment areas, which is generally discharged into relatively shallow water necessarily close to such boundaries as the river or estuary bottom and bank.

This technical note presents a similar type of simplified approach that is applicable to relatively shallow confined riverine water bodies. If the mixing-zone size as calculated using simple approximations is within mixing-zone guidelines specified by regulatory agencies, more precise calculations may not be necessary.

(Note: The mixing-zone calculations described in this technical note depend on a number of assumptions that are difficult to satisfy for estuaries and the tidally influenced portions of rivers. The difficulties are discussed after the presentation of the procedure to be used for a riverine environment.)

REGULATORY ASPECTS: The Federal regulations (Environmental Protection Agency 1980a) that apply to all discharges of dredged or fill material into waters of the United States are contained in pages 85336-85337 of the Federal Register, Volume 45. Since confined disposal area effluents are usually discharged into waters of the United States, nearly all containment area discharges fall under the jurisdiction of these regulations. In Part 230.3(m), the mixing-zone of a discharge is defined as "a limited volume of water serving as a zone of initial dilution in the immediate vicinity of a discharge point where receiving water quality may not meet quality standards or other requirements otherwise applicable to the receiving water." The guidelines recognize that it is not possible to set universal standards for the acceptable size of mixing zones since receiving water conditions vary so much from one location to another. The guidelines therefore instruct that, as part of the dredging permit process, the size of any proposed mixing zone should be estimated and submitted to the permitting authority. The permitting authority must then consider receiving water conditions at the proposed site and decide if the proposed mixing-zone size is acceptable.

Many state regulatory agencies may specify a limit to mixing-zone dimensions as a condition in granting the state water quality certification for a confined disposal operation. In this case the mixing zone necessary to meet applicable standards must be smaller than the specified limits.

Proposed amendments (Environmental Protection Agency 1980b) to the Section 404 regulations, given in pages 85360-85367 of the Federal Register, Volume 45, deal with the testing requirements for the specification of

dredged material disposal sites. If they are approved, they will replace Parts 230.60 and 230.61 in the current regulations. These proposed amendments give much more detailed guidance on how mixing-zone sizes should be estimated for permit applications. They recommend that effluent contaminant concentrations should be estimated by means of a modified elutriate test (Palermo 1986). There is also a general description of appropriate approaches to estimating mixing-zone size once the concentration of the most critical effluent pollutant has been estimated. The recommended approaches are as follows:

"(a) Mixing Zone Calculations. The perimeter of the mixing zone shall be defined by the applicable water quality standard of the contaminant requiring the greatest dilution volume or by 0.01 of the lowest 96-hour LC50 when a water column bioassay has been conducted.

(1) One of the following methods (provided in order of preference) shall be used to determine the volume and conformation of the zone required to achieve dilution and dispersal of contaminants to numerical limitations specified in Federal or State water quality standards or to 0.01 of the lowest 96-hour LC50, as indicated above.

(i) When field data on the proposed discharge are adequate to predict the initial dispersion and diffusion of the discharge plume, such data shall be used; or

(ii) When field data on the dispersion and diffusion of a discharge with similar characteristics are available, these data shall be used in conjunction with an appropriate mathematical model (acceptable to the permitting authority) to make the required determination; or

(iii) When the above methods are impractical, due to inadequate field data or the unavailability of an appropriate mathematical model, the zone of dilution and dispersion may be estimated by assuming particular geometrical shapes for the disposal plume."

The estimation techniques presented in this technical note would fall under class iii of the proposed amendments.

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## Analytical Method for Estimating Mixing Zones

The analytical solution technique for calculating mixing-zone size described in this section is based on theoretical and empirical relationships for dispersion as summarized by Fischer et al. (1979). Only equations for calculating mixing-zone size resulting from a single-point discharge are presented. Development of the specific equations used, additional equations for multiple points of discharge, and more detailed discussions of the solution techniques are presented by MacIntyre (in review).

A schematic illustrating a typical single-source effluent discharging into a receiving water body is shown in Figure 1. For such a condition, the mixing-zone length extends downstream and the body of the mixing zone remains close to the shoreline of the receiving water body.

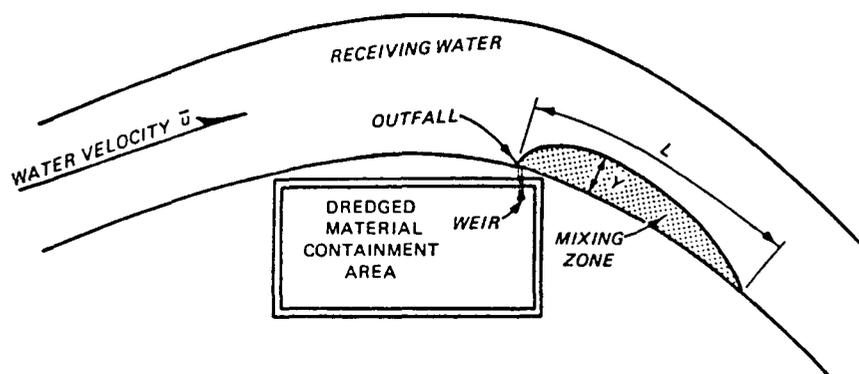


Figure 1. Schematic of a mixing zone for a single effluent source

### Data requirements

The following data are required for evaluating mixing-zone sizes for confined disposal area effluents:

- a. Effluent concentrations at the point of discharge and receiving water background concentrations for all contaminants of concern.
- b. Water quality standards applicable at the limit of the allowable mixing zone for all contaminants of concern.
- c. Depth, cross-sectional area, and current velocity of the receiving water body during expected low flow conditions.
- d. Effluent volumetric flow rate.

### Calculation procedure

The stepwise procedure for calculating mixing-zone sizes is summarized as follows:

Step 1. Check that the assumptions on which the equations depend are reasonable for conditions at the proposed discharge site.

Step 2. Use effluent, receiving water, and water quality standard concentrations of all contaminants of concern to identify the critical contaminant. The critical contaminant is the one that requires the greatest dilution, which will define the boundary of the mixing zone.

Step 3. Use receiving-water depth and velocity data to calculate a lateral mixing coefficient. This coefficient is a measure of how rapidly the effluent is dispersed through the receiving water.

Step 4. Calculate mixing-zone length.

Step 5. Check assumptions that depend on mixing-zone length.

Step 6. Calculate the maximum width of the mixing zone.

Step 1 - Assumptions. In order to apply the analytical solution described in this section, the following assumptions are required:

- a. No major change in cross-sectional shape, sharp bends, major inflows or outflows, or obstructions to flow exist in the receiving water body in proximity to the mixing zone.
- b. The receiving water body can be reasonably approximated by a shallow rectangular cross section.
- c. The confined disposal area effluent enters the receiving water as a point source at the bank with negligible horizontal momentum.
- d. Differences in density between the effluent and receiving water and in settling rates of suspended particles within the boundary of the mixing zone are negligible.
- e. The flow condition in the vicinity of the mixing zone can be approximated as a steady-state velocity flowing parallel to the bank of the receiving water.
- f. The major cause of dispersion in the receiving water body is the turbulence and shear flow associated with the horizontal water flow.
- g. The effluent plume is vertically well mixed, so that contaminant concentrations do not vary significantly with depth.
- h. The width of the effluent plume is small enough that its lateral dispersion is not restricted by the opposite bank of the receiving water body.

Step 2 - Identify critical contaminant. It is necessary to calculate the dilution required within the mixing zone in order to reach applicable water quality standards for all contaminants of concern. This requires an estimate of the effluent concentrations of regulated contaminants. The contaminant that requires the greatest amount of dilution will define the maximum boundary of the mixing zone.

Effluent from containment areas may contain chemical contaminants that are dissolved and/or adsorbed to suspended particles. The total effluent concentration of any contaminant consists of the sum of the concentrations of that contaminant in the dissolved and adsorbed states. This concentration should be estimated for each contaminant of concern by using a modified elutriate test. Procedures for use of the modified elutriate test and calculation of the required concentrations were published by Palermo (1986), with supplemental guidance on estimating the total suspended solids (TSS) concentration in the effluent. For each contaminant of concern, including TSS, the required dilution should be calculated as:

$$D = \frac{C_e - C_s}{C_s - C_b} \quad (1)$$

where

D = dilution factor required to dilute the contaminant of concern to the appropriate water quality standard  $C_s$ , vol/vol

$C_e$  = concentration of contaminant of concern in the effluent water,  $\text{mg}/\ell^{-1}$

$C_s$  = receiving water quality standard for the contaminant of concern,  $\text{mg}/\ell^{-1}$

$C_b$  = background concentration of the contaminant of concern in the receiving water,  $\text{mg}/\ell^{-1}$

The maximum boundary of the mixing zone will be defined as the isopleth (line of constant concentration) where the concentration of the most critical contaminant is reduced to the concentration specified by the appropriate water quality standard. It should be noted that if background concentrations exceed the water quality standard, the concept of a mixing zone is inapplicable.

Also, this approach for calculating required dilution is not applicable to turbidity (an optical property of water), which is reduced in a nonlinear fashion by dilution. A correlation curve for TSS versus turbidity (Earhardt

1984) should be used to define the TSS concentration corresponding to the water quality standard for turbidity. This TSS concentration can be used as a value for  $C_s$  and an appropriate TSS dilution calculated.

Step 3 - Estimate of lateral mixing coefficient.

Step 3.1. The depth of a simplified rectangular cross section for the receiving water body should be calculated as follows:

$$d = \frac{A}{W} \quad (2)$$

where

$d$  = average depth of the receiving water body channel, m

$A$  = cross-sectional area of the channel,  $m^2$

$W$  = surface width of the channel, m

Check to ensure that  $W$  is equal to or greater than 10 times the average depth  $d$ . If not, the estimate of a lateral mixing coefficient is likely to be inaccurate.

Step 3.2. Estimate the shear velocity by one of the following methods. In rivers where the mean channel slope is known, use:

$$u^* = \sqrt{gdS} \quad (3)$$

In rivers where the channel slope is not known, use:

$$u^* = 0.1 \bar{u} \quad (4)$$

where

$u^*$  = shear velocity in receiving water,  $m/sec^{-1}$

$g$  = gravitational acceleration,  $9.81 m/sec^{-2}$

$d$  = average channel depth, m (equation 2)

$S$  = slope of river bed (dimensionless)

$\bar{u}$  = average of instantaneous velocities across the channel cross section,  $m/sec^{-1}$ .

If the flow rate of the receiving water is known,  $\bar{u}$  can be calculated as the flow rate divided by the channel cross-sectional area. If the receiving-water flow rate is not known,  $\bar{u}$  must be determined from velocity measurements taken at the proposed site. It should be noted that  $\bar{u}$

should not be determined over a period of time during which velocity changes occur due to changes in the receiving-water flow rate.

Step 3.3. Estimate the lateral mixing coefficient by using one of the following equations.

In rivers:

$$E_t = 0.3 du^* \quad (5)$$

In estuaries:

$$E_t = 0.4 du^* \quad (6)$$

where

$E_t$  = lateral mixing coefficient,  $m^2/sec^{-1}$

$d$  = average channel depth, m (equation 2)

$u^*$  = shear velocity,  $m/sec^{-1}$  (equations 3 or 4)

The values of lateral mixing coefficient are derived from Fischer et al. (1979) and are based on experimental studies of dispersion in various rivers. Lateral mixing coefficients have been shown to vary widely from one location to another, and equations 5 and 6 give the lowest reasonable values so that estimates of mixing zone size will be conservative.

Step 4 - Estimate mixing-zone length. If the assumptions presented earlier are valid, the mixing zone will have a shape similar to the one shown in Figure 1. The length of the mixing zone (measured parallel to the bank) can be estimated as:

$$L = \left( \frac{1}{\pi E_t u} \right) \left[ \frac{Q_e C_e}{(C_s - C_b) d} \right]^2 \quad (7)$$

where

$L$  = mixing zone length, m

$Q_e$  = effluent volumetric discharge rate,  $m^3/sec^{-1}$

Step 5 - Check length-dependent assumptions.

Step 5.1. The flow in the water body near the mixing zone can be treated as a steady-state flow as long as:

$$L \leq \frac{\bar{u} T_c}{10} \quad (8)$$

where

$L$  = predicted mixing zone length, m (Equation 7)

$\bar{u}$  = cross-sectional average velocity (instantaneous or averaged over a few minutes),  $\text{m/sec}^{-1}$

$T_c$  = time taken for the observed value of  $\bar{u}$  to change by 10 percent, sec

Step 5.2. The lateral dispersion of the effluent plume will not be restricted by opposite bank of the receiving water body as long as:

$$W > \sqrt{\frac{8 E_t L}{\bar{u}}} \quad (9)$$

where  $W$  = surface width of receiving water channel, m.

Step 6 - Estimate maximum width of mixing zone. The maximum width of the mixing zone (measured perpendicular to the bank as shown in Figure 1) can be estimated as:

$$Y = \frac{0.484Q_e C_e}{\bar{u}(C_s - C_b)d} \quad (10)$$

where  $Y$  = maximum width of the mixing zone, m.

Example mixing-zone problem

Following is a hypothetical mixing-zone problem designed to illustrate the use of the mixing-zone estimation equations. A proposed dredged material containment area is expected to discharge into a river 480 ft (146.3 m) wide. From a study of US Geological Survey stream gage records, it is anticipated that while effluent will be discharged, the lowest river flow will be about 7,600  $\text{ft}^3/\text{sec}$  (212.8  $\text{m}^3/\text{sec}$ ) and that the river has a cross-sectional area of 4,000  $\text{ft}^2$  (371.6  $\text{m}^2$ ) at this flow rate. The local bed slope of the river is known to be very variable due to sediment transport. The containment area is expected to have a peak discharge of 15 cfs. The only effluent contaminant that exceeds water quality standards will be cadmium, which is expected to have an effluent concentration of 3.5  $\mu\text{g}/\text{l}$ . The background concentration of cadmium in the river is below the detection limit of 0.1  $\mu\text{g}/\text{l}$ , and the applicable cadmium water quality standard is 0.25  $\mu\text{g}/\text{l}$ . It has been specified that the maximum acceptable mixing-zone size is a 750-ft (228.6-m) radius centered on the effluent outfall. Is the size of the mixing zone likely to exceed this limit?

Step 1 - Assumptions. Since the purpose of this hypothetical problem is to demonstrate the use of the mixing-zone calculations, it has been defined so that all the assumptions on which the calculations depend are valid. Decisions on whether the assumptions are valid depend largely on the professional judgement of personnel familiar with the disposal site. More detailed guidance on which types of local conditions will satisfy the assumptions is given by MacIntyre (in review).

Step 2 - Identify critical contaminant. Cadmium is the only effluent contaminant that exceeds water quality standards. It is therefore unnecessary to use equation 1 to determine the critical contaminant, because cadmium is the only possibility.

Step 3 - Estimate lateral mixing coefficient.

Step 3.1. From the problem statement,

$$A = 4,000 \text{ ft}^2 \text{ (371.6 m}^2\text{)}$$

$$W = 480 \text{ ft (146.3 m)}$$

Calculate depth from equation 2:

$$d = \frac{A}{W}$$

$$d = \frac{371.6 \text{ m}^2}{146.3 \text{ m}} = 2.54 \text{ m}$$

Check that  $W \geq 10 d$ . It is.

Step 3.2. Since the local bed slope is known to vary due to sediment transport, the shear velocity should be estimated from the mean velocity. Calculate the mean velocity by dividing the river flow of  $7,600 \text{ ft}^3/\text{sec}$  ( $212.8 \text{ m}^3/\text{sec}$ ) by the cross-sectional area of  $4,000 \text{ ft}^2$  ( $371.6 \text{ m}^2$ ).

$$\bar{u} = \frac{7,600 \text{ cfs}}{4,000 \text{ ft}^2} = 1.90 \text{ ft/sec}^{-1} \text{ (0.579 m/sec}^{-1}\text{)}$$

From equation 4:

$$u^* = 0.1 \bar{u}$$

$$u^* = 0.1(0.579 \text{ m/sec}^{-1}) = 0.0579 \text{ m/sec}^{-1}$$

Step 3.3. In rivers, the lateral mixing coefficient should be estimated from equation 5:

$$E_t = 0.3 d u^*$$

$$E_t = 0.3(2.54 \text{ m})(0.0579 \text{ m/sec}^{-1})$$

$$E_t = 0.0441 \text{ m}^2/\text{sec}^{-1}$$

Step 4 - Estimate mixing-zone length. From the problem statement,

$$Q_e = 15 \text{ cfs } (0.425 \text{ m}^3/\text{sec}^{-1})$$

$$C_e = 3.5 \text{ } \mu\text{g}/\ell^{-1} \text{ (} 3.5 \times 10^{-3} \text{ mg}/\ell^{-1}\text{)}$$

$$C_s = 0.25 \text{ } \mu\text{g}/\ell^{-1} \text{ (} 2.5 \times 10^{-4} \text{ mg}/\ell^{-1}\text{)}$$

$$C_b < 0.1 \text{ } \mu\text{g}/\ell^{-1} \text{ (} 1.0 \times 10^{-4} \text{ mg}/\ell^{-1}\text{)}$$

In order to be conservative, it would be wise to assume that the background concentration is only just under the detection limit, rather than zero. Therefore use:

$$C_b = 1.0 \times 10^{-4} \text{ mg}/\ell^{-1}$$

Calculate mixing-zone length from equation 7:

$$L = \left( \frac{1}{\pi E_t \bar{u}} \right) \left[ \frac{Q_e C_e}{(C_s - C_b) d} \right]^2$$

$$L = \left[ \frac{1}{\pi (0.0441 \text{ m}^2/\text{sec}^{-1}) (0.579 \text{ m}/\text{sec}^{-1})} \right] \left\{ \frac{(0.425 \text{ m}^3/\text{sec}) (3.5 \times 10^{-3} \text{ mg}/\ell^{-1})}{[(2.5 - 1.0) \times 10^{-4} \text{ mg } \ell^{-1}] (2.54 \text{ m})} \right\}$$

$$L = 190 \text{ m (623 ft)}$$

Step 5 - Check length-dependent assumptions.

Step 5.1. Equation 8 requires that:

$$L \leq \frac{\bar{u}T_c}{10}$$

therefore

$$T_c \geq \frac{10L}{\bar{u}}$$

$$T_c \geq \frac{10(190 \text{ m})}{0.579 \text{ m/sec}^{-1}}$$

$$T_c \geq 3,280 \text{ sec (55 min)}$$

This is acceptable since the river flow will certainly not change by 10 percent in less than 1 hour.

Step 5.2. Equation 9 requires that:

$$W \geq \sqrt{\frac{8E_t L}{\bar{u}}}$$

$$W \geq \sqrt{\frac{8(0.0441 \text{ m}^2/\text{sec}^{-1})(190 \text{ m})}{(0.579 \text{ m/sec}^{-1})}}$$

$$W \geq 10.8 \text{ m}$$

This condition is amply satisfied since W equals 146 m.

Step 6 - Estimate maximum width of mixing zone. Estimate the maximum mixing zone width from equation 10:

$$Y = \frac{0.484 Q_e C_e}{\bar{u}(C_s - C_b)d}$$

$$Y = \frac{0.484 (0.425 \text{ m}^3/\text{sec}^{-1})(3.5 \times 10^{-3} \text{ mg/l}^{-1})}{0.579 \text{ m/sec}^{-1} [(2.5 - 1.0) \times 10^{-4} \text{ mg/l}^{-1}](2.54 \text{ m})}$$

$$Y = 3.3 \text{ m (10.7 ft)}$$

Since the mixing zone is predicted to have a length of 623 ft (190 m) and a maximum width of 10.7 ft (3.3 m), it is within the allowable limits of 750 ft (228.6 m) from the effluent outfall.

#### Multiple sources

A similar computational sequence has been devised for the case of multiple sources or points of discharge. This condition would exist if multiple weirs discharge simultaneously to the receiving water body. Detailed procedures for this case are given in MacIntyre (in review).

#### Tidal rivers and estuaries

The mixing-zone equations presented earlier depend on a number of assumptions that are more difficult to satisfy in estuaries and the tidally influenced portions of rivers. These difficulties are reviewed briefly below.

The assumption that velocities in the water body near the mixing zone can be represented by a single mean velocity parallel to the bank is usually a reasonable one in the tidally influenced portion of a river. However, it is not always acceptable in estuaries. Typically the downstream section of an estuary exhibits horizontal circulation patterns, so that the horizontal water velocity and direction vary with distance parallel to the bank, distance perpendicular to the bank, and time. Under these conditions, water near the mixing zone may not always travel parallel to the bank. Therefore, the simple mixing-zone equations presented in this technical note may not be applicable to the wide, open low-velocity sections of estuaries.

Also, the mixing-zone equations are not theoretically applicable as the mean velocity tends to zero. This is because the equations are dependent upon the process of advection, which does not exist in the absence of a flow velocity, and also because the primary source of dispersion is assumed to be the turbulence caused by the horizontal movement of water. However, in a real water body, as the velocity tends to zero, the primary sources of turbulence and dispersion are the wind and waves.

The rate of change of water velocity due to tidal effects can also cause problems. Step 5 presents an equation for steady-state flow conditions: the time taken for material to travel the length of the mixing zone should be an order of magnitude smaller than the time taken for a 10-percent change in the mean water velocity. It may be possible to satisfy this condition in the tidally influenced portion of a river, but it will probably not be possible to do so in most estuaries during a significant portion of the tidal cycle.

Methodologies for estimating the overall mixing-zone size under these conditions by superimposing a series of instantaneous mixing zones are discussed in MacIntyre (in review).

Another potential difficulty in estuaries is the phenomenon of stratification. Estuaries with low water velocities sometimes have a layer of relatively fresh water near the surface with a much more saline denser layer of water near the bottom and with quite a distinct interface between the two layers. The abrupt change of density at the interface tends to inhibit vertical mixing through the entire depth of the water column. Fischer et al. (1979) stated that the equation given for the lateral mixing coefficient in estuaries was derived from studies in the unstratified portions of estuaries. The methods of estimating mixing-zone size that are presented in this report are therefore not recommended for use under conditions in which strong vertical stratification is present in the immediate vicinity of the mixing zone.

#### Computer Modeling of Mixing Zones

The equations presented earlier were derived from a simplistic approach to the problem of estimating mixing-zone size that made it possible to use a combination of empirical and analytical solutions. The simplifications that make the calculations easily manageable are somewhat restrictive, and a more advanced set of similar empirical and analytical solutions could be used to estimate mixing-zone sizes under more complex conditions. The more advanced analytical solutions involve many more computations, and for this reason they are more easily dealt with by use of a computer. The simplicity and limited data requirements of analytical solutions make them an attractive tool. However, analytical solutions cannot be used for receiving water where there are complex hydrodynamic conditions, nor can they be applied under dynamic (unsteady) flow conditions. Where these conditions exist, a numerical model must be used, and numerical dispersion models are not susceptible to hand calculation. In addition to requiring a computer solution technique, numerical models generally require a much more detailed set of input data, and the collection of such data can be expensive.

Vanderbilt University (Saenz and Parker 1984) conducted a study of available computer models suitable for modeling mixing zones. Their report did not identify any models that were suitable for a broad range of

mixing-zone conditions. An updated literature review by MacIntyre (in review) came to the same broad conclusion: there are no readily available models suitable for modeling the first few hundred metres downstream from the discharge point. This is because the overwhelming majority of computer models are concerned with far-field solutions where concentrations can be adequately described by a two-dimensional or a one-dimensional model and the initial characteristics of the discharge are relatively unimportant. These models are generally inadequate in the immediate vicinity of a discharge, where a three-dimensional description of concentrations is often necessary and where the initial characteristics of the discharge can be highly significant. Within the first few hundred metres of the discharge, there are several different processes that may be significant, so a general model must be able to estimate each of the processes (for example, momentum, buoyancy, dispersion) and to identify the zones within which the processes are dominant. A general mixing-zone model must therefore be a series of submodels, each of which can handle a zone that is dominated by one of the principal mixing processes. The submodels must be capable of determining the limits of their applicable zones and passing concentration values at these limits on to other submodels so that the entire mixing zone may be estimated. The following tabulation presents a summary of the steady-state physical processes that might be suitable for inclusion as submodels in a general mixing-zone model. Sources that presently seem to present the most promising empirical and analytical solutions to these submodel processes are also presented in the tabulation.

<u>Physical Process to Be Handled by a Submodel</u>	<u>Source</u>
Momentum and/or buoyancy-dominated jets	Zeller et al. (1971) Mutz and Benedict (1972) Buhler and Hauenstein (1981) Jirka et al. (1981) Wright (1984)
3-dimensional dispersion in receiving water (receiving water channel must be idealized to have a trapezoidal cross section)	Prakash (1977) Fischer et al. (1979)
2-dimensional vertically averaged dispersion in receiving waters (model can handle real channel cross section)	Stefan and Gulliver (1978) Paily and Sayre (1978) Gowda (1984a, b)

It should be noted that a model for computing the fate of continuously discharged dredged material developed by Brandsma and Divoky (1976), and

modified by Johnson (1987) simulates all of the processes above within the framework of a single model. Although modifications would be needed to make the model suitable for a broad range of mixing zone conditions, it should be considered in future developmental efforts to provide a general mixing zone model.

### Summary

Estimation of mixing zones is a necessary step in evaluating discharges from confined dredged material disposal areas whenever contaminant concentrations at the point of discharge are above water quality standards for the receiving water. A simplistic two-dimensional calculation procedure may be used to estimate the mixing-zone size if certain assumptions regarding geometry and flow conditions within the receiving water body are met. For more complex conditions, a numerical model solution would be required. Although no model is readily available that meets all requirements, appropriate solution techniques have been identified.

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# *Environmental Effects of Dredging Technical Notes*



## REGULATORY IDENTIFICATION OF HYDROCARBON CONTAMINANTS IN DREDGED MATERIAL

**PURPOSE:** This note summarizes the findings of a workshop convened to assist Corps regulators in the evaluation of hydrocarbon contamination in dredged material. The workshop participants suggested a list of 15 compounds to be used in a tiered testing approach. The 15 compounds occur frequently in contaminated sediment, can be analyzed reliably, and are considered to be representative of hydrocarbons that are known in general to have toxic effects.

**BACKGROUND:** A 3-day workshop on regulatory evaluation of hydrocarbons in dredged material was conducted 13-15 May 1986 at the Waterways Experiment Station (WES). The workshop was requested by the US Army Engineer Districts, New York and Chicago. The purpose of the workshop was to provide the districts with assistance in identifying specific hydrocarbon compounds that would be the most appropriate to analyze in the regulation of dredged material disposal. The original focus was on petroleum hydrocarbons, but the scope of the workshop was broadened to consider any hydrocarbon contamination regardless of source.

Workshop participants from government agencies, private industry, and academia were selected for their expertise in analysis of hydrocarbons in sediment and in prediction of the potential environmental impacts of such substances. The workshop consisted of brief presentations and roundtable discussions focusing on various aspects of environmental chemistry and biological effects of hydrocarbons. The discussions culminated in the recommendation of specific hydrocarbons as indicator compounds and in the development of a suggested tiered testing approach for regulatory evaluation of hydrocarbon-contaminated dredged material.

**ADDITIONAL INFORMATION OR QUESTIONS:** Refer to the workshop proceedings (Clarke and Gibson 1986) or contact the authors, Ms. Joan Clarke (601) 634-2954 (FTS 542-2954) and Ms. Alfreda Gibson (601) 634-4027 (FTS 542-4027), or the EEDP Program Manager, Dr. Robert M. Engler, (601) 634-3624 (FTS 542-3624).

## The Nature of the Problem

Hydrocarbons comprise a large number of compounds, some of which pose potentially serious environmental threats. Hundreds of these compounds have been identified in sediment, water, and organism tissue samples. They span a wide range of water solubility, persistence, bioavailability, toxicity, bioaccumulation potential, carcinogenicity, and overall biological importance.

Hydrocarbons, by definition, are molecules composed only of carbon and hydrogen. Straight- or branched-chain hydrocarbons are called aliphatics; hydrocarbon ring structures are called cyclics. Hydrocarbons that contain only single bonds between adjacent carbon atoms are termed saturated (i.e., saturated with hydrogen atoms); the presence of double or triple bonds makes them unsaturated. Reactivity of hydrocarbons generally increases with degree of unsaturation. Aromatics, in the classic sense, are cyclics containing one or more 6-carbon rings in which the carbon atoms are joined by alternating double and single bonds (e.g., benzene and naphthalene). Heterocyclics are ring structures that include elements other than carbon and hydrogen (such as nitrogen, sulfur, or oxygen) and may or may not be aromatic. Cyclic hydrocarbons with side chains attached to a parent structure (nucleus) are alkylated. Examples of these structures are illustrated in Figure 1.

Hydrocarbons in aquatic systems can arise from several sources. Petroleum hydrocarbons originate only from petroleum products ranging from crude oil to highly refined products and often contain heterocyclics. Fuel-oil spills and miscellaneous disposal (e.g., municipal surface runoff) are major sources of petroleum contamination to the aquatic environment. Hydrocarbons may also be biogenic or pyrogenic in origin. Biogenic hydrocarbons are produced by living organisms and consist primarily of aliphatics. Pyrogenic hydrocarbons are generated by combustion or incineration of various organic substances including petroleum, coal, and wood products, and they enter aquatic systems mainly via atmospheric deposition.

In some circumstances, the potential for a toxic environmental impact of a dredged material may be determined by its particular mixture of hydrocarbon compounds. Analysis of hydrocarbons in a sediment sample as oil and grease or as total petroleum hydrocarbons or other summary type measures cannot provide sufficient information for accurate evaluation of potential environmental impacts. On the other hand, qualitative and quantitative analyses of all

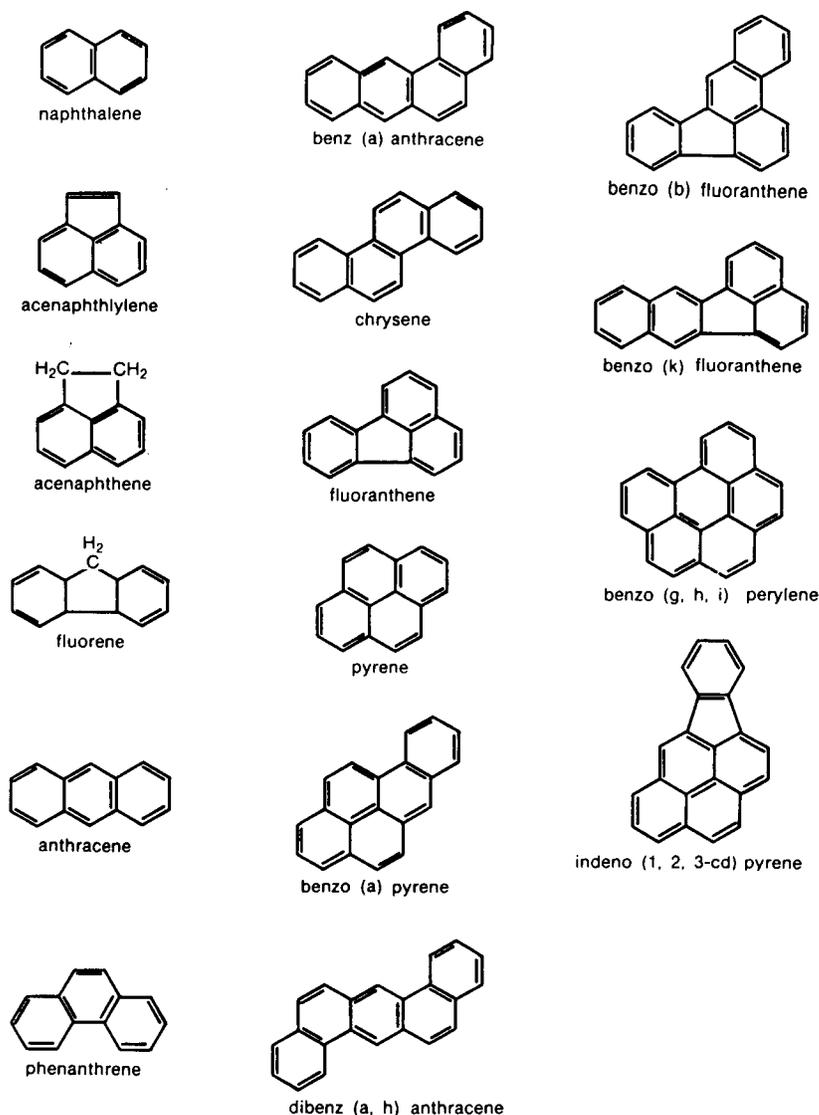


Figure 1. Examples of basic hydrocarbon and heterocycle structures

hydrocarbon compounds present would be virtually impossible. Even a relatively comprehensive analysis would be too time-consuming and expensive and would produce an excessive volume of data to be reviewed as part of the regulatory process. Thus, an intermediate approach is needed for adequate and informed regulatory evaluations of potential toxic impacts of hydrocarbons in dredged material.

The objective of the workshop was achieved by formulating such an intermediate approach based on identification of a limited number of hydrocarbon compounds to be used in a regulatory testing scheme.

## Recommendations of the Workshop

### Key compounds

Aromatics, particularly the polycyclic (or polynuclear) aromatic hydrocarbons (PAHs), are the most important class of hydrocarbon contaminants in dredged material from a regulatory standpoint. The PAHs are those aromatics having two or more fused rings. Not all PAHs are readily bioavailable, and most do not attain concentrations in water that are acutely toxic to aquatic organisms. However, concern over PAHs as environmental pollutants stems from the acute toxicity of some 2- and 3-ring compounds, such as anthracene and phenanthrene, and the chronic toxicity (especially carcinogenicity) of the higher molecular weight compounds.

Sixteen of the 129 toxic chemicals on the EPA priority pollutant list are PAHs (Richards and Shieh 1986, Keith and Telliard 1979). These 16 priority pollutant PAHs are illustrated in Figure 2. The workshop participants recommended 15 of the 16 priority pollutant PAHs as key compounds for regulatory evaluation of hydrocarbons in dredged material. Naphthalene was not included in the list because it is too water soluble to persist in sediment and too volatile to give accurate analytical results. The behavior, fate, and effects of the other 15 PAHs are representative of hydrocarbons that are known in general to have biological effects, and some are known carcinogens.

Aliphatics, another major class of hydrocarbons, need not be included in regulatory evaluations of dredged material because they generally do not cause major environmental impacts in the context of dredging and disposal.

### Testing approach

A tiered testing approach was suggested for regulatory evaluations of PAHs in dredged material. This type of evaluation would begin with a general assessment of the likelihood of contamination. If hydrocarbon contamination is considered likely, then the first testing tier would be conducted. This tier includes an acute toxicity test and analysis of the sediment for the 15 priority pollutant PAHs selected as key compounds. Acute toxicity tests exposing crustaceans (such as *Daphnia* in fresh water and *Mysidopsis* in salt water) to contaminated sediment are commonly done and have accepted standard procedures; the results are easy to interpret. If acute toxicity is seen, then regulators might decide to impose dredging or disposal restrictions without conducting any further tests.

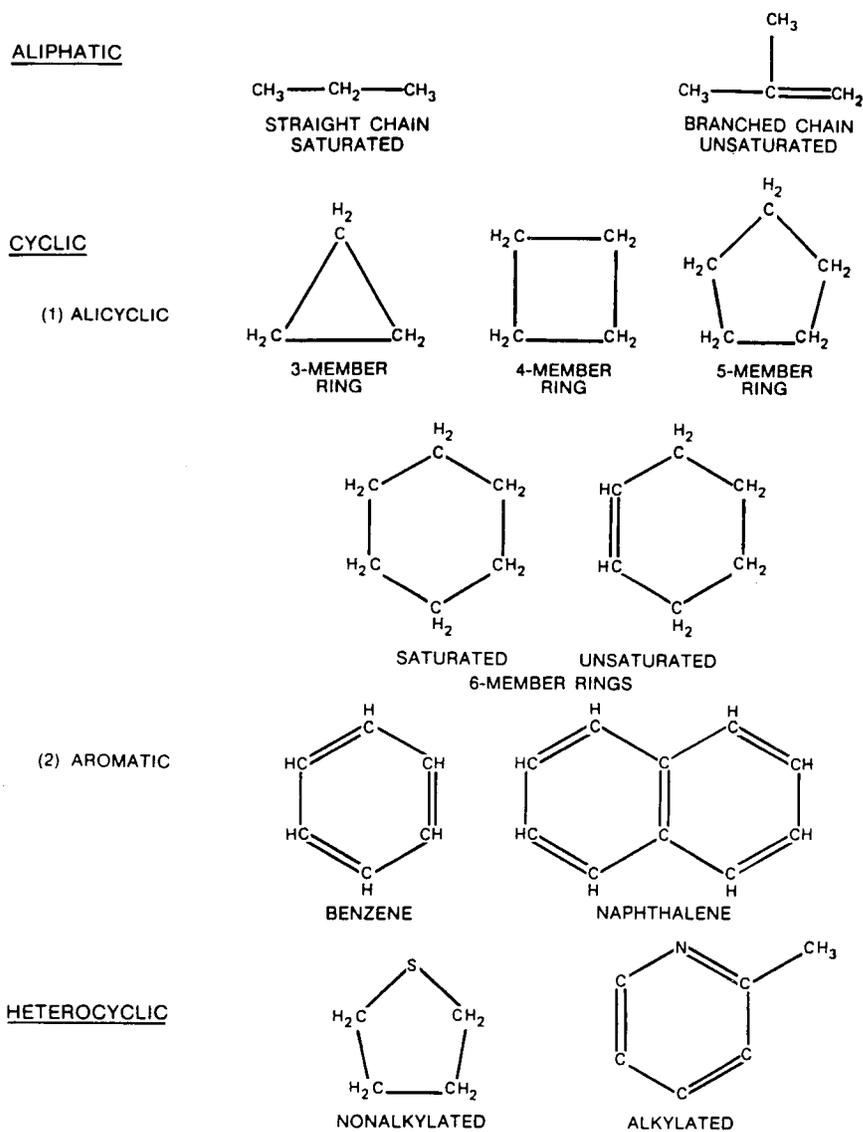


Figure 2. Structures of the 16 priority pollutant PAHs.  
 (Note: Naphthalene was not recommended for use as a key compound.)

The first testing tier would accomplish two main purposes. First, the acute toxicity tests would indicate any acute toxicity due to any contaminants in the dredged material, not just to the 15 priority pollutant PAHs. Second, the sediment analysis would signal levels of the 15 PAHs that were not acutely toxic but that might cause concern over chronic toxicity problems. Recommending scientifically defensible levels of concern for the 15 PAHs will be the primary objective of a second workshop to be held during 1987.

If concern over chronic toxicity is suggested by the results of the sediment analysis in the first testing tier, then a second tier test would be

conducted. The second tier would consist of a 10-day bioaccumulation test to demonstrate bioavailability. If any of the 15 PAHs is found in organism tissues after 10 days of exposure to the contaminated sediment, then that compound is bioavailable and may cause chronic effects. In the second-tier assessment, it is important to use organisms that have limited or no ability to metabolize PAHs. Appropriate saltwater species might include the clam *Mercenaria* or a suitable substitute bivalve or, in freshwater sediment, an amphipod such as *Pontoporeia*. Analysis of tissues for unmetabolized parent compounds is thus simplified. Although some metabolites may be more toxic to aquatic organisms than the parent PAH, analysis for metabolites of PAHs is not recommended for a routine regulatory program until more research is completed and analytical methods are better established.

Quality assurance/quality control (QA/QC) evaluations and procedures are a critical need, especially when a variety of laboratories are used by a regulatory agency for testing and review purposes. Strong QA/QC guidelines need to be developed.

#### Future Research

The problem of hydrocarbon contamination in dredged material is complex because hundreds of compounds may be involved, spatial heterogeneity can be great, and many environmental factors can influence bioavailability. The recommendations of this workshop represent only a starting point for informed and environmentally sound regulatory evaluation. Research is needed to develop standard analytical procedures and biological testing protocols for the evaluations of PAH metabolites and of toxic hydrocarbons other than the priority pollutant PAHs. These include some of the alkylated PAHs, heterocyclics, nitroaromatics, and aromatic amines. Bioaccumulation tests can demonstrate bioavailability under specific circumstances but cannot pinpoint chronic effects. Biological tests that need to be refined and standardized include assays for such chronic effects as carcinogenicity, genotoxicity, and reproductive effects.

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# *Environmental Effects of Dredging Technical Notes*



## ROUTINE AND ATYPICAL WETLAND DETERMINATIONS ACCORDING TO CE WETLANDS DELINEATION MANUAL

**PURPOSE:** This article summarizes the methods for delineating wetlands that have been published as the Corps of Engineers Wetlands Delineation Manual.<sup>\*</sup> It provides an abbreviated version of the manual and lists the wetland indicators and steps in the basic procedure for making routine and atypical wetlands determinations. This procedure does not replace that described in the manual, but serves as a reminder of steps required for making wetland determinations. The user should be familiar with both the manual and the terms used in this reference; many details and cautionary statements contained in the manual are omitted here. The user is referred to the manual for details. This abbreviated version is also being printed on waterproof paper in a size that will fit into the loose-leaf binder used for the Munsell soil color charts and will serve as a field reference.

**BACKGROUND:** Section 404 of the Clean Water Act gives authority to the Secretary of the Army, acting through the Chief of Engineers, to regulate the discharge of dredged or fill material into "waters of the United States." The term "waters of the United States" has broad meaning and incorporates both deepwater aquatic habitats and special aquatic sites including wetlands. Although all special aquatic sites are subject to provisions of the Clean Water Act, the manual considers only wetlands. The manual is a product of the Wetlands Research Program and was designed to provide precise and technically defensible methods for the delineation of wetlands, which is essential to regulatory, planning, and construction activities. The manual is presently being evaluated by the Corps Districts. The evaluation period is 1 year, and comments are expected in early 1988.

**ADDITIONAL INFORMATION OR QUESTIONS:** Points of contact for information concerning this article or questions relating to the Wetlands Research Program are Mr. Russell F. Theriot, Comm. or FTS (601)634-2733, or Dr. Robert M. Engler, Manager, Environmental Effects of Dredging Programs, (601)634-3624.

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\* Environmental Laboratory. 1987. "Corps of Engineers Wetlands Delineation Manual," Technical Report Y-87-1, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

## Wetland Indicators

Wetlands delineation is based on three environmental parameters: vegetation, soil, and hydrology. Positive wetland indicators of all three parameters will be present in wetlands except in atypical situations or abnormal environmental conditions. (Appendixes C and D and Data Forms 1 and 3 mentioned below are found in the manual.)

### Hydrophytic Vegetation

1. More than 50 percent of dominant species are classified as FAC, FACW, or OBL on regional lists of plant species that occur in wetlands (Appendix C).
2. Other indicators of hydrophytic vegetation include:
  - a. Visual observation of plant species growing in areas of prolonged inundation or soil saturation.
  - b. Morphological adaptations.
  - c. Technical literature indicates that dominants are hydrophytic.
  - d. Physiological adaptations.
  - e. Reproductive adaptations.

### Hydric Soil

1. Organic soil.
2. Histic epipedon present.
3. Sulfidic material present (rotten egg odor).
4. Aquic or peraquic moisture regime.
5. Anaerobic soil conditions for more than 7 consecutive days during the growing season.
6. Soil color:
  - a. Gleyed.
  - b. Matrix chroma of 2 or less\* in mottled soils.
  - c. Matrix chroma of 1 or less\* in unmottled soils.
7. Soil series on hydric soils list (Appendix D).
8. Iron and manganese concretions present.
9. Organic layer (3 inches or more) over sandy soil.
10. Organic pans in sandy soil.
11. Organic streaking of subsurface horizons in sandy soil.

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\* Colors should be determined in soils that have been moistened; otherwise, state that colors are for dry soils.

Wetland Hydrology

1. Recorded data from stream, lake, and/or tidal gages indicate that the area is periodically inundated during growing season.
2. Field indicators of wetlands hydrology include:
  - a. Visual observation of inundation.
  - b. Visual observation of saturated soil (upper 12 inches).
  - c. Watermarks.
  - d. Drift lines.
  - e. Sediment deposits including encrusted detritus.
  - f. Drainage patterns in low areas.

{ Consider recent rainfall

Methods

Preliminary Data Gathering and Synthesis\*

Extract information on vegetation, soil, and hydrology of the area from all available sources. The manual lists potential sources of information. Decide whether information for each parameter is sufficient to enable a determination without an onsite visit.

Three procedures for wetland determinations are described in the following pages: determinations where no site visit is necessary; procedure when an onsite visit is required; and procedure when the area has been recently altered (i.e., atypical situations). Terminal decisions are indicated by capital letters WETLANDS or NONWETLANDS.

Wetland Determinations (No Site Visit Necessary)

1. Identify community type(s). Must know dominant plant species of each type.
2. Compare data to list of hydrophytic vegetation indicators. One of the following will apply:
  - a. *No hydrophytic vegetation indicator present:* NONWETLAND.
  - b. *Hydrophytic vegetation indicator present; all dominants OBL and/or FACW:*
    - (1) Look for evidence of hydrologic alteration. If none, WETLAND.
    - (2) If hydrologic alteration has occurred, go to c.

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\* Omit the preliminary data gathering and synthesis if a decision has been made that it is more practical to conduct an onsite visit.

- c. *Hydrophytic vegetation indicator present; one or more dominant species FAC:*
- (1) Look for indicator of wetland hydrology. If absent, NONWETLAND.
  - (2) If present, look for indicator of hydric soil.
    - (a) Indicator present: WETLAND.
    - (b) No indicator present: NONWETLAND.
3. Complete Data Form 1 for each community type. If all are wetland types, the entire area is WETLAND. If not, determine community types that meet one of the above conditions. The boundary of these community types is the wetland boundary.

Wetland Determinations  
(Onsite Visit Necessary)

Determine the size of the area and whether there is evidence of alteration of one or more of the three environmental parameters used to identify wetlands (i.e., hydrophytic vegetation, hydric soil, and wetland hydrology). The following procedures are appropriate for typical areas of 5 acres or less, typical areas greater than 5 acres, and atypical situations when the area has been recently altered.

1. *Typical areas of 5 acres or less.* Complete the following actions.
  - a. Identify plant community types.
  - b. Select a representative observation point in each type.
  - c. Visually determine dominant plant species at each point and record on Data Form 1 (use separate form for each point).
  - d. Record indicator status of each dominant species from Appendix C.
  - e. Hydrophytic vegetation is present if more than 50 percent of dominant species are FAC, FACW, and/or OBL.
    - (1) If not, NONWETLAND.
    - (2) If all dominants are FACW and/or OBL and the community boundary is abrupt, go to f.
    - (3) If one or more dominant species is FAC, FACU, or UPL, or if the community boundary is not abrupt, go to g.
  - f. Look for evidence of recent hydrologic alteration (e.g., dikes, levees, drainage ditches, etc.).
    - (1) If not altered, WETLAND.
    - (2) If altered, go to g.
  - g. Look for indicator of wetland hydrology.

- (1) If present, area has wetland hydrology. Go to h.
  - (2) If not, NONWETLAND.
- h. If soil series is known, determine whether on list of hydric soils (Appendix D).
- (1) If so, WETLAND.
  - (2) If soil is not listed as hydric, dig a soil pit and examine soil at 10 to 12 inches (or below the A-horizon) for hydric soil indicator.
    - (a) If indicator found, WETLAND.
    - (b) If not, NONWETLAND unless abnormal environmental conditions are present. If so, go to i.
- i. Determine whether environmental conditions are normal.
- (1) If so, go to j.
  - (2) If not, interpret collected data considering normal environmental conditions. Go to j.
- j. Complete a data form for each point. Examine completed forms and combine all wetland community types into one unit. These are the wetlands of the area.
2. *Typical area greater than 5 acres.* The basic procedure for making a wetland determination at a given point was given for typical areas of less than 5 acres. The major procedural difference in areas greater than 5 acres is that observation points are established at points along transects. This procedure is as follows:
- a. Establish a baseline parallel to direction of major flow through the area and determine baseline length.
  - b. Divide baseline length by number of required transects. As a guide, use three transects when baseline length is 1 mile or less; three to five transects when baseline length is 1 to 2 miles; five to eight transects when baseline length is 2 to 4 miles; or locate transects at 0.5-mile intervals when baseline length exceeds 4 miles.
  - c. Establish transect starting point at midpoint of each baseline segment. If necessary, relocate one or more transects to include all community types.
  - d. Select observation point in first community encountered along first transect.
  - e. Apply procedure given for areas of less than 5 acres and determine whether point is in a wetland. Complete Data Form 1 for this and subsequent points.
  - f. Continue along transect until a second community type is encountered.
  - g. Apply procedure given for areas of less than 5 acres and determine whether this point is in a wetland.

- (1) If both points are in either wetlands or nonwetlands, go to the next community type along the transect and determine whether it is a wetland.
  - (2) If one point is in a wetland and the other is not, make additional wetland determinations between these points until wetland boundary is located. Complete Data Form 1 for this location and mark the boundary location on map.
- h. Locate observation points in all other community types along each transect and make wetland determination at each. Locate any wetland boundaries occurring between these points and mark them on the map.
- i. Connect points on the map that represent wetland boundaries, following contours when a contour map is available. Walk wetland boundary and adjust boundary line on map as necessary (this sometimes requires additional wetland determinations).
3. *Atypical Situations.* Apply the following procedures when evidence indicates recent unauthorized activities or natural events that preclude characterization of one or more parameters. Use Data Form 3 to record information.
- a. *Vegetation.*
- (1) Describe type of alteration (e.g., clear-cutting, selective removal of certain species).
  - (2) Document effect of alteration on vegetation.
  - (3) Characterize previous vegetation. Obtain all necessary supporting evidence. Some potential sources include:
    - (a) Aerial photography.
    - (b) Onsite inspection of remaining vegetation.
    - (c) Previous inspections.
    - (d) Adjacent vegetation.
    - (e) Soil Conservation Service records.
    - (f) Permit applicant.
    - (g) Public.
    - (h) National Wetlands Inventory maps.
  - (4) Record indicator status of dominant species from Appendix C.
    - (a) If more than 50 percent were FAC, FACW, and/or OBL, previous vegetation was hydrophytic.
    - (b) If condition in (a) is not satisfied, NONWETLAND.
    - (c) If previous vegetation could not be characterized, base decision on soil and hydrology (b and c, respectively).
  - (5) Complete vegetation section of Data Form 3.

- (a) Return to paragraph 1 and complete determination if no other parameter has been altered.
- (b) If either soil or hydrology has been altered, go to b or c.

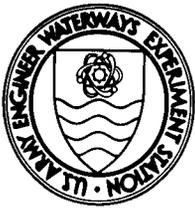
b. Soil.

- (1) Describe type of alteration (e.g., filled, surface layers removed, plowed).
- (2) Document effect of alteration on soil.
- (3) Characterize previous soil. Obtain all necessary supporting evidence. Some potential sources include:
  - (a) Soil surveys.
  - (b) Characterization of buried soil.
  - (c) Characterization of plowed soil.
  - (d) Adjacent unaltered soil. (Area must be in same topographic position and nearby.)
  - (e) Remnant profile (where soil layers have been removed).
- (4) Determine whether previous soil was hydric by applying indicators.
  - (a) If indicator found, hydric soil was formerly present.
  - (b) If no indicator found, NONWETLAND.
  - (c) If previous soil could not be characterized, base decision on vegetation and hydrology (a and c, respectively).
- (5) Complete soil section of Data Form 3.
  - (a) Return to paragraph 1 and complete determination if no other parameter has been altered.
  - (b) If either vegetation or hydrology has been altered, go to either a or c.

c. Hydrology.

- (1) Describe type of alteration (area leveed, diked, drained, etc.).
- (2) Describe effect of alteration on hydrology.
- (3) Characterize previous hydrology. Obtain all necessary supporting evidence. Some potential sources include:
  - (a) Stream, lake, or tidal gage data.
  - (b) Field indicators.
  - (c) Aerial photography.
  - (d) Historical records.

- (e) Floodplain management maps.
- (f) Public or local officials.
- (4) Determine whether wetland hydrology previously occurred by applying wetland hydrology indicators.
  - (a) If indicator found, wetland hydrology was formerly present.
  - (b) If no indicator found, NONWETLAND.
  - (c) If previous hydrology could not be characterized, base decision on vegetation and soil (a and b, respectively).
- (5) Complete hydrology section of Data Form 3.
  - (a) Return to paragraph 1 and complete determination if no other parameter has been altered.
  - (b) If either vegetation or soil has been altered, apply procedure in a or b.



# *Environmental Effects of Dredging Technical Notes*



## CORPS OF ENGINEERS' PROCEDURES AND POLICIES ON DREDGING AND DREDGED MATERIAL DISPOSAL (THE FEDERAL STANDARD)

**PURPOSE:** This note describes the Federal Standard pursuant to Corps' technical considerations and policies with regard to the disposal of dredged material in accordance with the Clean Water Act (CWA), which provides for selecting the least costly dredged or fill material disposal alternative, consistent with sound engineering practices and appropriate environmental quality standards. This approach also generally applies to assessments conducted in accordance with the Ocean Dumping Act, even though the following discussion centers on the CWA.

**BACKGROUND:** Navigable waterways of the United States have and will continue to play a vital role in the Nation's development. The Corps, in fulfilling its mission to maintain, improve, and extend these waterways, is responsible for the dredging and disposal of large volumes of sediment each year. Nationwide, the Corps dredges about 230 million cubic yards (c.y.) in maintenance and about 70 million c.y. in new dredging operations annually at a cost of about \$450 million. In addition, 100-150 million c.y. of sediments dredged by others each year are subject to permits issued by the Corps. In accomplishing its national dredging and regulatory missions, the Corps has conducted extensive research and development in the field of dredged material management (Engler, Patin, and Theriot 1988). Regulations, policies, and technical guidance prepared and used by the Corps are based on operating experience and results from extensive research programs. Federal expenditures on dredged material research, monitoring, and management activities have cumulatively exceeded \$100 million. Additional research regarding current issues relative to the Corps' national dredging program is an ongoing and dynamic process. Corps' policy is evolving as dredged material research provides a better understanding of the environmental impacts that can be anticipated from dredging and dredged material disposal. Corps' national policy is reflected in the final regulation for Corps' operation and maintenance dredging of Federal navigation projects published 26 April 1988 (33 CFR Parts 209, 335, 336, 337, and 338) and in the final rule for the Corps' regulatory program published 13 November 1986 (33 CFR Parts 320-330).

**ADDITIONAL INFORMATION:** This technical note was written by Dr. Robert M. Engler, Dr. Tom Wright, Dr. Charles R. Lee, and Dr. Tom M. Dillon. For additional information contact Mr. Dave Mathis (CECW-D), commercial or FTS: (202) 272-8843; or Dr. Wright (601) 634-3708; or Dr. Engler, manager of the Environmental Effects of Dredging Programs, (601) 634-3624.

## Corps Authorities and Responsibilities

The Corps has regulatory responsibility for all dredged material disposal activities that occur within the waters of the United States. This authority stems from Section 10 of the River and Harbor Act of 1899, Section 404 of the Clean Water Act (CWA) (Public Law 92-500, as amended), and Section 103 of the Marine Protection, Research, and Sanctuaries Act (Public Law 92-532, as amended). The Corps' regulatory responsibilities involve review of some 10,000-30,000 permit applications each year as well as appropriate maintenance of, and improvements to, the 25,000-mile congressionally authorized Federal navigation system serving 42 of the 50 states. Section 404 authorizes the Secretary of the Army to issue permits for the discharge of dredged or fill material into waters of the United States in accordance with the Section 404(b)(1) Guidelines (subsequently referred to as the Guidelines) and other requirements of Federal law as discussed below. The Guidelines require compliance with several conditions prior to allowing disposal of dredged material in waters of the United States. Compliance requires the avoidance of "unacceptable adverse effects" to the aquatic environment. The Guidelines specify four conditions of compliance ("restrictions on discharge" per 40 CFR 230.10):

1. There is no other practicable alternative that would have less adverse impact on the aquatic environment.
2. The disposal will not result in violations of applicable water quality standards after consideration of dispersion and dilution (40 CFR 230.10(b)(1)), toxic effluent standards, or marine sanctuary requirements, nor will it jeopardize the continued existence of threatened or endangered species.
3. The disposal will not cause or contribute to significant degradation of the waters of the United States.
4. All appropriate and practicable steps have been taken to minimize potential adverse impacts of the discharge on the aquatic environment.

Findings for compliance with condition 2 are based in large part on Section 401 of the CWA, which allows the individual states to establish State water-quality standards. All State-established standards must, at a minimum, be as stringent as established Federal water-quality criteria. However, the individual states have the option under the CWA, and several have so elected, to establish more stringent State standards to reflect the overriding priority

that these individual states have for environmental protection. Unless waived on a case-by-case basis by the State, or on such occasions overridden by critical factors in the national interest, State 401 Water Quality Certification must be obtained prior to initiation of any Federal or non-Federal dredged material disposal activity which occurs within navigable waters of the United States (40 CFR 230.10(b)(1)).

The findings of compliance with condition 3 are to be based, in part, on "evaluation and testing" of the proposed dredged material (Subpart G of the Guidelines). The assessment provided by Subpart G is used to determine the potential for significant\* adverse effects of dredged material disposal on the aquatic environment (factual determinations required by Part 230.11). According to the Guidelines (40 CFR 230.61), specific evaluation procedures, including chemical and biological tests to determine compliance with the Guidelines and State water-quality standards, are furnished by the Corps as the permitting authority.

The Corps' final decision on any proposed dredged material disposal activity, however, must be based on a broad public interest review which not only considers information derived from chemical and biological tests but which also considers an evaluation of the probable impact, including cumulative impacts of the proposed activity, on the public interest. In addition, embodied within this public interest review, is a Corps requirement to ensure that the substantive concerns of over 30 Federal environmental laws, Executive Orders (EOs), and other requirements are properly addressed, whenever applicable. These include the Coastal Zone Management Act, the Marine Protection, Research, and Sanctuaries Act, the Endangered Species Act, the Fish and Wildlife Coordination Act, EO 11990 for Federal projects (Protection of Wetlands), and EO 11988 (Floodplain Management). While each of these Federal Statutes (including the CWA) is generally resource specific in regard to environmental protection, the Corps' public interest review necessitates full consideration of all relevant information before rendering a decision.

The expected benefits resulting from the proposal must be balanced against its foreseeable detriments. All factors which may be relevant to the proposed activity will be considered, including conservation, economics,

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\* The term "significant" has no statistical relevance or connotation; it is used in the same general sense as "substantive."

esthetics, historic properties, fish and wildlife values, flood hazards, floodplain, national defense, water supply and conservation, water quality, energy needs, safety, food and fiber production, mineral needs, considerations of property ownership, and the general needs and welfare of the people.

The weight given to each factor is determined by its importance and relevance to a particular proposal. A specific factor may be given great weight on one proposal, while it may not be present or as important on another. The Corps' (District Engineer's) final decision will reflect the national concern for both protection and utilization of important resources. As such, the Corps is neither a proponent nor opponent of individual permit proposals, nor of congressionally authorized dredging projects.

Section 404(b)(2) allows the Corps to issue permits otherwise prohibited by the Guidelines, based on an overriding consideration of the economics of anchorage and navigation.

#### Federal Standard

The Corps, as agency policy, uses a Federal Standard philosophy and process in evaluating proposed dredged material disposal activities relative to the general public interest. This "Federal Standard" process is intended to meet environmental requirements at the least cost within a consistent national framework. The Federal Standard provides a reference point for Corps field offices in addressing regional issues in dredged material management. The intent of the Federal Standard is to ensure a necessary level of national consistency in the evaluation and undertaking of proposals for dredged material disposal (e.g., testing procedures), while also ensuring a necessary level of flexibility by the Corps field offices to account for region-specific considerations. However, significant deviations from national testing and evaluation guidance require consideration of cost, utility of information, and full technical explanation and documentation in the Section 103.

For proposed permit activities, Corps regulations (33 CFR 320-330) require that unnecessary testing procedures and regulatory controls be avoided, while simultaneously ensuring that overriding rights and interests of the general public are fully protected in the waters of the United States. Such rights include, but are not limited to, preservation of water quality, national security, and interstate commerce. These considerations are

discussed in more detail in a Corps Regulatory Guidance Letter of 19 August 1987, RGL-87-8, "Testing Requirements for Dredged Material Evaluation."

Permit activities

Evaluation of Section 404 permits, for which an application has been made to the Corps, normally will proceed concurrently with the processing of applications for permits for other Federal, State, and/or local authorities (33 CFR 320.4(g)), such as the State 401 Water Quality Certification. The applicant for a Section 404 permit will receive direction from the Corps as the permitting authority (40 CFR 230.61) concerning appropriate tests that must be conducted on material proposed for dredging. This note summarizes the Corps' national guidance given to its field offices on technically acceptable dredged material evaluation procedures. Also to be provided to permit applicants, where applicable and appropriate, are Corps recommended actions which can be undertaken to minimize any identified adverse effects of discharges of dredged material as provided under Subpart H of 40 CFR 230. Depending on the results of the general public interest review, the Corps may issue, issue with conditions, or deny individual permits. In those permit cases where denial of State Certification has occurred or is imminent or a state has not concurred in Coastal Zone Management concurrence, the Corps may either immediately deny the Section 404 permit without prejudice, or may continue processing the permit, concluding either in a denial as contrary to the public interest or denying without prejudice, noting that, except for the State 401 Certification denial or Coastal Zone Management nonconcurrence, the Section 404 permit could be issued.

Federal projects

For Federal projects, the Corps is required to use the Section 404(b)(1) Guidelines to determine the appropriate test and evaluation procedures for delineating the least costly, environmentally acceptable disposal alternative as well as to demonstrate compliance with applicable State water-quality standards.

The Corps submits its findings concerning project compliance with the 404 Guidelines and State water-quality standards to the State via the Public Notice process along with a request for State Water Quality Certification. The certification request also includes relevant information to demonstrate compliance with applicable State water-quality standards. The existing regulatory framework given in the CWA requires that a Corps-preferred alternative

be developed before the request for State Water Quality Certification. However, this does not preclude informal coordination with the State at a much earlier stage in the project evaluation, and indeed such informal coordination is fully encouraged, particularly if it will shorten the environmental compliance process for the Corps project.

The Corps Public Notice and Findings of Compliance or Non-Compliance with the Section 404(b)(1) Guidelines serves as a point of reference in any subsequent negotiations with the State on additional requirements or conditions which the State may require for Water Quality Certification.\* The Corps' District Engineer has the necessary discretionary authority to develop additional evaluative information requested by the State, which in the District Engineer's opinion, is technically justified and reasonably related to enforcement of the State's water-quality standards. The legislative record for the CWA provides congressional recognition that Federal project costs may be increased in some instances to address reasonable and technically appropriate State water-quality concerns. However, if the District Engineer determines that on a case-by-case basis a State's requirements are excessive or technically unjustified, he may request that the State or project sponsor fund the additional costs associated with any such requirement. In such cases where the State or project sponsor agrees to fund the additional costs, the District Engineer must also determine and notify the State and project sponsor that such additional costs may affect the continued economic viability of the Federal project in question. In the event that the State or project sponsor does not agree to fund the additional cost, the District Engineer may defer dredging while determining whether the dredging project is economically justified and is not contrary to the public interest.

For Federal dredging projects (where Congress has allocated Federal funds), the Corps is responsible, in developing dredged material disposal alternatives, for considering all facets of the dredging and disposal operation, including technically appropriate test and evaluation procedures, cost, engineering feasibility, overall environmental protection, and the no-dredging option. The alternative selected by the Corps should be the least costly alternative, consistent with sound engineering and scientific practices and

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\* This procedure is also followed for concurrence with certification of consistency for approved State Coastal Zone Management Programs.

meeting applicable Federal environmental statutes. This becomes the "Federal Standard."

### Corps of Engineers Technical Disposal Guidelines

The following paragraphs present the procedures by which the Corps regulates and manages the disposal of dredged material in the waters of the United States under its authorities and policies described above. These procedures, which evolved over the past decade, are subject to additional change and modification as new information and technology are developed and adequately evaluated.

Section 404 of the CWA provides that guidelines developed by the US Environmental Protection Agency (EPA) in conjunction with the Corps be applied by the Corps in selecting disposal sites and in the permit application review process. EPA published technical guidelines in 1975 and revised these in 1980 for use by the Corps in making the required ecological evaluation of a proposed discharge activity. The Corps issued final regulations for the Section 404 regulatory program in July 1977 to be used in evaluating proposed discharges of dredged or fill material into inland and ocean waters. In May 1976, the Corps issued an interim guidance manual as specified in the Federal Register to initiate technical implementation of the program.

The Section 404(b)(1) Guidelines as well as the 103 criteria are based on the following factors from Section 403(c) and 102(a) of the Clean Water and Ocean Dumping Acts, respectively:

1. The effect of disposal of pollutants on human health or welfare, including but not limited to plankton, fish, shellfish, wildlife, shorelines, and beaches.

2. The effect of disposal of pollutants on marine life including the transfer, concentration, and disposal of pollutants or their by-products through biological, physical, and chemical processes; changes in marine ecosystem diversity, productivity, and stability; and species and community population changes.

3. The effect of disposal of pollutants on esthetics, recreation, and economic values.

4. The persistence and permanence of the effects of disposal of pollutants.

5. The effect of the disposal at varying rates of particular volumes and concentrations of pollutants.

6. Other possible locations and methods of disposal and recycling of pollutants including land-based alternatives.

7. The effect of alternate uses of the oceans, such as mineral exploration and scientific study.

These "legal/technical" considerations form the framework from which the ecological evaluations must be developed.

The Section 404(b)(1) Guidelines recognize that compliance evaluation procedures will vary depending on the seriousness of the proposal's potential for unacceptable adverse impacts (40 CFR 230.10) and provide general guidance for evaluation and testing. Pursuant to the Guidelines, specific evaluation procedures, including chemical and biological tests, are furnished by the District Engineer on a case-by-case basis ("interim guidance by the permitting authority," 40 CFR 230.61).

To assist the Corps in the overall long-term management of the disposal of dredged material, a management strategy was developed by the US Army Engineer Waterways Experiment Station (Francingues et al. 1985). This strategy has been adopted as Corps policy and is incorporated by reference in 33 CFR Parts 209, 335, 336, 337, and 338, 26 April 1988 (Corps Dredging Regulation). The steps for managing dredged material disposal follow:

1. Evaluate contamination potential.
2. Consider potential disposal alternatives.
3. Identify potential problems.
4. Apply appropriate testing protocols.
5. Assess the need for disposal restrictions.
6. Select an implementation plan.
7. Identify available control options.
8. Evaluate design considerations.
9. Select appropriate control measures.

Following development, the management strategy was used as a framework for an example application for highly contaminated material at Commencement Bay, WA (a Superfund site), under the sponsorship of the State of Washington Department of Ecology, and the Corps (Peddicord et al. 1986). This example application considers all alternatives for disposal and provides detailed

technical rationales and flowcharts for evaluating disposal alternatives based on the results of appropriate testing.

Since the mid-1970's the Corps has been regulating the disposal of dredged material under the authority of 33 CFR Parts 320 through 330 and 40 CFR Part 230 and revised in 1980 for waters of the United States and under the authority of applicable sections of 40 CFR 220-229 (1973) and revised in 1977 for ocean dumping. In fulfilling the obligations and responsibilities mandated by those authorities, the Corps has conducted extensive research under the Dredged Material Research Program (Saucier et al. 1978) and continues to conduct research under the Environmental Effects of Dredging Programs (Engler, Patin, and Theriot 1988), and provides field assistance and management activities under the Dredging Operations Technical Support Program. In addition, it has published two guidance manuals, one for the CWA (Environmental Effects Laboratory 1976) and a joint manual with EPA for ocean dumping (Environmental Protection Agency/and US Army Corps of Engineers 1977); the latter provides much more detailed guidance than the former. Although these documents were state of the art when published, subsequent operational experience has led to changes in specific application. In particular, there has been a tendency for Corps coastal districts to use, depending on the subject of concern, portions or all of the testing procedures in the Ocean Dumping Implementation Manual for 404(b)(1) determinations whenever estuarine or marine waters are involved. Although a major reason for this is the detailed guidance, others include similarities between the requirements of the 404 Guidelines and those in Section 102(a) of Public Law 92-532 (the Ocean Dumping Act) and the fact that saline waters are involved. Additionally, shortly after the issuance of the Corps/EPA implementation manual on ocean dumping, the Corps and EPA were sued by the National Wildlife Federation. The suit was based on the technical validity of the testing procedures and interpretation of test results. Judgment was made in favor of the Corps and EPA and there has been no further challenge. Because of the above factors, the ocean dumping testing procedures and interpretive approaches have been in widespread use and have led to the informal adoption of the general testing and evaluation protocol from ocean dumping to 404(b)(1) evaluations.

This should not be construed to imply that the ocean dumping procedures/interpretation are "required" or "mandated" for 404(b)(1) evaluations. These procedures should be considered in light of project-specific concerns and,

where appropriate, may, in part or in whole, be used. However, they do, *de facto*, constitute an acceptable and widely used technique which has withstood court challenge and for which a major technical data base exists. That no absolute procedure exists for 404(b)(1) evaluations is further evidenced by cooperative efforts currently in progress between the Corps and EPA to establish standard testing and evaluation procedures.

### Tiered Testing and Assessments

The national comprehensive testing strategy supported by the Corps is a tiered approach (Table 1) with each successive tier being based on a "reason to believe" that there is potential for unacceptable adverse effects. Each tier is fully optional and may be subsequently eliminated if there is sufficient information available to provide an adequate assessment for that tier or if there is no reason to believe that there will be unacceptable adverse effects associated with that tier or disposal concern. Such multiple tests are clearly allowed by 40 CFR 230.4-1 ("No single test or approach can be applied in all cases to evaluate the effects of proposed discharges of dredged or fill material," and "Suitability of the proposed disposal sites may be evaluated by the use, where appropriate, of sediment analysis or bioevaluation."). However, such tests are subject to the condition that "In order to avoid unreasonable burdens on applicants in regard to the amounts and types of data to be provided, consideration will be given by the District Engineer to the economic cost of performing the evaluation, in light of the information expected and the contribution of that information to the final decision, and the nature and magnitude of any potential environmental effect."

The first tier of the existing approach consists of an initial evaluation of available information to establish whether there is a "reason to believe" that contaminants are or are not present. This tier is commonly referred to as the "exclusion clause" (40 CFR 230.4-1(b)(1)). If there is no reason to believe that contaminants are present and if certain other conditions are met, including grain size and chemical/physical similarity of the dredged material and the substrate at the disposal site, no further testing is required. If there is reason to believe that contaminants are present, or if sufficient information is not available, a second tier or evaluation may be conducted which consists of a bulk sediment analysis. Should sufficient

Table 1  
Comprehensive Testing Approach for Aquatic Disposal  
as Part of the Federal Standard\*

<u>Tier 1</u>	Initial evaluation of existing information and "reason to believe there is contamination."	
<u>Tier IIA</u>	Bulk sediment inventory. Reason to believe dredged material is more contaminated than disposal site sediment and potential unacceptable adverse effects may occur.	
<u>Tier IIB</u>	Elutriate analysis. Chemical analysis for contaminant(s) of concern, contrast to appropriate water-quality criteria and/or standard with consideration of mixing. Comparison to receiving water quality and/or bioassay when no standard exists.	
<u>Tier III</u>	Biological tests.	
<u>Tier IIIA</u>	Acute bioassay toxicity tests (as appropriate):	
	<u>Water Column (Elutriate)</u>	<u>Select Species</u>
	(Mixing considered)	(As necessary)
	Dissolved phase	Mysid shrimp
	Suspended solids phase	Grass shrimp
		Bivalve
		Fish
		Larva, bivalve
		Other
	<u>Benthic</u>	
	Solid phase	Mysid shrimp
		Amphipod
		Grass shrimp
		Clam
		Polychaete
		Other
<u>Tier IIIB</u>	Bioaccumulation.	
	<u>Water Column</u>	<u>Select Species</u>
	Suspended solids phase	Grass shrimp
		Clam
		Polychaete
		Other
	<u>Benthic</u>	
	Solid phase	Clam
		Polychaete
		Other

\* Table 1 presents the general types of tests and evaluations in a tiered and sequential basis where each tier (step) is, however, optional and may be eliminated or chosen as appropriate. Test species tested are not mandatory but are shown for consideration to a proposed disposal site region.

information be available from previous testing and evaluation, no additional chemical analyses are necessary.

The bulk sediment analysis is essentially an inventory of contaminants of concern and is used to compare the chemical composition of the dredged material to the composition of the material at the disposal site with emphasis generally placed on heavy metals, PCBs, PAHs, pesticides, and other substances of ecological or human health significance. If substantially greater concentrations are observed in the dredged material and there is reason to believe that the substances are bioavailable and sufficient information is not available, a third tier of testing may be required. This tier includes testing for water column impacts and/or benthic impacts.

If there is concern regarding water column impacts, an elutriate test may be performed to evaluate contaminant release into dredging or disposal site water. The results of the elutriate test are compared to water quality standards after consideration of mixing as described in the 404(b)(1) Guidelines. If there are no water-quality standards or the standards are thought to be inappropriate or inadequate, a water column liquid and/or suspended particulate phase bioassay may be conducted along with consideration of mixing. Again, depending on where the concern lies, the water column bioassay may address the dissolved constituents and/or the suspended solid particulate phase.

If there is concern regarding impacts to benthic organisms, a benthic bioassay may be conducted. In general, for a comprehensive assessment of potential impacts, three organisms are generally used: a filter-feeder, a deposit-feeder, and a burrowing species. These relate to potentially different ecological niches at the disposal site. In addition, a mysid shrimp may be considered and has been widely used as an internal standard and to form a basis for quality assurance.

If there is a reason to believe that bioaccumulation is of concern, a second component of the third tier consists of evaluating the potential uptake of contaminants. This may be done either in the field or in the laboratory, whichever is more appropriate. If done in the laboratory, it is customary to use survivors of the toxicity bioassays for bioaccumulation assessment if sufficient biomass is present in the survivors.

The tiered testing approach described above is essentially the procedure followed for the evaluation of the aquatic disposal alternative in the

development of the Federal Standard for a given dredging project. This approach should be applied consistently to each and every dredging project, Federal or permit. The approach is flexible to some extent in allowing consideration of the three phases of the aquatic environment (liquid, suspended solids, and solid), as appropriate, that potentially could be impacted by the discharge of dredged material. Testing of the appropriate phase is determined by the reason to believe that a potential for unacceptable adverse impacts in one or more phases could occur. Additional flexibility is incorporated in the approach in relation to the selection of bioassay species to be used in the tests. Species can be selected such as a bivalve, polychaete, and a crustacean (mysids, amphipods, shrimp) or other available, appropriate, developed and evaluated local species. The intent is to evaluate the potential impact on a deposit-feeder, a burrower, and a suspension-feeder representative of major ecological compartments.

The following discussion addresses in more detail the interpretation of bioassay test results from the tiered testing approach used to evaluate the aquatic disposal alternative portion of the Federal Standard. Additional detail on the evaluation of the aquatic disposal alternative can be found in Peddicord et al. (1986).

If there is reason to believe that the dredged material contains contaminants of concern at concentrations higher than those contained in the disposal site sediment and these contaminants are potentially bioavailable and could result in a significant\* adverse impact, then bioassay tests should be conducted. The bioassay tier testing is used to determine whether there is reason to believe contaminants in the dredged material will result in an unacceptable adverse impact to the water column and/or the benthic component of the aquatic disposal environment. The water column consists of a dissolved phase and a suspended solid particulate phase. An overwhelming preponderance of evidence from years of studies has demonstrated that the potential of water column impacts of contaminants released from dredged material disposal are generally negligible. While this evidence does not unequivocally prove that water column impacts will not occur with aquatic disposal, it does indicate that such impacts are sufficiently unlikely that the District Engineer

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\* The term "significant" has no statistical relevance or connotation; it is used in the same general sense as "substantive."

normally should conclude that it is appropriate to focus evaluation on the other issues rather than testing for potential water column impacts in association with disposal in aquatic sites where the majority of the material is deposited on the bottom and the remainder is subject to rapid dispersion and dilution.

In many cases it will be possible to assess the potential for water column impacts on the basis of previous water column testing and characteristics of the disposal site without conducting additional sediment-specific testing. However, there may be a reason to believe that the suspended solid particulate phase of the water column may result in a potential unacceptable adverse impact to the disposal environment. If this is the case, the suspended solids bioassays may be conducted. Likewise, if there is reason to believe that unacceptable adverse impact may occur in the solid phase, then a solid-phase bioassay should be conducted.

If the results of the bioassay tests show unacceptable toxicity to the test species, further testing may be required. In the case of suspended solids phase bioassay testing, consideration of a mixing zone at the disposal site should be evaluated to determine whether an acceptable mixing zone is available to eliminate significant adverse impacts due to potential toxicity at the disposal site. If unacceptable toxicity is shown in the solid phase test and mortality is sufficiently elevated above control and/or reference, a significant impact has been shown.

If unacceptable toxicity is not observed in the solid phase test species and there is reason to believe that there is a potential for bioaccumulation, or the results of the bioassays are not conclusive, further testing may be required. The surviving bioassay animals may be analyzed for bioaccumulation after exposure to the dredged material for an appropriate length of time.

Bioaccumulation by bioassay species exposed to the dredged material is compared to that of species exposed to disposal site sediment or an appropriate reference site in the disposal site environment.

The above discussion has addressed the first four steps of the Management Strategy (Francingues et al. 1985). Additional information on the need for restrictions and control measures for aquatic disposal and the evaluation of other disposal alternatives can be found therein and in Cullinane et al. (1986). A more comprehensive discussion of the interpretation of test results is provided by Peddicord et al (1987).

### Innovative Assessment Techniques

The enactment of Public Laws 92-532 (the Marine Protection, Research, and Sanctuaries Act of 1972) and 92-500 (the Federal Water Pollution Control Act Amendments of 1972) required the Corps to participate in developing guidelines and criteria for regulating dredged and fill material disposal. The focal point of research for these procedures is the Corps Dredged Material Research Program (DMRP), which was completed in 1978; the ongoing Corps Environmental Effects of Dredging Programs (Engler, Patin, and Theriot 1988) includes the Dredging Operations Technical Support (DOTS) Program, the Long-Term Effects of Dredging Operations (LEDO) Program, the Wetlands Research Program (WRP), and the recently completed Corps/EPA Field Verification Program (FVP).

While these research programs have allowed the Corps to develop an extensive and effective set of testing protocols and evaluation procedures, there continues to be a requirement for additional research. Less expensive, faster, and improved techniques for predicting the effects of disposal of dredged material are needed. Accordingly, innovative development of new and refined evaluation procedures are being undertaken through appropriate R&D programs of the Corps. However, until new procedures are proven through adequate documentation, existing techniques must be relied upon.

### Summary

The "Federal Standard" guidance serves as a consistent national framework and reference point for Corps field offices which provides for consideration of regional issues in dredged material management. In applying the process to different projects or regions of the country, it may be necessary to adopt specific testing procedures consistent with the Federal Standard Philosophy. Corps field office evaluations must be consistent with the national procedures, defensible in light of research results and scientific judgment, cost and time effective, and of direct use in decisionmaking.

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# *Environmental Effects of Dredging Technical Notes*



## GENERAL GUIDELINES FOR MONITORING EFFLUENT QUALITY FROM CONFINED DREDGED MATERIAL DISPOSAL AREAS

**PURPOSE:** This note provides general guidance for developing plans for routine field monitoring of the quality of the effluent from confined dredged material disposal areas for determining compliance with effluent standards. It also provides guidance on additional monitoring which can provide other useful information for the project designers and sponsors.

**BACKGROUND:** The Corps of Engineers must dredge about 300 million cu yd of sediments from the nation's harbors and navigation channels each year to accomplish its mission of keeping the waterways open for navigation. Some of this material, especially in industrial harbors, is contaminated by pollutants, either residual materials in treated discharges from cities and industries or materials washed from farms, streets, parking lots, or industrial areas by runoff. In many cases, contaminated dredged material may not be disposed of in open water, but must be placed on land in a confined disposal area. The effluent from these large sedimentation basins/storage areas is considered a discharge under Section 404 of the Federal Water Pollution Control Act, which requires a permit. Certification must be obtained from or waived by the state under Section 401 such that the effluent discharge will not violate applicable water quality standards. Section 401 also requires the certification to set forth necessary effluent limitations and monitoring requirements. A National Pollution Discharge Elimination System (NPDES) permit is not required, so NPDES monitoring should not be imposed. This note provides guidance for developing appropriate effluent quality monitoring programs.

**ADDITIONAL INFORMATION:** The authors of this technical note are Dr. Edward L. Thackston of Vanderbilt University and Dr. Michael R. Palermo of the US Army Engineer Waterways Experiment Station. For additional information, contact Dr. Palermo, (601) 634-3753, or the manager of the Environmental Effects of Dredging Programs, Dr. Robert M. Engler, (601) 634-3624.

### General Considerations

The quality of effluent during filling operations is of concern for confined disposal projects when the sediments being dredged are contaminated.

Modified elutriate tests (Palermo 1985) and column settling tests (Headquarters, US Army Corps of Engineers 1987) provide information for prediction of the quality of effluent for a given set of operational conditions. The Section 401 water quality certification may contain requirements for effluent quality monitoring to ensure that standards are met.

Depending on the situation and the extent of data collection, the data gathered in routine field monitoring can be used to (1) demonstrate permit compliance, (2) aid in control of the dredging contractor to ensure compliance, (3) aid in demonstrating the adequacy of the disposal area design, (4) document the water quality impact (or lack thereof) if there are public concerns, and (5) document the presence of contaminants in the disposal area in case there are concerns about later land use.

In developing an effluent quality monitoring program, the following considerations should be addressed:

1. Parameters to be monitored.
2. Sampling and analysis techniques.
3. Sampling locations.
4. Monitoring frequency.

These and other aspects of effluent quality monitoring programs are discussed in detail in the following paragraphs.

#### Parameters to be Monitored

Parameters of potential interest may be grouped or classified in different ways, but the parameters (followed by examples) usually of most interest in dredging can be classified as follows:

1. Physical and physicochemical--temperature, suspended solids (SS), dissolved oxygen (DO), pH, and turbidity.
2. Nutrients--total organic carbon (TOC),  $\text{NH}_3$ ,  $\text{NO}_3^-$ , and  $\text{PO}_4^-$ .
3. Metals--iron (Fe), manganese (Mn), nickel (Ni), zinc (Zn), cadmium (Cd), and chromium (Cr).
4. Organics--polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and pesticides.

Parameters to be monitored are site specific and should be chosen only after an analysis of all conditions relating to a project, including the bulk sediment analysis, the effluent prediction if one is made, the water quality

and hydrodynamics of the receiving water, and the requirements set forth by the state in the water quality certification. Contaminants should only be monitored if they are expected to be present. All parameters of concern need not be monitored at all locations at all times.

The first parameters to be selected are obviously the ones specified in the state certification and the discharge permit. They vary widely, depending on the site and the state, but usually SS, DO, some nutrients, and often some heavy metals are included. Specific toxic organics are normally not required to be monitored, unless there is evidence of their presence in the sediments in concentrations high enough to be of concern.

Suspended solids (or turbidity) should always be monitored, whether specified by the state or not, because it helps in management of the facility and evaluation of the design and is an indicator of other parameters. SS is the best indicator of overall performance of the disposal area, both for solids retention and for most other contaminants, which are strongly associated with SS by adsorption or ion exchange. Turbidity is a much more easily measured parameter than SS (it can usually be measured by the inspector in the field) and can often be used instead of SS routine monitoring after a correlation between the two has been established for the particular sediment and site. Earhart (1984) has described a method for correlating these parameters. Often, water quality standards are expressed in terms of turbidity, and thus it becomes the basic controlling parameter itself. Temperature, pH, and DO are easy to measure with a probe, but these parameters are rarely of concern, because dredging has little impact on them.

During the planning stages of all dredging projects in urban-industrial areas where contamination is likely, samples of the sediments to be dredged should be taken and analyzed for all contaminants which are reasonably expected to be present. Those which are potentially troublesome and which are found in the sediments in concentrations of concern should be monitored. These may include nutrients; toxic metals such as cadmium, chromium, nickel, or zinc; and toxic organics such as pesticides or PCBs.

#### Sampling and Analysis Technique

Standard procedures for sampling, preserving, and analyzing water samples should be followed for effluent quality monitoring programs. Detailed

guidance is contained in Plumb (1981), American Public Health Association (1985), and US Environmental Protection Agency (USEPA) (1979, 1980, 1982, and 1986). These documents are updated periodically, so the latest version should always be used.

### Sampling Locations

Several locations should be considered for monitoring, although not every monitoring location needs to be sampled each time for every parameter. Under Section 404, the effluent should meet applicable water quality standards within an acceptable mixing zone. Therefore, sampling should always be conducted at the edge of the mixing zone to determine permit compliance. Upstream or background receiving water should always be sampled to determine ambient conditions.

Several other locations may be sampled to provide information on the performance of the disposal area. Sampling at the overflow weir will provide data on the adequacy of the site design and the accuracy of laboratory tests used for effluent quality prediction. Sampling the influent to the disposal area (the dredge discharge) allows determination of the approximate removal efficiency of the disposal area and allows an estimation of the contaminant concentration of the stored dredged material.

In certain situations, there may be other desirable sampling locations. In multicell disposal areas with weirs between cells, the internal weir overflows should be sampled, at least for SS. This allows determination of the incremental removal efficiency of each cell as it changes during the project and is very useful information to the designer. In multipass disposal areas with internal baffles to reduce short circuiting, samples for SS at the openings may be useful. Where there is no direct discharge, but flow of supernatant water through porous dikes into a surrounding water body, the influent, the surface layer at a location away from the influent, and the water in the surrounding water body at the waterline adjacent to the dikes (in several locations) should be sampled.

### Monitoring Frequency

The desirable frequency of sampling varies widely, depending on permit

requirements, anticipated environmental impact, size and duration of project, progress of the dredge, the hydraulic retention time of the disposal area, and the funds available for analysis. Three samples should be the minimum number taken at any location, since three samples are required to determine a variance.

The maximum number of samples (or minimum sample spacing) is a function of the size or average retention time of the site. The sites act as mixing and equalization basins, damping out most sharp fluctuations in influent concentration, so the effluent concentrations are much less variable than are influent concentrations. As the average retention time increases, more mixing occurs and fewer samples are required to define the effluent characteristics. One sample per average hydraulic retention time is the maximum frequency that can be practically justified. The average retention time varies during the project, so the sampling frequency should vary also. Because most sites have an average retention time on the order of 24 hours, daily sampling for SS or turbidity is convenient and is recommended.

Sampling for nutrients, toxic metals, or organics, if required, can be less frequent, approximately once every two weeks. If frequent samples are analyzed for SS, which is easy and inexpensive to determine, less frequent samples for chemical contaminants are necessary, because variations in chemical concentrations are usually proportional to SS concentrations. Also, more frequent sampling does not necessarily provide more usable information, because analytical results for nutrients, metals, and organics frequently are not available for several weeks.

For a given average frequency of samples desired, sample spacing should be less (more frequent samples) when the dredge is moving rapidly, pumping consistently at a high rate, or is moving through highly contaminated areas. Sample spacing should be greater (less frequent samples) when the dredge is moving slowly, or is shut down often, or is moving through areas known not to be heavily contaminated.

Although water quality at the overflow weir is normally relatively stable, it can change very rapidly with changes in the weather. Therefore, samples should not be taken when the effluent from the disposal area is especially high in SS for short periods because of high winds, hydraulic surges from the dredge, weir problems, or other brief upsets, unless it is desired to document worst-case conditions. Such samples should be taken from

the first overflow following an extended period of zero outflow, because these samples will be uncharacteristically low in SS and other contaminants.

Composite samples may be more accurate indicators of the true average conditions at a point than grab samples, especially for situations in which conditions fluctuate greatly. This is the case for many confined disposal areas. Therefore, if conditions and resources allow it, composites should be used. Composite samples may be taken in many ways. If sampling personnel will be on site for several hours, several grab samples may be taken during this time and composited. Automatic samplers may also be used to obtain periodic grab samples which can then be composited. It may be especially desirable to use composites for samples taken only infrequently, such as the ones for nutrients, heavy metals, TOC, and organics.

#### Typical Monitoring Program

As an illustration, a sampling schedule is presented below for a typical project. This hypothetical project is in an industrial harbor where a sediment inventory has indicated the presence of toxic metals and organics. The project dredging will take 8 weeks, and the disposal site will have an average retention time varying from two days at the beginning of the project to about one day at the end. The permit specifies a mixing zone 1,000 ft long. For these conditions, the recommended sampling schedule is as follows:

1. At the point of permit compliance (downstream end of mixing zone).
  - a. SS--daily.
  - b. Nutrients, metals, and organics--once every two weeks.
2. Influent.
  - a. SS--twice per week.
  - b. Nutrients, metals, and organics--once every two weeks.
3. Background in receiving water.
  - a. SS--once per week.
  - b. Nutrients, metals, and organics--three samples.
4. At the weir.
  - a. SS--twice per week.
  - b. Nutrients, metals, and organics--once every two weeks.

If cost considerations require that the total number of samples be

reduced, the ratios of sampling frequencies should stay approximately the same.

#### Other Monitoring Requirements

In addition to taking water samples for analysis to determine concentrations of contaminants, other monitoring should be done to provide control over the quality of water discharged or to furnish background information to aid in the interpretation of the analytical results. This monitoring should be done by the resident engineer or inspector for the Corps.

On at least a daily basis, the inspector should observe and record the physical condition of the levees and discharge structure. He should note the condition of the weir boards, whether the weir is leaking, whether floating solids are caught on the weir, whether the weir is unlevel, and whether there are other unusual circumstances. Any change in weir elevation should be recorded.

The inspector should also note and record the visual quality of the effluent (whether clear, slightly turbid, or very turbid); any obvious flow patterns or changes, such as formation of deltas or obvious short-circuiting; and wind and weather conditions, especially the direction of the wind and relative wind velocity.

#### Cost

The cost of monitoring and analysis varies widely, depending on the length of the project, the number of locations sampled, and the parameters analyzed. Jacek (1986) reported that the cost of operational monitoring in the Detroit District varied from \$2,100 to \$15,500 per project and averaged \$5,900. Individual laboratories may charge more or less, depending on local conditions, the number of samples analyzed simultaneously, and the number of parameters analyzed for in each sample.

The sampling schedule shown for the previous example would cost about \$5,000 to \$10,000 for analysis alone. To this, expenses for the sampling itself must be added, but these should be minimal, since samples can be taken by the regular inspectors. This estimate is in line with the costs reported by the Detroit District. Jacek and Schmitt (1986) reported that, in the

Detroit District, about 20 parameters are generally monitored at about five to nine locations, about three or four times during each project. This results in slightly more parameters and locations, but fewer sampling times, than the typical project and the example.

Costs can run as high as \$100,000 per project if numerous organics which are present at very low concentrations are monitored or if the project is a lengthy one. However, this is unusual. The typical project monitoring cost is in the range \$5,000 to \$25,000.

### Monitoring Responsibility

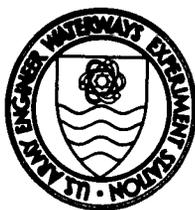
Responsibilities and duties vary from District to District and from project to project. However, usually the project sponsor (the Corps or other agency) is responsible for obtaining the state water quality certification and for meeting the Section 404 requirements (doing the Section 404(b)(1) analysis for a Corps project or actually getting a 404 permit if another agency is the sponsor), informing the dredging contractor of his responsibilities relative to water quality, taking the water samples, and transporting them (according to accepted standards of sample preservation) to the laboratory (in-house or commercial) selected to analyze them. The project sponsor is usually responsible for making those water quality measurements which can be done easily in the field with electronic probes or field instruments, such as DO, pH, temperature, and turbidity.

The laboratory is responsible for analyzing the samples for a preselected list of parameters (preselected by the project sponsor and listed in their contract or work order) and for reporting the results to the sponsor in a timely manner. The project sponsor is responsible for transmitting the results to appropriate state and local water pollution control agencies.

The Corps should be responsible for monitoring and recording the physical condition of the disposal area and the dredging operation and for documenting occurrences which might affect water quality or explain anomalies in the data.

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# *Environmental Effects of Dredging Technical Notes*



## PROCEDURES FOR EXAMINING THE RELATIONSHIP BETWEEN SEDIMENT GEOCHEMISTRY AND BIOLOGICAL IMPACTS OF CONTAMINANTS

**PURPOSE:** This note describes the development of procedures for examining the relationship between sediment geochemistry and biological impacts of contaminants. These relationships are illustrated by implementation of procedures in a laboratory experiment to assess the influence of polychlorinated biphenyls (PCB) contaminated sediment on bioaccumulation in estuarine clams.

**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, may profoundly affect US Army Corps of Engineers (USACE) dredging and disposal operations. Aquatic disposal of dredged material and selection of aquatic disposal sites may 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 sediment. The USACE is presently investigating the link between contaminant levels in sediment, sediment geochemistry, and 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.

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### Introduction

The relationship between sediment-bound contaminants and biological uptake of these contaminants is complex because of the many physical, chemical, and biological factors that can affect the relationship (McElroy and Means 1988). Operational and procedural problems encountered in determining how a sediment-associated contaminant affects aquatic organisms cause additional complications. If sediment quality criteria (SQC) are to be used to regulate dredged material disposal, prediction of biological responses based on changes in sediment

geochemistry, i.e., sediment physical and chemical properties, and sediment contaminant levels must be possible.

Radioactive tracers can be used to evaluate the effects of changing concentrations of sediment contaminants on aquatic organisms if the assumption can be made that the contaminant does not degrade during the study. Spiking a sediment with contaminants has generally been accomplished by the addition of organic solvent carriers containing the contaminant to the soil or sediment (Adams, Kimerle, and Mosher 1985; Word et al. 1987). Sediments in the aquatic environment are not usually exposed to contaminants dissolved in organic solvents, but to contaminants dissolved in water. However, most organic contaminants of interest exhibit such limited aqueous solubilities that spiking with aqueous solutions is impractical. Therefore, methods that closely simulate aqueous introduction of contaminants are desirable.

The laboratory experiments described in this note were designed to compare sediment amendment procedures; to determine the time necessary for added polychlorinated biphenyls (PCBs) to reach steady-state conditions with sediment interstitial water; to evaluate the utility of a small, simple bioassay apparatus; and to investigate the effect of different sediment PCB concentrations on PCB bioaccumulation by deposit-feeding clams.

### Materials and Methods

#### Sediment amendment and kinetics studies

Sediment from Oakland Inner Harbor, Oakland, CA, was amended with either 0.1 or 1.0  $\mu\text{g}$  PCB-52 ( $[^{14}\text{C}]2,2',5,5'$ tetrachlorobiphenyl)/g dry weight. The PCB-52 in a methanol carrier was either added directly to the sediment or indirectly by evenly coating the walls of 25-ml centrifuge tubes and allowing the methanol to evaporate to dryness prior to addition of water-sediment mixtures. Tubes were sampled at 1, 3, 7, 14, 30, and 45 days by centrifuging to remove particulates larger than 0.01  $\mu\text{m}$  from solution, then counting one ml of solution using liquid scintillation (LS). Ten millilitres of the solution was passed through a C-18 Sep-Pak cartridge (Waters Associates, Milford, MA) to separate the dissolved PCB-52 from both PCB-52 associated with suspended micro-particulates (smaller than 0.01  $\mu\text{m}$ ) and PCB-52 associated with dissolved organic macromolecules, such as humic acids (Landrum et al. 1984). One millilitre of

the solution or suspension that had passed through the Sep-Pak was counted by LS.

The experiment was repeated with higher sediment loading (20  $\mu\text{g}$  PCB-52/g dry weight sediment). A water-sediment ratio of 2:1 and sampling times of 2 hr, 6 hr, 24 hr, 3 days, and 7 days were used. The PCB-52 remaining on the glass container walls was also determined.

#### Clam bioaccumulation study

A sediment bioassay apparatus similar to that used by McElroy and Means (1988) was selected for initial bioaccumulation studies because of its small size and simplicity. Details of the apparatus are shown in Figure 1. Each bioassay apparatus was maintained in a water bath at 17.5° C, the temperature at which the clams were collected. Foam plugs (McElroy and Means 1988) were used to trap PCB-52 volatilized or stripped from the water by aeration. Oakland Inner Harbor sediment was amended indirectly with either 1 or 10  $\mu\text{g}$  PCB-52/g dry sediment. Five clams (*Macoma nasuta*) were introduced to each bioassay apparatus. Overlying water, interstitial water, foam plugs, and clams were sampled, processed, and counted by LS at each sampling period. Clam lipids and organism dry weight were also determined.

#### Data analysis

All statistical analyses were conducted using methods developed by the Statistical Analysis Systems Institute (Barr et al. 1976).

### Results and Discussion

#### Sediment amendment

Results of the sediment amendment and kinetics studies showed that desorption of PCB-52 from the container walls was rapid and virtually complete and that solution steady-state PCB-52 concentrations were reached within 24 hr after shaking was initiated. Therefore, the indirect spiking method was selected in order to avoid addition of solvent or carrier to the sediment and to simulate introduction of PCB to the sediment via water.

#### Bioassay

PCB-52 concentrations in interstitial water between sampling times did not substantially change in either the 1  $\mu\text{g}$  PCB/g or 10  $\mu\text{g}$  PCB/g treatments during the bioassay (Figure 2). Losses of PCB-52 from the system through volatilization increased as the experiment progressed, accounting for 0.54 and 1.44 percent of

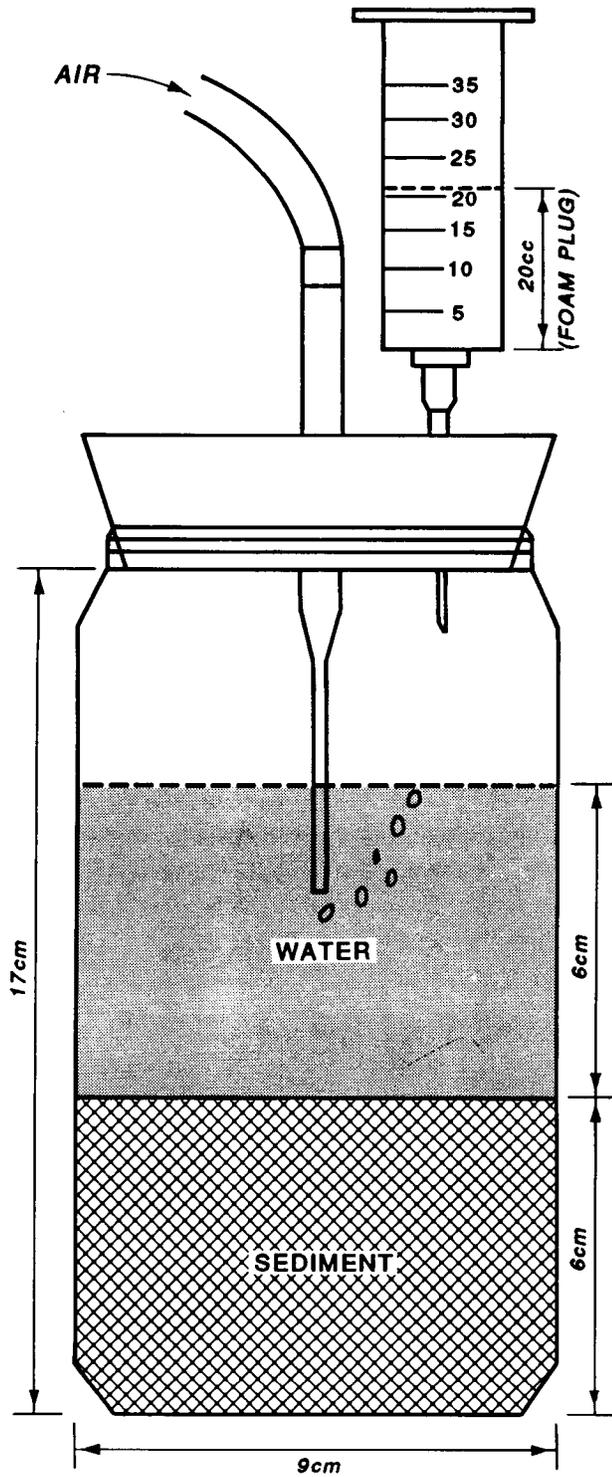


Figure 1. Testing apparatus, showing syringe body containing foam plug for collection of volatile PCB-52

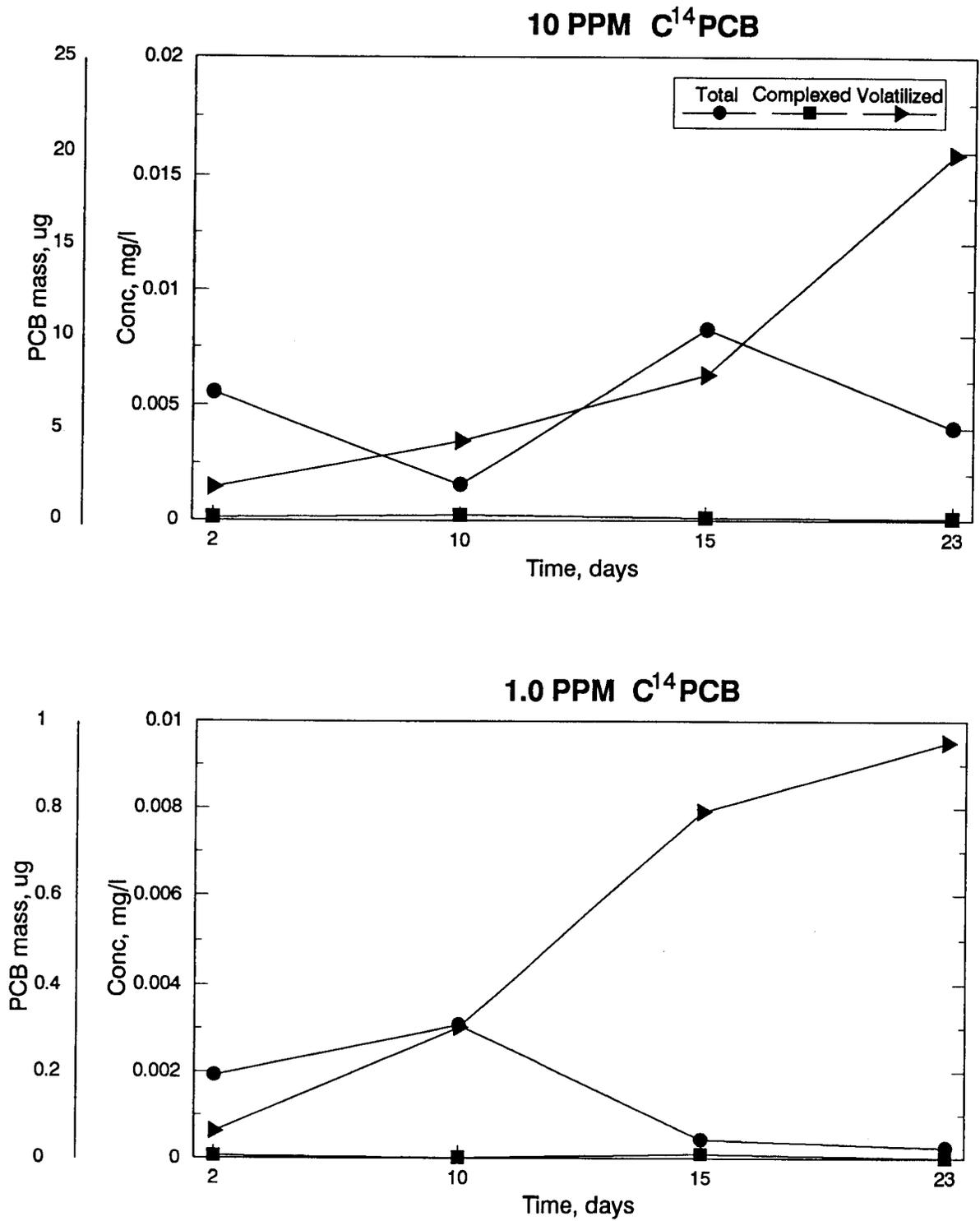


Figure 2. Concentrations (mg/l) of C<sup>14</sup> PCB-52 in sediment interstitial water (total and complexed) and mass ( $\mu\text{g}$ ) of PCB-52 lost through volatilization

the total mass of PCB-52 in the 1 and 10  $\mu\text{g}$  PCB/g treatments, respectively. Volatilization losses would affect bioassays with water column organisms if the organic contaminant is stripped from the water faster than it can be replenished from the sediment. This experiment used a deposit-feeding clam that is less affected by water column volatilization losses.

Tissue concentrations ( $\mu\text{g/g}$  wet weight) of PCB-52 increased steadily as exposure time increased in both the 1  $\mu\text{g}$  PCB/g and 10  $\mu\text{g}$  PCB/g treatments (Figure 3). Fewer than four replicates are shown at some sampling times because of sample loss through death of organisms, possibly because of disease.

An apparent preference factor (APF), a measure of the preference of neutral organic contaminants for organism lipids as opposed to sediment organic carbon, for each time point was calculated using the equation

$$\text{APF} = (\text{PCB}_s / \% \text{TOC}) / (\text{PCB}_o / \% \text{lipid})$$

where

$\text{PCB}_s$  = PCB concentration in sediment,  $\mu\text{g/g}$  dry weight

$\% \text{TOC}$  = percent total organic carbon, g/g dry weight

$\text{PCB}_o$  = PCB concentration in clams,  $\mu\text{g/g}$  wet weight

$\% \text{lipid}$  = percent lipid in organism extracts, g/g wet weight

This equation, taken from McElroy and Means (1988), is based on the thermodynamic bioaccumulation potential (TBP) equation of McFarland (1984) and the preference factor equation of Lake, Rubinstein, and Pavignano (1987). TBP gives the maximum theoretical concentration of a neutral organic compound that can be bioaccumulated from sediment. Percent TOC in the Oakland Harbor sediment was 1.06.

The values of the APFs calculated at 10 days and later in this study (2.3 for 10  $\mu\text{g/g}$  and 3.9 for 1  $\mu\text{g/g}$  PCB-52 treatments) (Figure 4) are similar to those for other empirical determinations reported in the literature (Feraro et al., in preparation; Clarke, McFarland, and Dorkin 1988) and are not greatly different from the theoretical preference factor (pf) of 1.72 calculated by McFarland and Clarke (1986). The observations in this study are consistent with and support previous results and indicate good correspondence between the laboratory results using spiked sediments and field studies involving natural sediments and assemblages of biota (Feraro et al., in preparation; Clarke, McFarland, and Dorkin 1988).

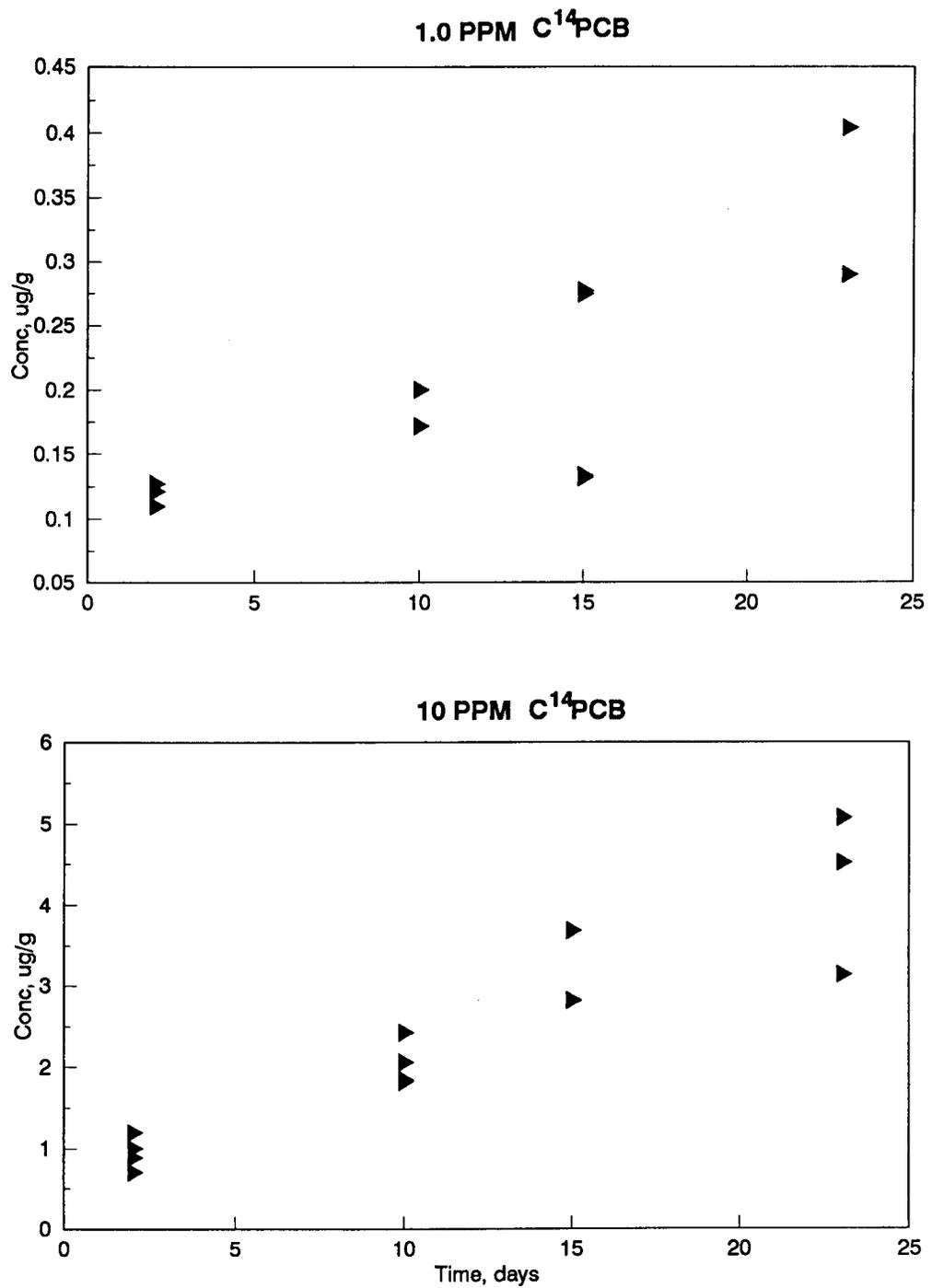


Figure 3. Replicate C<sup>14</sup> PCB-52 concentrations in clam tissue on a wet weight basis

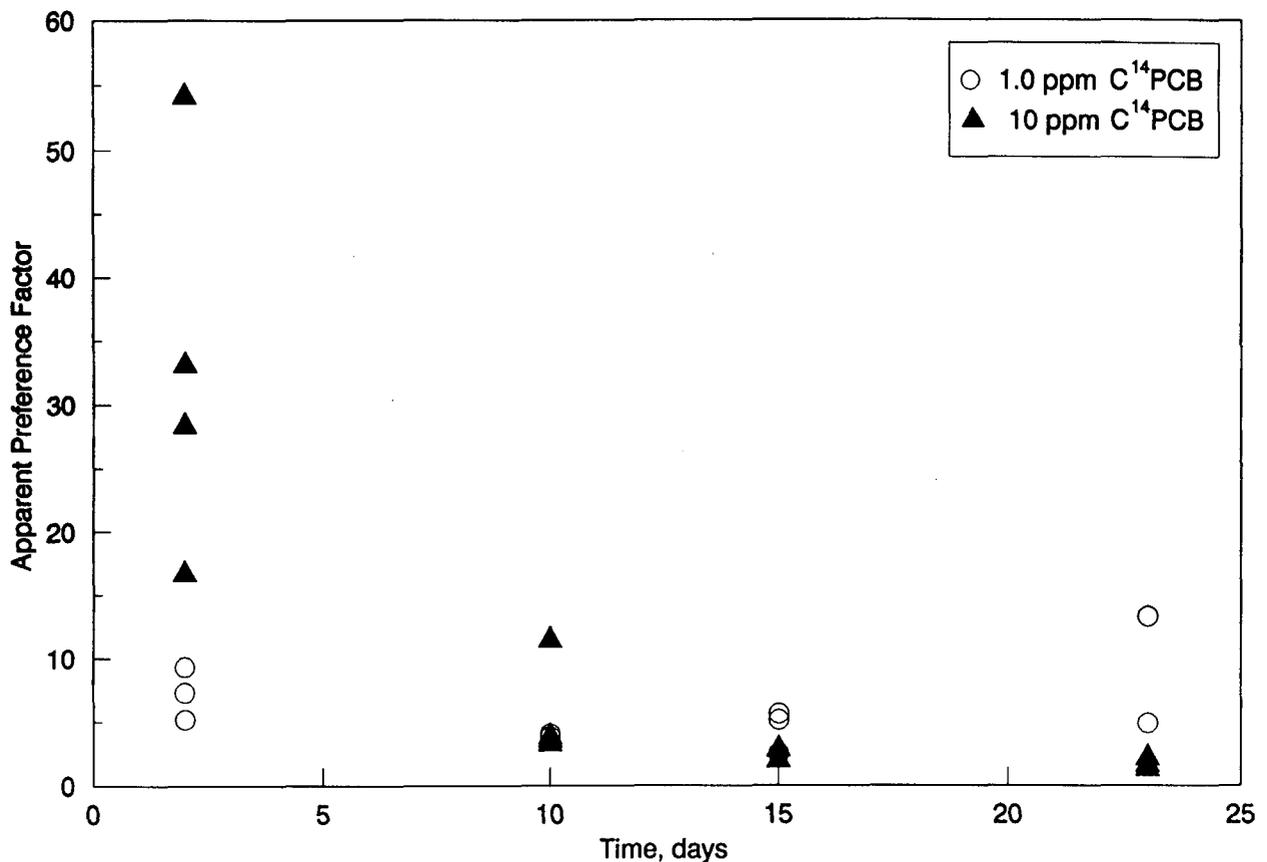


Figure 4. Replicate APFs for C<sup>14</sup> PCB-52 in clam tissue

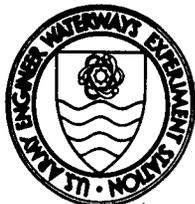
#### Summary of Findings

- The sediment bioassay apparatus has proved to be versatile experimental tool suitable for further studies.
- Sediment spiking with radioisotopes provides a means of examining sediment geochemistry/bioavailability relationships that favorably compares to results obtained with real-world sediment.
- PCB-52 volatilization losses from the bioassay experimental units raise questions about linking interstitial water PCB concentrations to biological effects observed in the water column.
- Equilibration of PCB-52 with lipids occurs rapidly in the deposit-feeding clam, *Macoma nasuta*; lipid normalization of contaminant data, as carried out in the equation for calculating APF, may obviate the need to carry out bioaccumulation tests for similar chemicals for longer than 10 days.

- The average (10, 15, and 23 days) APF for PCB-52 in the 10  $\mu\text{g}$  PCB/g exposures to spiked sediments (1.94) compares well to the reported (Feraro et al., in preparation; Clarke, McFarland, and Dorkin 1988) APFs of PCB-52 in natural sediments (0.52 to 2.1), and to the theoretical preference factor (1.72).

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# *Environmental Effects of Dredging Technical Notes*



## A PLANT BIOASSAY FOR ASSESSING PLANT UPTAKE OF HEAVY METALS FROM CONTAMINATED FRESHWATER DREDGED MATERIAL

**PURPOSE:** The Decisionmaking Framework (DMF) developed by Peddicord et al. (1986) provides a framework for evaluating sediments before dredging. This framework is comprised of several modules one of which is the Plant Bioassay for materials proposed for upland or wetland placement. The purpose of this note is to describe the methods and materials necessary to conduct such a plant bioassay.

**BACKGROUND:** The US Army Engineer Waterways Experiment Station (WES) has developed a plant bioassay using the freshwater plant *Cyperus esculentus* to evaluate phytotoxicity and potential mobility of contaminants from dredged material into the environment through plant uptake (Folsom and Lee 1981; Folsom, Lee, and Bates 1981). The plant bioassay procedure is an excellent tool for predicting bioaccumulation of heavy metals (e.g. zinc and cadmium) from freshwater sediments (Lee, Folsom, and Bates 1983). The bioassay was successfully evaluated using metal-contaminated Dutch sediments (Van Driel, Smilde, and van Luit 1983) as well as Welsh mining wastes (Folsom, Davis, and Houghton, 1988). Like the DMF, the Plant Bioassay Module is based on tiered testing. Tier I is a chemical extraction of test and reference sediments; Tier II is a laboratory/greenhouse plant bioassay. If the results of Tier I testing indicate a reason for concern, then the test sediment could be subjected to Tier II testing to verify the concern. Tier II testing consists of of a laboratory/greenhouse plant bioassay using test and reference sediments.

**ADDITIONAL INFORMATION:** This note was written by Dr. Bobby L. Folsom, Jr., and Mr. Richard A. Price, Environmental Laboratory. For additional information contact Dr. Folsom, (601) 634-3720, or the manager of the Environmental Effects of Dredging Programs, Dr. Robert M. Engler, (601) 634-3624.

**NOTE:** The contents of this technical note are not to be used for advertising, publication, or promotional purposes. Citation of trade names does not constitute an official endorsement or approval of the use of such commercial products.

## Sediment Collection and Preparation

Sediments to be tested are collected from a waterway using an appropriate sampler that can sample the entire vertical profile of the material to be dredged. The plant bioassay actually requires 16 gal of each sediment to be tested. Therefore, a 76- or 114-l (20- or 30-gal) drum would provide sufficient quantity of material to conduct the required testing; however, 208-l (55-gal) steel drums are generally easier to obtain. The drums are sealed with airtight lids and transported to the laboratory. Temperature during shipping should be maintained at  $4^{\circ} \pm 2^{\circ} \text{C}$ . Unless new, the drums should be steam-cleaned prior to use.

Before testing, the original, flooded sediments should be mixed thoroughly while in their respective drums using a mixer. All debris, such as cans, bottles, leaves, or twigs, is removed. The mixer is raised and lowered to thoroughly mix the sediment contained in each drum. Generally, one hour of mixing is required for adequate sediment homogenization. Testing using four replicates results in sample variability (coefficient of variation) of less than 10 percent. Subsamples from the drums represent original, flooded sediment and are used in the procedure described below.

### Flooded condition

A schematic diagram of the standard WES plant bioassay apparatus is shown in Figure 1. The mixed original, flooded sediment is placed one 500-ml scoop at a time to further minimize mixing variability, into each of four 7.6-l (2-gal) Bain-Marie containers. When the containers are filled with sediment, sediment diameter and sediment depth should be in a ratio of 1:1.5. Flooded conditions are maintained by keeping a 5-cm depth of deionized water or distilled water above the sediment surface of the inner Bain-Marie container. Percent moisture on an oven-dry weight basis (ODW) (oven temperature is  $105^{\circ} \pm 2^{\circ} \text{C}$  for 17 hr) is determined on small (5- to 10-g) subsamples of the flooded material in each replicate. A test sediment weight of 4,500 g (ODW) per replicate generally provides sufficient plant material for maximum plant growth. Since flooded and upland biomass production (yield) is one of the Decisionmaking Framework (DMF) comparisons, each replicate must contain the same quantity of sediment (ODW). The four flooded replicates are sealed with their lids and stored at  $4^{\circ} \pm 2^{\circ} \text{C}$  until the upland replicates (described below) are prepared for planting. Drying

and preparation of the upland replicates should be completed within three weeks of collection.

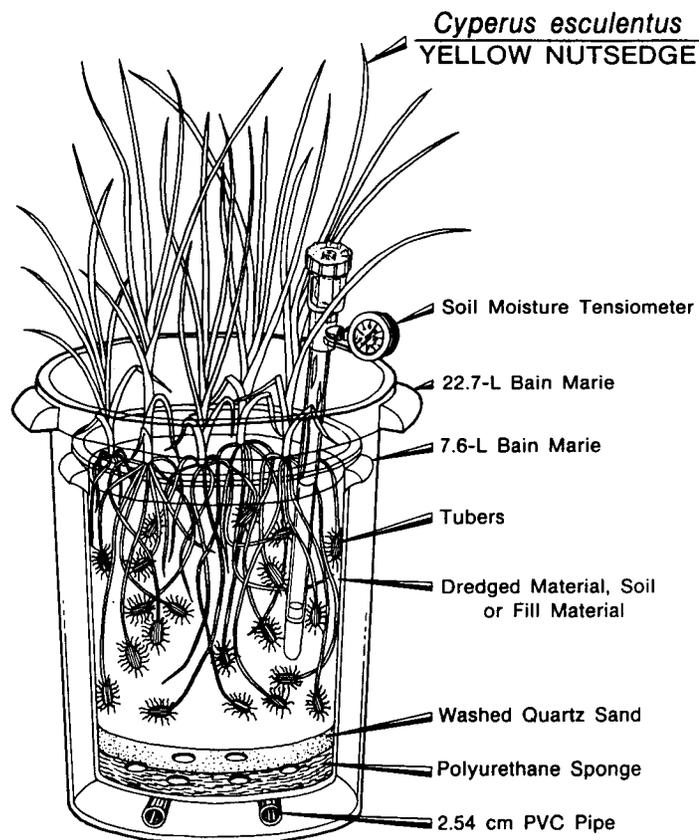


Figure 1. Schematic diagram of the Plant Bioassay Apparatus; sand layer and sponge are each 2.54 cm thick

### Upland condition

Four additional sediment replicates are prepared as described above for the flooded condition except that the sediment from each container is placed into an aluminum drying pan and allowed to air dry. The sediment must be turned and mixed daily with a large plastic spatula or Teflon-coated shovel to facilitate drying. All debris, such as cans, bottles, leaves, or twigs, is removed as the sediment dries. If large quantities of large rocks, gravel, and other materials are present, then a separation analysis should be conducted (Engineer Manual 1110-2-1906, Appendix V) (Headquarters, US Army Corps of Engineers 1970). After air-drying the sediment, most sediments form large bricklike clods that are extremely difficult to crush. Crushing and grinding of these clods is best

performed using a hammermill. Personnel operating the hammermill should wear appropriate respirators and protective clothing. One pass through the hammermill is sufficient for the material to pass a 2-mm screen (U.S. Standard Sieve No. 10). Greenhouse pot experiments generally use material that has been ground to pass a 2-mm screen to approximate field macroporosity (pore space affects particle surface area, drainage, gas movement, and other items) to estimate weathered sediment placed in an upland disposal site. The screened material is returned to a drying flat where it is remixed and subsampled for ODW analysis. Air-dried sediment (4,500 g ODW) is placed (one 500-ml scoopful at a time) into each of four 7.6-ℓ Bain-Marie containers prepared as before. The remaining air-dried sediment can be placed into an appropriate container (7.6-ℓ Bain-Marie bucket is a good choice), and stored until needed for subsequent chemical or physical analyses, if necessary. For air-dried replicates soil moisture is maintained between 0.04 and 0.06 MPa (field capacity is 0.00 MPa) by checking soil moisture tensiometers in each container daily. Plants are watered when tensiometers read greater than 0.06 MPa (generally every other day). When watering is necessary, the outer container is filled up to the sediment level of the inner container with distilled water. When tensiometers read less than 0.04 MPa, the water is siphoned out of the outer container.

Material from the reference or proposed disposal site is prepared in exactly the same manner as that described above for each disposal condition (i.e. flooded and upland).

#### Greenhouse Operation

The replicates are randomly placed on tables in a greenhouse. Day length of 16 hr is maintained by using light fixtures whose face is 130 cm from the top of the 22.7-ℓ Bain-Marie container. The 130-cm height allows potential maximum plant growth to occur without contacting the light fixture or becoming so close to the light that plant tissue is damaged from excess heat. Lights should be arranged in a pattern of alternating high-pressure sodium lamp and a high-pressure multi-vapor metal halide lamp. Alternating the lamps provides an even photosynthetic active radiation (PAR) distribution pattern. The PAR should be 1,200 microeinsteins per metre squared. The temperature of the greenhouse is maintained at  $32.2^{\circ} \pm 2^{\circ}$  C maximum during the day and  $21.1^{\circ} \pm 2^{\circ}$  C minimum at night to simulate a summer environment. Relative humidity should be maintained

as closely as possible to 100 percent, but never less than 50 percent.

### Planting and Growing Techniques

The plant used in the WES plant bioassay is *Cyperus esculentus*. Normally, *C. esculentus* (common name, yellow nutsedge) is considered a persistent major problem weed and causes yield reductions in many crops of the United States, Canada (Mulligan and Junkins 1976; Wills, Hoagland, and Paul 1980), and throughout the world (Holm et al. 1977). It is also considered a pioneer species that invades disturbed areas (e.g. dredged material disposal sites) readily (Mulligan and Junkins 1976). Although *C. esculentus* reproduces by seeds, tubers, bulbs, and rhizomes, tubers are the primary means of reproduction (Bell et al. 1962; Tumbleson and Kommedahl 1961), even though it may flower and produce seed under certain conditions (Mulligan and Junkins 1976). Therefore, *C. esculentus* was chosen as the plant bioassay index plant because of its natural tenacity and its ability to survive in both flooded and upland conditions and showed greatest potential for heavy metal uptake compared to other plant species (Lee, Sturgis, and Landin 1975). *Cyperus esculentus* also has a fairly short 45-day vegetative growth period under long days (Doty and Sweet 1970).

Each replicate of flooded and upland sediment is planted with four germinated tubers of *C. esculentus*. Suppliers of the tubers include Valley Seed Services (P.O. Box 8335, Fresno, CA 93791, phone: 209-435-2763) and Wildlife Nurseries (P.O. Box 2724, Oshkosh, WI 54903; phone 414-231-3780). Because germination of *C. esculentus* is close to 50 percent (Thomas 1969; Yip and Sweet 1978), twice as many tubers as needed for the experiment are set out for germination. Germination temperature is  $23^{\circ} \pm 2^{\circ}$  C in the light. The tubers are first rinsed in distilled water to remove substances that inhibit sprouting of buds on the tubers (Mulligan and Junkins 1976). The tubers are then placed between white paper towels and kept moist with distilled water until enough have sprouted to plant five tubers per container (usually seven to ten days). Sprouts should be approximately 3 cm long before planting. Plants are allowed to grow for 45 days from the time of planting.

### Harvesting

After 45 days, the aboveground plant material from each replicate is cut 5 cm above the sediment surface with stainless steel scissors and placed into

a labeled brown paper bag perforated by several holes to allow water vapor to escape during drying. Any flowers, stems, or seeds that may have developed are separated from the leaves, wrapped separately with white paper towels, and placed into the bag with the leaves. The bags containing the harvested plant material are dried to a constant weight in a forced-air drying oven at  $70^{\circ} \pm 2^{\circ} \text{C}$  (generally four or five days). All dried tissue is removed from the bags and weighed separately. Total plant yields are determined by weighing the oven-dried plant material.

### Digestion and Chemical Analysis of Plant Material

The dried leaves are ground in a small Wiley mill. Two grams (weighed to the nearest 0.0001 g) of the ground leaf tissue are digested using the tertiary acid digestion procedure except that 2.0 g ODW tissue are used rather than 1.0 g (Folsom and Houck in preparation). In some sediments, plant growth is not sufficient to provide 2.0 g of tissue. In these cases, whatever amount of tissue produced is digested. Chemical analysis of flowers, stems, and seeds is not conducted since their production is sporadic. Care should be taken during the initial digestion because excessive frothing of the nitric acid may occur, rendering that replicate useless. The diluted digestates are analyzed for heavy metals by atomic absorption (AA) spectroscopy or heated graphite analysis (HGA). Results of the digestion are calculated using the equation:

$$\begin{aligned} \text{Tissue metal concentration} &= \frac{\text{solution metal concentration} \times \text{dilution volume}}{\text{ODW leaf tissue digested}} \\ &= \frac{\mu\text{g metal/ml} \times 50 \text{ ml}}{\text{grams tissue}} \\ &= \frac{\mu\text{g metal}}{\text{grams tissue}} \end{aligned}$$

Plant uptake or organic compounds by *C. esculentus* has been limited to studies with 2,4,6-trinitrotoluene (TNT) (Folsom et al. 1988; Pennington 1988; and Palazzo and Leggett 1986) and polychlorinated biphenyl (PCB).\*

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\* Folsom, B. L., Jr. Unpublished laboratory results, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

literature review on potential contaminant migration pathways Brannon et al. (1989) suggested that plant uptake of organic compounds may be very important to cycling of organic compounds in confined dredged material disposal facilities and recommended further research be conducted.

### Data Comparisons

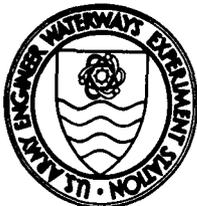
Plant heavy metal concentrations, total heavy metal plant uptake, and yield data are used to make the required DMF numerical comparisons. Plant heavy metal concentration, total heavy metal uptake, and yield are subjected to analysis of variance (ANOVA) and the Waller-Duncan K-Ratio t test to determine if test sediment mean values are different from reference sediment mean values. In cases where no plants survive in a replicate, the number of surviving replicates should be reported. Results of statistical analyses are then used to make the numerical comparisons (paragraphs B50 to B52, Peddicord 1986) in the DMF to help determine the most appropriate disposal option.

### Summary

Sediments are thoroughly mixed before testing. The plant bioassay procedure is generally conducted using four replicates of each disposal condition (i.e. flooded and upland) for each sediment or reference site considered. Flooded plant bioassay replicates are prepared and stored until sediment has been air-dried for the upland plant bioassay replicates. Replicates are placed into a controlled greenhouse environment and allowed to grow for 45 days. Aboveground plant tissue is harvested, weighed, acid digested, and analyzed for heavy metals by atomic absorption spectroscopy. Generally, heated graphite analysis is required to obtain the heavy metal concentration data. Test and reference sediment results are compared and conclusions are used in the Decisionmaking Framework.

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# *Environmental Effects of Dredging Technical Notes*



## A COMPUTERIZED PROCEDURE FOR PREDICTING PLANT UPTAKE OF HEAVY METALS FROM CONTAMINATED FRESHWATER DREDGED MATERIAL

**PURPOSE:** The Decisionmaking Framework (DMF) developed by Peddicord et al. (in preparation) provides a framework for evaluating sediments before dredging. This framework is made up of several modules, one of which is the plant bioassay for materials proposed for upland or wetland placement. Like the DMF, the plant bioassay module is based on tiered testing. Tier I is a computer simulation based on chemical extraction of test and reference sediments; Tier II is an actual laboratory/greenhouse plant bioassay. The purpose of this note is to briefly describe development and use of the computer simulation on a personal computer (PC).

**BACKGROUND:** The DMF uses the concept of tiered testing whereby all necessary information, but not more information than necessary, is used to determine regulatory compliance. Mobility of heavy metals into the environment through plant uptake could be significant in some circumstances. A quick screening test to predict potential plant uptake and mobility of heavy metals can be used during the initial sediment evaluation process. Data used in developing and verifying the US Army Engineer Waterways Experiment Station (WES) plant bioassay were collected, compiled, and subjected to predictive modeling techniques. A computer simulation of predicted plant uptake was developed. Methods for generating the required input data are also briefly described.

**ADDITIONAL INFORMATION:** Drs. Bobby L. Folsom, Jr., Environmental Laboratory, and Mark H. Houck, Purdue University, wrote this note. For additional information contact Dr. Folsom, (601) 634-3720, or the manager of the Environmental Effects of Dredging Programs, Dr. Robert M. Engler, (601) 634-3624.

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### Development of the Plant Uptake Program (PUP)

#### Data collection

Plant bioassay data collected during the past several years were compiled into one data set. The data were separated by sediment redox status (i.e.

flooded and upland) as sediment redox has been shown to strongly affect heavy metal availability to plants (Lee, Folsom, and Engler 1982; Lee, Folsom, and Bates 1983). They showed heavy metals extracted by an organic extract (diethylenetriaminepentaaceticacid) (DTPA) correlated well with plant uptake of heavy metals. The data were further subdivided by plant metal concentration and by total mass of metal taken up (i.e. total metal uptake equals plant tissue metal concentration times plant yield) since small plants may contain high heavy metal concentrations (Folsom, Lee, and Bates 1981). Data separation improved the strength of total plant uptake prediction and resulted in total uptake being independent of plant concentration.

#### Regression analysis

Regression analysis was used to determine estimators for the concentration and the total plant uptake of each of the heavy metals. Ordinary least-squares regression was used to find the best set of variables for the estimation equations. The estimators are based on linear and nonlinear functions of total and DTPA-extractable heavy metals, percent organic matter, sediment pH, and disposal condition (flooded or upland). The objective was to minimize the sum of the squared deviations of actual concentration or total uptake in the plant from predicted concentration or total uptake. The variables were not restricted. To make the predictions, each of these variables (except for total sediment content) should be determined on both original, flooded- and air-dried, upland sediment.

#### Installation and Initiation of PUP

The regression model was compiled and converted into a "user friendly" PC software program termed the Plant Uptake Program or PUP. PUP requires a 360-kilobyte low-density floppy disk drive and a 20-megabyte hard drive. The program will not require any other software except MS or PC DOS (3.0 or greater) as the PC operating system. Two 5.25-in. low-density 360-kilobyte floppy disks in a folder on the back page of this note contain PUP. PUP is installed on the hard drive and run using the following steps:

- Insert the PUP floppy disk into drive A.
- Type A:INSTALL C:  
(C: is the designation of the hard drive where the program will be installed; substitute your drive letter if different from C:).

- The program will now be copied and installed onto the hard drive. (This will not alter any existing files on your computer).
- Put the PUP floppy disks in a safe storage location.
- Type PUP and then press the RETURN key to start the program.

The opening menu (Figure 1) allows the user a choice of five options.

THESE OPTIONS ARE AVAILABLE NOW

1. Review the purpose and design of this program.
2. Review and possibly modify previously entered sample data.
3. Begin the estimation process for a new sample.
4. Compare a reference sample and a test sample.
5. Exit this program.

Please enter the number of the option you want:

Figure 1. Opening menu for the Plant Uptake Program (PUP)

The user can begin the estimation process by choosing option 3. Option 3 informs the user of required data input. If the data are available, the next PUP screen is the data entry screen. Here the user enters descriptive and chemical information about the sediment sample. PUP will calculate the estimations for whatever heavy metals are entered. Values below this detection limit (DL) are entered as one-half the value of the DL. The data are edited, if necessary, and then the program computes the estimations. At this point, several options (Figure 2) are presented to allow the user to review, analyze, print and store the results for future reference. Choosing option 2 allows the user the view the estimation results on screen. These results can be edited, printed, or stored in the data base by choosing the appropriate option. PUP initially contains several example data sets that the user can use to become familiar with the program. The user can delete them if he or she chooses. The user can then exit the program or continue to analyze other data.

Two measures of the quality of the estimations are provided. The first measure of the quality of estimation is the multiple correlation coefficient

THESE OPTIONS ARE AVAILABLE NOW

On Screen Options

1. Review information on interpreting estimates.
2. Review the estimation results.
3. Review field data used to develop estimation equations.

Printing Options

4. Print information on interpreting estimates.
5. Print the estimation results.
6. Print field data used to develop estimation equations.

Saving Data and Exiting Options

7. Store the estimation results in the database.
8. Return to the main menu to analyze other data or to exit the program.

Please enter the number of the option you want:

Figure 2. Available options for estimation results

( $R^2$ ). The multiple correlation coefficient is the proportion of the variation of the actual concentration or total plant uptake explained by the estimation equation: a value of 1.0 indicates an excellent fit between the estimation equation and the data used (i.e. data the user has input) in the estimation process; a value of 0.0 indicates the worst possible fit.

A general method in research is to conduct experimentation that allows separation of variables one variable at a time. Correlation coefficients are generally greater than 0.9 when only one pollutant at a time is tested at high concentrations. Most of the existing information used for setting environmental standards is based on this type of research data. Seldom, however, is only one element elevated. Sediments from Black Rock Harbor (Folsom et al. 1988), Indiana Harbor (Environmental Laboratory 1987a, b), and the Detroit River (Folsom, Lee, and Bates 1981) are several examples of multi-element contamination. The interactive effects encountered when one or more trace elements is in excess include synergisms, antagonisms, competition, protection, sequential additivity, and independence (Wallace 1989). In the absence of this knowledge and using an

empirical approach (as used in the present study) it is not surprising that frequently encountered regression coefficients are much less than 0.9. Such low regression coefficients may, indeed, be environmentally significant.

The second measure of the quality of estimation is the Standard Error of Prediction (SEP). SEP is the standard deviation of the distribution of the errors between the estimated and actual concentration or total uptake used in the estimation process. Single value estimates are provided for each concentration and total plant uptake. Because ordinary least-squares regression was used to develop the estimation equations, they may produce negative values of estimated heavy metal concentration or total plant uptake. The negative values usually imply that either the estimated value is close to zero or the current input values are sufficiently different from those used in the calibration of the estimation equations so as to make the estimates unreliable. A range of values is provided for each concentration and total uptake. This range is the 90 percent confidence interval which means there is a 90 percent chance that the actual value lies within the range. The development data summary (option 6) is presented to provide the user a frequency distribution of the sediment input data. For example, the user may have a test sediment zinc concentration of 450  $\mu\text{g/g}$ . The user can choose option 6 and see that a zinc concentration of 450  $\mu\text{g/g}$  is in the second quartile of the zinc concentration range. Allowing the user to view these data relationships may help determine whether there is "reason to believe" contamination exists (i.e. the first step in Tier I testing).

Currently, plant uptake of contaminants is appraised in the DMF from sediment extraction (DTPA) and actual plant uptake data. These data are obtained by subjecting the reference and test sediments to the WES plant bioassay. The user has to manually input and laboriously analyze the DTPA extraction and plant bioassay data. PUP frees the user from these tasks and rapidly performs the DTPA extraction comparison calculations. The DTPA comparisons are based on the decision criteria given in paragraph B47 of the DMF summarized below:

1. DTPA-extractable concentrations of all metals from the air-dried sediment are less than or equal to the reference (Case 1a) and less than or equal to the saturated sediment (Case 1b). This leads to a DECISION OF NO RESTRICTIONS to protect against contaminant impacts on plants colonizing the dredged material.
2. DTPA-extractable concentration of any metal from the air-dried sediment is less than or equal to the reference (Case 2a) and greater than the saturated sediment (Case 2b) or

3. DTPA-extractable concentration of any metal from the air-dried sediment is greater than the reference (Case 3a) and less than or equal to the saturated sediment (Case 3b). Condition b and c lead to a LOCAL AUTHORITY DECISION as discussed in paragraph B49.
4. DTPA-extractable concentration of any metal from the air-dried sediment is greater than the reference (Case 3a) and greater than the saturated sediment (Case 2b). This leads to a DECISION FOR FURTHER EVALUATION by conducting a plant bioassessment as discussed in paragraph B50.

To indicate the DTPA comparisons the user chooses option 4 (Figure 1) and is prompted to select the reference and test sample data that have already been entered into PUP. PUP then computes the comparisons using the criteria described in paragraph B47 of the DMF. Those heavy metals that exceed the criteria are cause for concern and require further evaluation using the WES plant bioassay (Folsom and Price in preparation). The number of heavy metals exceeding the criteria can then be used for further decisions on plant uptake (criteria given in paragraph B49 of the DMF). A schematic representation of the DTPA comparisons is shown in Figure 3. Figure 3 is an example output of DTPA comparisons made on hypothetical data. If actual data had been used, any metal that exceeded the criteria would appear as red letters labeled EXCD; those that did not exceed the criteria would be in white letters labeled DNEX. Those metals not entered appear as dashes. The reference and test sediments the user chooses to compare are also shown in Figure 3. Culmination of these comparisons completes Tier I testing.

#### Methods for Generating Required PUP Data

##### Sediment collection

Sediments to be tested are collected from the area to be dredged using a sampler that can sample the entire vertical profile of the material. A 3.785-ℓ (1-gal glass jar) composite sample of the vertical profile should be sufficient to conduct the testing. The sediment sample contained in the glass jar should be thoroughly mixed before conducting any testing. A 1-ℓ subsample of the material is removed for chemical analyses after mixing. This subsample represents original-flooded material. Another 1-ℓ subsample of the original sediment is placed into an aluminum drying pan and allowed to air dry. This subsample represents upland dredged material if it were placed in an upland disposal site. Total and DTPA-extractable metals, percent organic matter, and pH are determined on both flooded- and air-dried samples of each composited sediment sample. The procedures given below allow the user to obtain required data.

DTPA COMPARISON CASES

CASE	AS	CD	CR	CU	FE	HG	MN	NI	PB	ZN	No. Exceeded
1a	DNEX	EXCD	DNEX	DNEX	DNEX	EXCD	DNEX	DNEX	DNEX	DNEX	2
1b	DNEX	0									
2a	DNEX	0									
2b	DNEX	DNEX	DNEX	DNEX	DNEX	EXCD	DNEX	DNEX	DNEX	DNEX	1
3a	DNEX	0									
3b	EXCD	EXCD	DNEX	EXCD	DNEX	DNEX	DNEX	DNEX	EXCD	DNEX	4

SAMPLES USED IN COMPARISONS:

Upland or Air Dried Test Sample:  
 Flooded or Wet Test Sample:  
 Upland or Air Dried Reference Sample:

CASE	DNEX CRITERION
1a	DTPA conc of upland test sample <= DTPA conc of upland reference sample
1b	DTPA conc of upland test sample <= DTPA conc of flooded test sample
2a	DTPA conc of upland test sample <= DTPA conc of upland reference sample
2b	DTPA conc of upland test sample > DTPA conc of flooded test sample
3a	DTPA conc of upland test sample > DTPA conc of upland reference sample
3b	DTPA conc of upland test sample <= DTPA conc of flooded test sample

Would you like a printed copy of these results? (Y/N)

Figure 3. Results of reference and test sample comparisons

### Flooded DTPA and flooded pH

A 50.0-g (weighed to the nearest 0.0001 g) oven-dry weight basis (ODW) subsample of the original, flooded sediment is weighed into a 500-ml polycarbonate centrifuge bottle and centrifuged at 4° C and 13,701 × g's\* (9,500 rpm) for 30 min. The supernatant is decanted; pH is determined on the supernatant and is the flooded sediment pH. Two hundred fifty millilitres of 0.005 M DTPA + 0.01 M calcium chloride + 0.1 M triethanolamine solution (Lee et al. 1978) buffered at pH 7.3 is added to the sediment in the centrifuge bottle. The bottle is sealed and placed on a shaker for 24 hr and then centrifuged as before. The supernatant is poured into a polyethylene bottle and stored at 4° C until the time of chemical analysis. The supernatant is analyzed for heavy metals using atomic absorption (AA) spectroscopy. Heated graphite furnace analysis (HGA) is generally required to obtain the heavy metal concentration data.

### Upland DTPA

The procedure (as in the flooded DTPA extraction above) involves adding 250.0 ml of the DTPA extraction solution to 50.0 g (ODW) of the air-dried sediment in a 500-ml polycarbonate centrifuge bottle which is then shaken for 24 hr. Two hundred fifty millilitres of 0.005 M DTPA + 0.01 M calcium chloride + 0.1 M triethanolamine solution (Lee et al. 1978) buffered at pH 7.3 are added to the sediment in the centrifuge bottle. The bottle is sealed and shaken for 24 hr, then centrifuged as before. The supernatant is poured into a polyethylene bottle and the liquid is stored at 4° C until the time of chemical analysis. Heavy metal concentrations are determined using AA or HGA. Blanks are also subjected to the DTPA extraction procedure. Metal concentrations of the blank solution are subtracted from the DTPA extracting solution metal concentration before performing the calculation shown below.

DTPA-extractable heavy metals (both flooded and upland) are calculated using the following formula:

$$\text{DTPA metal concentration} = \frac{(\text{DTPA extracting solution metal concentration}) \times (\text{extracting solution volume})}{\text{weight of ODW soil actually extracted}}$$

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\* The 500-ml centrifuge head loaded with 500-ml centrifuge bottles containing sediment spun at 9,500 rpm for 300 min is equal to 13,700 times the acceleration due to gravity (g).

$$= \frac{\mu\text{g}/\text{m}\ell \times 250 \text{ m}\ell}{\text{g soil}}$$
$$= \frac{\mu\text{g metal}}{\text{g soil}}$$

Total heavy metal content

One gram (ODW) (weighed to the nearest 0.0001 g) of upland, air-dried sediment is placed into a 100-m $\ell$  micro-Kjeldahl flask. Ten millilitres of concentrated nitric acid are added to the flask and its contents and allowed to sit overnight. A tertiary acid mixture of a 2:1:5 ratio of perchloric acid (HClO<sub>4</sub>), sulfuric acid (H<sub>2</sub>SO<sub>4</sub>), nitric acid (HNO<sub>3</sub>) is prepared by adding the following volumes of acid to a 1- $\ell$  bottle, swirling to mix, and then putting into a repipet:

200 m $\ell$  conc HClO<sub>4</sub>  
100 m $\ell$  conc H<sub>2</sub>SO<sub>4</sub>  
500 m $\ell$  conc HNO<sub>3</sub>

CAUTION: THE FOLLOWING PERCHLORIC ACID (HClO<sub>4</sub>) DIGESTION SHOULD BE CONDUCTED IN A STAINLESS STEEL ACID-DIGESTION HOOD. Twenty millilitres of the tertiary acid mix are added to the contents of the flask, the flask swirled to wet its contents, and then the flask is placed on a digestion rack. Heat until the mixture starts to boil, and then increase the heat slightly. The nitric acid will distill as a yellowish gas. After the nitric acid is gone, the perchloric acid will distill off as a white gas. When all of the perchloric acid is gone, the digestion is complete and only the sulfuric acid will be left (approximately 2 m $\ell$ ). The flask is removed from the burner and allowed to cool to room temperature. Twenty millilitres of distilled water are added to the solution, which is then quantitatively filtered through a Whatman No. 42 filter paper contained in a long-neck funnel in a 50-m $\ell$  volumetric flask. Distilled water should be used to wash (while still on the filter paper in the funnel) any white, gelatinous precipitate that may have formed. This wash water should be allowed to filter into the 50-m $\ell$  volumetric flask as well. Dilute to volume with distilled water and then analyze for heavy metals. Blanks and National Bureau of Standards Standard Reference Material (NBS SRM) 1645 (river sediment) are also subjected to the perchloric acid digestion procedure. Metal concentrations of the blank solutions are subtracted from the solution metal concentration before performing

the calculation shown below. Metals concentrations are then corrected to 100 percent of the NBS SRM. Total heavy metal sediment concentration is calculated using the formula:

$$\begin{aligned} \text{metal concentration} &= \frac{\text{solution metal concentration} \times \text{dilution volume}}{\text{g ODW soil actually digested}} \\ &= \frac{\mu\text{g/ml} \times 50 \text{ ml}}{\text{g soil actually digested}} \\ &= \frac{\mu\text{g metal}}{\text{g soil}} \end{aligned}$$

#### Organic matter

Organic matter (OM) is determined by weight loss on ignition at 550° C on both flooded and upland sediment. Procedure No. 209E (American Public Health Association 1976) is used for this test. A 5-g subsample (ODW) of the sediment is weighed to the nearest 0.0001 g and dried at 105° ± 2° C until constant weight. Five grams (weighed to the nearest 0.0001 g) of the oven-dried sediment are then combusted at 550° ± 5° C for 24 hr in a muffle furnace. The sample is allowed to cool to room temperature and weighed to the nearest 0.0001 g. Weight loss on ignition is calculated and reported as percent organic matter (%OM). Use the formula below for the calculation of percent organic matter:

$$\%OM = \frac{\text{weight oven-dried sediment} - \text{weight combusted sediment}}{\text{weight oven-dried sediment}} \times 100$$

#### Upland pH

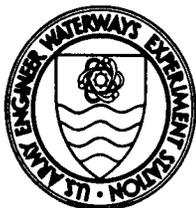
Ten grams (ODW) (weighed to the nearest 0.0001 g) of upland sediment are weighed into a tall 50-ml Pyrex glass beaker. Twenty millilitres of distilled water are added, and the mixture stirred with a polyethylene rod until all dry particles are wetted. The suspension is stirred for 1 min every 15 min for 45 min with a magnetic stirrer. After 45 min with the stirrer off, the pH electrode is placed into the solution above the surface of the sediment and the pH determined.

### Summary

Total and DTPA-extractable heavy metals, organic mater, and pH are determined on test and reference sediments prior to dredging and nonaquatic disposal. These data are entered in the Plant Uptake Program (PUP). PUP is Tier I of the Plant Bioassay module of the DMF. Results of Tier I testing (i.e. computer simulation) can be used as a quick screening tool to identify areas of concern. If the results of Tier I testing indicate a reason for concern, then the sediments may be evaluated with the Plant Bioassay module (Tier II) to verify the concern. Procedures for Tier II testing can be found in a companion Technical Note (Folsom and Price in preparation).

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# *Environmental Effects of Dredging Technical Notes*



## **Environmental Interpretation and Evaluation of Hydrocarbon Contaminants in Dredged Material**

### **Purpose**

This note summarizes recommendations of the second petroleum hydrocarbons workshop convened to assist Corps Districts in evaluating hydrocarbon contamination in dredged material.

### **Background**

On 15-17 March 1988, a workshop on environmental interpretation of petroleum hydrocarbons in dredged material was conducted at the US Army Engineer Waterways Experiment Station (WES). Participants represented government agencies, private industry, and academia, and were selected for their expertise in environmental chemistry and biological effects of petroleum hydrocarbons. The workshop was held at the request of US Army Engineer Districts, Chicago and New York, and followed an earlier (1986) workshop on regulatory evaluation of petroleum hydrocarbons in dredged material.

The purpose of the second workshop was to develop guidance on scientific interpretation of potential impacts of polycyclic aromatic hydrocarbons (PAHs) in dredged material. Participants in the 1986 workshop recommended a list of 15 priority pollutant PAHs for regulatory analysis of dredged material. They also recommended a two-tiered testing scheme consisting of first-tier acute toxicity tests and sediment analysis of the 15 PAHs in dredged material, and second-tier 10-day bioaccumulation tests. Roundtable discussions during the second workshop centered on a reexamination of the recommendations of the 1986 workshop, sediment analyses and biological testing for PAHs, and the biological effects of PAHs. Participants recommended no change in the list of 15 PAHs originally

selected for regulatory analysis of dredged material. The original two-tiered testing approach was expanded to a four-tiered approach that conforms to the Federal Standard for dredged material evaluation.

## **Additional Information or Questions**

Refer to the workshop proceedings (Clarke and Jarvis in preparation) or contact the authors, Ms. Susan Jarvis, (601) 634-2804, and Ms. Joan Clarke, (601) 634-2954, or the EEDP Program Manager, Dr. Robert M. Engler, (601) 634-3624.

## **Summary of Recommendations of the 1986 PAH Workshop**

The primary objective of the 1986 workshop was to identify from the myriad of petroleum hydrocarbons, specific compounds that would be most appropriate to analyze in the environmental assessment of dredged material placement. Participants agreed that PAHs are the most important class of hydrocarbons in dredged material due to their toxicity and persistence. Fifteen of the sixteen priority pollutant PAHs were recommended for the evaluation of dredged material: acenaphthene, acenaphthylene, anthracene, benz[a]anthracene, benzo[b]-fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenz[a,h]-anthracene, fluoranthene, fluorene, indeno-[1,2,3-cd]pyrene, phenanthrene, and pyrene.\* Naphthalene, the sixteenth priority pollutant PAH, was not included in the list because of problems in obtaining accurate analytical results. It is also very volatile and too water soluble to persist in sediments. The behavior, fate, and effects of the 15 selected PAHs were thought to be representative of hydrocarbons known to have biological effects.

A two-tiered testing approach was recommended based on the assumption of a reason to believe that a sediment is contaminated with PAHs. The first tier included acute toxicity testing and chemical analysis for the 15 selected PAHs. If Tier I results demonstrated acute toxicity, it would be unnecessary to continue to Tier II because the sediment would be considered unacceptable for unrestricted placement. If Tier I results indicated the presence of PAH contamination of the sediment, but no acute toxicity occurred, there could still be potential for unacceptable adverse biological effects. Tier II, bioaccumulation testing, would then be conducted to assess whether the 15 PAHs accumulate in the tissues of test organisms. If these compounds are not taken up by organisms exposed to the sediment (that is, are not bioavailable), then PAH-related biological impact would be unlikely to occur. Bioaccumulation testing would use organisms such as bivalves that have limited ability to metabolize PAHs and are thus capable of accumulating parent PAH compounds.

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\* Priority pollutants refer to a list of 129 toxic substances compiled by the US Environmental Protection Agency (USEPA). The list includes 16 PAHs.

Future research was recommended to develop analytical procedures and biological testing protocols for evaluating PAH metabolites, alkylated PAH, heterocyclic, nitroaromatics, and aromatic amines. Research and development needs included assays for carcinogenicity, genotoxicity, and reproductive effects. The recommendations and observations of the 1986 workshop are detailed in Clarke and Gibson (1987a, b).

## **Recommendations of the 1988 PAH Workshop**

### **Reevaluation of 1986 Workshop Recommendations**

Participants in the second workshop reexamined the list of 15 PAHs selected in the 1986 workshop for evaluation of dredged material, and generally agreed that the list should remain unchanged. Naphthalene was still excluded from the list because of potential problems in obtaining accurate chemical analysis of this compound from environmental samples. Other suggested additions to the list such as the alkyl-, nitrogen-, and sulfur-substituted PAHs, and benzo[e]pyrene were not accepted because of analytical problems, similarity in effect with PAHs already on the list, or because not enough is yet known about their behavior and biological effects in sediment.

### **Biological Effects of PAHs**

PAHs have been associated with a number of acute and chronic biological effects, including mortality, impairment of growth and reproductive processes, and carcinogenicity/mutagenicity. Mortality may occasionally result from high concentrations of the lower molecular weight, acutely toxic PAH. Acute toxicity from sediment-associated PAH is most likely to occur in aquatic organisms that feed at the sediment surface such as benthic fish, some crustaceans, or deposit-feeding polychaetes since these organisms receive maximum exposure to PAH in the sediment.

Chronic or sublethal effects may result from parent PAH or from biotransformation of the parent PAH compounds to more toxic metabolites. Fishes and some invertebrates generally have well-developed biotransformation capability for PAHs. Among the sublethal effects, adverse impacts on reproduction and growth will likely have the most ecological importance to a population of organisms over time.

PAHs may be linked to carcinogenicity or mutagenicity in susceptible organisms. Of the 15 recommended PAHs, benz[a]anthracene, chrysene, benzo[a]pyrene, dibenz[a,h]anthracene, benzo[b]fluoranthene, and benzo[k]fluoranthene have shown carcinogenicity in mammalian systems. PAHs that cause cancer in mammals may have a potential for causing cancer in other organisms, because the same mechanisms are involved. Nevertheless, cancer in aquatic populations may

not necessarily be caused only by PAHs, but also by other environmental stresses, especially in industrialized areas where other contaminants are present.

## Effects-Based Screening Guidelines

The Chicago and New York District sponsors requested effects-based numeric guidelines or thresholds for PAHs in sediments or in tissues. Each sponsor suggested that these guidelines serve as a screening tool for identifying sediments having sufficiently low or high PAH concentrations to eliminate the need for further testing. However, the workshop participants strongly emphasized that such threshold concentrations could not be recommended because there are no levels of concern for PAHs and not enough information to quantitatively link adverse biological effects with concentrations of PAHs either in sediment or in tissues. Scientifically sound evaluation of PAH-contaminated dredged material must be based on biological testing rather than numeric criteria because of a current lack of understanding of factors influencing bioavailability and toxicity of complex contaminant mixtures in sediments.

## Recommendations for a Tiered Testing Approach

An adequate environmental assessment program for dredged material placement should incorporate a suite of tests to assess the potential for various adverse effects of PAHs on species representative of those occurring at the placement site. By arranging the tests in tiers, the evaluator will be able to determine the number and progression of tests needed for a specific project evaluation.

The two-tiered testing approach recommended in the 1986 PAH workshop was expanded to four tiers (Figure 1). Tier I is the determination of a reason to believe that the dredged material is contaminated with PAHs and that the potential exists for unacceptable adverse biological effects as a result of dredging and placement. This assessment could use historic data, knowledge of point sources or spills, or any other relevant information. If there is a reason to believe, or insufficient information for any assessment, then the evaluation would proceed with chemical and biological testing.

Tier II involves chemical analysis of the sediment for the 15 selected PAHs to determine whether the dredged material is more contaminated than the sediment at the placement site environs. If Tier II indicates that there is a potential for unacceptable adverse effects to occur or if Tier II produces insufficient information to determine that potential, then Tier III would be conducted.

Tier III is the first biological testing tier and includes acute toxicity testing using sensitive organisms that are representative of organisms at the placement site environs. Appropriate species could include *Mysidopsis*, *Palaemonetes*, *Nereis*, *Rhepoxynius*, or *Ampelisca* in saltwater, and *Daphnia*, *Ceriodaphnia*, *Selenastrum*,

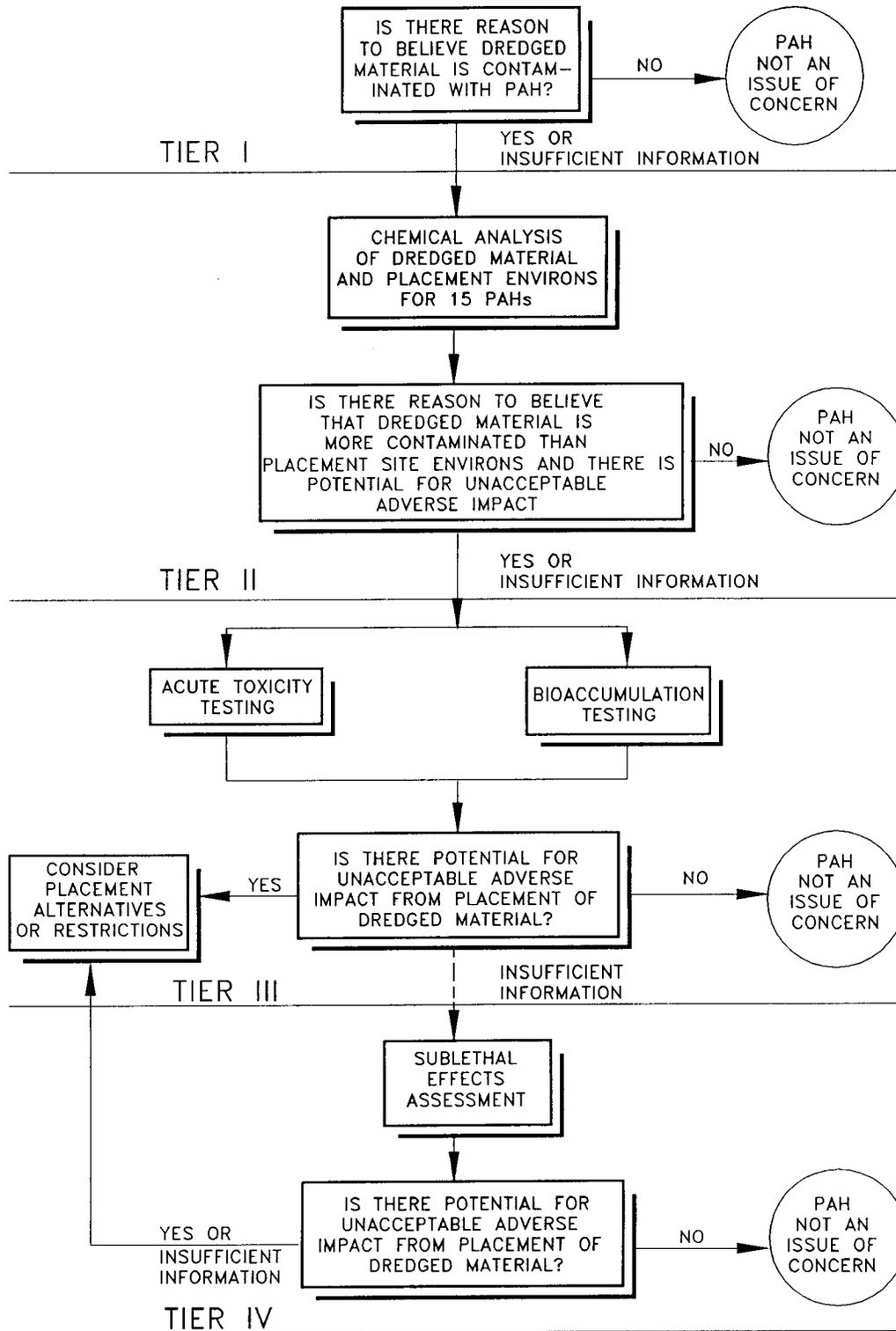


Figure 1. Suggested tiered testing approach, for evaluation of PAHs in dredged material

fathead minnows, *Pontoporeia*, *Chironomus*, or *Hexagenia* in freshwater. Tier III also includes bioaccumulation testing using deposit-feeding organisms that have little metabolic capability for PAHs. Bivalves such as *Macoma* or *Yoldia* are recommended for saltwater, while the Great Lakes amphipod *Pontoporeia* or another amphipod or *Hexagenia* are possibilities for freshwater bioaccumulation tests. As in Tier II, the significance of Tier III results is determined by comparing test results from dredged material to results from the placement site environs.

Results from Tier II and Tier III tests may be difficult to interpret for individual PAHs because of limitations in knowledge concerning the biological effects and relative importance of these individual compounds. A data base needs to be developed relating environmental levels of the 15 PAHs with biological effects. Presently, total PAH as the sum of the 15 PAHs could be used to compare and interpret results. The values generated for the 15 individual PAHs using this approach could be incorporated into the data base, but would not necessarily be used at this time in evaluation.

Tier IV would evaluate the potential for adverse impacts on sublethal effects such as reproduction and growth, perhaps using a partial or whole life-cycle test. The ability of organisms to reproduce successfully is an indication of fitness in the population. Environmental agencies and the scientific community are placing more emphasis on reproductive effects; therefore, reproductive bioassessment will likely become increasingly important in the future. Other possibilities for assessing sublethal effects include biochemical tests, such as enzyme induction, and assays for carcinogenicity or mutagenicity.

A definite need for research in the area of sublethal effects is clear. The workshop participants could not agree on any single test or suite of tests for sublethal effects. Nonetheless, they agreed that any tests adopted must be sensitive to the contaminants in the dredged material to be regulated, and site-specific to the extent that they assess the particular impacts known or suspected in the dredged material. At this time, no PAH sublethal effects tests are sufficiently standardized or verified to meet those criteria. Thus, the suggested four-tiered testing approach is not ready for full implementation. The first three tiers can be implemented now and correspond to the Corps' comprehensive testing strategy for dredged material placement as part of the Federal Standard (Engler and others 1988). Sublethal effects tests (Tier IV) require more research, development, and standardization before being adopted for evaluation of sediment.

The tiered testing approach arising from the PAH workshop should not be considered the final answer to evaluation of PAH-contaminated dredged material. However, it does supply a direction in which Corps Districts may proceed. More research and information are needed to develop a detailed, comprehensive testing approach for PAHs in sediment, particularly when chronic or sublethal effects are of concern. Progress in this direction is being proposed and initiated by the Corps under the Long-Term Effects of Dredging Operations (LEDO) program and the Water Quality Research Program. The ultimate goal is the development of technically sound and feasible guidance on PAHs as well as other contaminants of concern.

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# *Environmental Effects of Dredging Technical Notes*



## **Residue Effects Data Base on the Relationship Between Dioxin and Biological Effects in Aquatic Animals**

### **Purpose**

The purpose of this note is to present residue-effects data involving dioxins that are presented in the scientific literature. The information will be useful in developing regulatory guidance applicable to dioxin contamination.

### **Background**

Work Unit 31771, "Environmental Interpretation of Consequences from Bioaccumulation," of the Long-Term Effects of Dredging Operations (LEDO) Program is designed to provide interpretive guidance for evaluating data generated by Corps field offices for their permit applicants. This guidance results from identifying residue-effects relationships through laboratory experiments and literature reviews. Previous literature reviews conducted under this work unit have concentrated on heavy metals and chlorinated contaminants (Dillon 1984, Dillon and Gibson 1985). The present effort examines residue-effects relationships with dioxins as reported in the published literature.

Polychlorinated dibenzo-*para*-dioxins (PCDDs) are present as trace impurities in some manufactured chemicals and industrial wastes. Generally, the source for high levels of PCDDs is attributable to industrial discharge, hazardous waste dumps, or the application of PCDD-contaminated herbicide (Miller, Norris, and Hawkes 1973, Helder 1980, 1981). PCDDs are formed as a result of photochemical and thermal reactions in fly ash and other incineration products, as well as in high-temperature chlorination reactions. There are 75 PCDD isomers. The most toxic and most extensively studied is 2,3,7,8-tetrachlorodibenzo-*para*-dioxin (2,3,7,8-TCDD). In general 2,3,7,8-TCDD is the congener referred to by the term dioxin and is the congener discussed in this note.

## **Additional Information**

For additional information, contact one of the authors, Ms. Alfreda Gibson, (601) 634-4027, or Mr. Francis J. Reilly, Jr., (601) 634-4148, or the manager of the Environmental Effects of Dredging Programs, Dr. Robert M. Engler, (601) 634-3624.

## **Approach**

To conduct this review, 15 technical journals and 8 data base literature search services (for example, Tox-Line, Pollution Abstracts, and National Technical Information Service) were used. Additionally, the Dioxin Information Data Base under development by the US Army Engineer Waterways Experiment Station as part of the LEDO Program Work Unit 31772 "Toxic Substances Bioaccumulation in Aquatic Organisms" was searched. All pertinent publications were individually reviewed for the following information: test species, contaminant, exposure conditions, tissue concentration, and corresponding biological effect.

## **Summary**

An intensive review of the technical literature for both residue data and biological effects information showed that studies relating the two were, for the most part, nonexistent. Most of the papers reviewed contained biological effects information related to nominal (not measured) dosage, but with no associated body burden data. Only studies with both tissue residue levels and biological effects information relating to 2,3,7,8-TCDD are discussed in this note.

The majority of scientific studies focusing on dioxins in the aquatic ecosystem dealt with bioaccumulation. In addition, studies conducted to observe the biological effects of dioxins generally concentrated on species other than aquatic organisms (that is, birds and mammals, especially laboratory animals).

The lack of information on dioxin in the aquatic ecosystem is cause for general concern (Miller, Norris, and Hawkes 1973). Information is scarce or lacking on the biological properties of PCDD congeners (Walker and others 1991). No data are available on lethal and sublethal effects of any PCDD congener to aquatic organisms, except 2,3,7,8-TCDD (Eisler 1986).

Only two studies reported on residue-effects data. One was the exposure of carp to fly ash contaminated with 2,3,7,8-TCDD (Kuehl and others 1985). This is a relatively old reference with regard to dioxin research, and the chemical methodology employed by the authors has changed appreciably in the interim. The lowest tissue residue level measured was 1.2 parts per trillion ( $10^{-12}$  or pptr) and was associated with some gill tissue damage. A tissue residue of nearly 1,000 pptr, however, only resulted in fin discoloration, fin necrosis, and erosion. These data are presented in Table 1.

**Table 1**  
**Dioxin Residues Associated with Known Biological Effects**

Organism	Tissue Concentration, pptr	Exposure	Effect
<b>2, 3, 7, 8-TCDD</b>			
Carp* (1)**	1.8, 1.2	1 g fly ash (2,000 pptr) 15 day static	gill tissue damage
Carp* (2)	2.0, 2.2	5 g fly ash (2,000 pptr) 30 day static	gill tissue damage
Carp* (3)	200	1 g fly ash (2,000 pptr) 25 day static	fin discoloration, necrosis, and erosion
Carp* (4)	1,000	5 g fly ash (2,000 pptr) 38 day static	fin discoloration, necrosis, and erosion
Carp* (5)	6.1, 11	250 mg fly ash (2,000 pptr) 95 day flow through	gill tissue damage
Carp* (6)	20, 28, 32	1, 5, 10 g fly ash (160 pptr) 30 day static	gill tissue damage
Carp* (7)	20, 33	5 g fly ash (160 pptr) 10, 30 day flow-through	bioavailability not affected by semi-static conditions
Trout†	0-121	0-40 pptr eggs (static) 48 hr	no effect on hatching
Trout†	226-302	62-100 pptr eggs (static) 48 hr	significant reduction in hatching

\* Kuehl (1985).

\*\* Individual groups of carp were measured separately; the number in parentheses is the number of each separate group of carp studied.

† Walker (1991).

In the second study lake trout eggs containing 2,3,7,8-TCDD in concentrations from 0 to 100 pptr were observed through the fry stage (Walker and others 1991). The resultant tissue concentrations in the fry ranged from 0 to 302 pptr. The only adverse effect on hatching was in the two highest treatments (Table 1).

The results summarized in Table 1 were drawn from a very limited number of studies that made only a few measurements associated with biological effects. It should be noted that tissue residues of organisms exposed to 2,3,7,8-TCDD environmental contamination and caught in the wild have been found to have much higher tissue levels of 2,3,7,8-TCDD than the levels reported in Table 1 without any apparent adverse effects. Crabs taken from Newark Bay have been shown to have 3,670.4 and 6,238.2 pptr associated with their hepatopancreas (Rappe and others 1991). In the same study lobsters were found to have 1,611 pptr and striped bass flesh was found to have accumulated 733.9 pptr in the New York Bight (Rappe and others 1991). In another example the liver of a burbot from Sweden was found to have 469 pptr of 2,3,7,8-TCDD (de Wit and others 1990).

On the basis of this literature review, there are not enough published data on 2,3,7,8-TCDD tissue residues and associated biological effects to make definitive recommendations regarding the handling of sediments contaminated with 2,3,7,8-TCDD. There is a clear need for research in the area of 2,3,7,8-TCDD toxicity and additional information concerning residue levels and associated effects before any definitive guidance regarding 2,3,7,8-TCDD can be given.

This review also points out the clear lack of any information regarding tissue residue levels and concomitant biological effects for any of the other dioxin congeners. This information must be generated in a timely manner, as certain states and the US Environmental Protection Agency propose to regulate dioxins by measuring the concentrations of other dioxin congeners in addition to the 2,3,7,8-TCDD congener. The proposed methodology involves measuring all 2,3,7,8-substituted congeners, assessing their toxicity relative to 2,3,7,8-TCDD, and summing the results to achieve a Toxic Equivalency Quotient (TEQ). The TEQ methodology and guidance regarding its use are the subject of another publication (McFarland, Reilly, and Ferguson, in preparation). Without specific information regarding the toxic effects of 2, 3, 7, 8-TCDD and the relative toxicities of other dioxin congeners, providing meaningful guidance on dioxin regulation in the aquatic environment or in dredging and disposal activities will be impossible.

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# *Environmental Effects of Dredging Technical Notes*



## **New Technique for Sediment/Organism Equilibrium Partitioning Studies**

### **Purpose**

This technical note reports on the results of the initial experiment testing a new procedure that employs a kinetic model and a simple short-term fish/suspended sediment exposure system to assess equilibrium partitioning (EqP) of neutral organic chemicals.

### **Background**

Sediment Quality Criteria (SQC) based on EqP are under development by the U.S. Environmental Protection Agency (EPA). A stated purpose of SQC is the regulation of dredged material disposal. Long-term Effects of Dredging Operations (LEDO) Work Unit 32571, Relationships Between Sediment Geochemistry and Biological Effects, is researching methods to characterize the interactions between geochemical and physicochemical processes and contaminant uptake in, and effects on, aquatic organisms. This research will enable the U.S. Army Corps of Engineers (USACE) and other interested parties to best evaluate the adequacy of the EqP-based SQC as they are proposed and promulgated.

Theoretical bioaccumulation potential (TBP) calculations are based on the same thermodynamic principles as the EqP-based SQC. TBP is recommended as a screening procedure in the revised "Green Book," the national testing manual for ocean disposal of dredged material (EPA and USACE 1991). TBP is used in Tier II to indicate whether the presence of neutral organic chemicals in sediment is of little or no concern, or whether more definitive characterization by biological testing is necessary at Tier III or Tier IV levels. Revisions of the Green Book are expected to be made periodically. In the revisions, accuracy of TBP calculations may be increased by the use of empirically determined preference factors (*pf*) for specific chemicals. In sediment bioaccumulation tests, *pf* is

a constant that expresses the magnitude of the concentration difference at equilibrium, that is, the "preference" of the chemical for organism lipid versus sediment organic carbon. An alternative expression used by some workers is "accumulation factor" or AF. However, the meaning and usage are the same.

Research that further defines the geochemical, physicochemical, and physiological influences on EqP serves to improve the utility of TBP as well as to facilitate evaluation of EqP-based SQC intended to regulate dredged material disposal.

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## **Introduction**

Equilibrium partitioning (EqP) studies on bioaccumulation of organic chemicals in aquatic organisms typically require long-term exposures. In the case of highly hydrophobic neutral organic chemicals, such as polychlorinated biphenyls (PCBs), dioxins, and dibenzofurans, exposures of up to 6 months have been used (Pruell and others 1990). During long-term laboratory exposures, changes can occur in the condition of both organism and sediment. Sediments can be depleted of the bioavailable fraction of chemical, thereby reducing exposure. Nutrient quality and amount may decline and sublethal toxicity may occur, affecting the health of organisms and causing loss of lipids. Metabolic degradation of bioaccumulating chemicals may occur in long-term exposures, reducing bioaccumulation. Induction of metabolizing enzymes caused by the chemical under investigation or by other chemicals in the sediment may exacerbate this effect. Growth during the exposure period can dilute tissue concentrations causing reduced apparent bioaccumulation. Spawning and other seasonal changes also affect bioaccumulation (Lee and others 1989).

Kinetic modeling using short exposures provides an alternative to long-term exposures to achieve EqP. Short exposures greatly reduce or eliminate the difficulties inherent in long-term exposures. A drawback to using kinetic models and short exposures is the necessity of frequent sampling to define the uptake curve and enable a projection of steady state. However, when short exposures are used in a research mode, it is often possible to greatly reduce the cost of analyzing a large number of samples by using radiolabeled chemicals.

The purpose of this study was to test a new approach to EqP measurements that eliminates the necessity of obtaining steady-state conditions in the exposure. The procedure determines the proportional difference at projected steady

state between neutral chemical concentration in sediment and organism using a three-compartment closed kinetic model. The exposure matrix involves suspended sediment and fish, rather than the usual deposited sediment and sediment-processing infaunal organisms. The design permits manipulation separately of each of the principal variables (the chemical, organic carbon, and organism lipid) affecting the magnitude of the difference in concentration at equilibrium between chemical in sediment and in exposed organisms. This is accomplished while eliminating the sources of error inherent in conducting long-term exposures. This new technique is intended to be used as a research tool for understanding the relative contributions to EqP of geochemical and physicochemical properties, and physiological differences among organisms; and for the empirical determination of preference factors, *pf*.

## Materials and Methods

### Materials

PCB-52 ( $[^{14}\text{C}]$  2,2',5,5'-tetrachlorobiphenyl-UL) was obtained from Sigma Chemical Company. A 1:50 dilution with methanol of an 11.9  $\mu\text{Ci}/\text{mL}$  stock solution (409  $\mu\text{g}/\text{mL}$  in toluene) provided a working solution of 0.238  $\mu\text{Ci}/\text{mL}$  or 8.18  $\mu\text{g}/\text{mL}$  of PCB-52.

Japanese medakas, *Oryzias latipes*, were purchased from Carolina Biological Supply and were acclimated to laboratory conditions for at least 14 days at ambient temperature ( $\approx 23^\circ\text{C}$ ), in 120-L glass aquaria. Fish were maintained in filtered, aerated, dechlorinated tap water and were fed Aquarian<sup>®</sup> tablet food twice daily. A 12-hr dark/12-hr fluorescent light photoperiod was controlled by an automatic timer. Fish of both sexes were randomly used in the experiments. Air-dried and milled sediment from Barataria Bay, Louisiana, was stored at  $4^\circ\text{C}$  until use.

### Fish and Sediment Exposure to PCBs

A 2-L Florence flask was used as the exposure vessel (Figure 1). Two liters of dechlorinated water were placed in the flask containing 400 mg of sediment. Two medakas (0.5 to 1.2 g combined fresh weight) were placed in the flask. The sediment was suspended in the water by continuous stirring using a magnetic stir bar rotated at approximately 150 rpm's, and 0.1 mL of the PCB solution was added by pipet. Fish were not fed during exposures.

After various time intervals (0 to 120 hr), fish, sediment, and water were collected and analyzed for total radioactivity. Nine time intervals were used with three replications at each interval. One experimental unit (Florence flask, fish, water, and sediment) was taken down at each sampling. Fish were netted and rinsed with distilled water and their combined weight was taken. PCBs and lipids were extracted (Lake and others 1990) from the combined sample following homogenization in 20 mL acetone (x2) using a Brinkmann PCU-2-110 Polytron homogenizer. The acetone extract was partitioned between hexane and

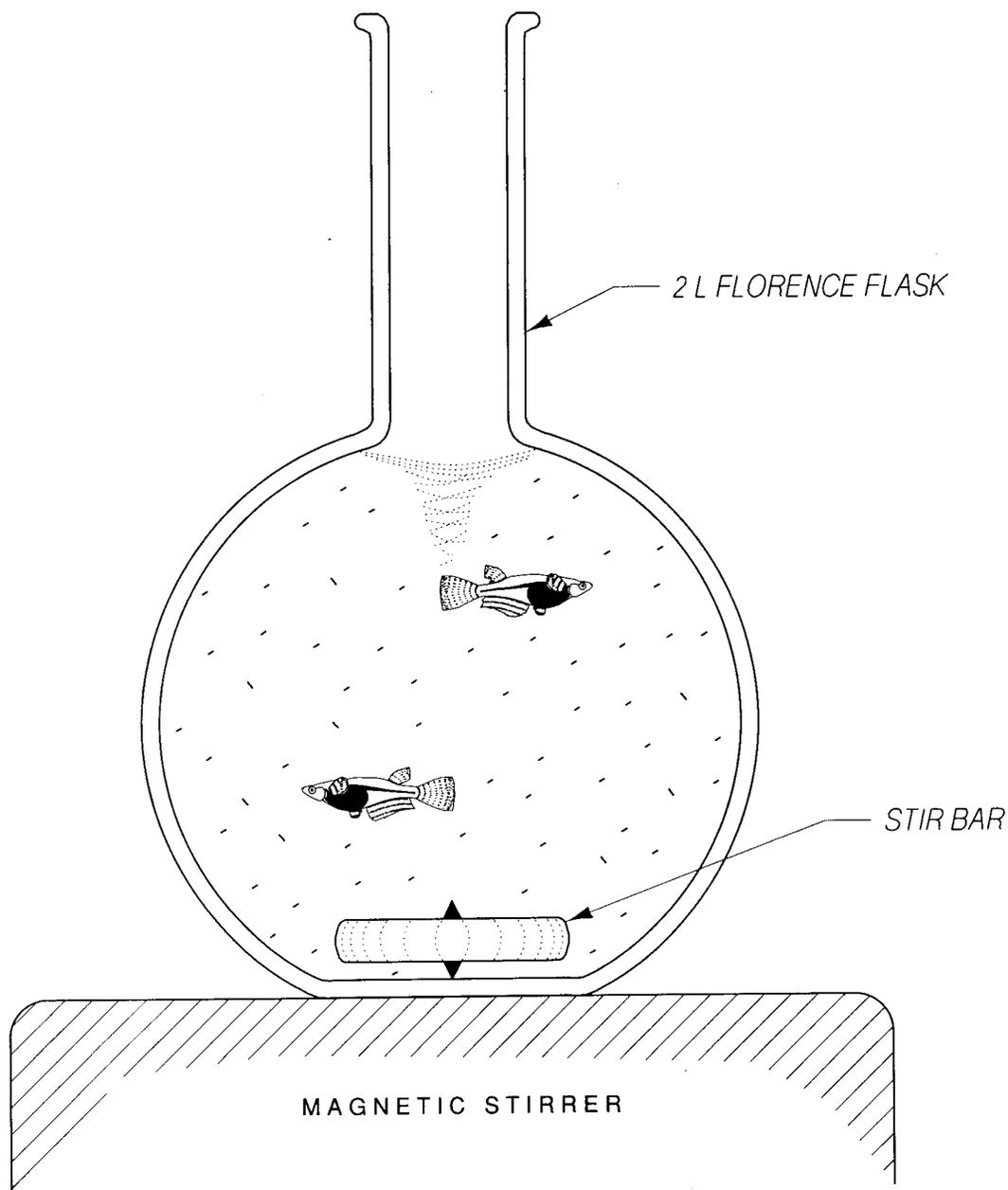


Figure 1. Exposure system, a Florence flask containing two medakas and 400 mg of sediment suspended in 2 L of water by means of a magnetic stirrer operated at 150 rpm's

water in a conical centrifuge tube. The hexane fraction was then split to provide separate aliquots for total lipids and total radioactivity measurements. The hexane extract for lipid determination was evaporated to dryness and total lipids were determined gravimetrically.

The sediment-water mixture was centrifuged and 100 mL of the supernatant were extracted with 25 mL hexane:acetone (4:1) followed by 20 mL hexane. The sediment pellet was extracted twice with 5 mL acetone and then partitioned between hexane and water. The hexane extracts of fish, sediment, and

water for radioactivity measurements were evaporated to near dryness in 20-mL scintillation vials; 15 mL of pseudocumene solvent (PCS) scintillation fluid (Amersham) was then added and the radioactivity counted on a Beckman Model 3801 Liquid Scintillation System.

Six 200-mL aliquots of the suspended sediment were centrifuged separately and the sediment pellets were analyzed by wet oxidation using the ampule method for total organic carbon (TOC) (Plumb 1981). The instrument used was an Oceanographic International Model 700 TOC Analyzer with infrared detection.

### Data Analysis

Computer modeling was conducted using PCNONLIN<sup>®</sup>, version 3.0 (Metzler and Weiner 1989).

### Results

The starting weight of fish in the exposure system varied from one exposure to another, while the mass of sediment put into suspension and the volume of water were constant. Therefore, the water and sediment radioactivity data for computation were adjusted based on a fish fresh weight of 1.0 g. The sediment TOC was 4.00 percent  $\pm$  0.578 (standard deviation). Lipid analyses produced highly variable results and the measurements taken were judged unusable for purposes of normalization of concentration data.

A three-compartment, closed kinetic model was used to represent PCB-52 distribution among water, fish, and sediment (Figure 2). In this model, water represents the central compartment (compartment 1) with PCB-52 being absorbed from the water by the sediment (compartment 2) and by the fish (compartment 3). Simultaneously, some of the chemical is desorbed back to the water from the sediment and eliminated to the water by the fish. These four processes are described by rate constants. Direct exchange between fish and sediment was considered insignificant and was not included in the model.

Equations for the rate of change of PCB in water, fish, and sediment are:

$$dC_w/dt = k_{21}C_s + k_{31}C_f - k_{12}C_w - k_{13}C_w \quad (1)$$

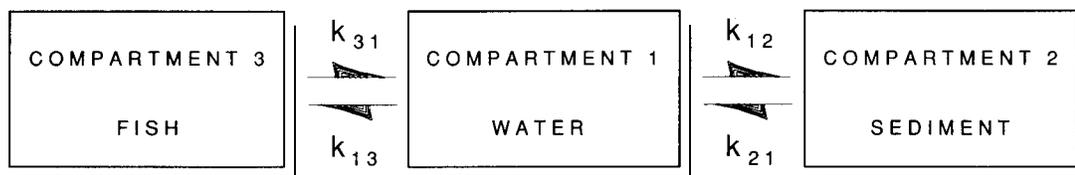


Figure 2. Three-compartment, closed kinetic model for phase-distribution of the chemical in the exposure system

$$dC_s / dt = k_{12}C_w - k_{21}C_s \quad (2)$$

$$dC_f / dt = k_{13}C_w - k_{31}C_f \quad (3)$$

where  $C_w$  is the adjusted mass of PCB-52 in the central compartment ( $\mu\text{g}/2\text{ L}$ ),  $C_s$  is the adjusted mass of PCB-52 in the sediment ( $\mu\text{g}/400\text{ mg}$ ),  $C_f$  is the concentration of PCB-52 in the fish ( $\mu\text{g}/\text{g}$ ), and  $k_{12}$ ,  $k_{21}$ ,  $k_{13}$ , and  $k_{31}$  are the rate constants for intercompartmental transfer.

The three differential equations (Equations 1 through 3) were fitted simultaneously to the experimental data by a nonlinear least-squares technique using PCNONLIN<sup>®</sup>. Values for the least-squares estimates of the rate constants are shown in Table 1. The analysis of variance for the fitted equations is shown in Table 2.

**Table 1. Least-Square Estimates of Rate Constants for Intercompartmental Transfer of PCB-52 for the Three-Compartment Model\***

Rate constant	Description	Least-squares Estimate (plus/minus standard deviation)
$k_{12}$	Water to sediment	$0.460 \pm 0.035$
$k_{21}$	Sediment to water	$0.108 \pm 0.016$
$k_{13}$	Water to fish	$0.066 \pm 0.008$
$k_{31}$	Fish to water	$0.005 \pm 0.002$

\* Units of rate constants are reciprocal hours ( $\text{hr}^{-1}$ )

**Table 2. Analysis of Variance for the Data Fitted to the Model Equations**

Equation	Source	df	SS	MS	F	P	r
$dC_w/dt$	Model	1	2.26	2.26	—	—	—
	Error	28	0.05	0.0018	1,255.6	<0.001	0.995
$dC_s/dt$	Model	1	0.89	0.89	—	—	—
	Error	28	0.24	0.0086	103.8	<0.001	0.877
$dC_f/dt$	Model	1	1.15	1.15	—	—	—
	Error	28	0.11	0.0039	292.7	<0.001	0.959

Figure 3 shows distribution of PCB-52 among the three compartments (water, sediment, and fish) over a 120-hr period. Model-generated lines are fitted to the data. Uptake of PCB-52 by sediment was rapid as was the decline of PCB-52 concentration in the water. By the 24th hour PCB-52 water concentration stabilized. The concentration of PCB-52 in the sediment shows a

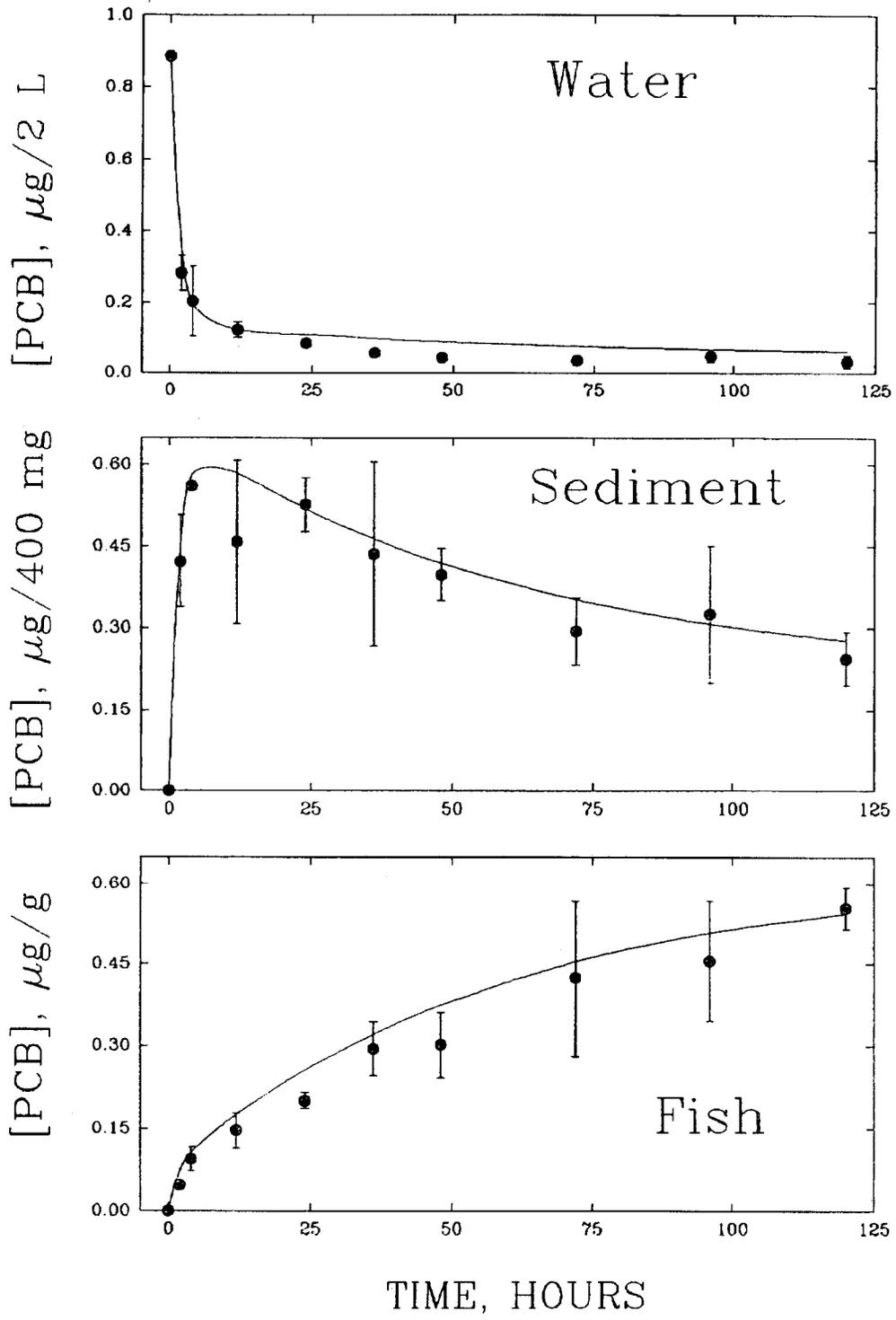


Figure 3. Distribution of PCB-52 among the three compartments — water, sediment, and fish over time; filled circles are means, vertical bars are plus/minus 1 standard deviation; lines are model-fitted estimates from the data

gradual decrease after the initial sorption phase, and this release of the chemical is reflected in the similarly gradual uptake of PCB-52 by the fish.

## Discussion

The slower uptake by the fish as compared with the sediment is typical of bioaccumulation or bioconcentration curves for hydrophobic neutral organic chemicals in general and can be explained by the existence of rate-limiting processes of a physiological nature (Karara and McFarland 1992). Equation 3 can be integrated to obtain the time required to reach a proportion (P) of steady-state concentration of chemical in the fish ( $C_{ss}$ ) when the exposure (water) concentration ( $C_w$ ) is constant:

$$t = -\ln(1 - P)/k_{31} \quad (4)$$

Solving Equation 4 for approximate steady state ( $P = 0.99$ ) results in a time of 921 hr or about 38 days. This time requirement is the same as would be expected for PCB-52 (or chemicals of similar  $\log K_{ow}$ ) in a standard bioconcentration or sediment bioaccumulation study, and is the reason why the Green Book (EPA and USACE 1991) requires at least a 28-day test for demonstration of bioavailability in Tier III testing and in Tier IV bioaccumulation tests. However, it is not necessary to carry a test of bioaccumulation to actual steady state if, as was done in the present experiment, the rate constants for uptake and elimination can be obtained by time-sequenced sampling over the first hours or days of exposure. Then, since steady-state chemical concentration in tissue is

$$C_{ss} = k_{13}C_w/k_{31} \quad (5)$$

the ratio of absorption and elimination rate constants is the bioconcentration factor,  $K_b$  (Branson and others 1975, Clarke and McFarland 1991):

$$k_{13}/k_{31} = C_{ss}/C_w = K_b \quad (6)$$

It has also previously been demonstrated that constant exposure conditions are likewise unnecessary if the exposure concentration decreases in a predictable first-order fashion (Karara and McFarland 1992).

As with  $K_b$ , the distribution coefficient for chemical partitioning between sediment and water,  $K_D$ , can be expressed as the ratio of rate constants:

$$K_D = C_s/C_w = k_{12}/k_{21} \quad (7)$$

When  $C_w$  is the same in Equations 6 and 7, the relationship of the two can be expressed as a proportion:

$$K_b/K_D = C_{ss}/C_s = (k_{13}/k_{31})/(k_{12}/k_{21}) \quad (8)$$

Equation 8 expresses the EqP relationship between sediment and organism, commonly referred to as the bioaccumulation factor, *BAF*. For the data of this experiment (Table 1):

$$BAF = (0.66/0.005)/(0.460/0.04) = 3.10 \quad (9)$$

Because *BAF* for a particular chemical will change depending on the organic carbon content of the sediment and the lipid content of the organism, the data of this type of experiment are usually normalized. Normalization on organic carbon and on lipid allows calculation of the *pf*:

$$pf = [(k_{13}/k_{31})/f_{lipid}]/[(k_{12}/k_{21})/f_{oc}] \quad (10)$$

where  $f_{lipid}$  and  $f_{oc}$  are the decimal fractions of lipid and organic carbon in exposed organisms and in sediment, respectively.

In the present study the lipid analyses were unusable and difficulties in the analysis are being resolved for future studies. For purposes of calculation here a default value of 6.0 percent lipid was selected based on previous studies in which lipid measurements were made on medakas (McFarland, Clarke, and Gibson 1985). Substituting the rate constants measured here (Table 1) and the decimal fractions of organic carbon (0.04) and lipid (0.06) into Equation 9 gives:

$$pf = [(0.066/0.005)/(0.06)]/[(0.460/0.108)/0.04] = 2.07$$

This value is in very good agreement with previously published *pf* values for PCB-52 using infaunal sediment-processing clams. Brannon and others (1989) exposed *Macoma nasuta* to sediment spiked with radiolabeled PCB-52 for up to 23 days and measured an average *pf* = 1.94. Sediment TOC was 1.06 percent. Ferraro and others (1991) exposed *M. nasuta* for 28 days to field-collected sediments having PCB-52 concentrations of 4.9 to 50 ng/g and having TOC contents ranging 0.84 to 7.37 percent. The *pf* values calculated were 2.1, 1.9, 2.1, 0.94 and 0.56.

## Conclusion

A three-compartment, closed kinetic model applied to data obtained with a simple fish/suspended sediment exposure system can be used to study EqP relationships for neutral organic chemicals. The approach eliminates the difficulties inherent in long-term testing required to reach steady-state conditions and the results agree well with previous work. The simplicity of the system permits a high degree of control to be exercised over the principal variables. This new technique can be applied both to determining reliable *pf* values for TBP calculations and, in a research mode, to gain further insights into the fundamental processes of equilibrium phase distribution of neutral organic chemicals. No kinetic projections can be made without some associated error, but in the present study that error appears to have been small judging by the high

correspondence between these results and the results of similar studies using long-term exposures.

In experiments to be begun in fiscal year 1993, the procedure described in this technical note will be used to test the limits of applicability of organic carbon normalization for neutral organic chemicals in EqP. The linearity of the normalization at low organic carbon concentrations has been clearly identified as a problem in defining both the utility of EqP-based SQC and in the practical usage of TBP calculations as a screening procedure. The experimental design described in this technical note appears well suited to address this problem.

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# *Environmental Effects of Dredging Technical Notes*



## **The $K_{oc}$ of Nonpolar Organic Compounds in Sediment**

### **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  $K_{oc}$  of selected PCBs and fluoranthene, and investigate the effect of time of contact between contaminants and sediment upon the value of  $K_{oc}$ .

### **Background**

The U.S. 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 U.S. 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 Corps 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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## Introduction

Sediment organic carbon has been identified as the most important factor controlling partitioning of nonpolar organic contaminants between sediment and water (Karickhoff 1981). Studies have also shown that partitioning of nonpolar organic compounds is strongly related to the octanol-water partitioning coefficient of the compound (Karickhoff 1981). Sediment concentrations expressed on a total organic carbon (TOC) basis have been used to predict concentrations of nonpolar organic compounds in sediment pore water for use as sediment quality criteria by the U.S. Environmental Protection Agency (EPA) (Brannon and others 1990).

The EPA approach for predicting interstitial water concentrations is called the Equilibrium Partitioning (EP) approach. The EP 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 chronic water quality criterion, the sediment is categorized as contaminated (Brannon and others 1990).

Prediction of pore-water contaminant concentrations based on values of  $K_{oc}$  (derived from  $K_{ow}$ , the octanol/water partition coefficient), total organic carbon content, and total contaminant concentration is likely to be incorrect. Measured pore-water concentrations have been shown to differ significantly from predicted values (Brannon and others 1991 and Steinberg, Pignatello, and Sawhney 1987). These differences can result from lack of equilibrium in the system or from factors related to organic matter composition (Gauthier, Seitz, and Grant 1987, Grathwohl 1990, Whitehouse 1985, and Schrap and Opperhuizen 1989) or other sediment factors affecting sorption capacity (Mingelgrin and Gerstl 1983). The source of organic matter affects the partitioning behavior of nonpolar organic compounds and can result in large variability in measured values of  $K_{oc}$ . Gauthier, Seitz, and Grant (1987) reported that  $K_{oc}$  can vary by a factor of 10 as a function of organic carbon aromaticity.

The laboratory experiments described in this technical note were designed to examine the relationships between sediment organic carbon and sediment interstitial water, the effects of sediment organic carbon on  $K_{oc}$  of selected polychlorinated biphenyls (PCBs) and fluoranthene, and the effects of time of contact between contaminants and sediment on interstitial water partitioning.

## Materials and Methods

Three sediments of varying TOC content were used in this study — Hamlet City Lake sediment (9.0 percent TOC) from Hamlet, North Carolina; U.S. Army Engineer Waterways Experiment Station (WES) reference soil (silt) (0.5 percent TOC) from Vicksburg, Mississippi; and sediment from Brown's Lake (2.8 percent TOC), a freshwater lake in Vicksburg, Mississippi. These sediments were

amended with 10 µg of either radiolabeled PCB 52 (2,2',5,5'-tetrachlorobiphenyl-UL-<sup>14</sup>C), PCB 151 (2,2',3,5,5',6-hexachlorobiphenyl-UL-<sup>3</sup>H), or fluoranthene (3-<sup>14</sup>C-fluoranthene) per g dry sediment weight using methods described previously (Brannon and others 1989). Sample bottles containing the sediments were stored upright to allow the sediments to settle and maintain a water cover. Samples were incubated for up to 180 days, with samples taken at intervals. At all sampling periods, concentrations of free and bound PCB 52, PCB 151, and fluoranthene were determined in the interstitial water using reverse phase partitioning (Landrum and others 1984).

## Results and Discussion

### Interstitial Water

Concentrations of free contaminants in interstitial water generally decreased or remained constant over time (Figure 1), even though the fraction of bound contaminant (contaminant complexed with dissolved organic carbon and microparticulates) remained essentially constant. For this reason, average values rather than time-dependent values of the bound contaminant fraction are given (Table 1). Decreases in concentrations of free contaminant are consistent with movement of nonpolar organic contaminants into interparticle organic matter pores over time (Brusseau and Rao 1989), resulting in decreased interstitial water concentrations.

**Table 1. Average (Standard Error) Fraction of Bound Contaminant in Interstitial Water during 180 Days of Testing**

<u>Contaminant</u>	<u>Brown's Lake</u>	<u>Hamlet City</u>	<u>Silt</u>
PCB 151	0.34 (0.03)	0.36 (0.02)	0.29 (0.03)
PCB 52	0.05 (0.009)	0.08 (0.02)	0.04 (0.004)
Fluoranthene	0.16 (0.03)	0.04 (0.004)	0.05 (0.017)

### Contaminant $K_{oc}$ Values

The ability of EP to predict interstitial water PCB 52, PCB 151, 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), PCB 52 (5.84), and PCB 151 (6.64) (Hawker and Connell 1988) in Equation 16 from Karickhoff (1981) that relates  $K_{ow}$  to  $K_{oc}$ . These calculations resulted in estimated  $K_{oc}$  values of 5.09, 5.43, and 6.22 for fluoranthene, PCB 52, and PCB 151, respectively. Measured values of  $K_{oc}$  were determined by dividing the TOC normalized sediment concentration of PCB

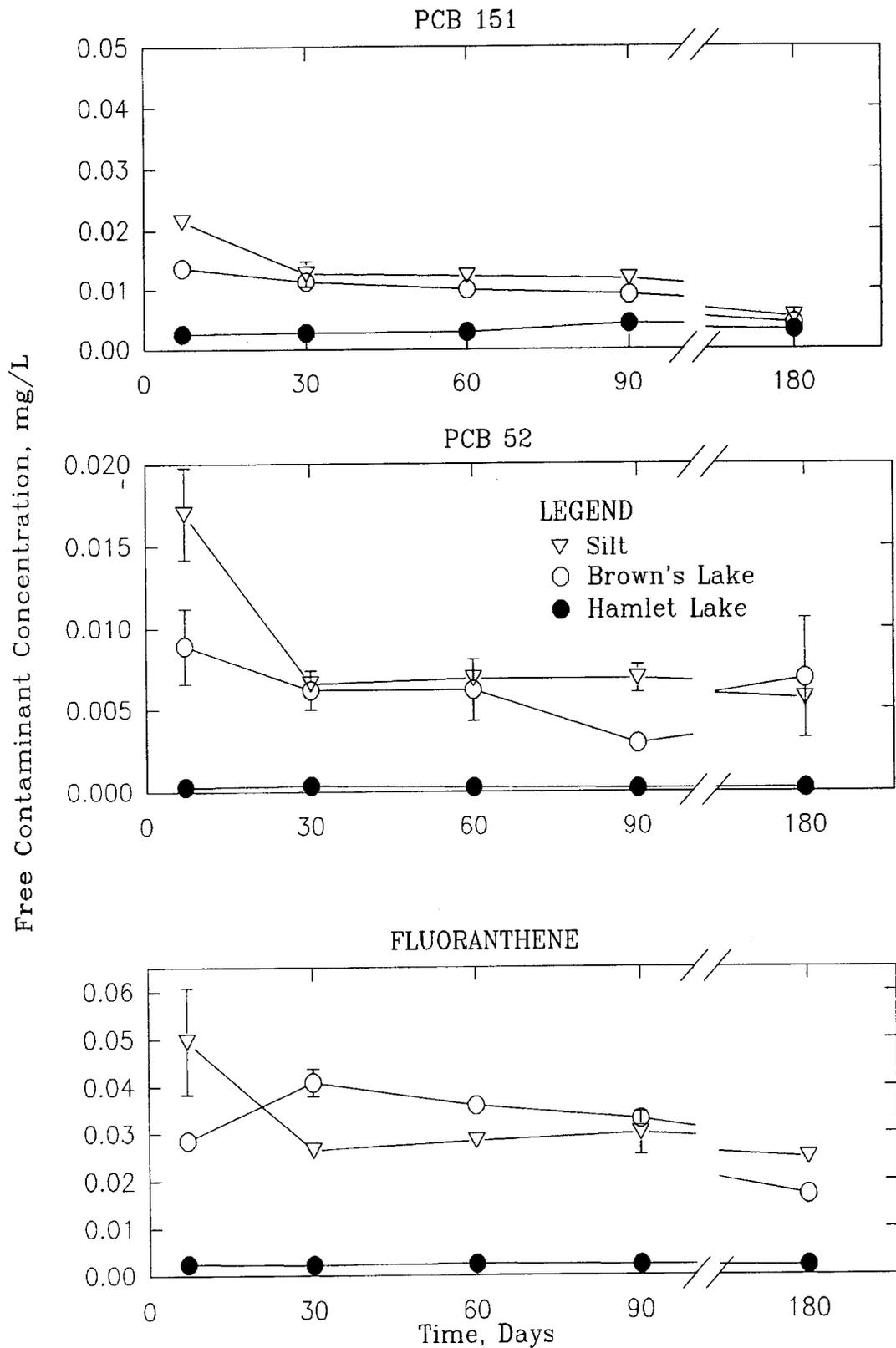


Figure 1. Concentrations of free PCB 151, PCB 52, and fluoranthene in interstitial water

52, PCB 151, or fluoranthene by the free interstitial water concentration of the respective compounds.

Measured  $\log K_{oc}$  values for PCB 151 ranged from approximately 4.2 to 4.9 and did not vary greatly between sediments (Figure 2). Furthermore,  $\log K_{oc}$  values for PCB 151 generally increased as exposure time in the sediment increased. Measured values of  $\log K_{oc}$  for PCB 151 were considerably lower than the estimated value.

Measured values of  $\log K_{oc}$  for PCB 52 and fluoranthene were in relatively close agreement for silt and Brown's Lake sediment, but were considerably higher (0.5-0.9 log units) in Hamlet Lake sediment (Figure 2). Measured values of  $\log K_{oc}$  for PCB 52 generally increased over time, similar to the results for PCB 151. Measured values of  $\log K_{oc}$  for PCB 52 were generally lower than the estimated value of  $\log K_{oc}$ , except for Hamlet Lake sediment, which was slightly higher. Measured values of fluoranthene  $\log K_{oc}$  generally increased over time and were lower than the estimated value, at times by as much as a full log unit.

Measured  $\log K_{oc}$  decreased as sediment TOC content decreased for PCB 52 (Figure 3). Fluoranthene  $\log K_{oc}$  varied in the same manner as did PCB 52 until TOC reached 2.8 percent, when no further change was noted. PCB 151 varied only slightly with sediment TOC. Ideally, measured values of  $\log K_{oc}$  should remain constant as TOC changes if EP reliably predicts free interstitial water concentrations of nonpolar organic contaminants. This situation was observed only for PCB 151. The deviation of measured  $K_{oc}$  values from estimated values and the dependence of measured PCB 52 and fluoranthene  $K_{oc}$  values on sediment TOC concentrations indicate that predictions of free interstitial water contaminant concentrations from EP are not reliable for all compounds. This may result from factors such as clay content and clay species increasing the sorption capacity of the sediment over that of the organic matter, resulting in a decrease in measured  $K_{oc}$ .

## Summary of Findings

Changes in  $K_{oc}$  over time and among sediments depended upon the compound tested. Values of  $K_{oc}$  measured using free interstitial water concentrations did not agree with estimated  $K_{oc}$  for all compounds and in all sediments. Estimated values of  $K_{oc}$  were generally substantially higher than measured values of  $K_{oc}$ . This was especially true of PCB 151 and fluoranthene in all sediments tested. For PCB 52, PCB 151, and fluoranthene,  $K_{oc}$  also generally increased as incubation time increased, indicating that sorption processes that reduced the free concentration of these compounds in the interstitial water were continuing. PCB 52 and fluoranthene  $K_{oc}$  also showed a marked dependence on sediment TOC concentration, generally decreasing as sediment TOC decreased.

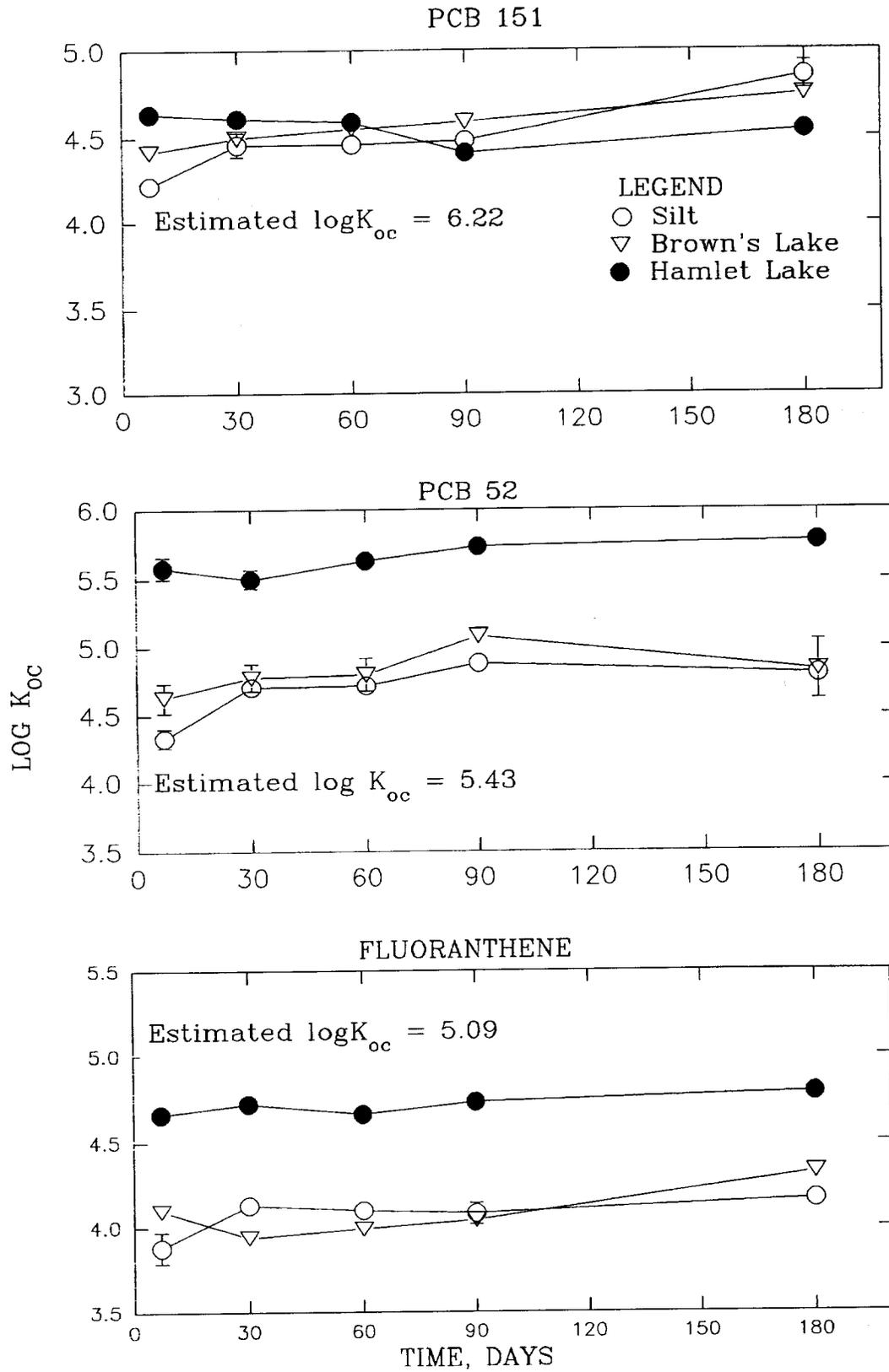


Figure 2. Measured  $\log K_{oc}$  for PCB 151, PCB 52, and fluoranthene

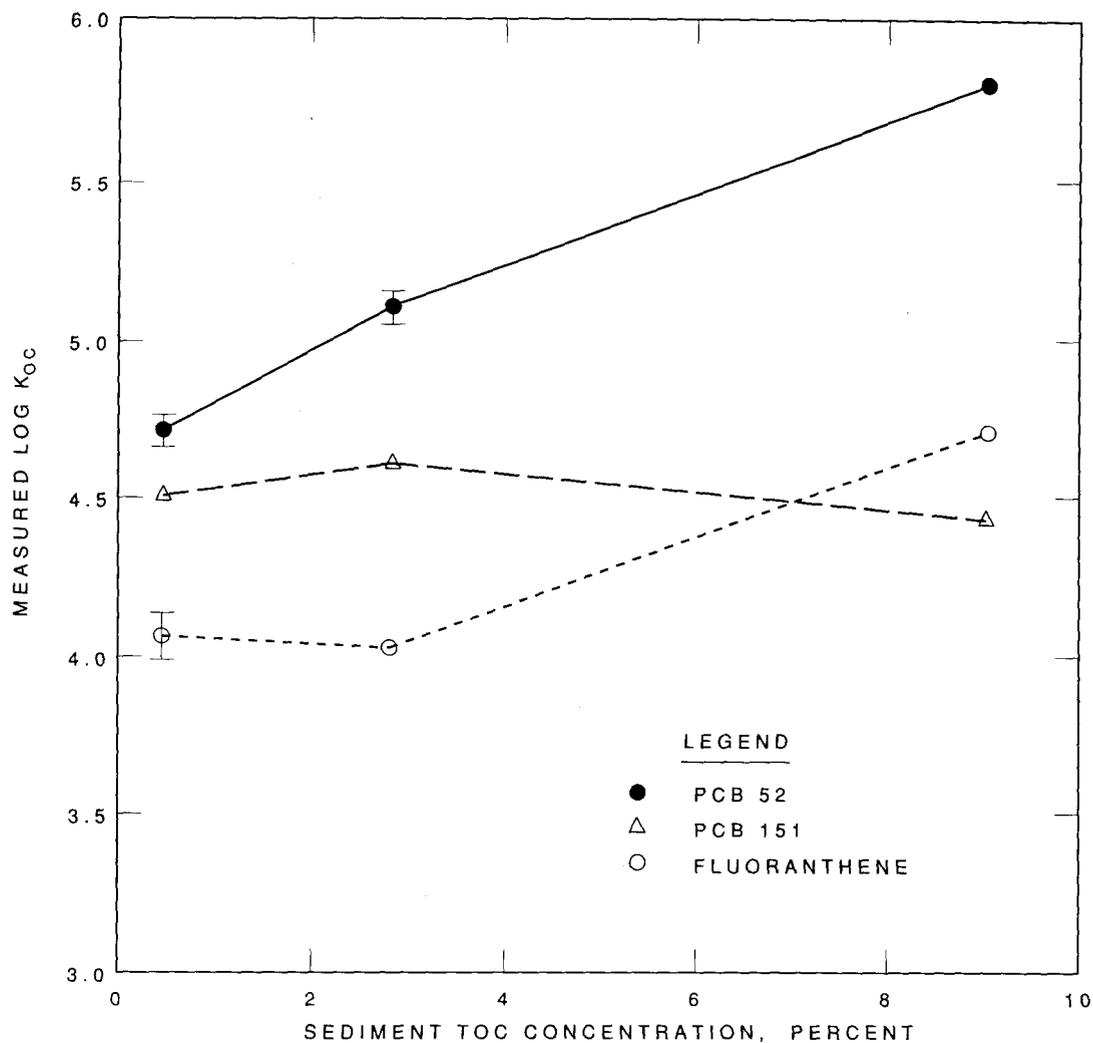


Figure 3. Measured log  $K_{oc}$  following three months incubation as a function of sediment TOC

The data indicated that a high degree of uncertainty exists for  $K_{oc}$  values among sediments and compounds. This can result in either overestimation or underestimation of free interstitial water contaminant concentrations when using equilibrium partitioning, estimated  $K_{oc}$  values, and TOC. Predictive methods with a high degree of uncertainty are not good regulatory tools. The factors affecting  $K_{oc}$  must be better understood before rigid regulatory criteria are promulgated.

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# *Environmental Effects of Dredging Technical Notes*



## **Critical Body Residue (CBR) Approach for Interpreting the Consequences of Bioaccumulation of Neutral Organic Contaminants**

### **Purpose**

This technical note describes a procedure for interpreting tissue residues of neutral organic chemicals generated in 28-day dredged material bioaccumulation bioassays. This interpretive guidance uses a critical body residue (CBR) of neutral organic chemicals reported for the fathead minnow, *Pimephales promelas*. The CBR is based on a very large U.S. Environmental Protection Agency (EPA) acute toxicity database and well accepted quantitative structure activity relationships (QSARs). Guidance in this technical note is not appropriate when xenobiotic metabolism of neutral organic contaminants is likely.

### **Background**

The evaluation of dredged material requires an assessment of "unacceptable adverse impacts." Testing to support this evaluation will often include sediment bioassays. One type of bioassay determines the bioaccumulation potential of sediment-associated contaminants. In this test, aquatic organisms are exposed to sediments for 10 or 28 days, depending on whether heavy metals or organic chemicals, respectively, are the contaminants of concern. Tissues of animals surviving the sediment exposure are chemically analyzed to evaluate bioaccumulation potential. Interpreting the biological importance of these bioaccumulation data (with regard to "unacceptable adverse impacts") has been problematic. Previous guidance to Corps field elements has been based on published peer-reviewed articles containing both contaminant tissue residues and the corresponding biological effects (see Bibliography). While this guidance is technically sound, its limited size and large test-to-test variations preclude broad generalizations.

The relationships among acute toxicity, level of exposure, and internal chemical dose have been examined in the fathead minnow, *Pimephales promelas* (McCarty and others 1985, McCarty 1986 and 1990). For a wide variety of neutral organic chemicals, the estimated internal body burden corresponding to acutely lethal exposures was remarkably constant — 4.4 mmol/kg wet weight\* (95 percent confidence interval (C.I.) = 3.7 - 5.2 mmol/kg, n = 150) (McCarty and others 1992). This conservative internal dose is referred to as the critical body residue (CBR). The CBR is based on a very large database of 96-hr LC<sub>50s</sub> generated by the U.S. Environmental Protection Agency Environmental Research Laboratory-Duluth (Brooke, Geiger, and Northcott 1984, Geiger and others 1985 and 1986, and Geiger, Call, and Brooke 1988); the bioconcentration QSARs of Mackay (1982); and the toxicity QSAR approach of Konemann (1981) and Veith, Call, and Brooke (1983). This technical note describes how the CBR reported for *P. promelas* can be used to interpret the biological consequences of bioaccumulation in dredged material bioassays.

## Additional Information

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## Approach

Using the CBR to interpret 28-day bioaccumulation data for neutral organic chemicals is simple and straightforward. The four-step procedure is described below and summarized in Table 1.

### Step 1

The first step is to express the original bioaccumulation data for each neutral organic chemical as milligram per kilogram wet weight. All contaminants must be considered even if the level of bioaccumulation was not statistically significant. If the original data are reported on a dry weight basis, multiply the concentration by (1.00 minus the proportion body water) to obtain wet weight-specific data. If percent body water of the test species is not known, 80 percent is a reasonable approximation (Lagler, Bardoch, and Miller 1962, Florey 1966, Emerson 1969, and Tucker and Harrison 1974). If concentrations are reported on a lipid basis, multiply by (1.00 minus the proportion of lipid) to

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\* In the original publications, tissue concentrations were reported as both mmol/kg and mmol/L. All residues are reported as mmol/kg in this technical note, which assumes an organism density of approximately 1.0.

**Table 1. Summary of Procedure for Using the Critical Body Residue (CBR) of Neutral Organic Contaminants in *P. Promelas* to Interpret Results of 28-day Bioaccumulation Bioassays**

Step 1	Express bioaccumulation data for all neutral organic chemicals as milligram per kilogram wet weight
Step 2	Convert the milligram per kilogram wet weight tissue concentrations from step 1 to millimoles per kilogram wet weight
Step 3	Multiply the millimoles per kilogram wet weight concentrations from step 2 by appropriate acute-to-chronic ratios to produce an estimated acute tissue concentration (EATC) for each neutral organic chemical
Step 4	Add up all EATCs from step 3. Compare this sum with the CBR for <i>Pimephales promelas</i> (4.4 mmol/kg). One of the following conclusions will emerge: If the sum of the EATCs is greater than CRB, "unacceptable adverse impacts" are likely. If the sum of the EATCs is less than CRB, "unacceptable adverse impacts" are unlikely.

obtain a weight-specific concentration. If percent lipid was based on a dry weight sample, convert to wet weight concentrations as above.

### Step 2

The second step involves converting each milligram per kilogram wet weight tissue concentration obtained in step 1 to millimoles per kilogram wet weight. To accomplish this, simply divide the molecular weight of each contaminant into its milligram per kilogram wet weight tissue concentration. One millimole of any chemical is equal to its molecular weight expressed in milligrams. Table 2 gives the atomic and molecular weights of many common elements and contaminants of concern. Additional atomic and molecular weights can be found in most chemistry textbooks or Verschueren (1983). If the molecular weight of the chemical of concern is not readily available, simply add up the atomic weights of all atoms in the molecule. The sum total of atomic weights is equal to the molecular weight.

### Step 3

Because bioaccumulation data generated in chronic (28-day) exposures are to be compared to a CBR which is estimated from acute (96-hr) exposures, some basis for normalizing this time difference is needed. One normalizing factor is the acute-to-chronic ratios published by the EPA (Table 3). The acute-to-chronic ratio is obtained by dividing the exposure concentration associated with chronic toxicity into the acutely lethal concentration; usually the 96-hr LC<sub>50</sub>. If no acute-to-chronic ratio has been calculated for the contaminant and test species of concern combination, a default value of 10 is recommended

**Table 2. Frequently Used Atomic and Molecular Weights**

<b>Atom</b>	<b>Atomic Weight*</b>	<b>Contaminant</b>	<b>Molecular Weight**</b>
Aluminum	227	Acenaphthylene	152
Arsenic	74.921	Acenaphthene	154
Barium	137.34	Anthracene	178
Boron	10.81	Aroclor 1016	257
Bromine	79.904	Aroclor 1221	192
Calcium	40.08	Aroclor 1232	221
Carbon	12.011	Aroclor 1242	261
Chlorine	35.453	Aroclor 1248	288
Fluorine	18.998	Aroclor 1254	327
Hydrogen	1.008	Aroclor 1260	372
Iron	55.847	Benzene	78
Magnesium	24.305	Benzo(a)anthracene	228
Mercury	200.59	Benzo(a)pyrene	252
Nickel	58.70	Benzo(e)pyrene	252
Nitrogen	14.007	Biphenyl	154
Oxygen	15.999	Chlorobenzene	113
Phosphorus	30.974	Bieldrin	381
Potassium	39.102	DDD	320
Silicon	28.086	DDT	355
Silver	107.87	Fluoranthene	202
Sodium	22.990	Mirex	546
Sulfur	32.06	Naphthalene	128
Tin	118.69	Perylene	252
Zinc	65.37	Phenanthrene	178
		Phenol	65
		Pyrene	202
		Tetrachloromethane	154
		Toluene	92

\* From Morrison and Boyd (1973).

\*\* From Verschueren (1983) and Mackay, Shiu, and Ma (1992a, 1992b).

**Table 3. Acute-to-Chronic (A:C) Ratios Published in the EPA Water Quality Criteria Documents (U.S. Environmental Protection Agency 1980) for Freshwater and Marine Organisms**

Contaminant	Test Species	A:C Ratio
Chlordane	<i>Daphnia magna</i>	3.6
	<i>Cyprinodon variegatus</i>	20
	<i>Lepomis macrochirus</i>	37
Chlorine	<i>Daphnia magna</i>	5.227
	<i>Menidia peninsula</i>	1,162
	<i>Gammarus pseudolimnaeus</i>	>37.18
	<i>Pimephales promelas</i>	6.162
Dieldrin	<i>Mysidopsis bahia</i>	6.2
	<i>Salmo gairdneri</i>	11
	<i>Poecilla reticulata</i>	9.1
DDT and metabolites	<i>Pimephales promelas</i>	65
Endosulfan	<i>Daphnia magna</i>	11
	<i>Mysidopsis bahia</i>	2.8
	<i>Pimephales promelas</i>	3.0
	<i>Cyprinodon variegatus</i>	2.4
Endrin	<i>Palaemonetes pugio</i>	19
	<i>Pimephales promelas</i>	2.2
	<i>Cyprinodon variegatus</i>	1.9
	<i>Jordanella floridae</i>	3.3
1,2-Dichloroethane	<i>Pimephales promelas</i>	5.9
1,1,2-Trichloroethane	<i>Pimephales promelas</i>	8.7
1,1,2,2-Tetrachloroethane	<i>Pimephales promelas</i>	8.5
Pentachloroethane	<i>Mysidopsis bahia</i>	1.4
	<i>Pimephales promelas</i>	6.6
Hexachloroethane	<i>Pimephales promelas</i>	2.8
Butylbenzyl phthalate	<i>Daphnia magna</i>	42
	<i>Pimephales promelas</i>	15
Heptachlor	<i>Pimephales promelas</i>	80
	<i>Cyprinodon variegatus</i>	3.9
Hexachloro-cyclohexane (Lindane)	<i>Daphnia magna</i>	33
	<i>Chironomus tentans</i>	63
	<i>Pimephales promelas</i>	7.5
Naphthalene	<i>Daphnia magna</i>	11
PCBs	<i>Gammarus pseudolimnaeus</i>	11
	<i>Pimephales promelas</i>	6.4
Pentachlorophenol	<i>Daphnia magna</i>	2.5
	<i>Pimephales promelas</i>	3.9
	<i>Cyprinodon variegatus</i>	6.9
Toxaphene	<i>Daphnia magna</i>	109.1
	<i>Mysidopsis bahia</i>	1.132
	<i>Pimephales promelas</i>	196
	<i>Cyprinodon variegatus</i>	1.540
	<i>Ictalurus punctatus</i>	28

(Kenaga 1982 and Mayer, Mayer, and Ellersieck 1986). Once the appropriate acute-to-chronic ratio has been identified, multiply it by the chronic bioaccumulation tissue concentration (obtained in step 3) to yield an estimated acute tissue concentration (EATC) in millimoles per kilogram wet weight for each neutral organic chemical.

#### Step 4

Add up all the EATCs obtained in step 3 and compare this sum to the CBR for neutral organic chemicals in *P. promelas* (4.4 mmol/kg). One of the following conclusions will emerge.

- If the sum of EATCs is greater than CBR, "unacceptable adverse impacts" are likely.
- If the sum of EATCs is less than CBR, "unacceptable adverse impacts" are unlikely.

An example calculation using hypothetical tissue residue data from a 28-day dredged material bioaccumulation bioassay is shown in Table 4.

#### Analysis

The above procedure is based on a number of assumptions. A major assumption is the validity of the CBR itself. One argument in its favor is the presumed mode of toxicity — nonspecific narcosis. This is "the reversible state of arrested activity of protoplasmic structures" (Veith and Broderius 1990). Neutral organic chemicals partition into the lipid portion of biological membranes because they are hydrophobic. Their presence as dissolved constituents in the lipid phase is believed to swell the membrane beyond a critical volume (Seeman 1972 and Franks and Lieb 1985). This swelling disrupts cellular structure and function and results in the overt symptoms of narcosis—lethargy, unconsciousness and death in extreme narcosis. This type of toxicity is called nonspecific narcosis because it affects biological membranes in general, not specific tissues; it has been observed in a very wide variety of organisms (plants, mammals, fish, and invertebrates); it can be induced by any neutral organic chemical; and the effects are additive. This mode-of-action suggests that the internal contaminant dose, expressed on a molar basis (that is, equal number of molecules), would be relatively constant for a variety of chemicals. This is precisely what is observed for the CBR estimated for *Pimephales promelas*.

Support for the validity of the CBR also comes from empirically determined acutely lethal tissue concentrations in aquatic organisms. Because the CBR is an estimated value, it is appropriate to compare it with empirically derived data gathered under the same or similar conditions (that is, acute exposures). Although such data are limited, the summary provided by McCarty and others (1992) indicates that acutely lethal tissue concentrations measured in crustaceans, insects, and other fish species agree reasonably well (that is, within single-digit range) with the estimated CBR for *P. promelas* (4.4 mmol/kg).

**Table 4. Example Calculation of Procedure Summarized in Table 1; Hypothetical Wet Weight Concentrations are from 28-day Dredged Material Bioaccumulation Bioassay with the Deposit-Feeding Marine Bivalve, *Macoma nasuta***

Contaminant	Molecular Weight	Tissue Concentration		A:C*	EATC**
		mg/kg	mmol/kg		
Phenol	65	0.1	0.002	10	0.02
Benzene	78	0.3	0.004	10	0.04
Toluene	92	0.2	0.002	10	0.02
Naphthalene	128	0.9	0.007	10	0.07
Biphenyl	154	0.6	0.004	10	0.04
Acenaphthene	154	1.4	0.009	10	0.09
Phenanthrene	178	2.1	0.012	10	0.12
Anthracene	178	1.8	0.010	10	0.10
Benzo(a)anthracene	228	3.7	0.017	10	0.17
Benzo(a)pyrene	252	4.2	0.017	10	0.17
Pyrene	202	5.9	0.029	10	0.29
Perylene	252	2.6	0.010	10	0.10
Chlorobenzene	113	0.7	0.006	10	0.06
Tetrachloromethane	154	0.1	0.001	10	0.01
Aroclor 1254	327	3.2	0.010	10	0.10
DDD	320	1.4	0.004	10	0.04
Dieldrin	381	0.8	0.002	10	0.02
Mirex	546	0.2	0.000	10	0.00

Sum of EATCs = 1.46

Sum of EATCs (1.46) < CBR (4.4)

Therefore, "unacceptable adverse impacts" are unlikely.

\* Acute-to-chronic ratio.

\*\* Estimated acute toxicity concentration.

A critical assumption in the procedure described in this technical note is that the toxicity and bioaccumulation potential of the freshwater fish, *P. promelas*, is representative of aquatic species in general. With respect to toxicity, Suter and others (1987) demonstrated that *P. promelas* is an acceptable surrogate test species for other freshwater fish. The EPA recommendation to evaluate toxicity of dredged material elutriates with *P. promelas* suggests that the agency believes it is an acceptable representative species. Both bioaccumulation and dose-response toxicity were reported for *P. promelas* following chronic exposures to

PCB-contaminated sediment (Dillon 1988). The advantages of using *P. promelas* in freshwater sediment bioaccumulation bioassays were discussed by Mac and Schmitt (1992). They also described in detail the bioaccumulation test procedure currently used for this species.

The toxicity and bioaccumulation QSARs used to estimate the CBR for *P. promelas* (4.4 mmol/kg) are based, in part, on the partitioning behavior of neutral organic chemicals between aqueous and lipid phases. These QSARs essentially treat aquatic organisms as "bags of lipids." Thus, lipid normalization tends to minimize differences among species. The CBR for *P. promelas* assumed a lipid content of 5 percent. If more divergent but realistic values are used (for example 3 and 8 percent), the mean CBR (95 percent C.I.) varies only slightly and remains within single-digit range; 2.6 mmol/kg (2.2 to 3.1) and 7.0 mmol/kg (5.9 to 8.3), respectively (McCarty and others 1992). Thus, the uncertainty introduced by interspecific differences in percent lipid appears to be minor.

There are, however, a number of reasons for questioning whether the CBR for *P. promelas* is representative of aquatic organisms in general. Suter and Rosen (1988), for example, demonstrated that extrapolating toxicity test results from fish to crustaceans introduces unacceptably large amounts of error. They speculated this may be due to interspecific differences in xenobiotic metabolism. Fish have highly developed contaminant metabolic capabilities. Other phylogenetic groups, such as mollusks, have very limited abilities. This is a major reason why deposit-feeding bivalve mollusks are frequently used in salt-water sediment bioaccumulation bioassays. A functionally equivalent freshwater mollusk has not been identified. A bioaccumulation test with the oligochaete, *Lumbriculus variegatus*, has recently been proposed (Call and others 1992). However, the capacity of this organism to metabolize xenobiotics has not been critically examined with regard to sediment bioassays.

If the validity of the CBR is accepted, then the major source of uncertainty in the procedure described herein is the link between acute and chronic toxicity. Acute-to-chronic ratios published by the EPA are used to establish this link. The acute-to-chronic ratio is obtained by dividing the chemical concentration associated with chronic toxicity into the acutely toxic concentration, usually the 96-hr LC<sub>50</sub>. The chronic value is based on results observed in partial or full life-cycle toxicity tests with aquatic organisms. It is derived from the lowest concentration where adverse biological effects were observed (lowest observed effects concentration or LOEC), the highest concentration where no adverse effects were observed (highest no effect concentration or HNEC), or the geometric mean of the LOEC and HNEC. The exact derivation varies with each chemical and each chronic laboratory experiment. If an acute-to-chronic ratio is lacking, a default value of 10 is recommended (Kenaga 1982 and Mayer, Mayer, and Ellersieck 1986). This default value is believed to be environmentally conservative for most organic chemicals.

Although acute-to-chronic ratios are empirical observations, there are some fundamental mechanistic reasons why acute and chronic toxicity should not, or in some cases, cannot be linked. The mode of acute toxicity of neutral organic

chemicals is believed to be nonspecific narcosis (see discussion above). While this mode of action can also produce chronic toxicity, other "specific" mechanisms may be more important in some animals. One mode of action requires bioactivation of the contaminant molecule via xenobiotic metabolism. The classic example is biotransformation of benzo(a)pyrene to the more toxic diol epoxide. Since xenobiotic biotransformation to toxic metabolites is not an important consideration in acute toxicity, the link between acute and chronic toxicity is lost if the species of concern has significant xenobiotic metabolizing capability. Another "specific" mechanism inducing chronic, but not acute, toxicity is associated with coplanar molecules such as 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and certain polychlorinated biphenyl (PCB) congeners. This receptor site-mediated mode of action does not require bioactivation by the xenobiotic metabolism system. For these reasons, it is recommended that the procedure outlined in this technical note not be used with isosteres of TCDD or if the test species has a well developed xenobiotic metabolizing system. In either case, the link between acute and chronic toxicity would be tenuous.

The conclusion reached in step 4 regarding the probability of "unacceptable adverse impacts" can never be a stand-alone criterion. That is, the decision regarding the acceptability of dredged material cannot be based solely on the results observed in step 4. Rather, it represents only one of many inputs to the technical evaluation of dredged material. Other considerations include the magnitude of bioaccumulation relative to the reference, the proportion of contaminants accumulated, sediment toxicity, volumes of material involved as well as potential management alternatives. The procedure in this technical note is simply an additional tool for evaluating the consequences of bioaccumulation in aquatic organisms.

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