EVALUATION OF DREDGED MATERIAL DISPOSAL ALTERNATIVES FOR US NAVY HOMEPORT AT EVERETT, WASHINGTON

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The US Navy has proposed to homeport a carrier battle group at Everett, Wash. Development of the homeport will involve dredging and disposal of approximately 1 million cu yd (765,000 cu m) of contaminated sediments and an additional 2.3 million cu yd (1.7 million cu m) of uncontaminated native material. The US Army Engineer District, Seattle, is providing technical assistance in developing a dredging and disposal plan for these sediments from the East Waterway. In addition, the Seattle District is a permitting agency under Section 10 of the River and Harbor Act of 1899 and Section 404 of the Clean Water Act.

The Seattle District requested that the US Army Engineer Waterways Experiment Station (WES) provide support for testing and evaluations required for its technical assistance role for the Everett project. The purpose of the WES studies was to evaluate

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the feasibility of alternatives from an environmental and related engineering standpoint. This report documents the results of these studies through September 1986.

Three major disposal alternatives were evaluated for disposal of the contaminated sediment: confined upland, confined nearshore, and contained aquatic disposal (CAD). The Navy identified CAD as a preferred alternative during the course of the WES study, and also as the selected alternative in all applications for a Section 404 permit.

The management strategy for disposal of dredged material, recently confirmed as Corps policy for such evaluations, was applied as a part of the WES study. Samples of the contaminated and uncontaminated East Waterway sediments were collected, and a series of environmental and related engineering tests and evaluations were conducted. Numerical modeling studies and analytical evaluations were also conducted to determine the physical behavior of the dredged material for the CAD alternative.

Evaluations of dredging equipment were made based on previous studies of the sediment resuspension characteristics of various dredge types and demonstrations of innovative equipment for dredging contaminated sediments. Site-specific feasibility determinations for identified disposal sites were made based on the available data.

Results of the WES study showed that CAD is feasible. However, CAD at the water depth under consideration and placement of cap by hydraulic pipeline without lateral confinement have not yet been attempted. Confined disposal at identified intertidal sites is feasible and involves known and proven technology. Upland disposal is feasible, but expensive contaminant controls would likely be required. Final designs are required for any of the alternatives under consideration.

SUMMARY

Background

The US Navy has proposed to homeport a carrier battle group at Everett,
Wash. Development of the homeport will involve dredging and disposal of
approximately 1 million cu yd (765,000 cu m) of contaminated sediments from
the East Waterway, Everett Harbor. An additional 2.3 million cu yd (1.7 mil-
lion cu m) of uncontaminated native material must also be dredged. The
US Army Engineer District, Seattle, was requested by the Navy to provide tech-
nical assistance in developing a dredging and disposal plan for these sedi-
ments from the East Waterway. In addition, the Seattle District is the
permitting agency under Section 10 of the River and Harbor Act of 1899 and
Section 404 of the Clean Water Act. The Seattle District requested that the
US Army Engineer Waterways Experiment Station (WES) provide support for test-
ing and evaluations required for its technical assistance role for the Everett
project. The purpose of the study reported herein was to evaluate dredging
and disposal alternatives for the Everett Homeport project from an environ-
mental and related engineering standpoint, using tests and evaluations con-
ducted by WES on the project sediments. This report documents the WES studies
conducted through September 1986 and is not intended to reflect subsequent
changes in the project.

Disposal Alternatives

Three major disposal alternatives were evaluated for disposal of contam-
inated Everett Harbor sediment: confined upland, confined nearshore, and con-
tained aquatic disposal (CAD). As defined for purposes of this report, CAD is
the placement of contaminated sediments in an open-water site and capping with
clean sediment either with or without lateral confinement. CAD was selected
as the preferred alternative by the Navy, and a potential CAD site was identi-
fied, the Deep Delta site located near the dredging in East Waterway. Two
nearshore sites were tentatively identified as alternatives, the Snohomish
Channel site and the East Waterway site. Site-specific feasibility determina-
tions for each of these sites were made as a part of this study based on the

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available data. In addition, an area for potential development of an upland site was identified at Smith Island, north of the homeport area. An effort was made to apply data to the Smith Island site to the maximum extent possible.

As this report was being written, the Navy's plans for disposal were evolving. The Navy proposed CAD with surface release of contaminated material using bottom dump barges and hydraulic pipeline discharge of clean material at the surface for cap placement. The use of a previously considered downpipe and subaqueous lateral confinement was eliminated. The dredging would be accomplished in two phases to accommodate other construction scheduling. The alternative as proposed is similar to conventional capping operations successfully demonstrated at other locations, although the proposed disposal site is in much deeper water.

Strategy for Evaluation of Alternatives

The WES has developed a Management Strategy for disposal of dredged material which describes a logical sequence for testing and evaluation of alternatives for disposal. A Decisionmaking Framework was developed for the Seattle District for application of the Management Strategy to other projects within the District. The Decisionmaking Framework provides a basis for comparison of test results with standards (or criteria) or reference information to determine if contaminant control measures are required in a given instance. These two documents serve as a basis for the testing and decisionmaking described in this report.

Samples of the contaminated and uncontaminated East Waterway sediments were collected by the Seattle District, and a series of environmental and related engineering tests were conducted by WES using the samples. Numerical modeling studies and analytical evaluations were conducted to determine behavior of the dredged material in each of the disposal environments. Contaminant concentrations for a reference water and water quality criteria were specified by the Seattle District for interpretation of the results. In addition, a performance goal of 5 percent for total mass release of contaminants for dredging and disposal was specified by the District. The reference, criteria, and performance goal were judged by the District to be a conservative means to indicate the potential need for contaminant controls.

Evaluation of Dredging Equipment

An evaluation of dredging equipment for the disposal alternatives was made based on previous studies of the sediment resuspension characteristics of various dredge types and demonstrations of innovative equipment for dredging contaminated sediments. For the CAD alternative, clamshell dredging and transport in split-hull barges is considered the most compatible dredging technique for the contaminated sediments. Hydraulic cutterhead dredging with direct pipeline transport is considered the best technique for dredging the uncontaminated (capping) material for the CAD alternative. Hydraulic cutterhead dredging with direct pipeline disposal is considered the best dredging technique for the intertidal alternatives. The use of conventional dredging equipment and techniques that have been successfully used in similar applications elsewhere is considered a reasonable approach for this project. Use of specialized dredging equipment due to the presence of contaminants is not considered necessary.

The estimated release of contaminants in the dissolved form during dredging is negligible. Estimated mass release was considered equal to the mass sediment release. Based on available data on sediment resuspension by dredges, a release of 2 percent for clamshell dredging and 1 percent for hydraulic cutterhead dredging was estimated. Control measures during dredging to reduce sediment resuspension and contaminant release are options to reduce total mass release. Implementation of those control measures which involve minimal additional cost should be considered. Such measures might include use of an enclosed clamshell bucket, operational controls, and selecting dredging sequences from north to south in the waterway to the extent practicable.

Evaluation of Contained Aquatic Disposal

The proposed CAD alternative involves level bottom capping of contaminated sediments with uncontaminated sediment. This alternative is similar to conventional capping operations that have been successfully demonstrated at other locations. However, capping has not yet been attempted at the water depths proposed, nor has capping been attempted using hydraulic pipeline placement of the cap without subaqueous lateral confinement. The CAD alternative should not be considered merely a variation of open-water disposal, but

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rather as an engineered approach with carefully considered design, care during construction, and monitoring to ensure that the design is adequate.

The following tests and evaluations were performed for the CAD alternative:

a. Standard elutriate tests for estimating dissolved contaminant release during placement.

b. Capping effectiveness tests to determine the required cap thickness to chemically and biologically isolate the contaminated material.

c. Numerical modeling to simulate behavior of the contaminated and capping material during placement.

d. Analytical evaluations of mounding behavior to estimate spread and height of the mound.

Capping effectiveness tests show that the Everett Harbor contaminated sediments should be capped with a minimum cap thickness of 80 cm to effectively isolate the material from the overlying environment. To allow for irregularity during placement, a 1-m cap thickness should be specified as an operational requirement.

Modeling results show that placement of a single bargeload of the contaminated sediments at the CAD site using surface disposal will result in an area of deposition on the bottom approximately 215 m in diameter. Approximately 1.9 percent of the material will remain in suspension longer than 1,800 sec and was assumed to be a mass release. Placement of the uncontaminated capping material using controlled surface discharge from a pipeline moving across the site would result in an area of deposition approximately 90 m in width. Multiple passes of the pipeline would be required to accumulate the required cap thickness.

Presently available models do not predict size and shape of the disposal mound after a large volume of material has been deposited; therefore, an estimate of the mound configuration was made based on field data collected at other sites. This evaluation indicated that the total volume of contaminated and cap material would accumulate in a mound with bottom radius of approximately 730 m and a final height of approximately 4 m. Final cap thickness would be approximately 1 m. The site dimensions previously defined at the Deep Delta site would have to be expanded to accommodate this mound configuration.

Standard elutriate testing indicated that contaminant release in dissolved form during placement of the contaminated material was below reference

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water concentrations or criteria for most parameters. Dilution of concentrations for remaining parameters to background or criteria can be accomplished within a short distance of the placement operation. Mass release during placement was considered directly related to sediment release and varied from 2.0 to 2.1 percent depending on the parameter. With the addition of the mass release due to clamshell dredging, the estimated total mass release for the CAD alternative is 4.1 percent. Therefore, no contaminant control measures are necessary to meet the performance standard of 5 percent.

A monitoring program for the CAD alternative should be implemented to include sediment resuspension and contaminant release during dredging and placement, configuration of the mound and cap during and after placement, and effectiveness of the cap.

Evaluation of Intertidal Disposal

Several options for using both the East Waterway and Snohomish Channel sites were identified by the Navy. Two options were considered environmentally representative and were evaluated in this study: (a) a 12.9-acre (50,000-sq m) configuration for the East Waterway site to be used in combination with a 100-acre (400,000 sq m) configuration for the Snohomish Channel site, and (b) a 155-acre (625,000 sq m) configuration for the Snohomish Channel site to be used alone.

The following tests and evaluations were performed for the intertidal alternatives:

a. Modified elutriate tests for estimating the quality of effluent discharged during filling operations.

b. Surface runoff tests for estimating the quality of rainfall-induced surface runoff.

c. Leachate tests for estimating the quality of leachate into ground water or seepage through dikes.

d. Settling tests to estimate the relationship between dredged and disposal area volumes and the suspended solids concentration in effluents during filling.

e. Chemical clarification test to determine effective polymers and dosages for removal of suspended solids from effluent or surface runoff.

f. Consolidation tests to determine the relationship of fill elevation and time after filling.

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g. Stabilization/solidification testing to determine effective chemical additives to immobilize contaminants and improve the engineering properties of the dredged material.

Modified elutriate test results show that the dissolved concentrations of contaminants in the effluent discharged during filling are below reference water concentrations or criteria for most parameters. Dilution of concentrations for remaining parameters to background or criteria can be accomplished within a short distance of the discharge. These results are applicable to both intertidal sites. Settling test and modified elutriate test results show that the mass release in effluent varies depending on the parameter. The maximum values were: 4.5 percent for East Waterway, 6.6 percent for the Snohomish (100 acres), and 5.4 percent for the Snohomish (155 acres).

Surface runoff test results show that the dissolved concentrations of contaminants in the runoff from a representative storm event are below reference water concentration or criteria for most parameters. Dilution of concentrations for remaining parameters to background or criteria can be accomplished within a short distance of the discharge. Mass release of contaminants in runoff during a 1-year period with typical yearly rainfall conditions is negligible. These results are applicable to both sites. It is assumed that a surface cap of sufficient thickness will be placed over the contaminated material within a year of disposal to prevent long-term release from surface runoff and potential uptake of contaminants by plants or animals that may colonize the site(s).

Drinking water standards were exceeded in the leachate for some parameters. Regional authority decisions regarding possible ground-water mixing zones or requirements for control measures would necessarily depend on the final site selection and design. An estimate of mass release in leachate based on modeling results and leachate test results showed that the mass release was negligible. These results are applicable to both sites.

With the addition of mass release due to cutterhead dredging, the estimated total mass releases for the intertidal alternatives are: 5.5 for the East Waterway, 7.6 for the Snohomish Channel (100 acres), and 6.5 for the Snohomish Channel (155 acres). Since the performance standard is exceeded for both alternatives, controls would be required to meet the standard. The most cost-effective controls would include reductions in sediment resuspension

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during cutterhead dredging and chemical clarification to reduce suspended solids and associated contaminants in the effluent during filling operations.

A monitoring program for intertidal disposal should be implemented to include: sediment resuspension and contaminant release during dredging and transport, effluent quality during filling, surface runoff quality for a representative storm event, and ground-water quality using monitoring wells.

Evaluation of Upland Disposal

An area for potential development of an upland site was identified at Smith Island, north of the homeport area. However, only limited information was available in September 1986, and a number of possible sizes and configurations for the upland site have been identified. Until a site configuration(s) is identified and additional data on site conditions are obtained, a site-specific evaluation for upland disposal similar to the evaluations performed for intertidal sites cannot be conducted. However, the results of settling, modified elutriate, surface runoff, and leachate tests are directly applicable to evaluation of upland disposal.

Comparisons of dissolved concentrations of contaminants in effluent as predicted by modified elutriate tests and water quality criteria are valid for any of the upland site configurations now under consideration for Smith Island. Mass release of contaminants in effluent is dependent on effluent suspended solids concentrations. Determination of mass release is therefore possible only for a specific set of site conditions. However, mass release in effluent would be similar to that determined for the intertidal sites under consideration. Based on the previous evaluations for the intertidal sites, controls for mass release in effluent would likely be required to limit the total mass release for the upland alternative to less than the 5-percent performance goal.

The final surface of the contaminated sediments placed in an upland site could be at elevations either above or below the water table. Comparisons of dissolved and particle-associated concentrations of contaminants in surface runoff under both anaerobic and aerobic conditions with water quality criteria are valid for an upland evaluation including Smith Island. Mass release of contaminants in surface runoff is directly proportional to surface area of the disposal site, since it can be assumed that rainfall occurrences would be the

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same for Smith Island as for the intertidal sites. Mass release was found to be negligible for the intertidal condition, and would similarly be negligible for the upland condition.

The prediction of leachate impacts is a function of ground-water movement at the site under consideration. Depending on the site selected and site conditions, contaminated dredged material may be placed above or below the water table. If contaminated material is placed below the water table, the leachate characteristics may be estimated using anaerobic leaching test results. Leachate from material placed above the water table may be estimated using aerobic results.

Anaerobic leaching data for lead and chromium exceeded the drinking water standards; therefore, a regional authority decision may require some type of control to prevent any contaminant migration from material placed below the water table because of the possibility of deterioration to potential receptors.

Aerobic leaching data indicate that cadmium, chromium, and lead exceed the drinking water standard by a much greater margin than the anaerobic test results. This may require a more extensive control measure for contaminated material placed above the water table than would be required for material placed below the water table. Again, site-specific conditions would dictate the type of control measure that would be necessary. The possibility of a ground-water mixing zone to provide the necessary dilution may be possible. Also, a shallow configuration for the containment area would make the installation of a liner a more viable control option. Depending on the size of the containment area, the amount of material to be dredged, and the site conditions, a practical disposal scenario would be to place the contaminated material below the water table, where the material would remain anaerobic, thereby releasing fewer contaminants. Cleaner material used as a surface cap could be placed above the water table.

Conclusions

Contained aquatic disposal (capping) of Everett Harbor sediments at the Deep Delta site is feasible. However, CAD at the water depth under consideration and placement of cap by hydraulic pipeline without lateral confinement have not yet been attempted. Confined disposal of Everett Harbor material at

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the Snohomish site or a combination of the Snohomish and East Waterway sites is feasible and involves known and proven technology. Disposal of Everett Harbor material at an upland site is generally feasible. Site-specific data are required for design of any of the alternatives under consideration.

PREFACE

This report describes an evaluation of dredging and disposal alternatives for the proposed US Navy homeport at Everett, Wash. The US Army Engineer District, Seattle, is assisting the Navy in preparing a plan for the dredging of approximately 1 million cu yd (765,000 cu m) of contaminated sediments, which is required as part of the project. This report presents the results of sediment testing and disposal modeling conducted by the US Army Engineer Waterways Experiment Station (WES) for the Seattle District. Earlier work by the WES included a March 1986 report describing design requirements for the project, a June 1986 report on evaluation of disposal alternatives, and a September 1986 technical supplement. These reports provided a partial basis for the project design, information in support of permit evaluation for the project, and information used in preparing a supplemental Environmental Impact Statement. This report describes environmental and related engineering evaluations of dredging and disposal alternatives, focusing on the sediment testing and modeling efforts performed by WES.

This report was prepared by the following personnel of the Environmental Engineering Division (EED) and the Ecosystem Research and Simulation Division (ERSD) of the WES Environmental Laboratory (EL) and the Estuaries Division (ED) of the WES Hydraulics Laboratory (HL): Dr. Michael R. Palermo, Mr. Rick A. Shafer, Mr. Clifford L. Truitt, Mr. Mark E. Zappi, Mr. Tommy E. Myers, Dr. D. M. Griffin, Jr., and Mr. Roy Wade, EED; Dr. James M. Brannon, Mr. John G. Skogerboe, Mr. T. C. Sturgis, Dr. Douglas Gunnison, Dr. Henry Tatum, and Ms. Susan Portzer, ERSD; and Mr. Steven A. Adamec, ED. Dr. Palermo acted as coordinator for the study. Technical reviews of various portions of the report were provided by Dr. Robert M. Engler, Manager, Environmental Effects of Dredging Programs, EL; Mr. Norman R. Francingues and Mr. M. John Cullinane, EED; Dr. Thomas L. Hart and Dr. Charles R. Lee, ERSD; Dr. Billy H. Johnson, Hydraulic Analysis Division, HL; and Messrs. John Malek, Dave Schuldt, Eric Nelson, Walt Farrar, and Robert Parker of the Seattle District. The report was edited by Ms. Jessica S. Ruff of the WES Information Technology Laboratory.

The report was prepared under the general supervision of Dr. Raymond L. Montgomery, Chief, EED; Mr. Donald L. Robey, Chief, ERSD; Mr. William H.

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McAnally, Chief, ED; Dr. John Harrison, Chief, EL; and Mr. Frank Herrmann, Chief, HL.

Commander and Director of WES was COL Dwayne G. Lee, CE. Technical Director was Dr. Robert W. Whalin.

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CONTENTS

CONVERSION FACTORS, NON-SI TO SI (METRIC)
UNITS OF MEASUREMENT

Non-SI units of measurement used in this report can be converted to SI (metric) units as follows:

Multiply	By	To Obtain
acres	4,046.873	square metres
cubic yards	0.7645549	cubic metres
feet	0.3048	metres
gallons (US liquid)	3.785412	cubic decimetres
pounds (force) per square inch	6.894757	kilopascals
pounds (mass)	0.4535924	kilograms
pounds (mass) per cubic feet	16.01846	kilograms per square metre
tons (force) per square foot	95.76052	kilopascals
tons (2,000 pounds, mass)	907.1847	kilograms
yards	0.9144	metres

EVALUATION OF DREDGED MATERIAL DISPOSAL ALTERNATIVES FOR US NAVY HOMEPORT AT EVERETT, WASHINGTON

PART I: INTRODUCTION

Background

1. The US Navy has proposed to homeport a carrier battle group at Everett, Wash. Development of the homeport will involve dredging and disposal of approximately 928,000 cu yd* of contaminated sediments from the East Waterway, Everett Harbor. An additional 2,377,000 cu yd of uncontaminated native material must also be dredged. The dredging work will be conducted in two phases to accommodate construction schedules and seasonal restrictions on dredging. The dredging volumes and phases are tabulated below. The project location, the dredging plan, and typical cross sections are shown in Figures 1-3.

	Dredging Volume, cu yd		
Phase*	Dredged as Contaminated**	Clean	Total
P-111	97,000	739,000	836,000
P-905	224,500	1,140,000	1,364,500
P-112	552,000	498,000	1,050,000
	54,500[†]		54,500
	928,000	2,377,000	3,305,000

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* Refer to Figure 2.
** Overdepth and prism tolerances included.
† Contaminated sediment below project depth in P-112.
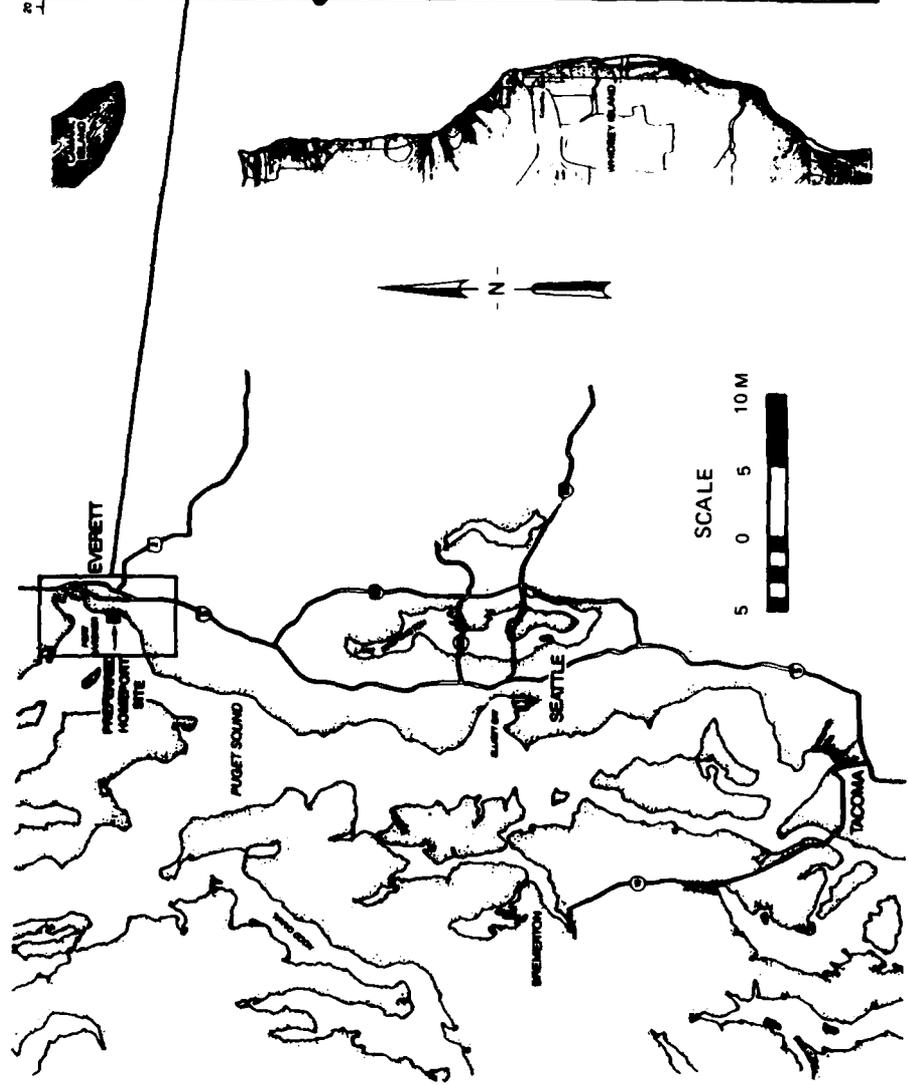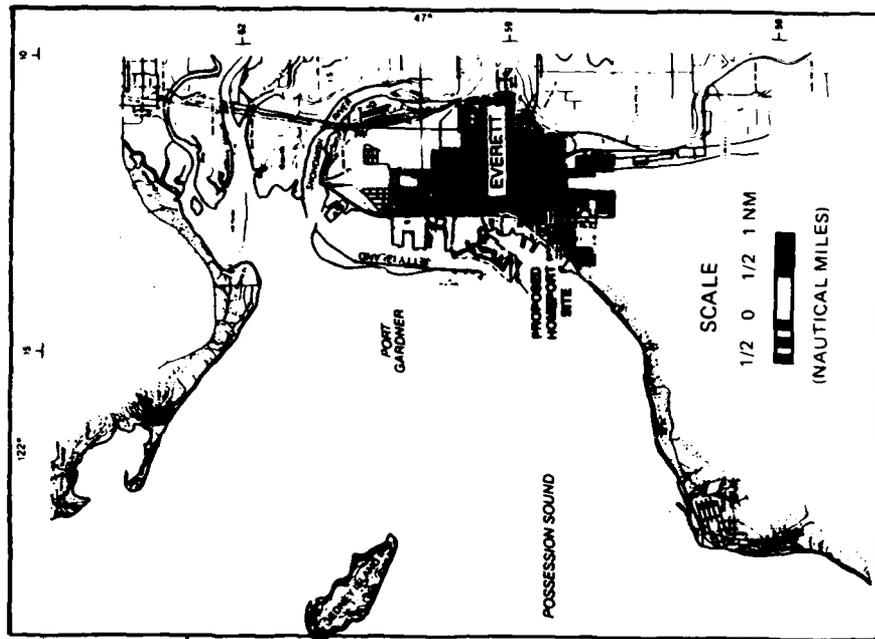
Figure 1. Location of homeport (US Navy 1985)

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LEGEND

SUITABLE FOR DEEP-WATER DISPOSAL

TOP 2 TO 3 FEET RESTRICTED DISPOSAL

TOP 6 TO 7 FEET RESTRICTED DISPOSAL

ALL EXCAVATION RESTRICTED DISPOSAL

b. Thickness of contaminated sediments
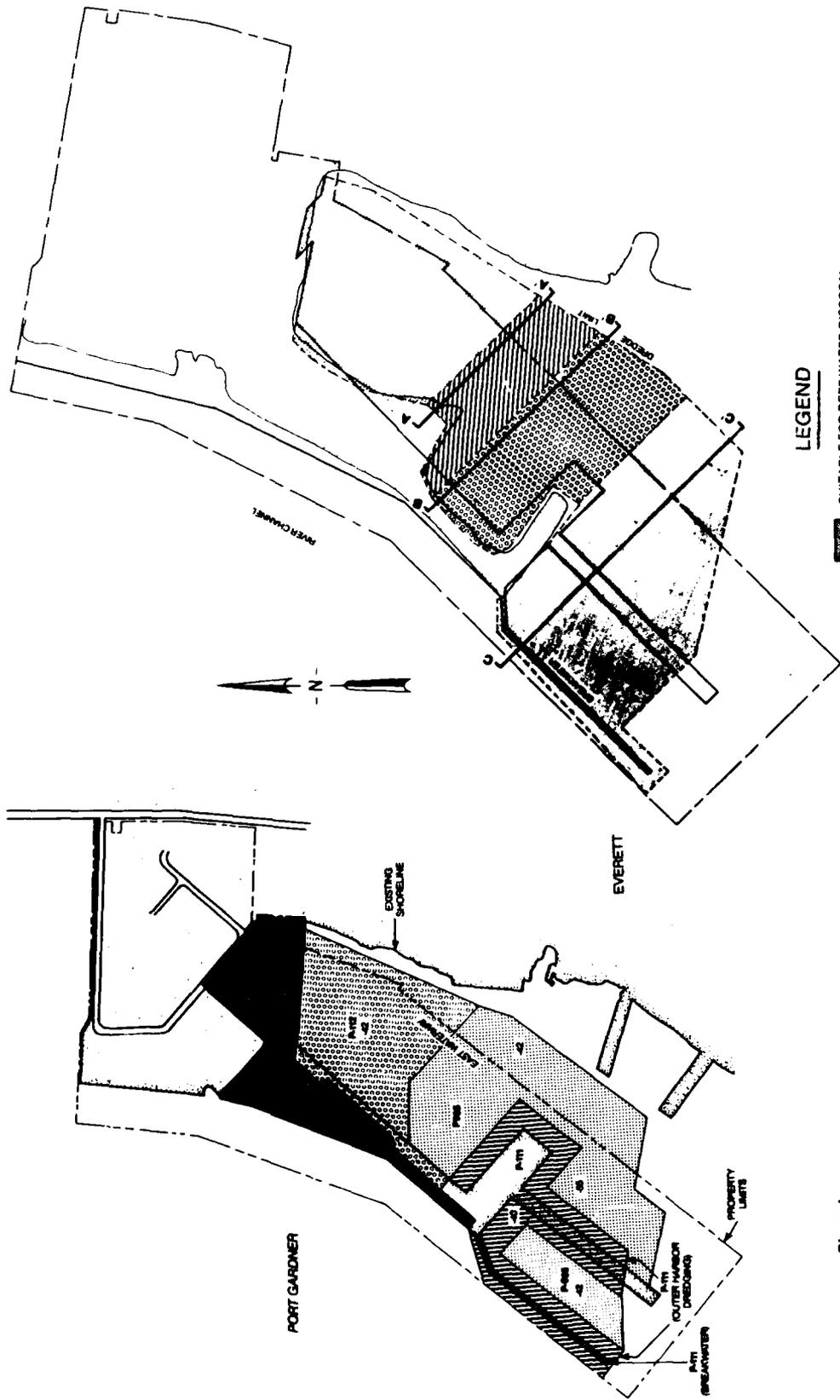
Phase I

P-111   Outer Harbor Dredging,
        Breakwater, and Mole

Phase II

P-112   Dredging Inner Harbor

P-905   Dredging Inner Harbor

a. Dredging depths

Figure 2.   Dredging plan (US Navy 1985)

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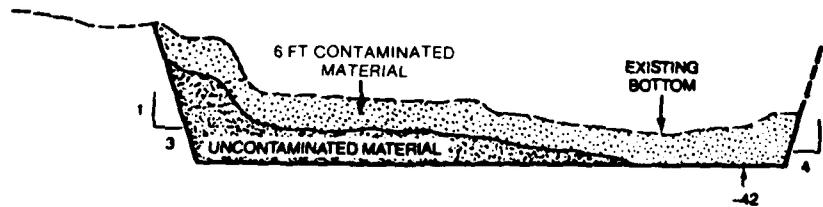
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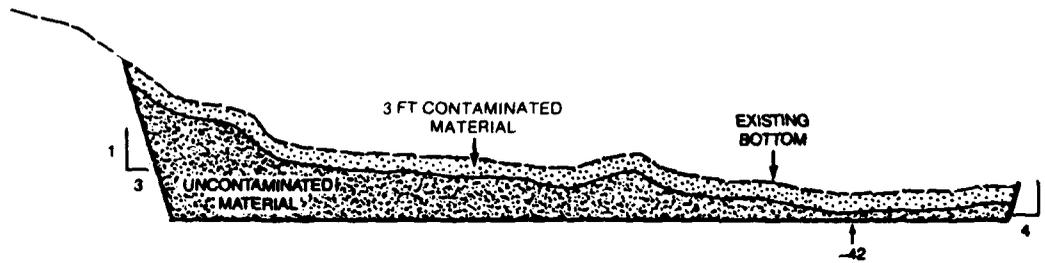
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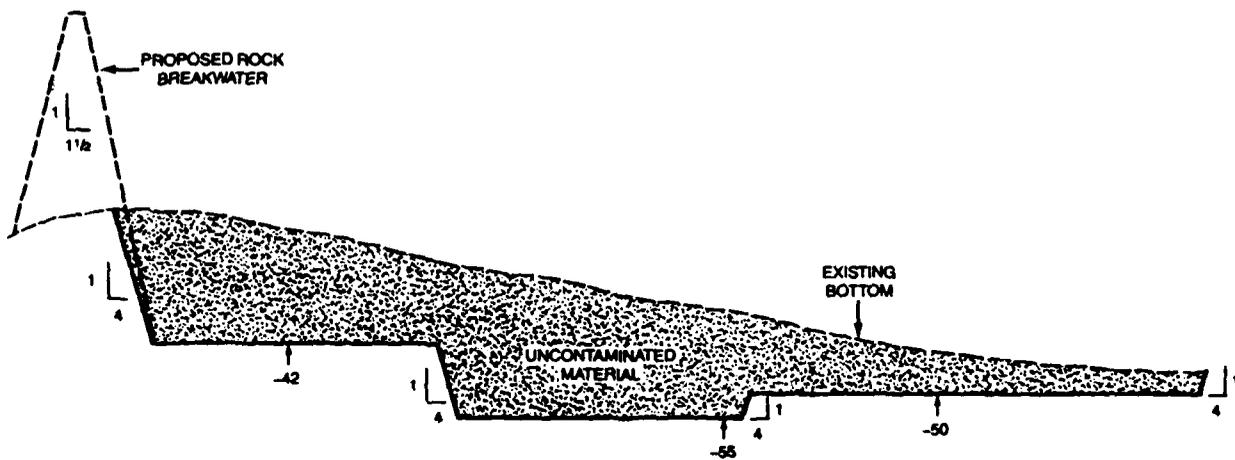
c. Section C-C'

Figure 3. Typical cross sections for dredging plan (US Navy 1985)

Technical Assistance Program

2. The Seattle District had identified the presence of contaminated sediments in the East Waterway during General Investigation studies for improvement of the existing Federal navigation project (Crecelius et al. 1984). Corps studies were suspended when the Navy announced its selection of Everett as the preferred homeport location. In June 1984, the Navy requested the technical assistance of the Seattle District for dredging and disposal components of the proposed homeport project.

3. The District's technical assistance program for the Navy included field sampling, chemical and biological analyses of the sediments, numeric modeling studies, and identification of problems and solutions associated with dredging and disposal designs. The program was conducted in three phases over 3 years (1984-1987) and was coordinated with key Federal and State agencies. The heart of the District's technical assistance was a contaminated sediments assessment program based upon a management strategy (Francingues et al. 1985) developed by the US Army Engineer Waterways Experiment Station (WES) and a decisionmaking framework (Peddicord et al., in preparation) developed by WES and the District as part of Commencement Bay Superfund remedial investigations. The management strategy describes a logical sequence for testing and evaluation of alternatives for disposal of dredged material. The decisionmaking framework provides a basis for comparison of test results with standards, criteria, or reference information to determine if contaminant control measures are needed. Accomplishments of each program phase are described below.

4. Phase I, initiated in June 1984 and completed in February 1985, characterized sediment and soils contamination (described later), estimated preliminary volumes of surface organic and underlying native sediments, and defined studies needed to predict environmental impacts and design requirements (US Army Engineer District (USAED), Seattle 1984). Chemical contamination in the upper, organic layer was found to exceed interim open-water disposal criteria for the Four Mile Rock site located in Seattle's Elliott Bay, the only criteria that existed at the time.

5. In Phase II, initiated in February 1985 and completed in May 1985, the District conducted biological tests on the native sediment and characterized aquatic and nearshore areas in Port Gardner that had been selected by

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the Navy as potential disposal areas (USAED, Seattle 1985). The report noted attenuation of contaminants in the upper layer of native material and recommended that this material be removed as overdepth dredging with overlying contaminated sediments. The remainder of the native material was judged to be acceptable for unconfined open-water disposal under then-current criteria. A nearshore site located along the Snohomish River Channel and an aquatic site, called the Deep Delta site, were found to exhibit the greatest potential for confined disposal of East Waterway sediments.

6. Phase III was initiated in May 1985 under management of the Seattle District. This phase was the most comprehensive, involving complex and detailed tests and studies. The Seattle District requested that WES provide support for the testing and evaluations required for its technical assistance role. The technical approaches used were developed cooperatively by the District and WES and have been proposed for application to other projects within the District that involve contaminated sediments. The following tabulation outlines the overall Phase III program. Detailed testing of East Waterway sediment was performed by the WES Environmental Laboratory. Additional chemical and biological tests of East Waterway sediments were performed by Battelle-Pacific Northwest Laboratory (PNL), which had conducted chemical and biological work during Phases I and II (Crecelius and Anderson 1986). Numerical dump model studies were performed by the WES Hydraulics Laboratory to evaluate aquatic disposal operations and options. Deepwater trawls of Port Gardner, to identify presence and numbers of important aquatic resources, were performed by the University of Washington School of Fisheries. Data results and interpretations were provided by the District to the US Navy as consolidated reports (USAED, Seattle 1986a, 1986b, 1986d) at specified milestones and formed the technical basis for project design by the Navy and its contractors. During Phase III, the program was constrained by the Navy design and anticipated construction schedule.

Activity	Responsibility
Program Management	Seattle District
Contaminated Sediment Testing	WES Environmental Laboratory
Sediment and water chemistry	

(Continued)

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Activity	Responsibility
Water quality tests	WES Environmental Laboratory

Standard elutriate test
Modified elutriate test
Surface runoff test
Leachate test

Engineering tests

Settling/sedimentation
Chemical clarification
Consolidation
Stabilization/liner evaluation

Chemical/Biological Investigations	Battelle-PNL

Chemistry cleanup
Bioassay/bioaccumulation
Sea surface microlayer

Numerical Dump Model Studies	WES Hydraulics Laboratory

Model verification
Material testing
Barge dump runs (contaminated sediment)
Vertical pipe runs (contaminated sediment)
Capping runs

Disposal Site Investigations	Univ. Washington, Fisheries

Deepwater trawls

Alternatives Evaluation	Seattle District and WES Environmental Laboratory

Dredge equipment/plan evaluation
Contained aquatic disposal
Confined nearshore/upland

Monitoring Plan Design	WES Environmental Laboratory and Seattle District

7. The Navy had prepared a Final Environmental Impact Statement (FEIS) for the project in 1985 (US Navy 1985). In addition to its technical assistance role, the Seattle District is the permitting agency under Section 10 of the River and Harbor Act of 1899 and Section 404 of the Clean Water Act. A Supplemental Environmental Impact Statement (EISS) was therefore required to provide information on which to base the permitting decision. Information produced under the technical assistance program provided key input to preparation of the EISS (USAED, Seattle 1986c, 1986e).

Disposal Alternatives

Available alternatives

8. Three major disposal alternatives were evaluated for disposal of contaminated Everett Harbor sediment: upland, nearshore, and contained aquatic. For purposes of this report, these alternatives have definitions as follows. Confined upland disposal is the placement of contaminated sediments in a diked upland site and capping with clean sediment or some other material. The same dredge and disposal methods could possibly be used for confined upland disposal as with confined nearshore disposal, i.e., hydraulic pipeline dredge or clamshell bucket dredge and barge. Confined nearshore disposal (also referred to as intertidal disposal) is the placement of contaminated sediments in an intertidal and/or shallow subtidal area and then capping with clean sediment. Dikes or berms are usually required to contain the disposed dredged material. Dredge and disposal methods can be either hydraulic pipeline or clamshell bucket and barge with some provisions for rehandling. Contained aquatic disposal (CAD) is the disposal of contaminated sediments at an open-water site with or without lateral confinement and then capping with clean sediment. This method has typically been applied to contaminated sediment dredged mechanically and then placed by bottom-dump barge for disposal.

Preferred alternative

9. Contained aquatic disposal was selected as a preferred alternative by the Navy. A potential CAD site was identified, and detailed data were collected at this site. Two nearshore sites were tentatively identified as alternatives, and limited site-specific data on these sites have been collected. Identification of an upland site occurred very late in the evaluation process. Site-specific evaluations for only three sites are included in this report--for the CAD site and the nearshore sites--based on the available site data. The sites evaluated in this report were specifically identified by the Navy from a larger list of alternatives.

Modifications to preferred alternative

10. As this report was being written, the Navy's plans for disposal were evolving. A total of 800,000 cu yd of contaminated material was originally proposed for dredging. This volume was later revised to 928,000 cu yd, accounting for removal of some contaminated material below depths required for navigation, allowances for overdredging, and refined estimates of the volumes

of contaminated material required for removal. In addition to an increase in the volumes to be dredged, a sequence for dredging related to various stages of construction has been proposed. Dredging related to breakwater construction would take place during an initial phase. Quantities proposed for this phase would include 97,000 cu yd of contaminated material and 739,000 cu yd of uncontaminated material. The remainder of the dredging would take place in a second phase. Quantities proposed for this phase would include 831,000 cu yd of contaminated material and 1,638,000 cu yd of uncontaminated material.

11. The Navy had identified CAD as the preferred alternative in its initial design efforts and permit application. A long downpipe leading from a disposal barge to near bottom was proposed for dredged material placement, and a subaqueous dike or berm was proposed for lateral confinement. Collection of additional detailed site information, evaluations performed by the Corps in its technical assistance role and by the Navy's design contractors, and the potential expansion of CAD site dimensions resulted in several modifications of the CAD alternative for the final project design.* The Navy's final proposed design included surface release of contaminated material using bottom dump barges and hydraulic pipeline discharge of clean material at or near the surface for cap placement. The use of a downpipe was eliminated. The Navy also eliminated the subaqueous berm but included provisions for construction of a subaqueous mound of clean material for lateral confinement as an added measure of conservatism. This modification required that the site dimensions be expanded to allow placement of the mound without total lateral confinement. The alternative as proposed in the final design is similar to conventional capping operations successfully demonstrated at other locations, although the proposed disposal site is in much deeper water. The location of the CAD site was also shifted three times during the course of this study to avoid environmentally sensitive resources. Such a shift in the CAD site location was not evaluated in detail in this report. Although this is a simplified project and parallels more closely existing experience, the CAD concept should not be thought of as merely a more elaborate version of conventional open-water "dumping." A CAD site is an engineered structure, and its successful performance depends on proper design and care during construction.

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* ABAM Engineers, Inc. 1986 (14 May). "Alternative Dredging and Disposal Methods," A-E Contract N62474-85-C-5366, Federal Way, Wash.

Purpose and Scope

12. The purpose of this report is to evaluate dredging and disposal alternatives for the Everett Homeport project. The evaluations are based on the results of sediment testing and open-water disposal modeling. The testing and modeling were designed to obtain the required technical data regarding the behavior of dredged material in the various disposal environments from environmental and engineering standpoints. Generic requirements for upland, near-shore, and contained aquatic alternatives are described. Feasibility determinations for nearshore and contained aquatic sites are given based on available site data provided by the Navy. This report is concerned only with the environmental and related engineering aspects of the project, and does not consider economic or other technical aspects. The information presented is based on studies conducted through September 1986 and is not intended to reflect subsequent changes in the project. The main body of this report contains descriptions of the testing and modeling results and evaluations of the alternatives. Detailed descriptions of the testing procedures and results are contained in the appendixes.

Sequencing of WES Reports

13. WES prepared three reports for the Seattle District during the course of this study. A report entitled "Dredged Material Disposal Design Requirements for US Navy Homeport at Everett, Washington" (Palermo et al. 1986a) was submitted to the Seattle District in March 1986. For simplicity, that report is referred to herein as the "Design Requirements" report. The Design Requirements report provided data on the environmentally related design requirements for the alternatives under consideration. WES prepared a second report entitled "Evaluation of Dredged Material Disposal Alternatives for US Navy Homeport at Everett, Washington" (Palermo et al. 1986b), which was submitted to the Seattle District in June 1986. That report is referred to herein as the "Disposal Alternatives" report. The Disposal Alternatives report provided site-specific evaluations of selected alternatives and provided data to support the EISS prepared by the Seattle District to support the Navy's permit application. A technical supplement to the Disposal Alternatives report (Palermo et al. 1986c) was submitted to the Seattle District in

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September 1986. That report is referred to herein as the "Technical Supplement" report. The Technical Supplement report completed the technical information provided by WES.

14. The Design Requirements, Disposal Alternatives, and Technical Supplement reports were prepared concurrently with ongoing design efforts conducted by the Navy's design contractors and with permit evaluations conducted by the Seattle District Regulatory Branch. These three reports were therefore based on the results of the WES studies at the time of their preparation. This technical report is an expanded compilation of the contents of the previous reports. Additional discussions or explanations have been added to provide links or transitions for material presented in the previous reports.

Strategy for Evaluation of Alternatives

15. The WES developed a Management Strategy for disposal of dredged material (Francingues et al. 1985) that describes a logical sequence for testing and evaluation of alternatives for disposal. A Decisionmaking Framework (Peddicord et al., in preparation) was developed for the Seattle District for application of the Management Strategy to other projects within the District. The Decisionmaking Framework provides a basis for comparison of test results with standards (or criteria) or reference information to determine if contaminant control measures are required in a given instance. These two documents serve as a basis for the testing and decisionmaking described in this report. For purposes of simplicity, they are herein referred to as the Management Strategy and the Decisionmaking Framework. The technical approach contained in these documents has been adopted as official Corps policy for studies involving disposal of contaminated sediments.*

16. The chemistry of contaminants in sediments, and thus their mobility and potential to adversely impact the environment, is controlled primarily by the physicochemical conditions under which the sediment exists. Fine-grained sediments that are saturated with water typically are anoxic, chemically reduced, and near neutral in pH. These conditions exist in sediments placed in mounds that form at typical nondispersive, open-water, aquatic dredged

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* BG P. J. Kelly. 1985 (17 Dec). "Policy Guidance Regarding Management and Disposal of Contaminated Dredged Material," Water Resources Support Center, Fort Belvoir, Va.

material disposal sites, and may exist in sediments used for marsh creation or nondispersively disposed in shallow water along shorelines. In this document, the term "aquatic disposal" is used in a general sense to refer to all disposal conditions in which fine-grained material remains water saturated, anoxic, reduced, and near neutral in pH. In contrast, when a fine-grained sediment is taken out of the water and allowed to dry, it becomes oxic and the pH may drop considerably. In this document, all disposal options in which a fine-grained sediment has these characteristics are referred to generally as "upland disposal," even though such conditions can occur on the surface of dredged material islands, the above-tide portions of fills, etc. Nearshore confined disposal sites could have a combination of anoxic, reduced conditions below tide elevation and oxic conditions in the dredged material placed above the tidal range (Peddicord et al., in preparation).

17. The Decisionmaking Framework contains test protocols to determine the potential release of contaminants from sediments in upland, intertidal, or aquatic disposal environments. Previous studies (USAED, Seattle 1984, 1985) have determined that the upper layer of Everett Harbor sediments was unsuitable for disposal in the open-water aquatic environment without control measures to isolate the material from sensitive aquatic resources. Capping, contained aquatic disposal, and confined disposal in an upland or nearshore site were therefore identified as potential disposal alternatives. The testing and data analyses in this report were limited to those necessary for evaluation of the available disposal alternatives. A schemetic illustrating the Management Strategy and the evaluations conducted for this study is shown in Figure 4.

Disposal Site Identification

18. Several potential sites had been identified by the Navy in its FEIS. These sites are shown in Figure 5. Information available during this study for these sites varied from cursory to more detailed. Although other sites underwent preliminary evaluation by the District, only those sites identified by the Navy were evaluated in this study. The following brief descriptions are adapted from the FEIS.
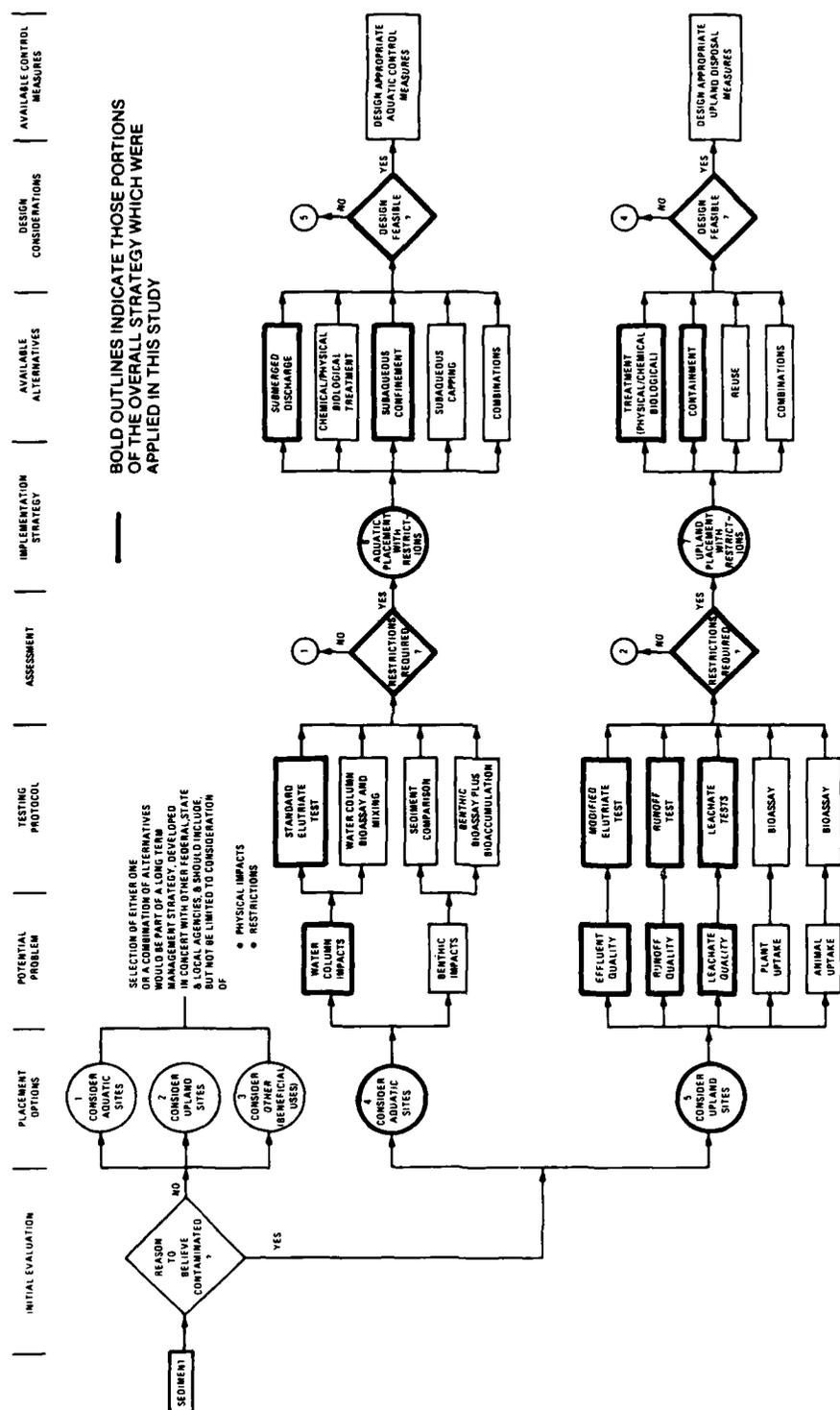
Figure 4. Management strategy flowchart
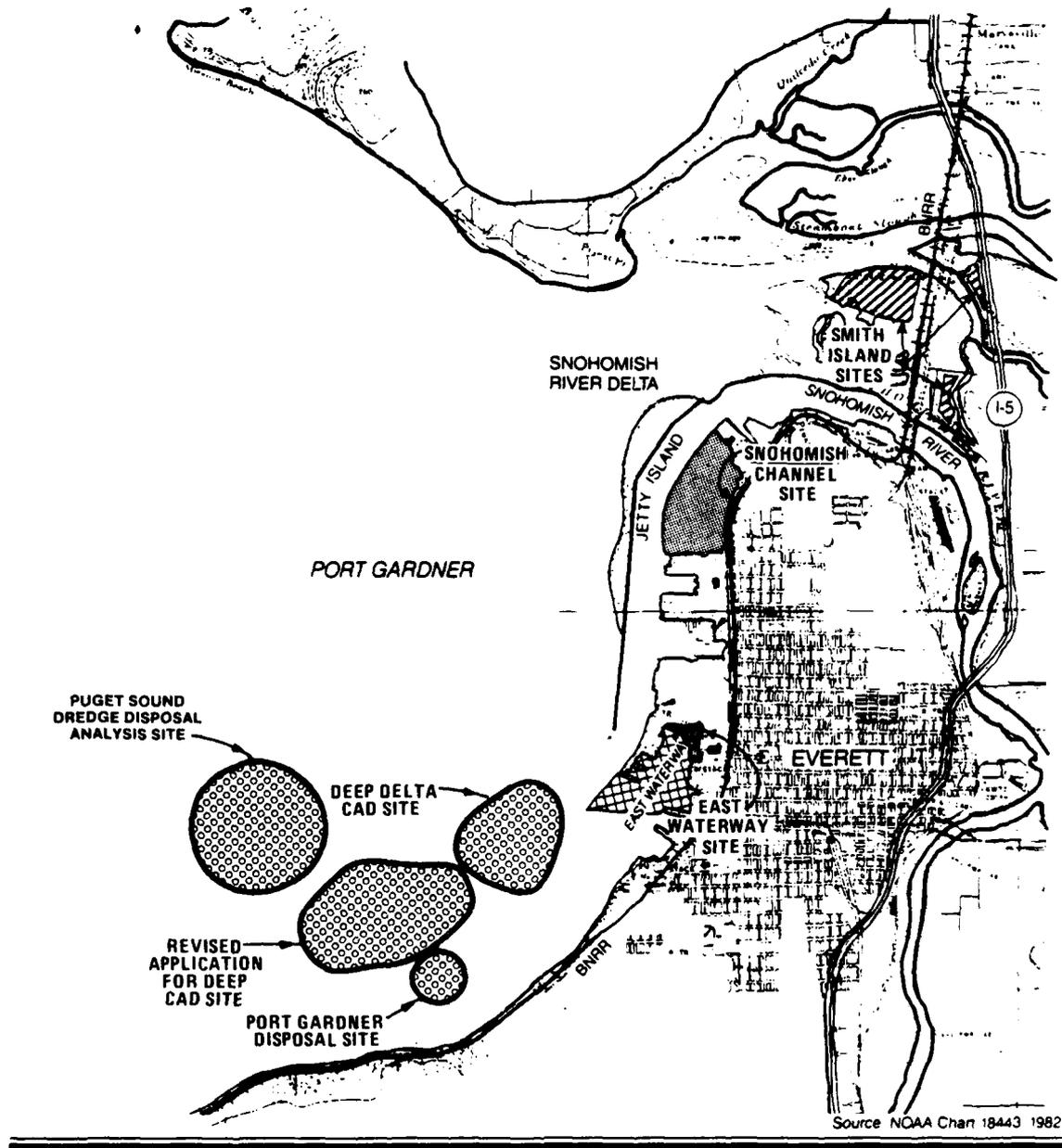
Figure 5. Location of known alternative disposal sites

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Deep Delta CAD site

19. Contained aquatic disposal at the Deep Delta site has been iden-
tified as the preferred alternative in the Navy's Section 10/404 permit appli-
cation. The Deep Delta deepwater disposal site is located west of the East
Waterway and Snohomish River channels. The site is located in approximately
250 ft of water. Bottom substrate consists of silts and sandy silts, which
indicates continual deposition from the Snohomish River. Chemical analysis of
the bottom sediments at the Deep Delta site indicates that they are cleaner
than Puget Sound background levels. Invertebrate sampling in the spring of
1985 showed moderate diversity of benthic infaunal species.

Snohomish Channel nearshore site

20. The Snohomish Channel nearshore site is located north of the East
Waterway on the east bank of the Snohomish River. The site is an intertidal
area of approximately 180 acres. Substrate is silty-sand to sandy-silt with a
heavy organic layer of wood chips, bark, and other organic debris. Chemical
analysis of sediments indicates moderate contamination of high molecular
weight polynuclear aromatic hydrocarbons. Heavy metals are at or below Puget
Sound background levels.

East Waterway nearshore site

21. A small site located at the north end of the East Waterway on the
Navy property has been identified as a potential nearshore site. The site is
within the area currently planned for dredging, and its use would result in a
reduced quantity of contaminated material requiring dredging. The substrate
is organically rich sandy silt and silt that has been identified as chemically
contaminated.

Confined upland disposal sites

22. Two sites located on Smith Island were identified by the Navy as
potential sites for upland disposal late in the evaluation process. Identifi-
cation of the upland sites was not accomplished in time for any site-specific
evaluations to be included in this study.

Criteria for Selection of Controls

23. Contaminant concentrations for a reference water and water quality
criteria were specified by the Seattle District for interpretation of test
results. In addition, a performance goal for total mass release of

29

contaminants for dredging and disposal was specified by the District. The reference, criteria, and performance goal were judged by the District to be a conservative means to indicate the potential need for contaminant controls.

Water quality

24. A reference water and water quality criteria were specified by the Seattle District for evaluation of elutriate, surface runoff, and leachate tests. Test results for elutriate and surface runoff were evaluated to determine whether the reference water concentrations (Table 1) were exceeded, and if so, to compare the test results with Federal water quality criteria for the protection of saltwater aquatic life (Table 2). The reference water was specified as Port Gardner background, and was considered equal to those concentrations determined for a water sample collected during September 1985 for purposes of conducting the elutriate tests. For the leachate tests, comparison of results with the US Environmental Protection Agency (USEPA) or State of Washington drinking water standards (Table 3) was specified.

Mass release

25. A performance goal of 5 percent for total mass release of contaminants from dredging and disposal was specified by the District. In this report, the term total mass release of contaminants refers to the total mass of in situ contaminants prior to dredging that is not placed in the disposal site or does not remain in the disposal site. The total mass release is the sum of mass release due to dredging and various mechanisms associated with disposal. The performance goal for total mass release is based on administrative agreement and has no technical basis with regard to environmental impact.

26. A direct computation of mass release of contaminants was possible for effluent, surface runoff, and leachate from confined sites for assumed operating conditions. Approximations of mass contaminant release during dredging and open-water placement for the CAD alternative were made based on estimates of the mass sediment release and elutriate data. Sediment release and contaminant release in the solids fraction are not completely equivalent, but the contaminants of concern are strongly adsorbed to the sediment particles, primarily the fine-grained silt, clay, and organic fractions. However, mass sediment release estimates include release of sandy material to which chemical contaminants tend not to be strongly absorbed. These estimates therefore allow only an approximate basis of comparison for test results and disposal options.

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Table 1

Everett Harbor Site Water Chemistry

Parameter	Concentration ppm
Arsenic	<0.005
Copper	0.007
Nickel	0.007
Cadmium	0.0006
Lead	<0.001
Zinc	<0.030
Chromium	0.004
Mercury	0.0067
PCB 1016	<0.0002
PCB 1221	<0.0002
PCB 1248	<0.0002
PCB 1232	<0.0002
PCB 1254	<0.0002
PCB 1242	<0.0002
PCB 1260	<0.0002
Acenaphthylene	<0.005
Naphthalene	<0.005
Acenaphthene	<0.005
Fluorene	<0.005
Fluoranthene	<0.005
Phenanthrene	<0.005
Pyrene	<0.005
Benzo(b)fluoranthene	<0.005
Anthracene	<0.005
Chrysene	<0.005
Benzo(k)fluoranthene	<0.005
Benzo(a)pyrene	<0.005
Benzo(g,h,i)perylene	<0.005
1-methylnaphthalene	<0.005
Indeno(1,2,3-c,d)pyrene	<0.005
2-methylnaphthalene	<0.005
Dibenzo(a,h)anthracene	<0.005

Note: Concentrations of this Everett Harbor site water sample were specified by the Seattle District for use as Port Gardner background or reference.

Table 2

USEPA Water Quality Criteria for the

Protection of Aquatic Life*

| | Criterion for Protection of Aquatic Life, $\mu g/\ell$ | | | |
| | Saltwater | | Fresh Water | |
Chemical**	24-hr Avg. (Chronic)	Maximum at Any Time (Acute)	24-hr Avg. (Chronic)	Maximum at Any Time (Acute)
Aldrin	--†	1.3	--	3.0
Arsenic (total trivalent)	--	--	--	440
Cadmium[1]	4.5	59		
50 mg/ℓ CaCO$_3$			0.012	1.5
100 mg/ℓ CaCO$_3$			0.025	3.0
200 mg/ℓ CaCO$_3$			0.051	6.3
Chlordane	0.0040	0.09	0.0043	2.4
Chromium[2] (total trivalent)	--	--		
50 mg/ℓ CaCO$_3$			--	2,200
100 mg/ℓ CaCO$_3$			--	4,700
200 mg/ℓ CaCO$_3$			--	9,900
Chromium (total hexavalent)	18	1,260	0.29	21
Copper[3]	4.0	23	5.6	
50 mg/ℓ CaCO$_3$				12
100 mg/ℓ CaCO$_3$				22
200 mg/ℓ CaCO$_3$				43
Cyanide (free)	--	--	3.5	52

(Continued)

* Federal Register, Vol 45, No. 231, Friday, 28 November 1980, pp 79318-79357.

** Superscript numbers 1-7 are defined at the conclusion of the table under "Note."

† Criterion not established.

(Sheet 1 of 3)

Table 2 (Continued)

| Chemical | Criterion for Protection of Aquatic Life, μg/ℓ | | | |
| | Saltwater | | Fresh Water | |
	24-hr Avg. (Chronic)	Maximum at Any Time (Acute)	24-hr Avg. (Chronic)	Maximum at Any Time (Acute)
Dieldrin	0.0019	0.71	0.0019	2.5
DDT	0.0010	0.13	0.0010	1.1
TDE	--	--	--	--
DDE	--	--	--	--
Endosulfan	0.0087	0.034	0.056	0.22
Endrin	0.0023	0.037	0.0023	0.18
Heptachlor	0.0036	0.053	0.0038	0.52
Lindane	--	0.16	0.080	2.0
Lead[4]	25	668		
50 mg/ℓ $CaCO_3$			0.75	74
100 mg/ℓ $CaCO_3$			3.8	170
200 mg/ℓ $CaCO_3$			20	400
Mercury	0.025	3.7	0.00057	0.0017
Nickel[5]	7.1	140		
50 mg/ℓ $CaCO_3$			56	1,100
100 mg/ℓ $CaCO_3$			96	1,800
200 mg/ℓ $CaCO_3$			160	3,100
PCB (total)	0.030	0.030	0.014	0.014
Selenium inorganic selenite	54	410	35	260
Silver[6]	--	2.3		
50 mg/ℓ $CaCO_3$			--	1.2
100 mg/ℓ $CaCO_3$			--	4.1
200 mg/ℓ $CaCO_3$			--	13

(Continued)

Table 2 (Concluded)

Chemical	Criterion for Protection of Aquatic Life, $\mu g/\ell$			
	Saltwater		Fresh Water	
	24-hr Avg. (Chronic)	Maximum at Any Time (Acute)	24-hr Avg. (Chronic)	Maximum at Any Time (Acute)
Toxaphene	--	0.070	0.013	1.6
Zinc[7]	58	170	47	
50 mg/ℓ CaCO$_3$				180
100 mg/ℓ CaCO$_3$				320
200 mg/ℓ CaCO$_3$				570

Note: Criteria for some metals in fresh water are hardness-dependent and are derived from the following equations, where h is hardness in milligrams per litre as CaCO$_3$, and e is the natural logarithm base.

Metal	24-hr Avg.	Maximum at Any Time
[1]Cadmium	$e^{1.05 \ (\ln h) - 8.53}$	$e^{1.05 \ (\ln h) - 3.73}$
[2]Chromium (total trivalent)	--	$e^{1.05 \ (\ln h) + 3.48}$
[3]Copper	(main table)	$e^{0.94 \ (\ln h) - 1.23}$
[4]Lead	$e^{2.35 \ (\ln h) - 9.48}$	$e^{1.22 \ (\ln h) - 0.47}$
[5]Nickel	$e^{0.76 \ (\ln h) + 1.06}$	$e^{0.76 \ (\ln h) + 4.02}$
[6]Silver	--	$e^{1.72 \ (\ln h) - 6.52}$
[7]Zinc	$e^{0.83 \ (\ln h) + 1.95}$	(main table)

(Sheet 3 of 3)

Table 3

Contaminant Concentrations in Drinking Water Standards

Parameter, mg/ℓ (Unless Otherwise Noted)	Drinking Water Standards	
	Federal	State of Washington
Arsenic	0.05	0.05
Barium	1.0	1.0
Cadmium	0.010	0.010
Chromium	0.05	0.05
Lead	0.05	0.05
Mercury	0.002	0.002
Selenium	0.01	0.01
Silver	0.05	0.05
Fluoride	1.4-2.4	1.4-2.4
Nitrate (as N)	10.0	10.0
Endrin	0.0002	0.0002
Lindane	0.004	0.004
Methoxychlor	0.1	0.1
Toxaphene	0.005	0.005
2,4-D	0.1	0.1
2,4,5-TP Silvex	0.01	0.01
Trihalomethanes	0.1	0.1
Turbidity (JTU)	1.0	1.0
Coliform bacteria - membrane filter test (1b/100 ml)	1.0	1.0
Gross alpha (pCi/ℓ)	15.0	15.0
Combined Radium 226 and Radium 228	5.0	5.0
Beta and photon particle activity (Mrem/year)	4.0	4.0
Sodium	Monitor	250.0
Chloride	250.0	250.0
Color (units)	15.0	15.0
Copper	1.0	1.0
Corrosivity	Noncorrosive	Noncorrosive
Foaming agents	0.5	0.5
Iron	0.3	0.3
Manganese	0.05	0.05
Odor (threshold No.)	3.0	3.0
pH (units)	6.5-8.5	6.5-8.5
Sulfate	250.0	250.0
Total dissolved solids	500.0	500.0
Zinc	5.0	5.0

PART II: SEDIMENT SAMPLING AND TESTING

Sample Collection and Preparation

Sediment

27. The sediments to be dredged were extensively sampled and char-
acterized* prior to this study (USAED, Seattle 1984, 1985). The results of
these evaluations indicated two distinct layers of sediments. The upper layer
was characterized as a marine silt with significant presence of wood chips and
organic material. All sediments considered contaminated were within the upper
portions of this layer (see Figure 3). Underlying the marine silt was a layer
of sandy silt with little organic material. This layer was characterized as a
native sediment and is not contaminated. Chemical analysis of the sediments
(USAED, Seattle 1984, 1985) indicated that the contaminant concentrations were
similar throughout the volume of contaminated sediments. For this reason, the
Seattle District made a decision that one composite sediment sample from the
contaminated marine silt would be collected for WES testing. Approximately
8 cu yd of sediment was needed to perform the entire range of required tests.
The Seattle District collected representative samples of the contaminated
sediment and the native clean sediments and shipped the samples to WES.

28. Sediment sample collection took place on 6 June 1985. Sediment was
collected and composited from the more contaminated portions of the waterway
using a small clamshell dredge operated from the Corps vessel *Puget*. Although
the sediment was generally sampled from the top 3 ft, efforts were made to
penetrate to 4 to 5 ft at those locations where the contaminated sediment
layer was thickest (see Figure 6). Sufficient sediment to fill from one to
four steel drums was collected from each of 14 stations in the East Waterway
roughly corresponding to previous coring stations (see Figure 6). Later the
same day, the filled drums were emptied into a hopper, and the material was
pumped into a cleaned concrete mixer. The sediment was mixed for a total of
45 min, 30 min during placement in the concrete mixer and for an additional

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* Hart Crowser and Associates, Inc. 1985 (Dec). "Geotechnical Engineering
Design Report, NAVSTA Puget Sound, Everett, Washington," Contract
N62474-85-C-5233, Seattle, Wash.
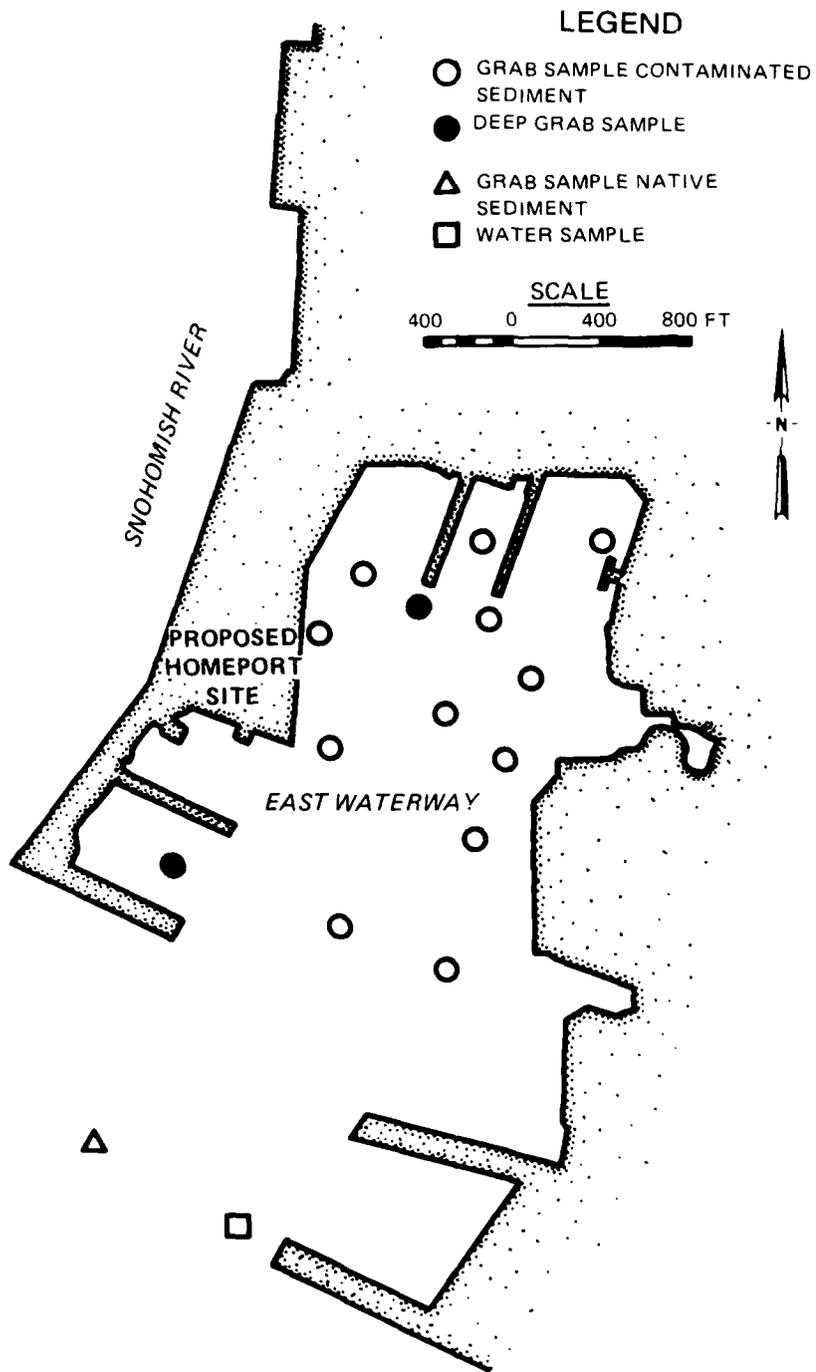
Figure 6. Sampling locations for WES composite sample

15 min to complete the mixing. No water was added to the sediment. The homogenized sample was then poured into 29 cleaned drums and carefully sealed. Six barrels of a clean or native sediment considered appropriate for use as a capping material were also collected at the site but not homogenized (see Figure 6). All 35 barrels were placed into a refrigerated truck and transported to WES. On arrival, the samples were stored in a cold storage facility until required for a particular test.

Water

29. Samples of Port Gardner water were collected off the Port dock in the outer waterway on 27 September 1985 for use in the elutriate testing (see Figure 6). These samples also served as reference background water quality samples. The samples were collected from near bottom at depths of 25 to 35 ft so as to simulate the water that would be entrained with the sediment during dredging operations. This water was considered representative of the Deep Delta disposal site as well. The samples were immediately placed in a cooler and transported to WES for testing. On arrival, the samples were stored in a cold storage facility until required for a particular test.

Sediment and Water Characterization

Physical

30. Physical and engineering characterization of the homogenized composite sample sediment was conducted by the WES Geotechnical Laboratory. The characterization consisted of natural water content, specific gravity, Atterberg limits, grain size distribution, and Unified Soil Classification. The contaminated sediment composite was a black, sandy, organic silt (OH) with a high percentage of wood chips. Properties of the contaminated marine silt and native sediment* are compared with the WES composite in the following tabulation. Note that water content refers to the geotechnical engineering term and is the ratio of the weight of water in a sample to the dry weight of solids, expressed as a percentage. Grain size distribution ranges for the marine silt (both contaminated and uncontaminated) and native sediments are shown in

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* Hart Crowser and Associates, Inc. 1986 (Mar). "Detailed Aquatic Site Study, Confined Aquatic Disposal Site, NAVSTA Puget Sound, Everett, Washington," Contract N62474-85-C-5366, Seattle, Wash.

Figure 7. Note the influence of wood chips on the grain size distribution of the marine silt.

Parameter	WES Composite Contaminated Sediment	Contaminated Marine Silt	Uncontaminated Native Sediment
Water content, percent	157	130-375	27-56
Specific gravity	2.44	2.38-2.39	2.67-2.72
Percent passing #200 sieve	88	40-95	30-100
Liquid limit, percent	116	62-134	28-61
Plasticity index, percent	59	26-79	3-25

In situ channel water content

31. The in situ channel water content is an important parameter in design and evaluation of disposal alternatives. A clamshell dredge can effectively remove sediment from the bottom at near the in situ water content. Hydraulic dredges can remove sediment at varying densities up to a maximum limited by dredging depth, pipeline length, etc., and a large volume of water will be added. Regardless of the dredging method, the material will undergo an additional change in water content (and therefore in density) during the disposal process. When material is finally placed in either a CAD site or confined intertidal or upland site, it may occupy a volume that is significantly different than that occupied in the channel. The water content of the disposed material will be dependent on the dredging method, disposal process, and site characteristics.

32. The measured in situ water content of the WES composite sample is within the range of other water content data reported as a part of the Everett investigations conducted by the Navy's contractors.* An average water content of 208 percent was determined from 14 core samples taken from the upper 3 ft of East Waterway sediment. This value is considered representative of the in situ contaminated sediment. A value of 250 percent was determined from one station from the upper 1 ft of sediment. This value is considered representative of the highly organic surface layer of material. An average water content of 73 percent was determined from 20 core samples taken in the upper

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* Hart Crowser and Associates 1986, op. cit.
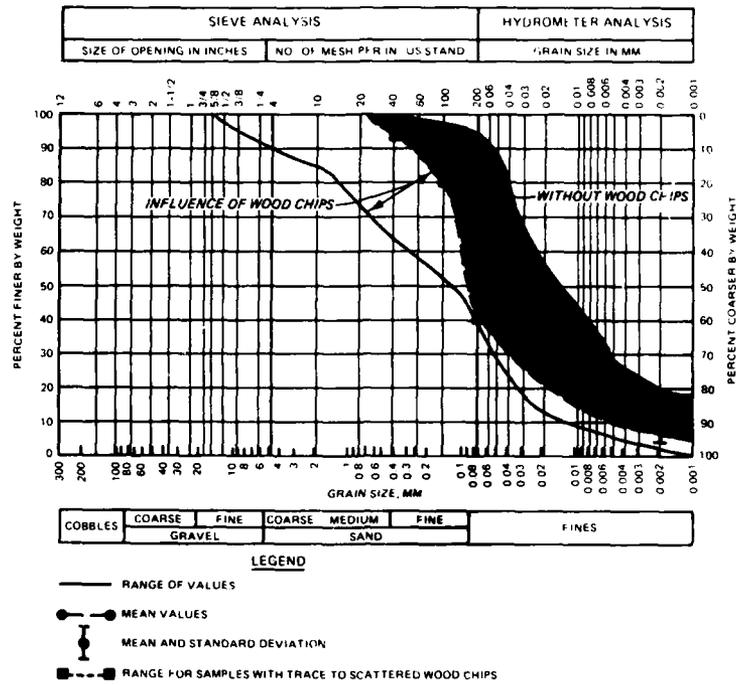
a. Marine silt (both contaminated
and uncontaminated)
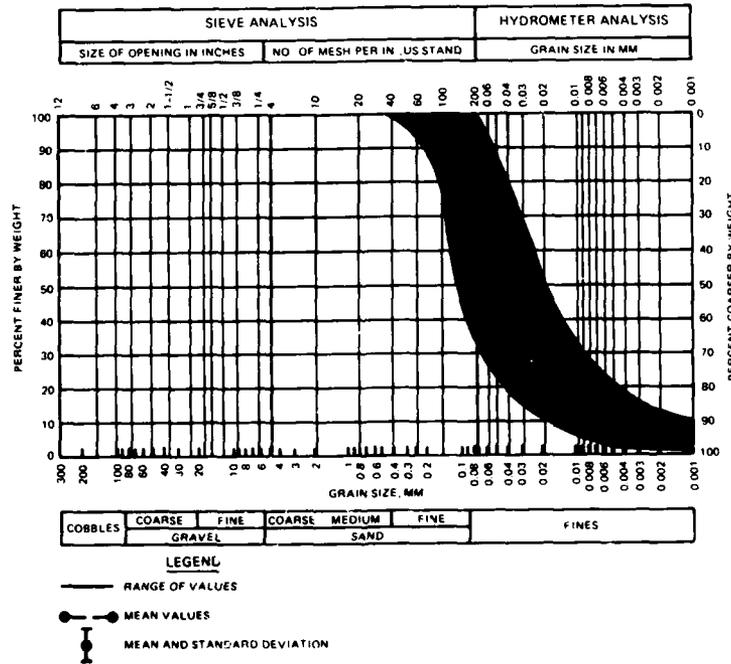
b. Native sediments

Figure 7. Grain size distribution ranges for sediments

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layers of native sediments. This value is considered representative of the native sediment that must be dredged along with the in situ contaminated sediment and treated as contaminated for purposes of disposal. A volumetrically weighted average of all core samples shows that the water content of the material that must be dredged and treated as contaminated is 131 percent. This weighted average was only slightly lower than the WES composite sample value of 157 percent.

33. A water content value of 250 percent was used in the open-water disposal modeling described in Part III of this report. This high value is considered the most conservative, i.e., would result in a higher estimate for dispersion and bottom spread of material placed in open water. An in-channel water content of 157 percent was used in the estimates of volume initially occupied by material placed in confined intertidal disposal sites. This value is considered representative of the total volume of material to be dredged as contaminated material. Since the design of disposal sites is directly dependent on representative in situ water contents, a more precise determination of the variation in in situ conditions is needed for the final design. This determination should consider the final dredging sequence, equipment, and disposal method.

Chemical

34. The Everett composite sample was analyzed for bulk concentration of priority pollutants. Results are summarized in Table 4. These results correlated well with independent analyses performed by Battelle Marine Research Laboratory and served to confirm that the composited sample was representative of the more contaminated areas in East Waterway identified by previous studies (USAED, Seattle 1984). Most compounds were at or below detection limits. It should be noted that different analytical methods were used in the Battelle analysis, and the Battelle results were consistently higher than the values in Table 4.

35. Based on the results of this analysis, a list of selected representative parameters or specific compounds of concern was developed for the study in consultation with the Seattle District. The parameters of concern were: chromium (Cr), nickel (Ni), copper (Cu), zinc (Zn), arsenic (As), lead (Pb), cadmium (Cd), mercury (Hg), polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and 1- and 2-methylnaphthalene.

Table 4

Bulk Sediment Analyses of Composite Sediment Sample

Parameter	Concentration mg/kg	Parameter	Concentration mg/kg	Parameter	Concentration mg/kg
Arsenic	5.73	4-Nitrophenol	<1.	1 4-Dichlorobenzene	<1.
Copper	73.4	Chloromethane	<0.025	1 2 4-Trichlorobenzene	<1.
Nickel	21.4	Chloroethane	<0.025	Hexachlorocyclopentadiene	<1.
Cadmium	3.30	2-Methyl-4 6-dinitrophenol	<10.	1 2-Dichlorobenzene	<1.
Lead	48.1	Bromomethane	<0.025	Naphthalene	8.2
Zinc	148.5	Methylene Chloride	<0.025	2-Chloronaphthalene	<1.
Chromium	39.7	Pentachlorophenol	<1.	Hexachloroethane	<1.
Mercury	0.201	Vinyl Chloride	<0.025	Hexachlorobutadiene	<1.
Iron	7864.	1 1-Dichloroethene	<0.025	Acenaphthylene	<1.
Aldrin	<0.0002	1 1-Dichloroethane	<0.025	Dimethyl Phthalate	<1.
Manganese	237.	1 2-Dichloroethane	<0.025	Diethyl Phthalate	<1.
G-BHC	<0.0002	Bromodichloromethane	<0.025	4-Bromophenyl Ether	<1.
Total Phosphorus	789.	Trans-1 2-Dichloroethene	<0.025	Acenaphthene	2.0
A-BHC	<0.0002	1 1 1-Trichloroethane	<0.025	4-Chlorophenyl Phenyl Ether	<1.
D-BHC	<0.0002	1 2-Dichloropropane	<0.025	Hexachlorobenzene	<1.
Ammonia Nitrogen	167.	Chloroform	<0.025	Fluorene	2.2
B-BHC	<0.0002	Carbon Tetrachloride	<0.025	N-Nitrosodiphenyl Amine	<1.
Chlordane	<0.002	Trans-1 3-Dichloropropene	<0.025	Phenanthrene	5.7
PPDDD	<0.0002	Trichloroethene	<0.025	Anthracene	1.5
Dieldrin	<0.0002	1 1 2-Trichloroethane	<0.025	Pyrene	4.0
Endosulfan Sulfate	<0.0002	Bromoform	<0.025	Benzo(A)Anthracene	2.1
PPDDE	<0.0002	Dibromochloromethane	<0.025	Dibutylphthalate	<1.
A-Endosulfan	<0.0002	Benzene	<0.025	Butylbenzylphthalate	<1.
Endrin	<0.0002	1 1 2 2-Tetrachloroethane	<0.025	Bis(2OEthylhexyl)Phthalate	<1.
PPDDT	<0.0002	Cis-1 3-Dichloropropene	<0.025	Fluoranthene	4.5
B-Endosulfan	<0.0002	2-Chloroethylvinylether	<0.025	Chrysene	1.8
Endrin Aldehyde	<0.0002	Tetrachloroethene	<0.025	Di-N-Octylphthalate	<1.
Heptachlor	<0.0002	Toluene	<0.025	Benzo(B)Fluoranthene	2.5
Heptachlor Epoxide	<0.0002	Acrolein	<0.25	Indeno(1 2 3-C D)Pyrene	<1.
PCB 1221	<0.002	Bis(2-Chloroisopropyl)ether	<1.	Total Organic Carbon	71,540
PCB 1248	<0.002	Chlorobenzene	<0.025	Benzo(K)Fluoranthene	2.5
PCB 1232	<0.002	Acrylonitrile	<0.25	Dibenzo(A H)Anthracene	<1.
PCB 1254	0.25	N-Nitroso-di-n-propylamine	<1.	Benzo(A)Pyrene	1.4
PCB 1016	<0.002	Ethylbenzene	<0.025	Benzo(G H I)Perylene	<1.
PCB 1242	<0.002	N-Nitrosodimethylamine	<1.	Total Phenol	17.1
PCB 1260	<0.002	Nitrobenzene	<1.		
Toxaphene	<0.002	Isophorone	<1.		
2-Nitrophenol	<1.	2 4-Dinitrotoluene	<1.		
4-Chloro-3-Methylphenol	<1.	3 3-Dichlorobenzioine	<5.		
Phenol	<1.	Bis(2-Chloroethoxy)Methane	<1.		
2 4-Dimethylphenol	<1.	2 6-Dinitrotoluene	<1.		
2 4 6-Trichlorophenol	<1.	Benzidine	<10.		
2-Chlorophenol	<1.	1 3-Dichlorobenzene	<1.		
2 3 Dichlorophenol	<1.				
2 4 Dinitrophenol	<10.				

36. The Port Gardner water sample was also analyzed for the parameters of concern. Results are shown in Table 1. All parameters were below detection in the site water sample except for Cu, Ni, Cd, Cr, and Hg. The site water concentrations equaled or exceeded the Federal water quality criteria for the protection of saltwater aquatic life for Cu, Ni, and Hg.

Standard Elutriate Tests

Procedures

37. The standard elutriate test is used to estimate dissolved contaminant release in dredge hoppers or pipelines and does not consider subsequent mixing and dilution during the disposal process. Standard elutriate tests were conducted using the composite sediment and dredging site water samples. These tests were used to estimate the degree of dissolved contaminant release due to placement of the sediments in open-water sites for the CAD alternative. Additional standard elutriate tests (total concentration) were conducted to gain qualitative data on total concentrations which might aid in evaluating the particle-associated contaminant release (mass release). However, this change in the procedure has not been laboratory developed or field verified. Procedures and results are discussed in detail in Appendix A.

Results

38. Standard elutriate test results (dissolved concentrations) were compared with the background water concentrations and water quality criteria in accordance with the Decisionmaking Framework (Peddicord et al., in preparation). The tests results are an estimate of the dissolved release of contaminants during placement of dredged material in open water for the CAD alternative. Nickel, Cd, Pb, Cr, and PCB 1254 exceeded the background concentrations. Both Cd and Cr concentrations were below the chronic and acute exposure values given in the Federal water quality criteria. The only remaining parameters of concern are Ni, Pb, and PCB 1254.

39. Ni concentration in the elutriate was above the chronic exposure value given in the Federal water quality criteria but was far below the acute exposure value. It should be noted that the Port Gardner background concentration equals the chronic criteria for Ni. The standard elutriate

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concentration cannot be diluted to the chronic criteria by mixing.* However, the elutriate concentration is well below the acute criteria.

40. Lead concentrations slightly exceeded the chronic exposure values given in the Federal water quality criteria. A dilution factor of less than one was calculated using procedures in the Decisionmaking Framework that would dilute the standard elutriate value to the chronic exposure value. Size and configuration of the mixing zone would depend on site-specific information not yet available. However, such a minimal mixing and dilution could easily be achieved within a short distance of the open-water disposal operation.

41. The PCB 1254 concentrations exceeded both the chronic and acute exposure values given in the Federal water quality criteria. A dilution factor of 13 was calculated using procedures in the Decisionmaking Framework that would dilute the standard elutriate value to the chronic exposure value. Size and configuration of the mixing zone would depend on site-specific information not yet available. However such a minimal mixing and dilution could easily be achieved within a short distance of the open-water disposal operation.

42. Only 7 of 33 contaminants of concern were detected in the standard elutriate tests. Only five parameters exceeded Port Gardner background concentrations and only three parameters, Ni, Pb, and PCB 1254, exceeded the Federal water quality criteria. These parameters were all of low concentration, and dilution to background concentrations or criteria could easily be accomplished within a short distance of the disposal operation. Based on these data, there appears to be no need for controls from the standpoint of contaminant release in the dissolved form during placement of the sediments for the CAD alternative.

43. The concentrations of contaminants detected in all the elutriate samples were low. Analytical variability for such low concentrations can mask the differences in dissolved and total concentrations that would normally be expected. Total standard elutriate concentrations were equal or lower that filtered concentrations for Cu, Cd, Cr, Hg, and PCB 1254. The total results were elevated in comparison to dissolved results for nickel and lead. Total standard elutriate concentrations were not used in calculating mass release.

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* Calculation of mixing zone dimensions was performed by the Washington Department of Ecology.

Modified Elutriate Tests

Procedures

44. Modified elutriate tests were conducted to predict the quality of water discharged as effluent during active disposal operations. These tests define the dissolved and particle-associated concentration of contaminants in the effluent and account for the settling behavior of the dredged material, retention time of the containment area, and chemical environment in ponded water during active disposal. Detailed procedures and results are presented in Appendix A.

Results

45. Modified elutriate test results (dissolved) were compared with the background water concentrations and water quality criteria in accordance with the Decisionmaking Framework. Comparison of elutriate data to Port Gardner background is only appropriate where return water is to the marine system (i.e., East Waterway site). Although the Snohomish Channel site is tidally influenced, the regime is primarily freshwater and would be best compared to Snohomish River background water. No such sample was collected for this study. The dissolved test results are an estimate of the dissolved concentrations of contaminants that can be expected in the effluent discharged from a confined disposal site. Only Ni and PCB 1254 exceeded the background concentrations. Ni exceeded the chronic exposure value but was below the acute exposure value given in the water quality criteria. PCB 1254 exceeded both the chronic and acute exposure values.

46. It must be noted that the Port Gardner background concentration equals the chronic criteria for Ni. The modified elutriate concentration cannot be diluted to the chronic criteria by mixing. However, the elutriate concentration is well below the acute criteria.

47. The PCB 1254 concentrations exceeded both the chronic and acute exposure values given in the Federal water quality criteria. A dilution factor of 13 was calculated using procedures in the Decisionmaking Framework that would dilute the modified elutriate value to the chronic exposure value. Size and configuration of the mixing zone would depend on site-specific information not yet available. However, such a minimal mixing and dilution could easily be achieved within a short distance of the effluent discharge.

48. Only 5 of 33 contaminants of concern were detected in the dissolved fraction in the modified elutriate tests. Only two parameters, Ni and PCB 1254, exceeded Port Gardner background concentrations and the Federal water quality criteria. These parameters were of low concentration, and dilution to background concentrations or criteria can easily be accomplished within a short distance of the disposal operation. Based on these data, there appears to be no need for controls for removal of dissolved contaminants from effluents discharged from confined sites during filling operations (either upland or intertidal).

49. Total unfiltered modified elutriate concentrations were equal to or lower than concentrations for the dissolved samples for Cd, Cu, and Ni. This could possibly be due to scavenging of dissolved metals by adsorption and coprecipitation with hydroxides that form on the surface of particles under oxidizing conditions. The total results were elevated in comparison to dissolved results for Cr and PCB 1254. These results were used for calculations of mass release of contaminants in the effluent.

Surface Runoff Tests

Procedures

50. The purpose of this test is to predict the water quality of rainfall-induced surface runoff from a confined upland or nearshore (above water table) dredged material disposal site. When sediment is taken from aquatic environments and placed in an upland condition, dramatic physicochemical changes can occur. As the sediment dries and oxidizes, it may become acidic when large amounts of sulfides and organic matter and small amounts of neutralizing compounds are present. This drop in pH can further result in mobilization of soluble heavy metals in surface runoff. Decisions on disposal site selection and containment control measures require information on the effects of these physicochemical changes on rainfall-induced runoff water quality. A laboratory rainfall simulator-lysimeter system was used to evaluate the potential surface runoff water quality from a confined upland dredged material disposal site prior to dredging and disposal of the material. Detailed procedures and results are presented in Appendix B.

Results

51. Surface runoff water quality problems from the East Waterway sediment during the wet, anaerobic stage will be primarily in the form of high suspended solids concentrations. This is a typical problem that occurs when dredged material is first placed in upland disposal sites and is easily controlled by allowing the suspended solids to settle out of the runoff before release from the disposal site. During this period, contaminants such as heavy metals remain tightly bound to the sediment and will be removed from the runoff as the suspended solids are removed. Concentrations of PCBs in surface runoff water were all below detectable limits, and PAH concentrations were low. Filtered runoff concentrations were significantly below the USEPA acute criteria values and slightly below the Port Gardner reference water quality values.

52. Dredged material often forms very hard-crusted surfaces with extensive cracking. Metals such as Cd, Cu, Ni, Zn, and manganese can become very soluble in surface runoff with filtered concentrations equaling unfiltered concentrations. Because of a high concentration of organic material, the East Waterway sediment reacted differently than previously tested sediments. Instead of forming hard, crusted surfaces with extensive cracking, this sediment formed a very light fluffy surface that was highly erosive. Suspended solids concentrations in the surface runoff remained very high (1,000 mg/ℓ) causing unfiltered metal concentrations to also remain high. Filtered concentrations of Cd were not significantly different from unfiltered concentrations, and Zn and Cu were also present in significant concentrations in the filtered samples. Filtered concentrations of Cd were substantially greater than the USEPA acute criteria and the Port Gardner reference site. A dilution factor of 18 was calculated to be required to dilute the runoff concentration to the criteria.

Leachate Prediction Tests

Procedures

53. When contaminated dredged material is placed in a confined near-shore or upland disposal facility, the potential exists to generate leachates having adverse impacts on ground water and surface water quality. Subsurface drainage and seepage through dikes may reach adjacent surface and ground

waters, resulting in contamination of ground water and deterioration of surface water quality.

54. At present, there is no routinely applied laboratory testing protocol capable of predicting leachate quality from confined dredged material disposal sites. Newly developed testing procedures to predict leachate quality were therefore being used to evaluate the confined disposal alternative for Everett Harbor dredged material. These leaching techniques have been used only once before; therefore, the procedures are in an early stage of development, and results have been interpreted with caution. When properly applied, these techniques should allow determination of the potential impacts of using a nearshore or upland site. This information is needed to develop cost-effective site designs.

55. Appropriate testing procedures were evaluated and applied for estimating leachate contaminant levels from Everett Harbor sediment for the nearshore and upland disposal alternatives. Laboratory leaching tests used for predicting short- and long-term leachate quality included sequential batch leaching tests and permeameter testing, a modified form of column leaching. Results from these tests were combined with a mass transport equation to provide an integrated approach for predicting contaminant concentrations from a confined site. Details of the integrated approach and its application to Everett Harbor sediment are provided in Appendix C.

Results

56. Batch testing. The intrinsic release characteristics of Everett Harbor dredged material for As, Cd, Cr, Cu, Ni, Hg, Pb, Zn, PAHs, and PCBs were determined using sequential batch leaching tests. Tests were also conducted to determine shaking time required to reach steady-state concentration values, the proper liquid-solids ratio at which to conduct batch tests, and the effects of varying salinity on metal concentrations in leachate.

57. Desorption isotherms were developed using data from the sequential batch leaching tests. The sequential batch leaching tests involved shaking sediment with successive inputs of fresh distilled-deionized water and analyzing the leachate. Procedures used in the anaerobic sequential batch leaching tests are described in Appendix C. From the desorption isotherms, the mass of contaminant leached, and where possible, the distribution coefficients, K_d , were obtained. The desorption isotherms for metals and organics fall into four distinct groups. These groups consisted of: (a) desorption isotherms

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with leachate values that were near the detection limit for the parameter, (b) desorption isotherms that produced a linear relationship between steady-state sediment and leachate concentrations, (c) desorption isotherms that showed a double-valued relationship between steady-state sediment and leachate concentrations, and (d) desorption isotherms that did not show a well-defined relationship between steady-state sediment and leachate concentrations.

58. Desorption isotherms for anaerobic metals fit into all four of these categories. Hg was not detected in any of the leachates and fell into category (a). Copper and Pb fell into category (b). Arsenic and Ni fell into category (c), and Cd, Cr, and Zn fell into category (d). For aerobic sequential leaching, Hg and As fell into category (a). Nickel and Zn fell into category (b), and the remainder of the metals fell into category (d).

59. Releases of organic contaminants from anaerobic sediment were measurable for only 8 of 33 compounds analyzed during sequential leaching. Compounds that were detected fell into category (a), as all were near the detection limit. This can be expected if the distribution coefficient is large. Distribution coefficients for organic contaminants were calculated by computing the average from all the point estimates provided by the data from the sequential batch leach tests.

60. **Permeameter testing.** Continuous flow column leaching studies were conducted in divided-flow stainless steel permeameters using anaerobic and aerobic sediment. Column effluent was analyzed for As, Cd, Cr, Pb, Zn, and the organic compounds listed in Table C2. (The specific details of permeameter loading and operation are presented in Appendix C.) Data from the anaerobic columns show concentrations of As below detection limits. Concentrations of Cd, Cr, Pb, and Zn were at or above detection limits. Metal leachate concentrations from aerobic columns were generally higher and showed greater variation than metal leachate concentrations from anaerobic columns. Leachate concentrations of PCBs from anaerobic and aerobic columns were low, and no PAHs were detected.

61. **Integrated approach.** Application of the integrated approach to anaerobic leaching of PCBs from Everett Harbor sediment showed that predicted values agreed well with observed values and that because of the high distribution coefficients for PCBs, pore water concentrations in the field can be predicted using a simple equilibrium equation. The integrated approach was not applied to the leaching of metals from anaerobic Everett harbor sediment

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because most of the metal desorption isotherms fell into categories (a), (c), and (d). Unless a metal desorption isotherm is a category (b) isotherm, the mass transfer equation developed thus far cannot be used to predict column elution curves. Therefore, an approximate method, based on equating liquid-solids ratios in batch and column tests, was developed and used to predict column leachate concentrations using batch leaching data. Using the approximate method, the general shape of column elution curves was well predicted for anaerobic leaching of As, Cd, and Zn. Less agreement was observed for Cr and Pb. Comparison of predicted to observed values was limited because of the small region of overlap between batch and permeameter data.

62. The integrated approach was not used to predict elution curves for aerobic metals. Previous work with sediment from Indiana Harbor has demonstrated that leaching conditions in aerobic batch tests and aerobic column tests are not comparable. Therefore, there is no basis for prediction. (Additional discussion is provided in Appendix C.)

63. Summary. The intrinsic contaminant release characteristics deter-mined in batch and column leaching tests for Everett Harbor sediment indicate that mobility of metals and organic contaminants is low under anaerobic conditions. Low mobility under anaerobic conditions is consistent with previous experience with other sediments. Under aerobic conditions some metals are mobilized in large quantities. The fraction of metals that was resistant to anaerobic leaching in batch tests was generally greater than 90 percent of the bulk concentration. Under aerobic conditions, over 85, 56, and 49 percent of the Zn, Ni, and Cd was mobilized in batch tests. The higher metal release observed during aerobic testing is related to the pH reached under test conditions.

64. Differences were also noted between the pH values observed in the aerobic batch testing (3.5 to 4.8) for Everett Harbor sediments and those reported from runoff testing. Theoretically, the pH of the sediment in the surface runoff tests should reach pH levels similar to that reached in the aerobic batch leaching tests, once the sediment reaches a comparable oxidation level. However, the sediment in the surface runoff test is in a static, unmixed state and a longer time will be required to reach an oxidation status comparable to that observed in the batch testing.

65. There are potential ground-water problems with PCBs in both anaerobic and aerobic leachates. Other organic contaminants should pose no

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Table 5

Contaminant Leachate Concentrations

(milligrams/litre) for Seepage Analysis

| Contaminant | Drinking Water Standards, mg/ℓ | | Anaerobic | Aerobic |
	Federal	State		
As	0.05	0.05	0.039	<0.005
Cd	0.01	0.01	0.010	0.034
Cr	0.05	0.05	0.080	2.27
Cu	--	--	0.096	0.023
Ni	--	--	0.052	0.449
Pb	0.05	0.05	0.058	0.210
Zn	5.0	5.0	0.181	3.5
PCB	--	--	0.0036	0.00176

problems since they were not consistently measured in both the batch and column leachates as were PCBs. Restrictions due to PCB release from Everett Harbor sediment would need to be imposed if the attenuation capacity of the underlying soil was exceeded, an evaluation that could be conducted only for site-specific conditions. Site-specific factors will determine the type of leachate control strategy, if any, that is appropriate. Table 5 provides a summary of leachate contaminant concentrations for use in computing seepage. The use of these concentrations for predictions of contaminant release in leachates is discussed in Part IV.

Tests for Capping Effectiveness

66. Tests for capping effectiveness were conducted using the composite sample of contaminated sediment and samples of the native Everett Harbor sediments intended for use as capping material for the CAD alternative. Detailed procedures and results are presented in Appendix D. A small-scale (22.6-ℓ) reaction column was used to predict the cap thickness required to chemically

isolate contaminated Everett Harbor sediment from the overlying water column. Dissolved oxygen depletion rates and release rates of ammonium-nitrogen and orthophosphate-phosphorus were used as contaminant surrogates in the predictive test. Because of their high concentration and chemical behavior, ammonium-nitrogen and orthophosphate-phosphorus proved to be the best contaminant surrogates for Everett Harbor sediment in this test. There was not a significant difference (p = 0.05) between the dissolved oxygen depletion rate of contaminated Everett Harbor sediment and the native sediment used for capping; these rates were 628 $mg/m^2/day$ and 635 $mg/m^2/day$, respectively. The native sediment is a relatively clean material that underlies the more organically rich, contaminated Everett Harbor sediment.

67. The small-scale study did not address bioturbation. To confirm the results of the small-scale tests in the presence of bioturbation, a large-scale test was conducted using three different kinds of animals. Clams and polychaetes were added to the sediment surface to assess the effect of capping on benthic and infaunal organisms and to provide a source of bioturbation. Mussels were suspended in the water column to determine potential contaminant movement through the cap and into the water column. The cap thickness used in the large-scale tests was 50 cm. This thickness was determined based on the 30-cm cap thickness, which has shown to provide effective chemical seal in the small-scale tests, plus an additional 20 cm of capping material to account for burrowing activity by the animals. Results demonstrated that a 50-cm cap of native sediment overlying Everett Harbor sediment was effective in preventing the transfer of heavy metals, PAHs, and PCBs from the contaminated sediment into the overlying water and biota in the presence of bioturbation.

68. The large-scale test did not include geoduck, an organism occurring in Puget Sound that burrows to a depth of 0.5 m or greater. To prevent exposure of this organism to contaminated sediment, it is recommended that a 50-cm safety margin be added to the thickness required to achieve a chemical seal (30 cm). Thus, the recommended effective cap thickness for contaminated Everett harbor sediment is 80 cm. This thickness does not include any additional material that may be needed to compensate for cap erosion, consolidation of the cap, or incorporation of the cap material in the underlying contaminated Everett harbor sediment during placement.

Settling Tests

69. Settling tests were performed to define the sedimentation characteristics of the sediment to be dredged. These tests determine the required disposal area ponding depth and surface area required for effective retention of suspended solids during the dredging operation, and are used to predict the concentration of suspended solids in the effluent resulting from gravity settling within the disposal area. The tests were conducted using 8-in-diam settling columns. Detailed procedures and results are presented in Appendix E.

70. The behavior of contaminated Everett Harbor sediment at a slurry concentration equal to that expected for inflow to a confined site was governed by zone settling processes. The sediments exhibited a clear interface between settled material and clarified supernatant water as expected for saltwater conditions. The settling data for zone and compression settling of the slurry mass and flocculent settling of fine particles in the clarified supernatant were used to calculate the estimated suspended solids concentrations in the effluent and the relationship between dredged volumes and disposal area volumes as described in Part VI of this report.

Chemical Clarification Tests

71. Chemical clarification testing was conducted to screen chemical polymers to determine their effectiveness and dosage requirements in removing suspended solids from effluent or surface runoff waters not easily removed by gravity settling alone. These tests provide the data necessary to design a chemical clarification system for use at intertidal or upland sites, should there be a need to include such a system in the final design.

72. Various commercial polymers were tested using samples of slurry representative of effluent or surface runoff from confined sites. The polymers showing effective removal of suspended solids were subjected to further testing to define optimum dosage. Low viscosity, highly cationic polymers were found to be the most effective. Optimum dosage for the recommended polymer, NALCO 603, was determined to be 25 mg/ℓ. Detailed results are presented in Appendix F.

Consolidation Tests

73. A consolidation test was conducted using the composite sample of contaminated sediment to provide data for evaluation of filling and settlement rates for confined sites. The test results are applicable for evaluation of both intertidal and upland sites. The test were conducted using standard odometers and procedures developed specially for soft sediments (Cargill 1983). If a confined site is selected for disposal, the test results can be used to determine the fill surface elevation as a function of time. This information will be useful in determining the appropriate timing for placement of a surface cap of cleaner material and the surface elevation behavior of the capped disposal site. The test results are presented in Appendix G.

Sediment Stabilization Tests

74. One promising technique for immobilizing contaminants, providing a disposal site liner, and improving the engineering properties of dredged material is solidification/stabilization. Solidification/stabilization involves the addition of a setting agent(s) to the dredged material. Various setting agents have been used to treat hazardous industrial wastes and flue gas desulfurization sludges. These include cement, lime, kiln dust, fly ash, blast furnace slag, sodium and potassium silicates, and various combinations of these materials. The resulting product has improved engineering properties (lowered permeability and increased bearing capacity) and reduced contaminant mobility.

75. The technical feasibility of reducing contaminant mobility in Everett Harbor sediment by solidification/stabilization was investigated in a series of laboratory-scale applications of selected solidification/ stabilization processes. This state of the art is evolving rapidly, and new setting agents are constantly being developed. Testing for this study was not intended to be an exhaustive evaluation of all possible agents. The processes selected were portland cement, portland cement with Firmix (a proprietary addition), Firmix, and lime with fly ash. Samples of solidified/stabilized products were prepared and cured for physical and chemical testing. Unconfined compressive strength was investigated as a key test for physical stabilization. Samples of solidified/stabilized dredged material were subjected to

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laboratory leaching tests. The data from these tests were used to assess the potentia⸴ for contaminant release from the various products. Detailed procedures and results are presented in Appendix H.

76. The unconfined compressive strength data showed that sediment from Everett Harbor can be physically stabilized by a variety of solidification/ stabilization processes. There are no major technical obstacles, such as chemical interference, when applying solidification/stabilization technology to Everett Harbor sediment. The technology has the flexibility and versatility to meet specifications for physical stability ranging from primarily immobilizing sediment solids in a low-strength product to producing a material suitable for end uses typical of low-strength concrete.

77. The chemical leach data showed that solidification/stabilization reduced the leachability of selected metals. Arsenic and zinc were completely immobilized by the processes included in this study. Data were not available to evaluate the potential of solidification/stabilization technology to reduce the leachability of specific organic contaminants.

PART III: OPEN-WATER DISPOSAL MODELING AND MOUNDING
CHARACTERISTICS

78. Numerical modeling was conducted by the WES Hydraulics Laboratory to predict the behavior of the contaminated dredged material during placement for the CAD alternative. The processes that occur during placement of dredged material at an open-water site are shown in Figure 8. Of particular interest was the estimation of the portion of the total sediment disposed that would remain in suspension in the water column. Also, the degree of sediment spread (area of deposition) on the bottom was of interest. Modeling was conducted to simulate disposal of one bargeload of contaminated material at the surface and disposal of one bargeload of contaminated material through a vertical pipe (submerged discharge point), as shown in Figure 9. Model runs were also conducted for hydraulic pipeline discharge of capping material by discharging at the surface and through vertical pipes of various lengths. The model used was specifically designed for instantaneous discharge from a barge or scow (Johnson and Holliday 1978).

79. The model did not have the capability to predict mounding of material as it accumulates on the bottom. Therefore, an analytical evaluation of mounding characteristics for the contaminated material and the capping material was made, based on available data from other sites.

80. Detailed descriptions of procedures, results of all model runs, and evaluation of mounding configuration will be presented in a separate report (Adamec et al. 1987).

Disposal of Contaminated Material at the Surface

Assumptions

81. Basic assumptions for modeling the disposal of contaminated material at the surface were:

    a. Disposal takes place in approximately 265 ft of water.

    b. A total load of 4,000 cu yd of material will be dumped from the barge.

    c. The disposed material has a bulk density of 1.25, a void ratio of 0.8, and is composed of 25 percent wood chips, 22 percent sand, and 53 percent silt-clay. Four cohesive clumping factors were used: 0, 30, 50, and 70 percent. The distribution of material by size in the clumps was equal to the total distribution.

56
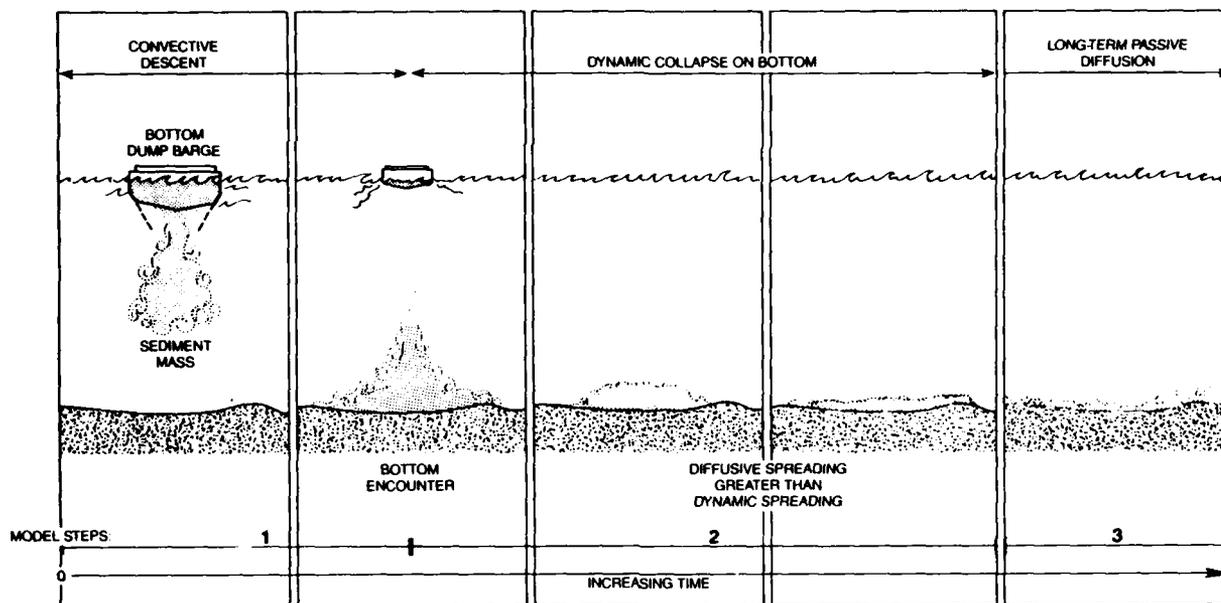
Figure 8. Processes occurring during surface placement for CAD alternative
(USAED, Seattle 1986c)

82. Seven model runs were made with varying current and material compositions as shown in Table 6. The assumed current direction was from southeast to northwest. The long-term computation grid consisted of 21 by 21 cells, with each cell being 200 ft square. Model coefficients for bottom friction and diffusion were estimated based on results of a limited calibration with Elliott Bay data performed as part of this study. The coefficients were slightly modified to reflect the change in water depth and current conditions. Each model run simulated a period 3,600 sec (1 hr) in length, and 300-sec time steps were used.

83. Note that results presented in Table 6 are for conditions after 1,800 sec. The extremely low current velocities measured at the site would indicate that a majority of material remaining in suspension at 1,800 sec would subsequently accumulate on the bottom within the site boundaries. However, these concentrations are so small that they are at the limit of modeling accuracy. Therefore, that percentage which remained in suspension after 1,800 sec was considered as sediment release for purposes of estimating total mass contaminant release. This is considered a conservative assumption.
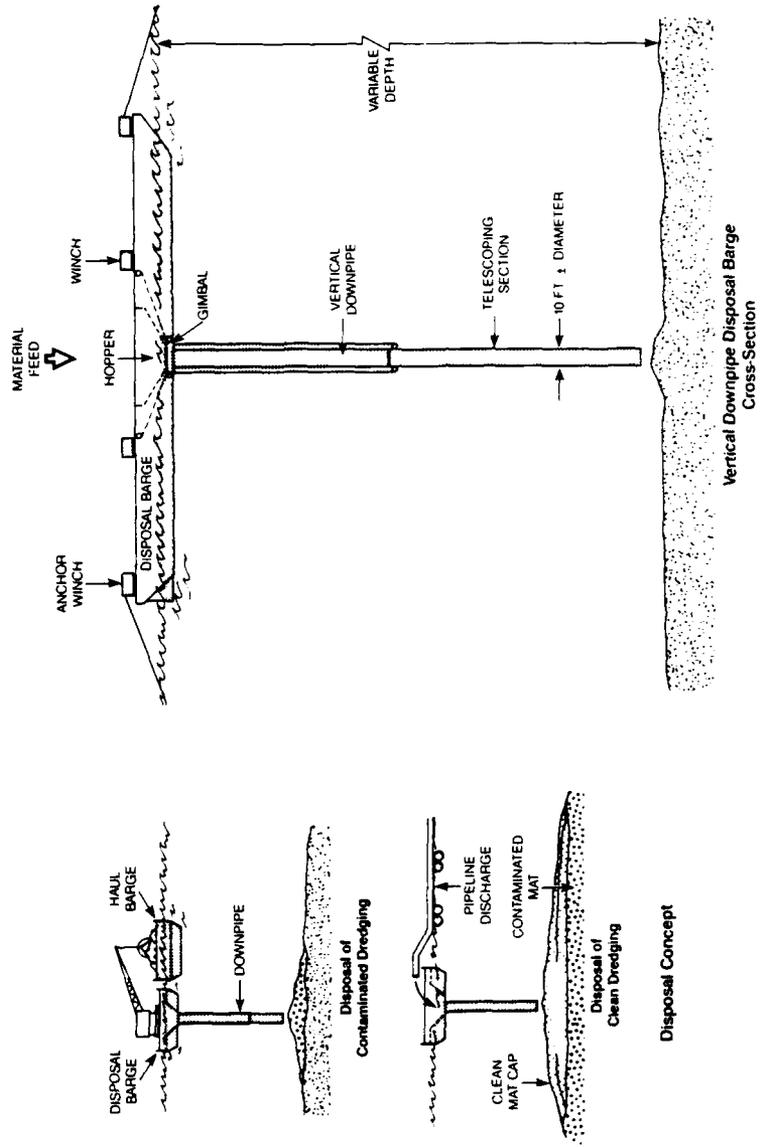
Figure 9. Schematic of vertical downpipe considered for contaminated material placement (USAED, Seattle 1986d)

Table 6

Results of Model Runs for Disposal of One Bargeload
of Contaminated Material at the Surface

| Run | Current fps | Clumping percent | Percentage Remaining in Suspension | | | | Deposition | |
			Sand	Silt-Clay	Wood Chips	Total	Area ft	Maximum Thickness ft
1	0.1	0	3.4	2.2	0	1.9	800 × 1000	0.17
2	0.5	0	12.7	3.5	0	4.6	800 × 800	0.16
3	0.1	30	3.3	2.3	0	1.9	800 × 800	0.26
4	0.5	30	10.7	3.6	0	4.3	600 × 800	0.24
5	0.1	50	3.2	2.4	0	2.0	600 × 600	0.60
6	0.1	70	2.8	2.6	0	2.0	600 × 600	0.49
7	Stratified	0	3.4	2.2	0	1.9	800 × 800	0.17

Results

84. **Sediment remaining in suspension.** The predictions of percent of the sediment fractions and the total percent remaining in suspension 1,800 sec (30 min) after disposal are presented in Table 6 for a range of assumed conditions. For all sets of current and material compositions, the total percentages of sediment remaining in suspension longer than 1,800 sec were less than 5 percent of the total. After 3,600 sec (1 hr), the total percentages of sediment remaining in suspension were less than 2 percent of the total. These estimates do not include any material that may be stripped from the descending cloud; however, past field observations have shown this amount to be extremely small.

85. Based on current data collected at the Deep Delta CAD site,* the medial current speeds vary from approximately 0.26 fps at the surface to 0.11 fps near the bottom. These site-specific current data were used in model run 5 as listed in Table 6. The sediment remaining in suspension longer than 1,800 sec was 1.9 percent.

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* Nortec Technical Services, Inc. 1986 (Mar). "US Navy Deep-Delta Confined Aquatic Disposal Site Current Monitoring Study (Draft)," Seattle, Wash.

86. _Area of sediment deposition._ The areas with deposition of material on the bottom in excess of 0.01 ft were similar for all conditions modeled. The areas (delineated by grids) range from 600 by 600 ft to 800 by 1,000 ft and are shown in Table 6. These areas are for one 4,000-cu yd dump. The thickness at the center of the mound varies from 0.16 to 0.60 ft. The area of deposition for run 5 was 800 by 800 ft as established by the model grid or approximately 700 ft in diameter. These data indicate a tendency for the material to remain in a mounded configuration after impact.

Disposal of Contaminated Material Through a Vertical Pipe

Assumptions

87. Basic assumptions used for modeling the disposal of contaminated material through a vertical pipe were:

  _a._ A 10-ft-diam pipe will extend 250 ft below the water surface.

  _b._ A total load of 4,000 cu yd of material will be dropped into the pipe at the rate of 10 cu yd per minute.

  _c._ The ambient velocity near the bottom was specified to be either 0.1 or 0.5 fps.

  _d._ The disposed material has a bulk density of 1.25, a void ratio of 0.8, and is composed of 22 percent sand, 25 percent wood chips, and 53 percent silt-clay.

88. Since this disposal operation is actually a series of small instantaneous dumps, the instantaneous disposal model was employed with a superposition of results to yield the final deposition pattern on the bottom. This was accomplished through a series of eight individual model runs. Results from each run were then used to represent 50 drops of approximately 10 cu yd each.

89. At the end of the first run, the model results showed that the material was deposited on the bottom in a circular pattern with a radius of approximately 23 ft. At the end of 50 drops, it was assumed that the thickness of the bottom deposit would decrease by 25 percent of its initial value due to consolidation. At the end of the next 50 drops, the thickness would decrease another 25 percent. However, for the remainder of the disposal operation, consolidation factors were not applied.

90. Once the deposition pattern for the first 50 drops was established, the model was rerun but with a nonzero bottom slope determined by the thickness of the deposit and the bottom spread. This resulted in a greater spread of

material on the bottom for the second run. Although the numerical model cannot simulate the actual flow of material down the sides of a bottom mound, this approach seems reasonable as an attempt to simulate the effect of the mound. This same procedure of consolidating the previous 50 drops, determining a bottom slope, and rerunning the model was carried out eight times to represent a total of 400 drops of material through the pipe. No entrainment was allowed in the convective descent phase.

Results

91. <u>Sediment remaining in suspension.</u> Since the material is subjected to ambient current conditions for only 15 ft of descent to the bottom, displacement of the cloud during descent is insignificant. Once the bottom collapse phase begins, the ambient current does transport small clouds as they are formed. However, since settling takes place during each time step in the model, before the transport, material from these runs was always deposited on the bottom before it could be transported by the current. No erosion of material deposited on the bottom is considered in the model. The only other way that the ambient current can influence model results is through its effect on the estimated rate of vertical diffusion, which can sometimes be the deciding factor in terminating the collapse phase. However, neither current condition was large enough to influence the collapse termination in these runs. Therefore, the results presented hold for both currents assumed.

92. <u>Area of sediment deposition.</u> Results from the vertical pipe disposal operation modeling showed that the final deposition of material on the bottom resulting from a single 4,000-cu yd bargeload is contained within a radius of approximately 50 ft from the end of the disposal pipe. The maximum thickness was computed to be approximately 10 ft under the pipe, with a gradual tapering of the bottom thickness to about 3 ft at the outer boundary of the deposited mound. These results hold for velocity conditions of both 0.1 and 0.5 fps.

Disposal of Uncontaminated Material

93. Hydraulic dredging and direct pipeline transport is proposed as the method for placement of the uncontaminated material as a cap for the CAD alternative. Additional model runs were made to evaluate the behavior of the capping material under these conditions. These runs simulate placement using

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three downpipe configurations: 150 ft by 10-ft-diam, 50 ft by 10-ft-diam, and contained surface discharge. Contained surface discharge is defined as use of a scatter plate to reduce momentum and short curtains or downpipe-like barge configuration to reduce surface turbidity, minimize the effects of higher surface currents, and direct the flow of material downward in the water column. Each configuration was modeled at four typical hydraulic dredge production rates: 20, 30, 40, and 50 cu yd per minute. The surface discharge point was assumed to move across the center of the disposal area at a speed of 0.5 fps, moving across the area in 2,820 sec per pass. The effective discharge radius after hitting the scatter plate at the end of the discharge pipe was assumed to be 20 ft.

94. The modeling results indicate that for the confined surface discharge, the majority of the deposition occurred within a 300-ft swath along the line of movement of the discharge pipe. Maximum cap thickness for a single pass of the surface discharge pipe was approximately 0.09 ft (at the 30 cycles/min discharge). A 1-ft cap thickness would therefore accumulate at any specific point in the site with approximately 11 passes.

95. The discharge was modeled in a stationary mode for the downpipe runs, assuming that the pipe could not be moved with the dredge operating. For the 50-ft downpipe runs, the maximum cap thickness for a 1-min discharge was approximately 1.8 ft, within a radius less than 100 ft. For the 150-ft downpipe runs, the maximum cap thickness was 2.0 ft. These results indicate a 1-ft cap would be generated at a point within approximately 30 sec.

96. The total percentage of capping material remaining in resuspension, tabulated below, varied from 0.4 to 32.0 percent, depending on the method of placement. For surface discharge and a production rate of 30 cu yd per minute, 9.4 percent remained in suspension. This indicates that over 90 percent of the capping material will be deposited in the cap.

Disposal	Discharge, cu yd/min			
	20	30	40	50
Surface	11.1	9.4	15.5	32.0
50 ft downpipe	3.2	4.2	10.9	26.3
150 ft downpipe	0.5	0.4	1.6	9.3

Modeling Summary

97. Under the site-specific current and material conditions modeled for surface disposal of contaminated material at the Deep Delta site, 1.9 percent of the total sediment disposed will remain in suspension longer than 1,800 sec. The area of sediment deposition in excess of 0.01 ft for surface disposal of one bargeload of material was approximately 700 ft in diameter. These data indicate a tendency of the material to mound.

98. For disposal through a vertical pipe, the model results indicated that all sediment would be deposited on the bottom prior to transport by currents. The area of deposition on the bottom would be within a radius of 50 ft from the vertical pipe for one bargeload of material. These data indicate a strong tendency for the material to mound under conditions for the vertical pipe disposal.

99. The uncontaminated material, when hydraulically dredged and disposed at the surface, will be deposited within a 300-ft swath as the discharge pipe is moved across the water surface. The diameter of deposition for a stationary downpipe is less than 100 ft.

Modeling Results for Alternate CAD Sites

100. The CAD Deep Delta site, originally identified as the Navy's preferred site, is located in approximately 265 ft of water. Detailed modeling runs for this study were made for conditions at this site. An alternate site in slightly deeper water is now being considered for CAD to offset potential impacts to biological resources. This site is called the Revised Application for Deep CAD (RADCAD) site and is located adjacent to the Deep Delta Site, as shown in Figure 5. The site has a water depth of approximately 310 to 430 ft.

101. Use of the alternate site at deeper depth would mean a proportionally higher sediment mass remaining in suspension. Model runs for the Deep Delta site at a depth of 265 ft indicate 1.9 percent of the material remains in suspension after a time period of 1,800 sec (conservatively considered a mass release). A single model run has also been conducted for a surface dump of contaminated material in a 400-ft depth. These results indicate 3.6 percent of

the material remains in suspension after a time period of 1,800 sec. Inter-
polation for a 370-ft average depth at the RADCAD site yields approximately
3.2 percent remaining in suspension. It should be noted that all these figures
are essentially at the accuracy limit of the currently available models.

102. Deposition patterns for the 400-ft run showed little change over
the 265-ft runs. This would indicate that the "bottom footprint" used for the
mounding evaluation as described below would be approximately the same for both
the Deep Delta and deeper sites.

103. No model runs for hydraulic placement of the capping material have
been made for the 400-ft depth conditions. However, it is anticipated that
results would be similar to those generated for the 265-ft depth, i.e., dis-
crete particle settling behavior. The processes governing the gradual buildup
of the cap would therefore be the same for the deeper depth.

104. Additional model runs for a range of depth conditions up to 800 ft
have been conducted for the Puget Sound Dredge Disposal Analysis (PSDDA)
(Trawle and Johnson 1986). Since the conditions for the Everett study area are
similar to those used in the PSDDA study, the generic model runs performed for
PSDDA can be used to qualitatively evaluate material behavior at deeper water
sites being considered for the Everett project.

Analytical Evaluation of Mounding Characteristics

General

105. An evaluation of mounding characteristics is an essential part of
CAD design. The purpose of this evaluation is to generate a conservative
estimate of the extent of spread or occupied bottom surface area of the mound
and to determine if sufficient capping material is available to place the
design thickness over the occupied surface area. It is recognized that the
Navy design for the CAD site is evolving and that other configurations for the
mound are feasible from a design standpoint.

106. The modeling described in the Disposal Alternatives report and in
the above paragraphs delineates the area of deposition of one 4,000-cu yd
bargeload of contaminated material and the short-term deposition character-
istics of hydraulically dredged cap material. However, the model is not capa-
ble of simulating the effects of mounding or settlement after a large volume of

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material from multiple dumps has been deposited. Therefore, an evaluation of mounding characteristics was made based on existing data at other disposal sites.

107. Two major processes must be evaluated in estimating mounding behavior: the tendency of the material to flow due to momentum transfer during placement and the tendency of the material to form a stable angle of repose. Both processes are influenced by the method and rate of dredged material placement and the mechanical condition of the material resulting from the dredging. The tendency to flow will largely be offset by the tendency of the material to mound. The 1V on 50H bottom slope at Port Gardner is not great enough to induce gravity flow of the disposed material. There would be some tendency for successive impacts of the contaminated material to spread previously placed material, but bottom friction forces would quickly dampen the spread. Naturally occurring bottom undulations and clumps within the disposed material, characteristic of clamshelled material, would also inhibit the tendency for the material to flow.

108. A major factor in estimating mound configuration is the slope or angle of repose taken by the contaminated material and cap. No analytical method has been developed for prediction of mound size or slopes in a subaqueous condition. Some insight can be gained by examining data on existing mounds. However, data on mound slopes exist for only a few sites. The change in void ratio due to entrainment of water and the subsequent settlement of mounds due to consolidation are also major considerations. As with the slopes, no analytical method has been developed for prediction. Therefore, conservative assumptions for this behavior were made for this evaluation.

109. The tendency for clamshelled material to remain in clumps and the nature of the existing bottom at the CAD site are factors that would cause the material to mound and would reduce the need for lateral confinement. The modeling runs for this project and experience with capping projects to date indicate that mechanically dredged, reasonably cohesive material can be placed into discrete mounds using carefully controlled and monitored, but otherwise conventional, equipment and techniques (Bokuniewicz et al. 1978, Semonian 1983, Truitt 1986a). Clamshelled material will exhibit significant clumping and cohesion, increasing stability. Under these conditions, local differences in the slope of mounds should be expected. The assumption of clumping and cohesion for clamshelled material is a major consideration in this evaluation and

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is based on the assumption that the material will be dredged in essentially its present in situ condition and will not be significantly disturbed during debris removal (i.e., only large logs evident by surface probing will be removed prior to dredging and the bottom will not be "raked").

110. The relatively soft bottom at the CAD site would tend to absorb impact energy during placement of the clumps, and the displacement of existing bottom sediments could form some degree of lateral confinement. Although the average slope at Port Gardner is 1V on 50H, the bottom is likely composed of a series of irregular ridges and swales that would increase the tendency of material to maintain steeper mound slopes.

Data for existing mounds

111. Data from mounds in Long Island Sound indicate that silty material that is clamshelled and released at the surface exhibits a clearly defined central mound with steep slopes surrounded by a much lesser volume of more fluid material with much flatter slopes. Estimates of the slope of the central mound vary from approximately 1V on 15H to 1V on 25H. Localized slopes as steep as 1V on 10H are evident from survey data for these mounds (Semonian 1983). This steepness is indicative of a high degree of cohesion and clumping of cohesive blocks of material, and little entrainment of water during descent. However, the small portion of the material that entrained water during descent exhibited a more fluidlike behavior than the majority of the deposit. This portion of the material was deposited as an apron with flatter slopes surrounding the central mound. Data from the Long Island Sound monitoring indicate that the portion of the mound that is involved with the apron is approximately 20 percent by volume (Semonian 1983). Since the apron material is less dense than the material comprising the central mound, the percentage of material comprising the apron by weight would be a lesser value. The slopes of the apron are expected to be flatter than 1V on 20H and may be less than 1V on 60H (Bokuniewicz, Cerrato, and Hirschberg, in preparation).

112. Data from other sites in which the material was deposited from a slurry, as from a hopper dredge, indicate a much flatter slope for the mounds (Bokuniewicz, Cerrato, and Hirschberg, in preparation). For example, in the New York Mud Dump Site, the average slope is approximately 1V on 100H (Suskowski 1983). This slope is also the result of dumping at multiple disposal points. The material comprising the mound had differing characteristics,

ranging from soft claylike materials to silts and fine sands. Local slopes at
the site were as steep as 1V on 10H. Data from a site in Tampa Bay show a
slope of approximately 1V on 100H (Williams 1983). This material was a fine
sandy material that would exhibit little or no clumping or cohesion.

113. All available data on mound slopes indicate that a slope of 1V on
25H or steeper can be attained by fine-grained cohesive material that is
dredged by clamshell and disposed from a barge. These data served as the basis
for estimates of mound slopes for the Everett contaminated sediments, which
would also be dredged by clamshell and disposed from a barge.

Assumed mounding behavior

114. **Placement.** Placement of material for the contaminated mound was
assumed to be by bottom dumping from a stationary position at a designated
point, likely marked by a taut-line buoy or some other fixed point. However,
it also was assumed that the tendency for the contaminated material to form a
discrete mound would require that the disposal point be moved periodically. It
may be necessary to spread the material in a mound with a relatively flat top
amenable to later placement of the cap. Actual placement will depend on the
results of construction monitoring. A flatter mound will also aid in main-
taining overall mound stability. The placement of the cap by hydraulic dis-
charge at or near the surface will involve a continually moving discharge point
using a predetermined, monitored pattern.

115. **Contaminated material characteristics.** The in-channel water con-
tent of the contaminated material is approximately 130 percent, equivalent to a
void ratio of 3.5.* It was assumed that some water would be entrained during
placement and the average void ratio after placement would be 4.5. This is
considered a conservative assumption.

116. **Cap material characteristics.** The in situ water content of the
uncontaminated material to be used for capping is approximately 50 percent,
equivalent to a void ratio of 1.3. This material would be hydraulically
dredged and placed by pipeline discharge at the surface. The resulting void
ratio upon deposition in the cap was assumed to be 4.5. Cap placement using
hydraulic placement from the surface should result in a sedimentation behavior
similar to natural sedimentation. That is, because of the water depths, no jet

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* Hart Crowser and Associates, Inc., 1986, ibid.

or momentum effects will be evident in the lower water column, and the material will ultimately settle as discrete or flocculating particles.

117. **Disposal sequencing.** Since the Navy's proposed dredging plan extends over a period of two dredging seasons, the sequence of disposal operations was taken into consideration. All dredged material quantities discussed are approximate based on the above assumptions for material characteristics. This sequence was assumed to include initial placement of 100,000 cu yd of contaminated material and immediate capping with uncontaminated material. After 9 months, an additional 800,000 cu yd of contaminated material would be placed and then capped with 1,500,000 cu yd of uncontaminated material. The area of deposition for individual bargeloads of contaminated material and passes of the pipeline for capping material were assumed to be equal to those determined by the modeling described in Part III.

118. **Mound slopes.** In developing a conceptual mound configuration, it was assumed that both the contaminated and capping material would be deposited on the bottom in a circular pattern with radius of 500 ft or less, corresponding to the deposition pattern indicated by the modeling runs. It was further assumed that as the mound develops, it would roughly assume the form of a truncated cone, with the top of the cone equal in radius to the area of deposition of the material. As the material accumulates, it would cause spreading to occur with side slopes of 1V to 100H relative to the bottom slope. This results in an angle of repose on the downslope side of approximately 1V on 30H. This slope is within the experience of the Long Island mounds, which were formed with similar materials and dredging methods. It was assumed that spreading in the upslope and cross-slope directions would be governed by similar slopes; however, movement of the disposal point as described above may be necessary to maintain a mound with a relatively flat surface and uniform spread in all directions.

119. The behavior of clamshelled silt material when disposed in open water exhibits a well-defined central mound with side slopes of 1V to 30H or steeper. However, a small portion of the material in each discrete barge disposal will entrain water during descent and will behave in a more fluidlike manner than the majority of the deposit. It was assumed that this apron material would tend to deposit with flatter slopes, approximating the 1V on 50H slope of the existing bottom surrounding the mound proper. Local variations in the mound surface due to discrete dumps will tend to reduce any

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tendency of the apron material to flow. The large surface area of the mound and the overall mound slope will also provide the opportunity for deposition of the apron material on the contaminated mound proper. However, without lateral confinement, a portion of the apron material may move off the contaminated mound proper in the downslope direction due to gravity flow or spreading from subsequent dumps. The final diameter of the capped mound must exceed the diameter of the contaminated mound. This is necessary to provide the required cap thickness over the entire contaminated mound. The overall diameter of the cap defines the required size of the disposal site that will be capped. In effect the capped site diameter provides a zone in which the majority of apron material flowing off the contaminated mound proper would be capped.

120. It was assumed that the slopes of the capping material would conform to the slopes taken by the underlying contaminated material since the cap is gradually built up by settling of discrete particles in a manner similar to natural sedimentation. Natural slopes in the general area of the site vary in steepness but appear to be stable at the slopes assumed for the contaminated material. Similar slopes would therefore appear reasonable for the capping material as it accumulates on the mound.

121. **Mound consolidation.** The layers comprising the mound and cap were each assumed to undergo 50-percent consolidation. This is considered a very conservative assumption for the capping material. Capping thickness in excess of the design requirement of 1 m is maintained even with the assumed consolidation of 50 percent. Actual consolidation is expected to be much less, based on consolidation data for the WES composite sample (contaminated material).

122. **Mound configuration.** A plan view of the mound for the assumed conditions and a conceptual cross section parallel to the slope showing the mound configuration are presented in Figures 10a and 10b. The point of disposal for the second dredging phase is shown offset to the upslope direction with respect to the initial mound formed from the first dredging phase. In this way, the first mound could provide a toe for the larger mound and could result in some degree of lateral confinement. Conceptual cross sections perpendicular to the slope showing the layering resulting from disposal sequencing and mound consolidation are shown in Figure 11.

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MOUND #2 WITH CAP
DIAMETER = 4800 FT

MOUND #1 WITH CAP
DIAMETER = 3000 FT

DOWNSLOPE DIRECTION
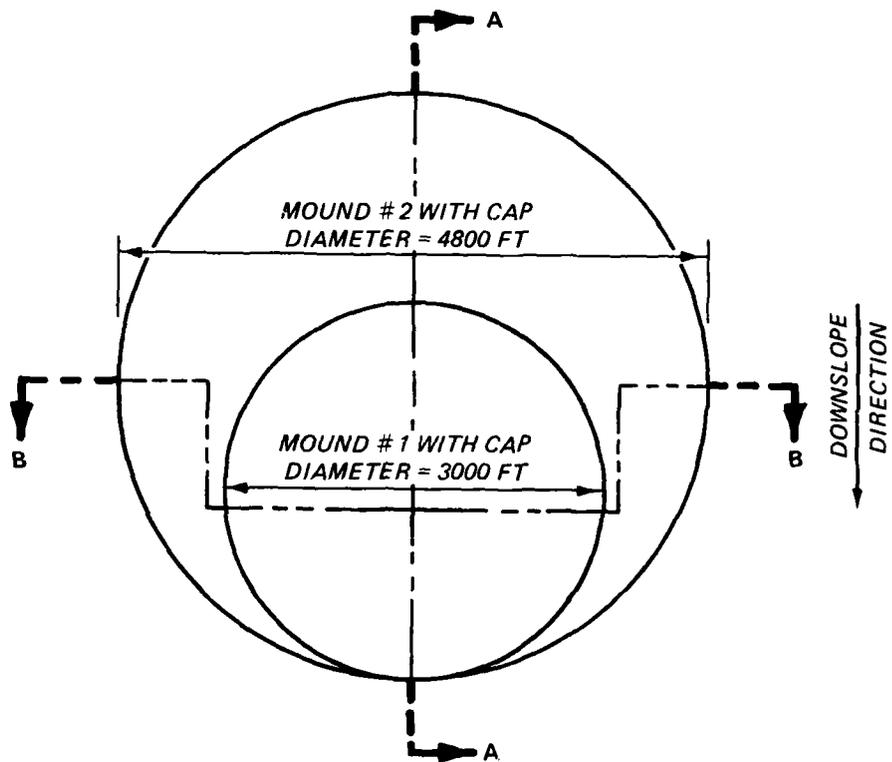
a. Conceptual plan view

NOTE: VERTICAL SCALE FOR MOUND LAYERING
GREATLY EXAGGERATED. LAYERING SHOWN
FOLLOWING CONSOLIDATION.

4800 FT

3000 FT

BOTTOM SLOPE = 1V ON 50H
SHOWN TO SCALE

LEGEND

▨ CONTAMINATED MATERIAL

▩ CAPPING MATERIAL
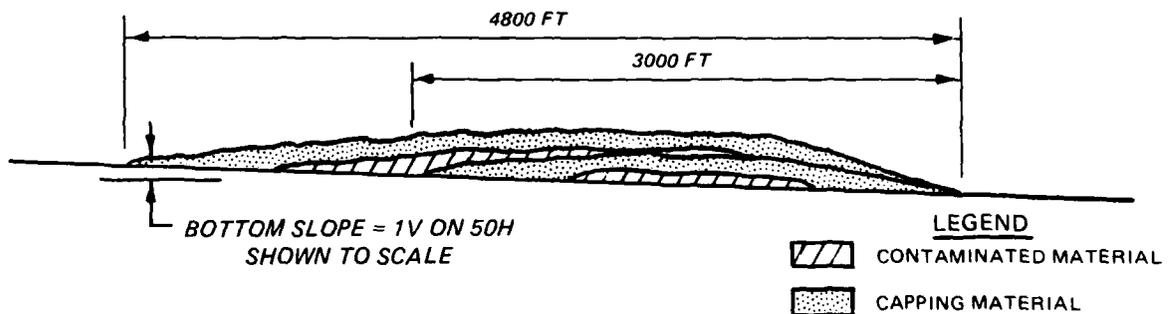
b. Conceptual cross section A-A

Figure 10. Plan and cross section of CAD site

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NOTE: TRUNCATED CONE
MOUND IMMEDIATELY AFTER
INITIAL DISPOSAL OF 100,000 CY
OF CONTAMINATED AND 500,000 CY
OF UNCONTAMINATED MATERIAL

NOTE: MOUND AFTER 9 MOS CONSOLIDATION

NOTE: MOUND IMMEDIATELY AFTER FINAL DISPOSAL
OF 800,000 CY OF CONTAMINATED, AND
1,500,000 CY OF UNCONTAMINATED MATERIAL
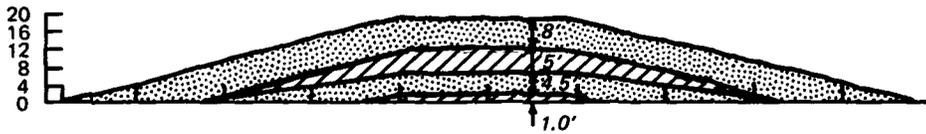
NOTE: FINAL MOUND CONFIGURATION

DISTANCE, FT

SECTION B-B

UNCONTAMINATED

CONTAMINATED
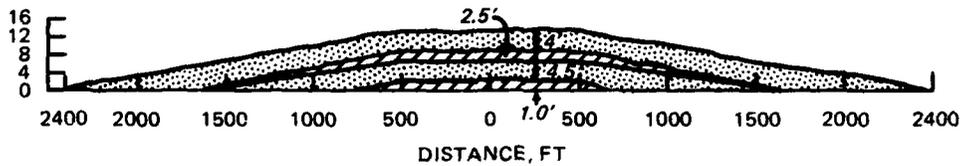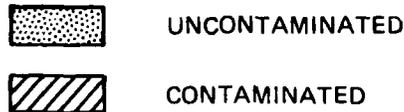
Figure 11. Conceptual sectional views of disposal mound
configuration showing disposal sequencing and mound
consolidation

123. Since the deposition area for each bargeload of material is smaller than that required for the final configuration of the disposal mound, the overall site dimensions appear to be governed by the total quantity of dredged materials disposed and their mounding characteristics. Assuming that the uncontaminated capping material is adequately "slurried" and that disposal locations are carefully controlled, the total dredging quantity of approximately 3 million cu yd will result in a disposal mound that is approximately 2,400 ft in radius and approximately 12 ft high. If the dredging plan allows for the final placement of 1.5 million cu yd of uncontaminated material, the entire site will be covered by a cap that is approximately 4 ft thick.

PART IV: DREDGING EQUIPMENT EVALUATION AND SELECTION

124. The dredging equipment evaluation presented in this part of the report is based on previous studies of the sediment resuspension character-istics of various dredge types (Raymond 1984; Hayes, in preparation) and recent demonstrations of innovative equipment for dredging contaminated sediments.

Performance Goals

125. Selection of dredging equipment for the contaminated Everett Harbor sediments should be based on the following performance goals:

a. Dredging equipment must be compatible with the disposal alter-native under consideration.

b. Dredging equipment should be capable of removing the sediments at a reasonable cost.

c. Sediment resuspended during the dredging operation should be minimized.

Considering the performance goal that an acceptable dredging and disposal alternative should result in no more than a 5-percent combined total contami-nant release, minimizing resuspension will be of utmost importance.

Equipment for Contained Aquatic Disposal Alternative

Proposed dredging equipment

126. The WES Design Requirements report evaluated a three-phase dredg-ing approach under consideration at that time by the Navy's consultants. The use of a submerged vertical downpipe for placing contaminated sediment and the construction of a laterally confining submerged dike or berm were part of that approach, but were determined to be overly conservative based on subsequent tests and modeling. Under the current proposal, contaminated sediments would be mechanically dredged with clamshell dredge(s) (Figure 12), transported to the disposal site in split-hull barges, and placed using conventional surface-disposal techniques. Uncontaminated sediments would be removed using a hydraulic cutterhead dredge (Figure 13) and transported by a pipeline opera-tion directly to the disposal site. Capping, using these cleaner sediments, would be accomplished by discharge from the pipeline at, or just below, the
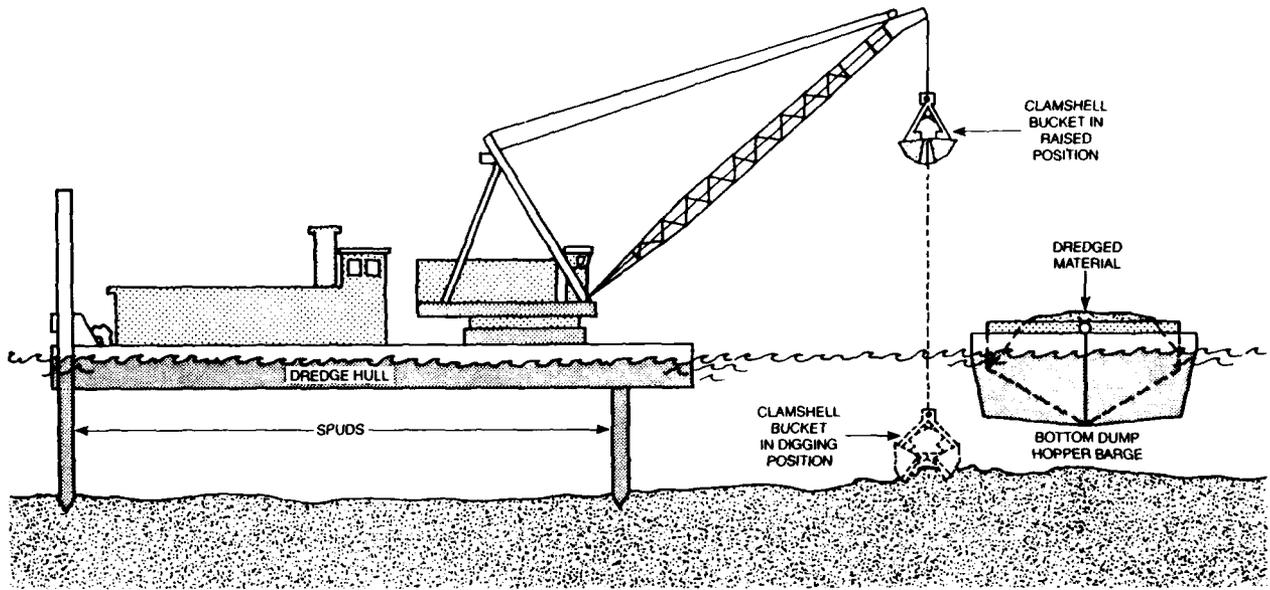
Figure 12. Clamshell dredge (USAED, Seattle 1986c)
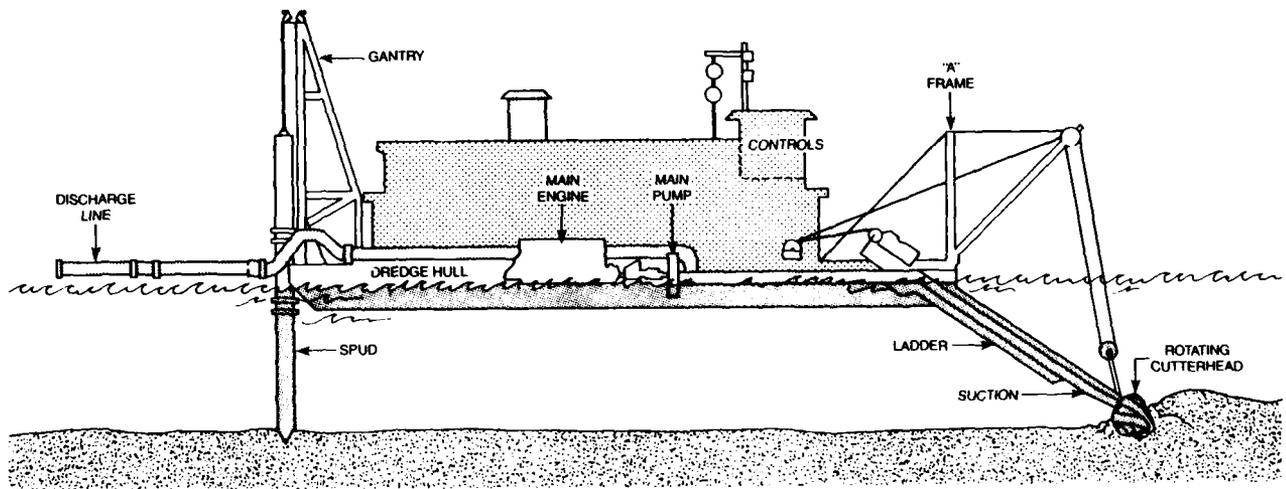
Figure 13. Cutterhead dredge (USAED, Seattle 1986e)

water's surface. With the elimination of the downpipe and subaqueous berm, conventional dredging equipment and approaches are now proposed.

Evaluation of effects of
dredging equipment on disposal option

127. **Subaqueous berm.** Preliminary modeling results had indicated that the completed disposal mound would occupy an area larger than that originally delineated at the site. A system of confining dikes was originally proposed to reduce downslope spreading and to compartmentalize the disposed material. Such dikes are not generally necessary to prevent the contaminated sediment from leaving the site during the short-term disposal process. Model results and the experience with capping to date (e.g., Bokuniewicz et al. 1978; Morton, Parker, and Richmond 1984; Sumeri 1984; Truitt 1986a) affirm that mechanically dredged, reasonably cohesive sediments can be placed into discrete mounds using carefully controlled and monitored, but otherwise conventional, equipment and techniques. Dikes can, however, provide some economy of materials (at the possible expense of lost disposal capacity from intruding dike area) by reducing the total area over which capping sediment must be applied. By providing lateral confining pressures for the soft contaminated sediments, dikes may also allow capping materials to be applied at faster rates and in thicker lifts than on unconfined mounds.

128. Since the predicted mound radius of approximately 2,400 ft can be accommodated by enlarging the site boundaries over those originally proposed, the confining dikes will be unnecessary. This is a reasonable approach and is consistent with predictive results and available experience. It is recommended that the results of bathymetric and supporting remote sensing monitoring during placement of the initial phase sediments be reviewed immediately to verify that the sediment behavior and mound configuration are as predicted. If monitoring indicates that the spread is greater than that estimated, the berm could then be constructed as a contingency measure.

129. As originally proposed,* the confining dike or berm would be constructed in cells using the uncontaminated sediments dredged by hydraulic cutterhead dredge, with placement through a vertical pipe. However, it is

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* ABAM Engineers, Inc. 1985 (Nov). "35% Submittal, Basis of Design, Outline Specifications, Preliminary Cost Estimate, Preliminary Engineer Calculations," Federal Way, Wash.

questionable whether subaqueous confining dikes can be constructed to the side slopes proposed using slurried material, even if the material contains a significant sand fraction. If a berm should be required, using clamshelled material for the construction in a manner similar to that proposed for the contaminated material placement would be advantageous because of the cohesion and clumping normally associated with clamshelled material. Construction of a dike section with cohesive clumps is considered less of an uncertainty. A submerged discharge as originally proposed is desirable for controlling the placement of material used to construct this dike. However, construction of the dike by surface disposal of clamshelled material is also a viable option. Other options for berm construction are discussed in Part V.

130. **Contaminated material placement.** Clamshell dredging for the contaminated sediment is considered the most compatible dredging method for the CAD disposal alternative. Based on the results of the predictive modeling and the additional area available at the disposal site, conventional surface disposal of the contaminated material can be performed without the need for rehandling or use of a submerged conduit. Since one of the objectives of a level-bottom capping operation is to accurately place material in a dense, discrete mound, it is important to maintain any cohesive properties of the sediment and to minimize entrainment of additional water. In general, mechanical dredging tends to impart less energy into the sediment so that disruption of any cohesive bonds is reduced, little additional site water is added, and some opportunity is available for limited consolidation in the barge prior to disposal. Mechanical dredging of the contaminated sediments is, therefore, considered the most physically compatible dredging method for the capped disposal option.

131. **Cap placement.** Hydraulic dredging of uncontaminated material for subsequent use as capping material is recommended as proposed. Modeling results suggest that placement of this material can be accomplished by discharging slurry from a pipeline at the water's surface. However, in order to use a pipeline for disposal at the Deep Delta site, pipeline sections will need to be submerged to cross navigation channels, and at least one booster pump station will be necessary along the required pipeline length. Operational difficulties will need to be addressed, including anchorage of the submerged section of pipeline and the terminal and booster station barges in

deeper water and more difficult than normal sea conditions, and prevention of leakage plumes at pipe joints. Also, the discharge should be directed into a scatter plate to reduce momentum. Use of curtains or downpipe-like barge configuration would reduce surface turbidity normally found with above-surface discharges. The ability to use a pipeline and near-surface discharge does, however, eliminate the need for rehandling the sediment, and by increasing entrainment during descent, will allow the cap to be placed over the relatively soft, underlying contaminated material with less impact or potential for displacement.

Equipment for Confined Disposal

132. Dredging equipment most suited for placement of material in a confined site is the conventional hydraulic cutterhead dredge. This dredge resuspends less material at the point of dredging than other conventional dredges. Also, the direct hydraulic placement of material in the confined site by pipeline avoids double handling of the material. All confined sites now under consideration are within reasonable pumping distances.

Sediment Resuspension and Contaminant
Release During Dredging

General

133. During dredging operations, all dredge plants, to differing degrees, disturb bottom sediments and create a plume of suspended solids at the dredging site. The suspended solids plume can result in relatively low concentrations in the upper water column or high concentrations near the bottom, or both, depending on the type of sediment and the amount of energy introduced into the sediment by the dredge. The major problem from suspended solids will occur when dredging the contaminated sediments. These sediments may release contaminants into the water column through resuspension of the sediment solids, dispersal of interstitial water, or desorption from the resuspended solids. Clearly, the control of sediment resuspension during dredging will reduce the potential for release of contaminants.

Resuspension during mechanical dredging

134. **Operation and source of losses.** The specific mechanical dredge type recommended for removing the contaminated sediments will likely be a clamshell bucket dredge mounted on a flat-bottom barge. Conventional clamshell operations usually excavate a heaped bucket of material, but a portion may be washed out (resuspended) by turbulence during hoisting. Once the bucket clears the water surface, additional losses may occur through rapid drainage of entrapped water and slumping of material heaped above the rim. Loss of material is also influenced by the fit and condition of the bucket, the hoisting speed, and the properties of the sediment. It is common in conventional maintenance operations to drag the bucket across the bottom of a completed cut section to smooth out the irregular surface. This practice can produce substantial bottom turbidity. Finally, accidental and/or deliberate overflowing of disposal barges and spillage and leakage from the barges can result in sediment losses and elevated suspended solids concentrations in the vicinity. In summary, a mechanical dredging operation presents a number of opportunities for release of sediment and associated contaminants into the water column.

135. **Mass release.** Very little definitive information is available to quantify the actual rate of sediment resuspension from the above sources. Those studies that have been performed, however, suggest that the mass resuspended during clamshell dredging is a small percentage of the total mass of sediment removed. Tavolaro (1984) reported that approximately 2 percent of the total mass of dredged material was resuspended and lost at the dredging site during a conventional operation. This estimated 2 percent included 1.2 percent resuspended by the dredge itself and 0.8 percent due to scow overflow and spillage. It is unlikely that deliberate overflowing of scows would be permitted during the dredging of contaminated sediments. However, for a conservative estimate of the potential mass loss at the dredge site, the total 2 percent will be used in this analysis.

Resuspension during hydraulic dredging

136. **Operation and source of losses.** The hydraulic cutterhead dredge is a commonly used dredge plant, and is generally considered the most efficient and versatile. The rotating cutter loosens sediment and makes it available for removal by the suction pipe. Resuspension at the dredge point can be viewed as the difference between the amount of sediment loosened or disturbed

from the bottom and the amount actually entrained and removed by the suction process. Accordingly, those factors that affect the loosening and the removal process will influence the resuspension. These include the rotational speed of the cutter, swing speed of the ladder, and depth of burial of the cutter below the sediment surface. Other more minor contributions to resuspension are the sloughing of partially cut portions of the swing, the penetration into the bottom and removal of the spuds, the swing wire anchors, and leakage around pipe connections.

137. **Mass release.** No total mass release (or balance) study is available at this time for a cutterhead operation. However, WES studies (Raymond 1984; Hayes et al., in preparation) have provided data on the concentration of suspended solids near cutterhead dredging. These data show that the concentration levels from cutterhead dredges are generally less than for conventional clamshell dredges. Since the average concentrations are lower, it is reasonable to expect that the total mass release from the cutterhead is also lower. An estimate of 1 percent loss of the total mass dredged appears appropriate and was used in this analysis.

Contaminant release

138. No laboratory-developed and field-verified approach exists to predict contaminant mass release at the point of dredging. However, conservative estimates can be based on the mass release of sediment plus an evaluation of the dissolved release.

139. Not all the sediment that is resuspended will necessarily be sediment release as defined for the purposes of this report. For example, if dredging of the contaminated sediments proceeds from north to south, a large portion of the sediment initially resuspended will be redeposited and later dredged. No method currently exists to quantify the portion of such sediments affected, but the total sediment released will be less than the total resuspended. For this reason, the estimate of mass contaminant release based on mass sediment release is considered conservative.

140. The approach previously described for estimating mass release of dissolved contaminants from an open-water disposal operation was assumed applicable to dissolved contaminant release due to sediment resuspension at the point of dredging. When this approach was applied to the higher value of 2 percent, estimated to be resuspended during mechanical dredging, the resulting dissolved contaminant mass release was negligible.

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141. The total mass contaminant release due to the dredging operations is, therefore, considered equal to the mass sediment resuspended. This is 2 percent for clamshell dredging (CAD alternative) and 1 percent for cutter-head dredging (nearshore alternative).

Methods to reduce sediment resuspension

142. Techniques and equipment are available as options that may be implemented if desired to reduce overall sediment resuspension and contaminant release. The following examples are taken from the general guidance found in Raymond (1984).

143. **Clamshell dredges.** The following methods can be considered to reduce the sediment resuspension and subsequent loss when clamshell dredges are used:

a. Use of a specifically designed enclosed clamshell bucket (Figure 14) to reduce spillage and leakage.

b. Controlling the drop, hoist speed, and swing of the bucket to reduce spillage and resuspension due to the impact and removal of the bucket at the bottom, and preventing sweeping of the bottom by the bucket.

c. Preventing scow or barge overflow.

144. **Cutterhead dredges.** The following methods can be considered to reduce the sediment resuspension and subsequent loss when cutterhead dredges are used:

a. Controlling swing speed, cutter rotation speed, and depth of burial.

b. Use of modified stepping methods or a spud carriage system to minimize overlapping of cuts.

1 COVER
2 COVER
3 RUBBER PACKING OR WELDED TONGUE AND GROOVE
4 ROD
5 SHELL
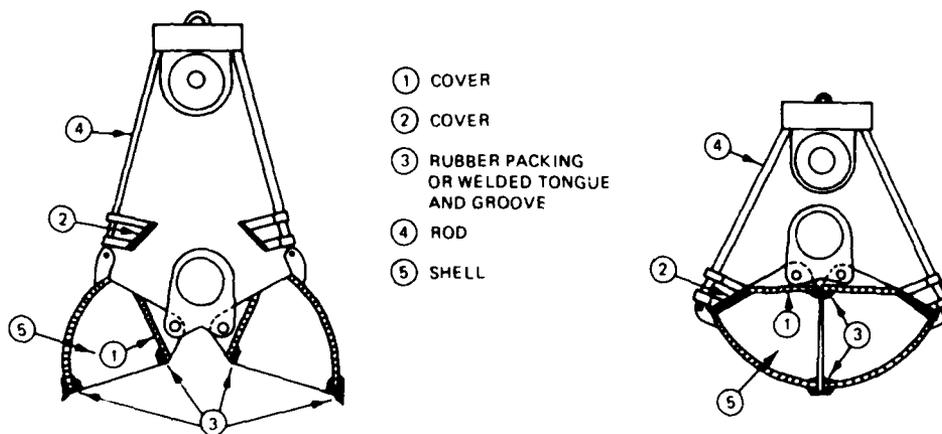
Figure 14. Open and closed positions of the watertight bucket

PART V: EVALUATION OF CONTAINED AQUATIC DISPOSAL

145. Contained aquatic disposal has been identified by the Navy as the preferred alternative for disposal of the contaminated Everett Harbor sediments. The CAD or capping concept can be summarized as three basic components: controlled, accurate, subaqueous placement of the dredged material; isolation of the material from the receiving environment (typically with some type of covering or cap); and monitoring and maintenance of the site. There are a number of variations in techniques, equipment, and materials that can be combined to produce different configurations or to accommodate differing requirements. Figure 15 presents schematics of two types of capping projects. The first is generally referred to as level bottom capping and the second, as contained aquatic disposal, although the term CAD has been used in this report to include all types of capping. As the name suggests, level-bottom capping projects attempt to place the contaminated material on the existing flat or to gently slope the bottom in a discrete mound. Capping is then applied over the mound by one of several techniques, but usually in several disposal sequences to ensure adequate coverage. Where the mechanical conditions of the contaminated material and/or bottom conditions (e.g., slopes) require a more positive lateral control during placement, confining options may be applied. These might include the use of an existing depression, preexcavation of a disposal pit, or construction of one or more confining, submerged dikes or berms.

146. As discussed in Parts III and IV, the CAD alternative as originally proposed utilized confining dikes as well as submerged discharge. The concept has been modified to a simpler, level-bottom capping project. The Navy proposes surface release of contaminated material using bottom dump barges and hydraulic pipeline discharge of clean material at or near the surface for cap placement. The use of a downpipe and subaqueous confinement would be eliminated. This modification will require that the site dimensions be expanded to allow placement of the mound without confinement. The alternative as now proposed is similar to conventional capping operations successfully demonstrated at other locations. Although this is a simplified project and more closely parallels existing experience, the CAD concept should not be thought of as merely a more elaborate version of conventional open-water "dumping." A CAD site is an engineered structure, and its successful performance depends on proper design and care during construction.

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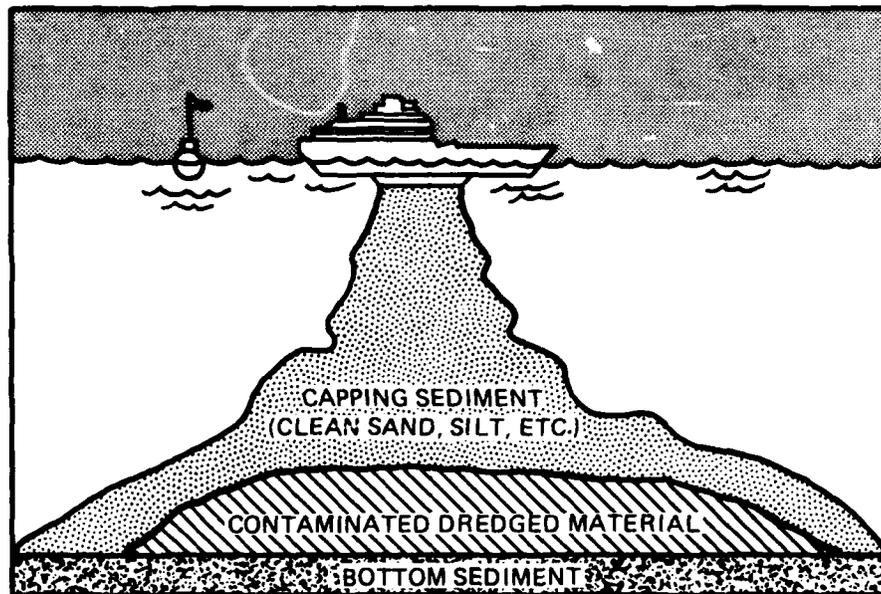
a. Level bottom capping
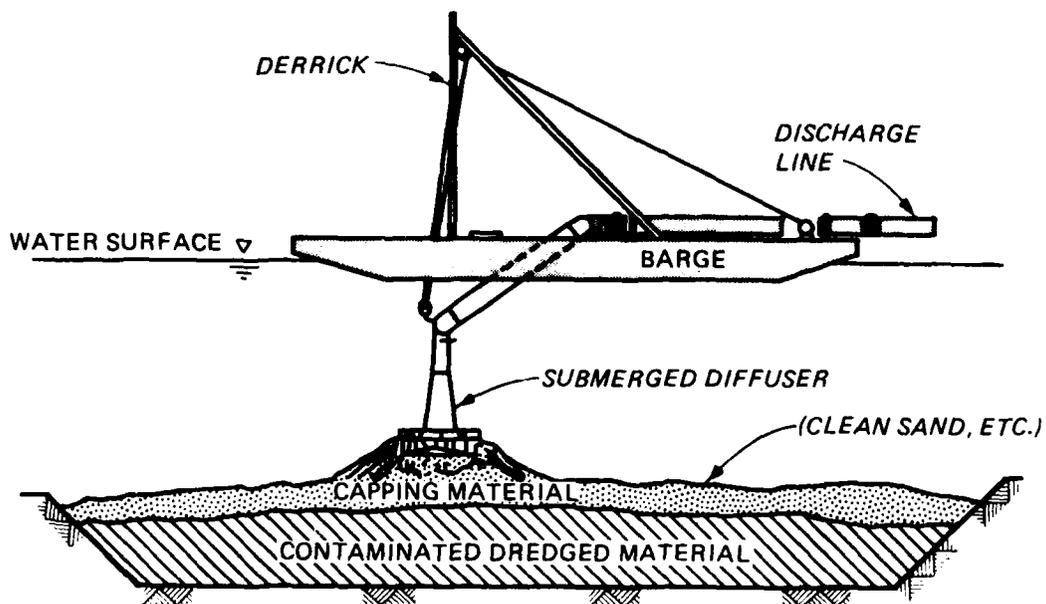
b. Contained aquatic disposal

Figure 15. Schematic of CAD project showing use of
a submerged diffuser for placement

Site Description

147. The Deep Delta site shown in Figure 16 was identified as the CAD
site in the applicant's original Section 10/404 permit application. Detailed
information on the site characteristics, including bathymetry, currents, and
geotechnical properties of in situ sediments, has been collected and recently
provided.*,** In summary from those documents, the Deep Delta site is located
west of the mouth of the Snohomish River and channel. The area originally
identified as available for disposal was approximately 3,000 by 4,000 ft with
a gentle to moderate southerly slope to bottom elevations averaging
-250 ft (mllw). The substrate sediments at the site extending to a thickness
of 10 to 40 ft consist of clayey silts to fine sandy silts, and are likely the
result of the continual deposition of the river sediment load. Chemical anal-
ysis of these sediments indicates that they are cleaner than Puget Sound back-
ground levels.

148. Local currents at the site result from a combination of forcing
function, including wind stress, tidal oscillations, and density gradients in
the water column produced by freshwater discharge. The current monitoring
work performed by the Navy* covered a 31-day period during which a number of
storm events passed through the Puget Sound area. A statistical frequency
characterization of these storms has not been performed, but they appear to
represent major wind speed and discharge events, and the resulting currents
measured should represent "average worst" conditions. The reported data indi-
cate that the current structure at the site is vertically stratified. The
median instantaneous current speed below a depth of 100 ft was 3.5 cm/sec, and
the maximum instantaneous speed at the bottom was 18 cm/sec. The 31-day
vector-averaged currents in the lower water column had velocities in the range
of 1.5 to 4.0 cm/sec and tended toward the northwest, generally paralleling
bottom contours. Salinity structure has not been reported.

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* Nortec Technical Services, Inc. 1986 (Mar). "US Navy Deep-Delta Confined
  Aquatic Disposal Site Current Monitoring Study (Draft)," Seattle, Wash.
** Hart Crowser and Associates, Inc., 1986, ibid.
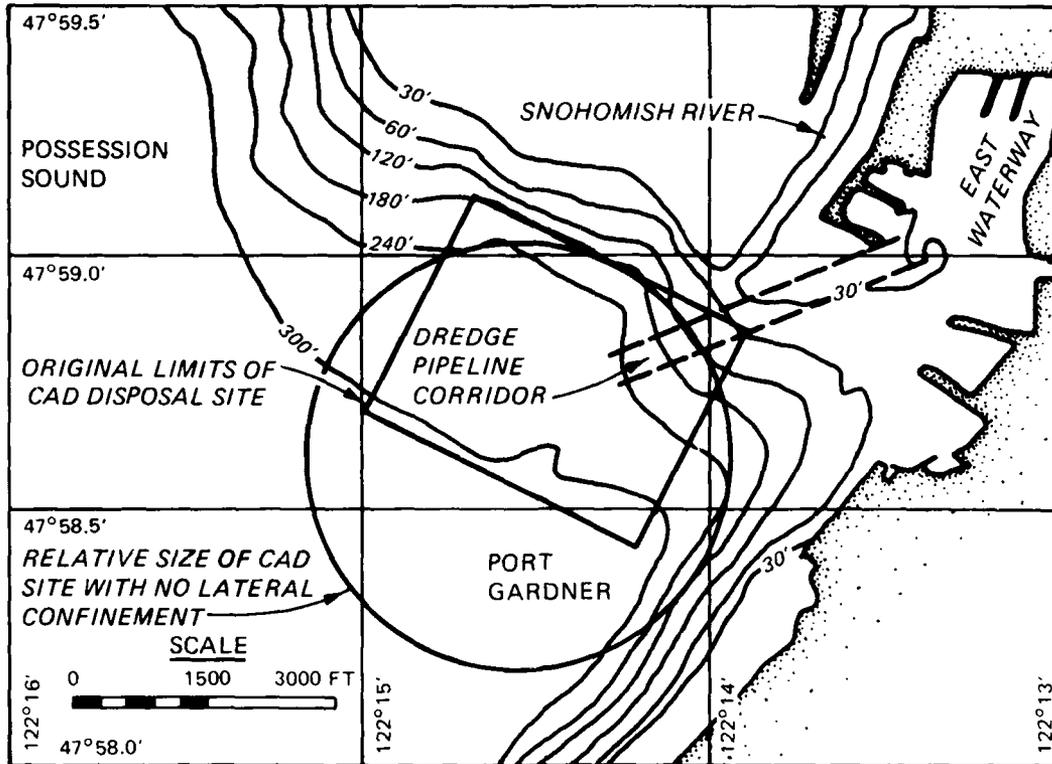
Figure 16. CAD Deep Delta site

Controlled Placement of Dredged Material

General

149. The CAD alternative previously under consideration called for the
controlled placement of uncontaminated dredged material for a confining struc-
ture, placement of contaminated material within the structure, and the place-
ment of uncontaminated material to form a cap of specified thickness. The CAD
alternative involves placement of material from the surface within a desig-
nated bottom boundary and capping to a minimum thickness. These operations
have been successfully performed in water depths less than 100 ft. However,
the water depth at the CAD site under consideration is in excess of 250 ft,
and controlled placement of capping material required for CAD has not yet been
attempted at such depths. Also, hydraulic pipeline placement of capping mate-
rial in such depths has not yet been attempted. Confined aquatic disposal
should be technically feasible at such depths; however, the deep water depth
will require additional provisions for precise positioning of equipment and
monitoring of the operation while in progress. With the elimination of the

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downpipe and confinement berm, the basic cost of the CAD operation at this site would be similar to that demonstrated in other locations.

150. As shown in Part III, the behavior of a dredged material placed at an open-water site by instantaneous release from a barge can be described as three or four distinct transport phases or stages generally paralleling the physical forces or processes that dominate during each period. A number of factors affect this descent, including the mechanical properties of the sediment and conditions in the water column and at the site bottom. The following are brief descriptions of the nature and magnitudes of the effects produced by the more important of these factors (see Bokuniewicz et al. 1978 and Truitt 1986a).

Factors affecting placement

151. Currents. The principal influence of currents in the receiving water is to displace the point of impact of the descending jet of material with the bottom (by a calculable amount). Even very strong currents (up to 10 times the average at the Deep Delta site) at some disposal sites studied did not significantly affect the accuracy of the placement. Somewhat greater dispersion during placement in higher currents is likely and is reflected in the results of the model runs described in Part III. Although, as shown in those generally conservative trends, the relationship between current velocity and suspended sediment is not linear, i.e., a fivefold increase in current speed resulted in a threefold increase in suspended load durir , the short simulation period.

152. Water depth. Aside from the effect depth has on current speeds, there appears to be little additional short-term influence on the actual disposal process using instantaneous surface dumping. The same general stages of descent have been observed at sites (including Elliott Bay) with water depths ranging from approximately 50 to 220 ft. The very cohesive fraction of mechanically dredged material (the clods or clumps) attains terminal speed quickly after release from a barge, and does not accelerate further with depth. The bottom surge does not spread at a faster rate, but the initial thickness of the surge has been shown to be a function of water depth because of additional entrainment. The total water depth at a site has more impact on stability over long time periods (usually favorable) than on placement processes. Certainly, operational and monitoring problems may be more severe at deeper sites. In addition, even though open-water disposal has taken place

and has been monitored at deep-water sites, experiences with capping are limited to approximately 100 ft. Depth does have a more pronounced effect on the descent of the jet resulting from pipeline discharge since sediment cohesive bonds are destroyed, and entrainment of site water with depth is increased. Real-time feedback to verify both placement and capping progress and behavior during construction will be critical.

153. Density stratification in water column. A severe density gradient in sufficiently deep water can result in arrest of the descending mass of material from a barge. The depth at which that might occur can be calculated. In addition to the relative densities of the water column layers, the depth to the interface of the pycnocline (not the total depth) and the initial volume of the released dredged material are the important terms. Density structures were considered as part of the Puget Sound Dredge Disposal Analysis (PSDDA) studies and were judged to not be significant factors at Port Gardner. Although no data have been provided on the density structure of the water column at the site, the barge volumes anticipated (4,000 cu yd) are sufficiently great that the descending mass should have enough momentum to penetrate to the bottom. However, the jet from the pipeline discharge could easily entrain sufficient site water to reach a density so low that jet integrity is lost. The only practical result is that discrete particle settling begins, and the sediment becomes more susceptible to the effects of currents and depth.

154. Navigation and positioning. Accurate navigation to the disposal site and precise positioning during material placement are obvious requirements; however, their importance at this deep-water site cannot be over-emphasized. State-of-the-art equipment and techniques should be employed to ensure accurate point disposal. A positioning study was conducted as part of the PSDDA studies. This information was used in project design considerations by the District and the Navy. Taut-moored buoys, mooring barges, various acoustical positioning devices, and computer-assisted, real-time helmsman's aids should be investigated. In all cases, barges or scows must be required to come to a complete halt before release (static dump) and accurately keep station during the release. In general, for the clamshelled, contaminated sediments, rapid release with high insertion speed (from the static barge) is desirable. This allows the material to reach terminal speeds quickly, minimizes contact time with the water column, and most closely approximates the modeling assumptions.

155. **Submerged discharge.** The use of a submerged discharge or closed
conduit of some type to place the dredged material is an option that has been
discussed in the homeporting project as well as others. Several conduit tech-
nologies are available, including a submerged diffuser, gravity-fed downpipe
or tremie, and various pumpdown systems. In general, a conduit is used, pri-
marily to ensure more accurate placement of the material and to reduce exit
velocities during formation of the bottom surge. These effects are demon-
strated in the results of the comparative modeling. A conduit extending from
the surface to the bottom will certainly chemically isolate the material from
the water column during descent, significantly reduce entrainment, and mini-
mize the effects of currents and stratifications. The use of a conduit is a
conservative measure that could be employed to address one of these specific
problems. However, as discussed below, the use of a submerged discharge is
not required from an environmental standpoint.

Contaminant Release During Placement

Water quality

156. Standard elutriate test results (dissolved concentrations) were
compared with the background water concentrations and water quality criteria
in accordance with the Decisionmaking Framework. The test results are an
estimate of the dissolved release of contaminants during placement of dredged
material in open water for the CAD alternative. The results indicated that
there was little release of dissolved contaminants. Only Ni, Cd, Pb, Cr, and
PCB 1254 exceeded the background concentrations. The Cd and Cr concentrations
were below both the chronic and acute exposure values given in the Federal
water quality criteria. The only remaining parameters of concern are Ni, Pb,
and PCB 1254.

157. The Ni concentration in the elutriate was above the chronic expo-
sure value given in the Federal water quality criteria but was far below the
acute exposure value. It must be noted that the Port Gardner background con-
centration equals the chronic criteria for Ni. The standard elutriate con-
centration cannot be diluted to the chronic criteria by mixing. However, the
elutriate concentration is well below the acute criteria.

158. The Pb concentrations slightly exceeded the chronic exposure
values given in the Federal water quality criteria. A dilution factor of less

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than 1 was calculated using procedures in the Decisionmaking Framework that would dilute the standard elutriate value to the chronic exposure value. Size and configuration of the mixing zone would depend on site-specific information not yet available. However, such a minimal mixing and dilution could easily be achieved within a short distance of the open-water disposal operation.

159. PCB 1254 concentrations exceeded both the chronic and acute exposure values given in the Federal water quality criteria. A dilution factor of 13 was calculated using procedures in the Decisionmaking Framework that would dilute the standard elutriate value to the chronic exposure value. Size and configuration of the mixing zone would depend on site-specific information not yet available. However, such a minimal mixing and dilution could easily be achieved within a short distance of the open-water disposal operation.

160. Only 7 of 33 contaminants of concern were detected in the standard elutriate tests. Only five parameters exceeded Port Gardner background concentrations, and only three parameters, Ni, Pb, and PCB 1254, exceeded the Federal water quality criteria. These parameters were all of low concentration, and dilution to background concentrations or criteria can easily be accomplished within a short distance of the disposal operation. Based on these data, there appears to be no need for controls from the standpoint of contaminant release in the dissolved form during placement of the sediments for the CAD alternative.

Mass release

161. Mass release during the placement for the CAD alternative was assumed to be directly related to the loss of solids plus an estimation of the dissolved release based on elutriate data. The modeling efforts described in Part III of this report indicate that the sediment remaining in suspension longer than 1,800 sec and assumed to be released during open-water disposal would be 1.9 percent. The mass release for contaminants in the dissolved form was estimated based on the standard elutriate data, and an estimate of the total water entrained and released during open-water disposal. It was assumed that the total volume of water entrained during clamshell dredging operations would be equal to 30 percent of the total volume dredged. It was further assumed that the total entrained volume would be released with contaminant concentrations equal to the dissolved standard elutriate concentrations. The same assumptions were applied for disposal at the surface and disposal through a vertical pipe. These calculations indicated that all dissolved mass

releases were negligible except Cd with 0.1-percent release and Hg with 0.2-percent release.

162. The total mass release for disposal at the surface will therefore range from 2.0 to 2.1 percent, depending on the parameter. For disposal through a vertical pipe, total mass release will be negligible except for dissolved Cd with 0.1-percent release and dissolved Hg with 0.2-percent release.

Capping Material Thickness and Placement

Capping requirements

163. One of the principal design decisions in a CAD project is the nature and thickness of the capping material placed over the dredged material mound. The capping material provides the isolation necessary to control the movement of contaminants out of the dredged material and into the overlying water column, and to prevent direct contact between the aquatic biota and the contaminated material. The cap will also perform the important physical function of stabilizing the material and protecting it from transport or dispersion away from the site. The design of the cap must, therefore, consider both grain size and thickness.

164. The results of laboratory testing described in Appendix D have indicated that the uncontaminated sediments from the homeport site can be used to provide an effective cap for the contaminated material. An effective thickness of 80 cm of this material is required to sequester the contaminated sediments from the overlying water column. The placement of a uniform "blanket" of this thickness in 250 ft of water using surface pipeline discharge is not operationally practical. Application of a 1-m or greater cap thickness should therefore be specified as an operational requirement. This will allow for some irregularity in thickness and would be more consistent with the resolution and accuracy of monitoring equipment. This thickness also can be easily achieved with the quantity/volume balance suggested in Part III. It should be noted that actual quantities placed will initially create a much thicker cap.

Effect of dredging sequence on capping

165. As discussed in Part III, the first-phase dredging to include that required for breakwater construction involves the removal of approximately 97,000 cu yd of contaminated and 739,000 cu yd of uncontaminated material

(approximately an 8 to 1 ratio). The remaining dredging requires the removal of approximately 831,000 cu yd of contaminated and 1,638,000 cu yd of uncontaminated material (approximately a 2 to 1 ratio). The ratios above reflect in situ channel volumes to be removed. The in situ density of the capping material is higher than the contaminated material, and the capping material will be hydraulically dredged. These factors will result in ratios of capping to contaminated material in the mound of 16 to 1 for Phase I and 6 to 1 for Phase II (see mound configuration in Figure 11). However, there is an imbalance of capping material available for the initial dredging phase as compared with the latter phase.

166. Several options could be considered to offset this imbalance. First, the dredging sequence as proposed could be altered to remove a greater quantity of contaminated material in the initial dredging contracts. This would allow more effective use of capping material that must be removed for the breakwater construction. A second option (ultimately proposed by the Navy) would be to provide some lateral confinement using more simplified techniques, as described below, than those originally proposed in the 35-percent design submittal.* A third option is to proceed with the proposed sequence, specifying a minimum applied cap thickness of 1 m. If monitoring determines that the 1-m thickness has not been achieved, additional material from other sources could be placed to increase the cap thickness. Some combination of the first and second options would likely add little cost to the overall project. The third option could potentially involve significant additional costs.

Criteria for Successful Capping

167. Capping will be completely successful if all contaminated material reaching the bottom is capped with a thickness of uncontaminated material in excess of 80 cm. However, a small percentage of the contaminated material apron as described above may not remain on the mound during the mound formation process. The overall diameter of the capped site as described above will provide a means for this material to be capped within the designated boundaries of the disposal site. It any movement of the apron material outside the designated site is found by the monitoring, the capping operations could be

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* ABAM Engineers, Inc., 1985, op. cit.

modified to ensure the material is capped. The placement of a confining berm could be considered as an added measure to minimize any downslope movement of the apron material.

168. The mounding configuration described above indicates that sufficient capping material is available to place a 1-m cap over the contaminated mound, and the procedures for cap placement as proposed are designed for a uniform capping thickness. However, local variations in bottom topography, contaminated mound surface, and the actual application of capping material will all result in local variation in the final cap thickness. Monitoring data should define the final configuration of the contaminated mound and the applied cap thickness after initial placement and consolidation.

Subaqueous Confinement

169. Use of a subaqueous laterally confining structure or depression was recommended in the Design Requirements report for the CAD alternative proposed in the initial design. As then recommended, subsequent modeling and analytical evaluations based on data from other sites were conducted as described in Part III. These analyses indicated that the dredged material mound ˜nd cap can be placed within a bottom radius of 2,400 ft with no lateral confinement.

170. However, a subaqueous berm is one option that should be considered to offset a potential shortage of capping material for the final cap. The berm could be constructed using surface dumping of material in one of two ways. First, the point(s) of disposal for the initial dredging phase could be specified to form a crescent-shaped alignment along the southern edge of the CAD site. The exact location would be governed by the estimated spread of the mound resulting from the initial dredging phase. This would result in the initial capped mound located in such a way as to provide some later confinement on the downslope side for the subsequent larger mound. Another method would involve the construction of a berm by clamshell dredging and surface disposal techniques (identical to that proposed for the contaminated material) with the excess uncontaminated material dredged in the initial phases of the project. This material could be disposed at the surface in a circular arc along the south side of the disposal site. Any mounding of this material along the downslope edge of the disposal site could provide lateral

confinement of the contaminated material dredged in subsequent phases of con-
struction. This option would involve use of clamshell in lieu of cutterhead
dredges for removal of a portion of uncontaminated sediment, which may add to
dredging costs. However, some degree of lateral confinement would be pro-
vided, and the spread of contaminated material would be reduced in the later
stages of the project. This may help to provide a thicker final cap thickness
with the remaining capping material available.

Submerged Discharge

171. Standard elutriate test results as described previously indicate
that contaminants released in the dissolved form are either below background
(or criteria) or can be diluted to background (or criteria) within a short
distance of the disposal operation. Requiring a submerged discharge for
placement of the contaminated material from the standpoint of water quality
is therefore not justified.

172. The modeling results discussed in Part III of this report indicate
that use of a submerged discharge point will allow all sediment to quickly
reach the bottom at the disposal site with little or no resuspension. How-
ever, the model results also indicate that surface discharge will not result
in sediment loss that would cause the performance standard for total mass
release to be exceeded. Therefore, use of a submerged diffuser or downpipe is
not required to meet the mass release performance standard.

173. Use of a submerged discharge is not required from the standpoint
of contaminated dredged material or cap placement, assuming no confining dike
is constructed and the site dimensions are expanded to accommodate spread.
The only application where use of submerged discharge may have advantages is
for construction of a submerged berm, if such construction is determined to be
required. A submerged discharge would allow more precise placement and econ-
omy of material used for construction of the berm. However, construction of
the berm using surface disposal of clamshelled material as described above is
also a viable option.

Summary of Mass Release for the CAD Alternative

174. The performance objective specified that the total mass release of contaminants should not exceed 5 percent for any disposal alternative. The following tabulation summarizes the estimated releases for each of the potential mechanisms.

Release Mechanism	Estimated Release
Dredging (clamshell)	<2.0 percent
Transport	Negligible
Water column (disposal at surface)	<2.1 percent
Total	<4.1 percent

175. Based on conservative estimates of mass release, the CAD alternative with surface disposal and conventional clamshell dredging is within the 5-percent performance objective for all parameters. Implementation of control measures to reduce sediment resuspension and contaminant release during the dredging process is a viable control measure to reduce the total mass release.

Feasibility Determination

176. Use of the proposed CAD site without lateral confinement is feasible if the dredged material mound will form and spread with slopes of 1 to 100 relative to the bottom slope or steeper (approximate angle of repose of 1V on 30H). However, it should be stressed that CAD has not been attempted at these depths, and there are some uncertainties associated with the placement of the CAD mound on a sloping bottom. Therefore, monitoring during placement of the contaminated material and cap should be conducted for both disposal phases to ensure that material behavior and mound configuration are constructed in accordance with the final design. If monitoring of the initial phase indicates that placement of material or cap is not satisfactory, construction of a berm at the site, placement of additional capping material, or shifting disposal operations to an alternate site could be considered as contingencies. Incorporation of a confining berm as a part of the design is considered an additional measure of conservatism.

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177. Precise placement of the material during the entire CAD operation will be important. The disposal barges used for placement of the contaminated material should be stationary during the release of each dump. This will assist in keeping the dredged material mass in a clumped condition during descent and the resulting mound spread within the estimated limits. Control for the point of discharge should be incorporated in the plans and specifications. Taut-line buoy or real-time electronic positioning with onboard computer printout are possible methods that could be used. For the capping operation, electronic positioning would be appropriate for determining the rate of movement of the pipeline discharge.

178. The shifting of the CAD site to a deeper site has been proposed to avoid sensitive biological resources. If an alternate site is selected, consideration should be given to locating the site so that existing bottom topography is as flat as possible. This would serve to reduce or eliminate the uncertainties associated with CAD on a sloping bottom.

Monitoring Requirements

179. The following monitoring requirements are recommended for the CAD alternative:

a. Sediment resuspension and contaminant release during the dredging and transport operation.

b. Sediment remaining in suspension and contaminant release during placement.

c. Configuration and density of confining dike (if built), contaminated sediment in place, and cap.

d. Migration of contaminants through the cap.

e. Mound densification and cap erosion.

Monitoring plans are given in Appendix I.

PART VI: EVALUATION OF NEARSHORE DISPOSAL SITES

180. The physicochemical conditions controlling contaminant mobility for the nearshore disposal of dredged material will be a combination of those conditions occurring under aquatic and upland disposal. Three distinct physicochemical environments will develop after the filling of a nearshore dredged material disposal site that can be described as:

a. Upland - dry, unsaturated layer.

b. Intermediate - partially or intermittently saturated layer.

c. Flooded - totally saturated layer.

181. Initially, all of the dredged material will be saturated, anaerobic, and reduced when placed in a nearshore disposal site. After the filling operation is completed, the upper surface layer of dredged material above the high-tide elevation will become upland. The layer of dredged material between the high- and low-tide elevations will become an intermediate layer with a moisture content varying between saturated and unsaturated. The degree of moisture will depend on the rate of water movement in, through, and out of this layer. The layer of dredged material at and below the low-tide elevation will remain saturated. The potential pathways of contaminant migration and the three physicochemical environments that would develop at a nearshore disposal site are illustrated in Figure 17.

182. The test protocols for predicting contaminant mobility at nearshore disposal sites should address the pathways for contaminant migration illustrated in Figure 17. The contaminant migration pathways and associated test protocols for the Everett nearshore dredged material disposal sites are tabulated as follows.

Pathway of Contaminant Migration	Test Protocol
Effluent discharge	Modified elutriate test
Surface runoff quality	Surface runoff test
Leachate	Leachate test
Seepage	
Soluble diffusion, seepage	
Soluble convection via tidal pumping	
Capillary	
Mobility between layers	
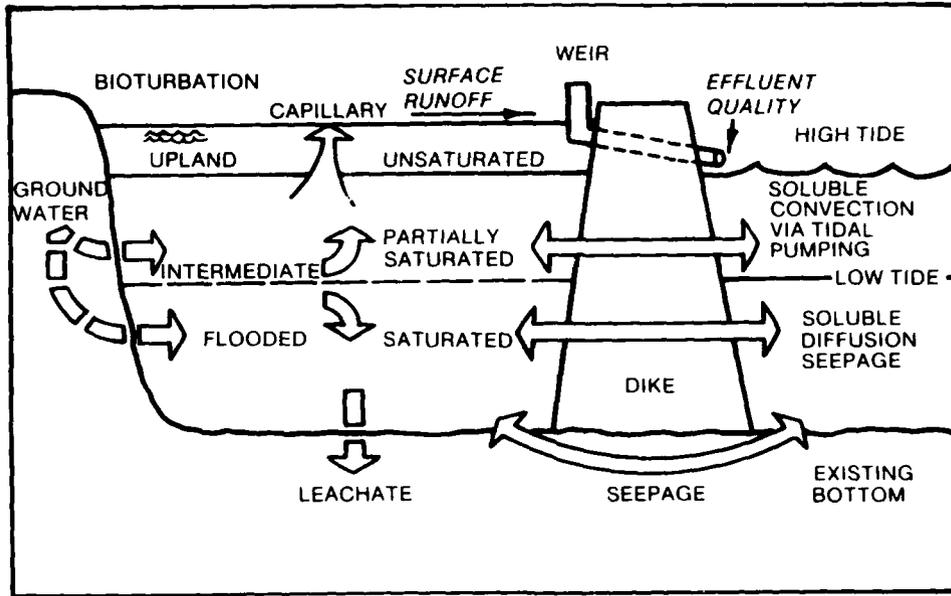
Figure 17. Nearshore disposal site migration pathways

The results of these test protocols provide appropriate information to determine the requirements for controls at the nearshore disposal sites.

Testing Protocols and Contaminant Pathways

183. When dredged material is placed in an upland or nearshore environment, drastic physicochemical changes occur. As soon as the dredged material is placed in a containment area and exposed to the atmosphere, oxidation processes begin. The influent slurry water from a hydraulically dredged site will initially be dark in color and reduced with little oxygen as it is discharged into the containment area. Mechanically dredged sediments will have sediment pore water that is dark in color and reduced initially. As the slurry water passes across the confined disposal site and approaches the discharge weir, the water becomes oxygenated and will usually become light gray or yellowish light brown. The color changes indicate further oxidation of iron complexes in the suspended particulates as they move across the containment area. Once active disposal operations are completed, dredged material consolidation will continue to force pore water up and out of the dredged material, and it will drain toward the discharge weir. This drainage water will continue to become oxidized and lighter in color. After the surface pore

water has drained off, the surface of the dredged material will become oxidized and lighter in color. As the dredged material dries, the pores are emptied of water, and oxygen diffusing through the pore spaces will further oxidize the material. Salt will accumulate on the surface of the dredged material, especially on the edge of cracks that form during drying. Rainfall events will dissolve and remove the salt accumulations in surface runoff. This surface runoff can also include certain metal contaminants that become more soluble as the dredged material dries out. Organic complexes become oxidized and decompose rapidly during this drying process. Sulfide compounds become oxidized to sulfate salts, and pH may drop to very acidic conditions. This chemical transformation could release complexed contaminants to surface runoff, soil pore water, and leachate through the material. Contaminant mobility through the migration pathways illustrated in Figure 17 will be significantly controlled by the physicochemical changes that occur during drying and oxidation of the dredged material. Any test protocol used to predict contaminant mobility should account for the physicochemical changes occurring in the dredged material when placed in the specific disposal environment.

Confined Disposal Design Requirements

184. Basic design requirements for storage of the dredged material and retention of solids during the disposal process are similar for sites constructed in nearshore or upland areas. Requirements for volumetric storage, minimum surface area, effluent suspended solids, and weir length were determined using the settling data described in Appendix E and procedures given in Palermo, Montgomery, and Poindexter (1978). These requirements would also apply if hydraulic rehandling from barges was used for placement of material in confined sites. For mechanical placement, the requirements would be conservative.

Volumetric requirements

185. Volumetric requirements for confined disposal of 800,000 cu yd of contaminated sediments are dependent on the dredging method and rate and the compression settling characteristics of the sediment. For representative hydraulic dredged sizes and dredging rates and settling characteristics described in Appendix E, the total volumes occupied by the dredged material at the completion of the disposal operation are tabulated below. These

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volumetric requirements were calculated assuming dredge operation of
12 hr/day.

Dredge Size, in.	Volumetric Storage, cu yd
12	964,000
24	1,154,000
36	1,278,000

186. Total dike heights required to contain these volumes will be a
function of the area diked. Additional dike height is required to accommodate
a minimum ponding depth of 2 ft and a minimum of 2 ft freeboard. The feasi-
bility of constructing the dikes to the total required height will be depen-
dent on geotechnical evaluations.

Surface area requirements

187. Available surface area of confined disposal sites is an important
factor in their capability to contain dredged solids and associated contami-
nants. Surface area requirements for effective settling are a function of the
dredging flow rate and the zone settling characteristics of the sediments.
The minimum surface areas required for representative dredge sizes are tabu-
lated below.

Dredge Size, in.	Minimum Surface Area, acres
12	28
24	92
36	196

Effluent suspended solids

188. The effluent suspended solids concentrations are dependent on the
dredging flow rate, the effective retention capacity of the disposal site, and
the flocculent settling characteristics of the sediments. The effluent sus-
pended solids concentrations for the minimum surface areas in the preceding
paragraph and corresponding dredge sizes are tabulated below. These values
are computed assuming that a minimum ponding depth of 2 ft is maintained
during the disposal operation.

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Dredge Size, in.	Effluent Suspended Solids, mg/ℓ
12	69
24	82
36	112

Weir design

189. Effective weir length required to discharge the carrier water without resuspension is a function of the dredging flow rate and the settling characteristics of the sediments. Effective weir lengths required for representative dredge sizes are tabulated below.

Dredge Size, in.	Effective Weir Length, ft
12	15
24	55
36	120

Effluent controls

190. Based on the modified elutriate test results, no controls are required for removal of dissolved contaminants if mixing to a dilution factor of approximately 13 can be achieved within a mixing zone of acceptable size. Contaminants associated with the suspended solids in the effluent can be controlled by good site design and operation for retention of suspended solids. Mass release of contaminants during filling operations, including both dissolved and particle-associated, was calculated to be less than 1 percent for all parameters except PCB 1254 with a loss of 3.2 percent. If mass release from other sources causes the 5-percent performance goal to be exceeded, chemical clarification of effluent to remove additional suspended solids and associated contaminants could be required as a control measure.

Surface runoff controls

191. Based on the results of surface runoff testing, no controls are required for removal of dissolved contaminants if mixing to a dilution factor of approximately 18 can be achieved within a mixing zone of acceptable size. Mass release of contaminants in kilograms from a representative 100-acre disposal site and representative storm events was calculated as described in

Appendix B. When compared with the total mass of sediment placed in the confined site, these values are negligible.

Surface cap

192. Although no surface cap of clean material is required as a control measure for surface runoff quality, placement of such a cap is recommended for the confined disposal alternatives. A surface cap of clean material would ensure that no problems would arise with contaminant uptake by plants or animals that might colonize the site, or with any future use of the site.

Leachate controls

193. There are potential problems with metals release in both the anaerobic and aerobic leachate. Under aerobic conditions there is a possibility that the attenuation capacity of the underlying soils may be exceeded if metal mobilization is as high as expected. This condition would indicate the potential need for restrictions to be placed on leachate/seepage generation from Everett Harbor sediment. Site-specific factors will determine the type of leachate control strategy that can be implemented. Potential leachate control strategies include site selection, site controls (both chemical and physical), and dredged material modification (fixation of contaminants, liming the sediment, etc.). Specific leachate control technologies, if needed, cannot be recommended or designed until a site is identified because of the site-specific nature of leachate controls.

Nearshore Disposal Site Descriptions

194. Two nearshore sites are being considered: (a) the East Waterway site and (b) the Snohomish Channel site. The locations of the sites are shown in Figure 5. These sites, as well as site information, were provided by the Navy from a larger list of alternatives. The Navy is considering several alternative design and operational scenarios for these sites. Two of the proposed scenarios considered most representative were selected for evaluation in this report: (a) East Waterway site with a 12.9-acre configuration to be used in combination with the Snohomish Channel site in a 100-acre configuration, and (b) Snohomish Channel site to be used alone in the 155-acre configuration.

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The following descriptions of the sites and proposed design and operational
conditions are based on information provided by the Navy.*

East Waterway site

195. The scenario evaluated for the East Waterway nearshore site is
illustrated in Figure 18. Due to the limited area at the East Waterway, the
extreme dike height requiring very expensive construction techniques, and the
infringement on turning areas for the homeport ships, the alternative with the
smallest surface area was considered for this evaluation. This option would
use a 12.9-acre surface area for the disposal site and would have a volumetric
capacity of 287,000 cu yd below el +7 for contaminated sediments. Once this
capacity is filled, the remainder of the contaminated sediments would be dis-
posed of in the Snohomish Channel site (100-acre configuration). Site-
specific hydrogeological data are limited since the East Waterway site has not
previously been investigated as a dredged material disposal site. Containment
structures will have to be constructed prior to dredged material placement.
This disposal site is seaward of the northerly and easterly shorelines of the
East Waterway. Site elevations vary from +15 to -36 mllw. Historically the
site has been used for shipping and log handling. Along the easterly shore-
line are the Scott Paper Company industrial facilities and the Naval Reserve
Center Pier and facilities. The site receives waters from upland storm drain
systems and from the Scott Paper Company treatment plant. The outfall system
from the treatment plant is located in the northeast portion of the disposal
area.

196. In general, the subsurface conditions at this site are anticipated
to be similar to those encountered across the general homeport site. Soils
consist of deltaic deposits of silty sand, sandy silt, and clayey silt, and
have moderate to low strength and moderate to high compressibility. Ultimate
settlement of the surface for the filled area is estimated to range between
8 and 12 ft with as much as 2 ft occurring during construction. These values
include both foundation settlement and consolidation of the dredged sediments.

197. Studies to date on the contaminated sediments indicate that the
contaminants are sediment bound as long as the sediments are maintained in a
saturated condition. For this reason, a criterion for disposal in a nearshore
site such as East Waterway (and Snohomish Channel) is that the upper surface

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* ABAM Engineers, Inc., 1986, op. cit.
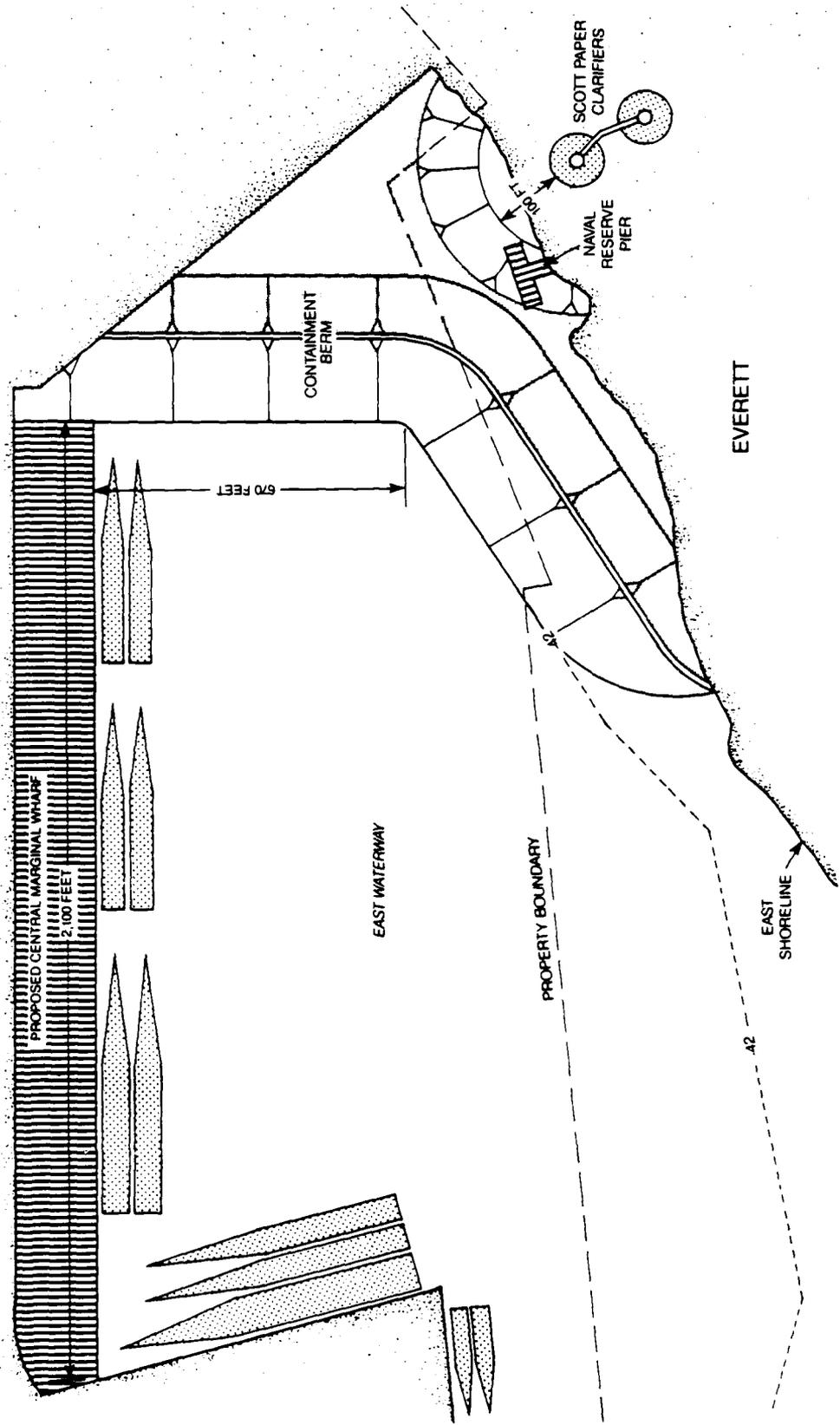
Figure 18. East Waterway nearshore disposal site option (ABAM Engineers, Inc. 1986)

PROPOSED CENTRAL MARGINAL WHARF
2,100 FEET

670 FEET

CONTAINMENT BERM

EAST WATERWAY

PROPERTY BOUNDARY

A2

EAST SHORELINE

A2

EVERETT

SCOTT PAPER CLARIFIERS

NAVAL RESERVE PIER

100 FT

of contaminated material be placed no higher than the ground-water elevation or midtide level, even when deposited as a pipeline slurry.

198. The ground-water level is estimated to be at or above an elevation of 7 ft* and is essentially independent of tidal fluctuation. This level is only an estimate based on ground-water levels observed in nearby wells. The level is expected to vary across the disposal site and is influenced to some degree by both offshore conditions and onshore flow from the east. Typical ground-water elevations across the Norton Avenue terminal range from 10 to 14 ft. The levels decrease to about 7 ft near the shoreline and throughout the central and south moles. Therefore, a level of 7 ft would be expected in outward portions of the disposal site, with higher levels expected while moving inland. The approximate mean tide level was also used as the design contaminated disposal level at a similar nearshore disposal site, the T-91 offshore confined disposal site for contaminated soils at the Port of Seattle. The capacity of the East Waterway site will also be a function of settlement that will occur during placement of the dredged material. Assuming relatively rapid sedimentation, the capacity of the East Waterway site may be determined assuming 1 ft of foundation settlement during placement.

199. Several concepts for the disposal of sediments using the East Waterway and for constructing the retaining structure are being considered. These concepts vary from placing all of the contaminated and uncontaminated sediments in the East Waterway site, to placing only part of the contaminated sediments and using the Snohomish Channel site and CAD site for the remaining dredged material. All concepts have +19 mllw as finished grade shoreward and -42 mllw as final dredged depth seaward. The general plan alignment is per-pendicular to the northern end of a further central marginal wharf, east 630 ft, thence southeast to the existing shore. Structural center-line posi-tion was dictated by maintaining -42 mllw depth for the full 630 ft. The length of the central marginal wharf is assumed to be 2,100 ft.

200. Based on available data, the following assumptions regarding the design and operation scenario for the East Waterway site are made:

a. Debris will be removed by a barged-mounted clamshell and eventually transported by truck to an upland disposal site.

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* Hart Crowser and Associates, Inc., 1986, ibid.

b. Contaminated and uncontaminated sediments will be removed with a hydraulic pipeline dredge (effective flow rate restricted to 5 cfs).

c. Containment height will be at el +20 (mllw).

d. Freeboard will be 2 ft.

e. Ponding depth is 2 ft, minimum (1 ft was proposed).

f. The contaminated sediments will be placed below +7 (mllw) and uncontaminated material to +18 (mllw).

g. The foundation settling during dredged material placement is 2 ft.

h. The amount of contaminated material dredged from the project for disposal in the East Waterway site was 255,300 cu yd.

i. The volume occupied by the contaminated sediments is 287,000 cu yd in the East Waterway disposal site.

j. A 90-ft-thick cap of clean material is placed on the contaminated material.

k. The surface area is 12.9 acres.

Snohomish Channel site

201. Several alternatives are being considered for disposal in the Snohomish nearshore sites. For the purposes of this report, it is assumed that the total amount of contaminated material to be dredged is 928,000 cu yd. The amount remaining after disposal in the 12.9-acre East Waterway site would be placed in a 100-acre Snohomish Channel site. If used alone, the Snohomish Channel site would encompass 155 acres. The site plan with dike alignments is illustrated in Figure 19. The site is part of a water tidal mudflat. The adjacent areas have been filled for industrial and recreational development. The Soil Conservation Service's "Snohomish County Soil Survey, 1947" shows the site as "coastal beach." According to this document, "the soil type consists of gray sand and gravel forming sloping beaches...and is subject to continual washing by waves during periods of storm or high tide." The site has received deposits of sawdust, bark, and other materials as a result of log rafting and periodic waste dumping in recent years. Two basic types of vegetation exist on the site determined by the topography. The zone from the fill to the mudflat supports a band of salt marsh vegetation while the mudflat supports a typical algae vegetation.

202. The subsurface conditions at this site are anticipated to be similar to those encountered across the general homeport site. The soils consist of deltaic deposits of silty sand, sandy silt, and clayey silt. These soils
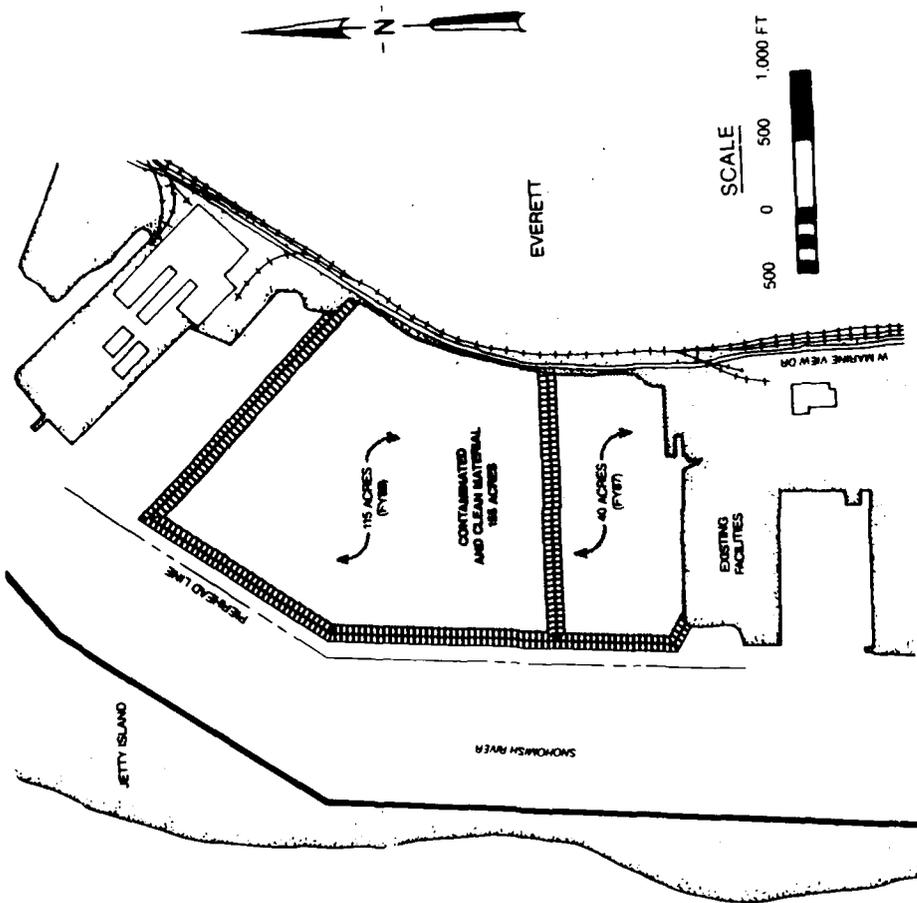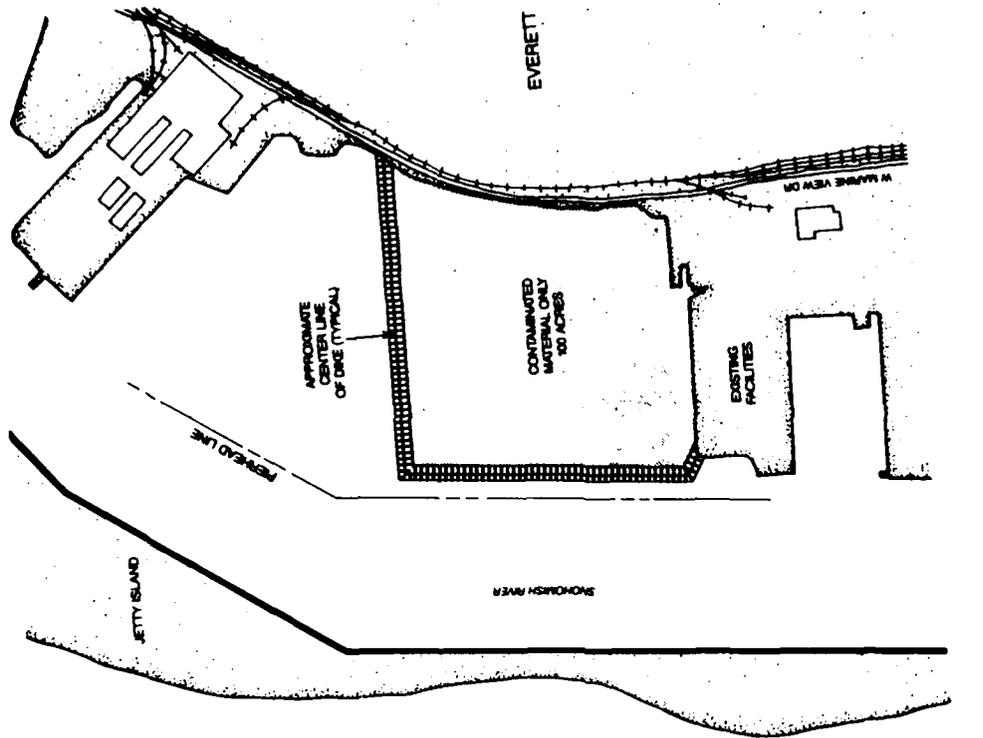
Figure 19. Dike layouts for Snohomish Channel nearshore disposal sites (ABAM Engineers, Inc. 1986)

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have moderate to low strength and moderate to high compressibility. The design may be similar to the Norton Avenue terminal fill, which included a waste rock containment dike. The dikes are anticipated to have slopes of 2H:1V adjacent to the river channel and 1.5H:1V elsewhere. The dikes will probably be set back from the river channel by at least 25 ft. Due to uncertain conditions at this site, settlement will not be considered for capacity determination. As with the East Waterway site, capacity will be partially dependent on the expected ground-water levels. Data from the Norton Avenue terminal site were used to estimate expected conditions. The wells in the area indicate ground-water elevations between 10.5 and 14.5 ft over a year-long monitoring period. Again, the levels did not appear to be significantly influenced by tidal fluctuations or mean tide levels. It has been assumed that the water levels at the Snohomish Channel site will be above el 10 ft over most of the site. It is possible that water levels near the river channel could more closely reflect the mean tide level near an elevation of 7 ft. In addition, the fine-grained nature of the expected fill materials may, through capillary action, maintain saturation a few feet above the hydrostatic water levels. This was seen at the Norton Avenue terminal site as the site remained saturated to the surface for as long as 2 years following construction as the soils consolidated.

203. Depending on the disposal alternative used, several containment diking concepts are being considered. One dike alignment concept will provide for a minimum of 155 acres of containment, while another concept provides 100 acres. These dike layouts are illustrated in Figure 19. The dikes will be constructed to elevation +21 from imported materials. The effective dike height will be +20 (mllw), which assumes sufficient fine material in the narrow top of the dike to control turbid leaching from the settlement pond. Dike slopes will be constructed at 2H:1V or steeper.

204. Based on the available data, the following assumptions regarding the design and operational scenarios for the Snohomish Channel site are made:

a. Debris will be removed by a barge-mounted clamshell and transported to the Snohomish site.

b. Contaminated and uncontaminated sediments will be removed with a 26-in. hydraulic pipeline dredge.

c. Contaminated sediments will be deposited in the site below +7.

d. Uncontaminated sediments will be deposited in the site to el +18.

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e. Dike height will be to el +20 ft msl.

f. Freeboard is 2 ft.

g. The ponding depth is 2 ft, minimum (1 ft was proposed).

h. The foundation settling during placement is 1 ft.

i. The amount of contaminated material dredged from the project
    for disposal in the Snohomish Channel site was 672,700 cu yd
    for the 100-acre configuration and 928,000 cu yd for the
    155-acre configuration.

j. The volume occupied by the contaminated sediments in the dis-
    posal site was 756,400 cu yd for the 100-acre configuration
    and 1,043,000 cu yd for the 155-acre configuration (calculated
    values).

k. The surface area is 100 acres or 155 acres, depending on the
    disposal option.

l. A 9-ft-thick cap of clean material is placed on the contami-
    nated material.

Feasibility Evaluation for Nearshore Disposal

East Waterway site

205. Solids retention and initial storage. The surface area assumed
for the East Waterway nearshore disposal site was 12.9 acres. It was assumed
that this site has volumetric capacity of 287,000 cu yd below el +7 ft. The
Automated Dredging and Disposal Alternatives Management System (ADDAMS) (Hayes
et al., in preparation), a family of computer programs, was used to determine
relationships between channel volumes, disposal site volumes, and effluent
suspended solids concentrations. Based on ADDAMS calculations, this site
volume could accommodate approximately 255,000 cu yd of in situ channel con-
taminated material. The dredging inflow must be limited to approximately
5 cfs to maintain effective zone settling due to the small surface area. This
could be most effectively accommodated by using a "Y" valve to apportion the
flow between the East Waterway and Snohomish Channel sites if the sites can be
filled simultaneously. If the sites cannot be filled simultaneously, the flow
to the East Waterway site must be limited by using a small dredge or by opera-
ting the proposed 26-in. dredge intermittently. Required weir length for this
site is approximately 10 ft. The predicted effluent suspended solids concen-
tration was 186 mg/ℓ. This estimate was used in the following section to
predict effluent quality for the disposal site during material placement.

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206. **Effluent quality.** Results of the modified elutriate test have been presented and discussed in Part II and Appendix A. Only 5 of 33 contaminants of concern were detected in the dissolved fraction by the modified elutriate tests. Only two parameters, Ni and PCB 1254, exceeded Port Gardner background concentrations and the Federal water quality criteria. Dilution to background or criteria can be achieved within a short distance of the effluent discharge.

207. The modified elutriate particle-associated concentrations and the predicted effluent suspended solids concentration of 186 mg/ℓ were used to determine the contaminant mass release from the East Waterway site effluent. A summary of the contaminant concentrations and mass release is presented in Table 7. The mass release in effluent can be effectively reduced by chemical clarification of the effluent.

208. **Surface runoff.** Results of the surface runoff tests are presented and discussed in Part II and Appendix B of this report. Only contaminants associated with particles are of concern in the wet, unoxidized condition. However, for dry, oxidized conditions, runoff concentrations of dissolved Cd, Cu, and Zn exceed the water quality criteria, but can be diluted to criteria or standards within a short distance of the runoff discharge. The runoff water yearly mass releases for the 12.9-acre East waterway site are presented in Table 8 and are negligible.

209. Surface runoff tests also indicate that very high suspended solids concentrations can be anticipated in surface runoff from aerobic material. Surface runoff should therefore be retained in a pond until suspended solids have settled out to the greatest degree possible. This will effectively reduce any mass release associated with the runoff. All nearshore disposal alternatives propose the placement of contaminated material below the ground-water level and the placement of at least 9 ft of clean material on top. Placement below the ground-water level would tend to keep the material from oxidizing on the surface, reducing the potential of soluble releases in runoff. The installation of the 9-ft cap would act as an effective control measure to control any long-term contaminant release from surface runoff at the East Waterway site and will prevent any chance of uptake by plants or animals that may colonize the site. If possible, this surface cap should be placed a few months after placement of the contaminated material.

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Table 7

Summary of Effluent Concentrations and Mass

Release for the East Waterway Site

Parameter	Dissolved Concentration mg/ℓ	Site Water Concentration mg/ℓ	USEPA Quality Criteria, mg/ℓ		Effluent Concentration mg/ℓ	Mass Release %
			Chronic	Acute		
Copper	0.006	0.007	0.004	0.023	0.006	0.05
Nickel	0.018	0.007	0.007	0.140	0.018	0.6
Cadmium	0.0002	0.0006	0.0045	0.059	0.0002	0.04
Chromium	0.003	0.004	0.018	1.2	0.024	0.59
PCB 1254	0.0004	<0.0002	0.00003	0.00003	0.0012	4.5

Table 8

Estimates of Yearly Mass Release of Filtered Contaminants

from Surface Runoff of Dry, Oxidized Sediment for

East Waterway Site

Parameter	Mass Release kg	Mass Release %
Cd	0.032	0.002
Cu	0.012	0.00003
Zn	0.598	0.0008
Pb	0.004	0.00002

210. _Leachate._ The predicted maximum leachate values are presented in Table 9. These values are based on the anaerobic batch leach tests. The anaerobic values are being used since all disposal alternatives assume that the contaminated sediments will be placed below the water table, resulting in a saturated, anaerobic environment. The USEPA Hydrologic Evaluation of Landfill Performance (HELP) model (Schroeder et al. 1984) was used to predict the amount of leachate that would be expected from the 12.9-acre East Waterway site. This model assumed a 9-ft cap with good vegetation and local climato-logical data. The volume of leachate and the predicted quality were used to determine the mass release of contaminants from the East Waterway disposal site. The percent mass release of contaminants from leachate for the East Waterway site is negligible. The results are presented in Table 9.

211. Since drinking water standards for Pb and Cr were exceeded by the test results, a regional authority decision may require some type of control to prevent any contaminant migration because of the possibility of deteriora-tion to potential receptors. Discussions with District personnel indicate that there are no drinking water wells in the area, nor any sensitive ecolog-ical areas. Also, indications are that the underlying sediments for the East Waterway site are already contaminated. If the Regional Authority Deci-sion (RAD) determines that a control would be warranted, several control options are available. The site may be lined with a synthetic or natural liner. A capping system to prevent infiltration could also be installed in concert with the liner. Leachate collection and treatment in place of lining and capping could also be considered; however, Cu and Pb concentrations from the leaching tests are increasing over time, which would necessitate long-term operation of a leachate treatment system and the associated long-term expense of operation and maintenance. In situ stabilization of the sediments after disposal could also be considered as a remedial measure should contaminant release increase in the future. Stabilization during disposal operations to fix the entire slurry mass and chemical admixing to contain specific contaminants are possible control options; however, any solidification/ stabilization process would be expensive.

Snohomish Channel site

212. _Solids retention and initial storage._ If 255,300 cu yd of contam-inated sediments will be dredged and placed in the East Waterway, the

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Table 9

Predicted Maximum Leachate Values and Mass Release for the

East Waterway Site

| | | USEPA Water Quality Criteria, mg/ℓ | | Drinking Water Standards, mg/ℓ | | Mass |
Parameter	Predicted Leachate Value, mg/ℓ	Chronic	Acute	Federal	Washington State	Release %
As	0.039	--	--	0.05	0.05	0.0007
Cd	0.010	0.0045	0.059	0.010	0.010	0.0001
Cr	0.080	0.018	1.2	0.05	0.05	0.000004
Pb	0.058	0.025	0.668	0.05	0.05	0.001
Zn	0.181	0.058	0.170	5.0	5.0	0.00007
Cu	0.096	0.004	0.023	--	--	0.0001
Ni	0.052	0.007	0.140	--	--	0.002
PCB 1254	0.0036	0.00003	0.00003	--	--	0.010

remaining 672,700 cu yd of contaminated sediment will be dredged and placed in the Snohomish Channel site (100-acre configuration). Based on ADDAMS calculations, this channel volume would occupy 756,400 cu yd in the Snohomish Channel site. For the 155-acre configuration, the total of 928,000 cu yd of contaminated sediments would occupy 1,043,000 cu yd in the disposal site. Both the 100- and 155-acre configurations could accommodate the anticipated flow rate for a 26-in. dredge. The required weir length for this site is approximately 60 ft (for both configurations). Predicted effluent suspended solids concentrations were 300 mg/ℓ for the 100-acre configuration and 236 mg/ℓ for the 155-acre configuration. These estimates were used in the following section to predict effluent quality from the disposal sites during material placement.

213. Effluent quality. The results of the modified elutriate test were discussed for the East Waterway site and are applicable to the Snohomish Channel site. The predicted contaminant concentrations and the predicted effluent suspended solids concentrations of 300 mg/ℓ and 236 mg/ℓ for the two possible

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configurations were used to determine the contaminant mass release from the Snohomish Channel site effluent. Summaries of the contaminant concentration and mass release for the 100- and 155-acre configurations are presented in Tables 10 and 11, respectively.

214. The dissolved concentrations of contaminants of concern are low, and dilution to background concentration or criteria could be accomplished within a short distance of the disposal operation. The mass release of contaminants in the effluent can be effectively reduced by chemical clarification.

215. **Surface runoff.** Results of the surface runoff test for dissolved parameters were discussed for the East Waterway site and apply to the Snohomish site. Release of dissolved contaminants can be diluted to background or criteria within a short distance of the discharge. The runoff water quality and yearly mass releases for the 100- and 155-acre Snohomish Channel site configurations are presented in Tables 12 and 13, respectively, and are negligible. As with the East Waterway site, retention of runoff in a pond prior to release will reduce any mass loss associated with the suspended particles. Placement of contaminated material below the water table elevation and placement of a surface cap as proposed will be effective control for long-term release from surface runoff and will prevent uptake by plants and animals.

216. **Leachate.** The predicted maximum leachate values and mass releases for the Snohomish Channel site 100- and 155-acre configurations are presented in Tables 14 and 15, respectively. As with the East Waterway site, contaminated dredged material will be placed below the water table and capped, resulting in a saturated, anaerobic environment. For this reason, anaerobic batch leach test results were used in the evaluation. The HELP model was used to predict the amount of leachate that would be expected from each of the site configurations. The model assumed a 9-ft cap with good vegetation and local climatological data. The volume of leachate and the predicted quality were used to determine the mass release of contaminants from the Snohomish Channel site. The percent mass release of contaminants from leachate for the Snohomish Channel site is negligible.

217. Since drinking water standards for Pb and Cr were exceeded by the test results, a RAD may require some type of control to prevent any contaminant migration because of the possibility of deterioration to potential receptors. Discussions with District personnel indicate that there are no

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Table 10

Summary of Concentration and Mass Release

of Site Effluent for Snohomish Channel (100 acres)

Parameter	Dissolved Concentration mg/ℓ	Site Water Concentration mg/ℓ	USEPA Quality Criteria, mg/ℓ Chronic	Acute	Effluent Concentration mg/ℓ	Mass Release %
Copper	0.006	0.007	0.004	0.023	0.006	0.05
Nickel	0.018	0.007	0.007	0.140	0.018	0.6
Cadmium	0.0002	0.0006	0.0045	0.059	0.0002	0.04
Chromium	0.003	0.004	0.018	1.2	0.035	0.91
PCB 1254	0.0004	<0.0002	0.00003	0.00003	0.0017	6.6

Table 11

Summary of Concentration and Mass Release

of Site Effluent for Snohomish Channel (155 acres)

Parameter	Dissolved Concentration mg/ℓ	Site Water Concentration mg/ℓ	USEPA Quality Criteria, mg/ℓ Chronic	Acute	Effluent Concentration mg/ℓ	Mass Release %
Copper	0.006	0.007	0.004	0.023	0.006	0.05
Nickel	0.018	0.007	0.007	0.140	0.018	0.6
Cadmium	0.0002	0.0006	0.0045	0.059	0.0002	0.04
Chromium	0.003	0.004	0.018	1.2	0.028	0.73
PCB 1254	0.0004	<0.0002	0.00003	0.00003	0.00145	5.4

Table 12

Estimates of Yearly Mass Release of Filtered Contaminants
from Surface Runoff of Dry, Oxidized Sediment for the
Snohomish Channel Site (100 acres)

Parameter	Mass Release kg	Mass Release %
Cd	0.388	0.006
Cu	0.150	0.0001
Zn	7.220	0.0025
Pb	0.043	0.00005

Table 13

Estimates of Yearly Mass Release of Filtered Contaminants
from Surface Runoff of Dry, Oxidized Sediment for the
Snohomish Channel Site (155 acres)

Parameter	Mass Release kg	Mass Release %
Cd	0.388	0.006
Cu	0.150	0.0001
Zn	7.220	0.0025
Pb	0.043	0.00005

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Table 14

Predicted Maximum Leachate Values and Mass Release for the

Snohomish Channel Site (100 acres)

	Predicted Leachate	USEPA Water Quality Criteria, mg/ℓ		Drinking Water Standards, mg/ℓ		Mass Release
Parameter	Value, mg/ℓ	Chronic	Acute	Federal	Washington State	%
As	0.039	--	--	0.05	0.05	0.0032
Cd	0.010	0.0045	0.059	0.010	0.010	0.0006
Cr	0.080	0.018	1.2	0.05	0.05	0.00002
Pb	0.058	0.025	0.668	0.05	0.05	0.0001
Zn	0.181	0.058	0.170	5.0	5.0	0.0004
Cu	0.096	0.004	0.023	--	--	0.009
Ni	0.052	0.007	0.140	--	--	0.0001
PCB 1254	0.0036	0.00003	0.00003	--	--	0.0005

Table 15

Predicted Maximum Leachate Values and Mass Release for the

Snohomish Channel Site (155 acres)

	Predicted Leachate	USEPA Water Quality Criteria, mg/ℓ		Drinking Water Standards, mg/ℓ		Mass Release
Parameter	Value, mg/ℓ	Chronic	Acute	Federal	Washington State	%
As	0.039	--	--	0.05	0.05	0.0036
Cd	0.010	0.0045	0.059	0.010	0.010	0.0007
Cr	0.080	0.018	1.2	0.05	0.05	0.0001
Pb	0.058	0.025	0.668	0.05	0.05	0.0069
Zn	0.181	0.058	0.170	5.0	5.0	0.0004
Cu	0.096	0.004	0.023	--	--	0.008
Ni	0.052	0.007	0.140	--	--	0.0130
PCB 1254	0.0036	0.00003	0.00003	--	--	0.050

drinking water wells in the area or sensitive ecological areas. However, in comparison to the East Waterway, the underlying sediments at the Snohomish Channel disposal site are relatively clean. Without additional data, a determination as to the possibility of a ground-water mixing zone to provide the necessary dilution cannot be made. The relatively shallow configuration of the Snohomish Channel site would make installation of a liner, either natural or synthetic, a more viable control option than with the East Waterway, if needed. Also, the large volume of contaminated materials placed in the Snohomish Channel would make the solidification/stabilization control option very expensive and probably not a viable alternative.

Summary of Mass Release for Nearshore Alternatives

218. Summaries of the total contaminant mass release for the nearshore alternatives are given in Tables 16-18. Mass release varies with parameter, with total mass release less than 2 percent for all parameters except PCB 1254. The largest contributor to mass release is effluent during filling operations.

219. The East Waterway site has a total mass contaminant release of 5.5 percent, slightly exceeding the performance goal of 5 percent. However, this holds true for only that portion of the material going to that site. The Snohomish Channel 100-acre configuration would accept the majority of the contaminated material when the East Waterway site would be used, and the total mass release for that site is 7.6 percent. The Snohomish Channel site 155-acre configuration has an estimated total mass release of 6.5 percent, also exceeding the performance goal.

220. Control measures would therefore be required to bring the total mass release within the performance goal for all nearshore alternatives. Chemical clarification is recommended as a cost-effective control measure to reduce the suspended solids and associated contaminants in the effluent. Controls during dredging to reduce the sediment resuspension and contaminant release are also a viable option.

Table 16

Summary of Mass Release for East Waterway Site

Parameter	Dredging	Site Effluent % Release	Surface Runoff % Release k	Leachate % Release	Total % Release
Cd	1.0	0.04	0.002	0.0001	1.0421
Cu	1.0	0.05	0.00003	0.0001	1.05013
Zn	1.0		0.0008	0.00007	1.00087
Pb	1.0		0.00002	0.001	1.00102
As	1.0		0.0007	1.0007	2.0
Cr	1.0	0.59		0.000004	1.59
Ni	1.0	0.6		0.002	1.602
PCB 1254	1.0	4.5		0.010	5.5

Table 17

Summary of Mass Release for Snohomish Channel Site (100 acres)

Parameter	Dredging	Site Effluent % Release	Surface Runoff % Release k	Leachate % Release	Total % Release
Cd	1.0	0.04	0.005	0.0006	1.0456
Cu	1.0	0.05	0.00009	0.009	1.05909
Zn	1.0		0.00002	0.0004	1.00042
Pb	1.0		0.00004	0.0001	1.00014
As	1.0			0.0032	1.0032
Cr	1.0	0.91		0.00002	1.91
Ni	1.0	0.6		0.0001	1.6001
PCB 1254	1.0	6.6		0.0005	7.6

Table 18

Summary of Mass Release for Snohomish Channel Site (155 acres)

Parameter	Dredging	Site Effluent % Release	Surface Runoff % Release k	Leachate % Release	Total % Release
Cd	1.0	0.04	0.006	0.0007	1.0467
Cu	1.0	0.05	0.0001	0.008	1.0561
Zn	1.0		0.0025	0.0004	1.0029
Pb	1.0		0.00005	0.0069	1.00695
As	1.0			0.0036	1.0036
Cr	1.0	0.73		0.0001	1.73
Ni	1.0	0.6		0.0130	1.613
PCB 1254	1.0	5.4		0.050	6.5

Monitoring Requirements

221. The following monitoring requirements are recommended for confined disposal:

   a. Sediment resuspension and contaminant release during the dredging and transport operations.

   b. Effluent quality during filling operations.

   c. Surface runoff during a storm event.

   d. Ground-water quality and quality of seepage through dikes.

Monitoring plans to meet these requirements are given in Appendix I.

PART VII: EVALUATION OF UPLAND ALTERNATIVES

Background

222. The information in this part stresses the applicability of test results in evaluating upland disposal alternatives. As for intertidal disposal, an upland disposal site may involve placement of material in one or more disposal environments. The sediment testing results described in this report are directly applicable in evaluating upland disposal alternatives.

223. An area for potential development of an upland site has been identified at Smith Island, north of the homeport area. It should be noted that this site was identified very late in the study, and all site information was provided by the Navy. Limited information regarding site conditions was available in September 1986. Further, a number of possible sizes and configurations for the upland site have been identified. Until a site configuration(s) is identified and additional data on site conditions are obtained, a site-specific evaluation for upland disposal similar to those performed for intertidal sites and described in the Disposal Alternatives report cannot be conducted. However, a description of the applicability of test results for representative upland disposal conditions is given in the following paragraphs. An effort has been made to apply data to the Smith Island site to the maximum extent possible.

Applicability of Test Results

Solids retention and initial storage

224. The configurations now under consideration for the Smith Island area vary from 35 to 89 acres in surface area. Data on required surface area for various dredge inflow rates, required volumetric storage capacities, and relationship of effluent suspended solids as a function of flow rate were presented in Part VI. This information is directly applicable to evaluation of sites at Smith Island. The allowable inflow rate to maintain effective solids retention and the required volumetric storage will be in direct proportion to the final surface area available for the site.

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Effluent quality

225. Comparisons of dissolved concentrations of contaminants in effluent as predicted by modified elutriate tests and water quality criteria are presented in Appendix A. These comparisons are valid for any of the upland site configurations now under consideration for Smith Island.

226. Mass release of contaminants in effluent is dependent on effluent suspended solids concentrations. Determination of mass release is therefore possible only for a specific set of site conditions. However, mass release in effluent would be similar to that determined for the intertidal sites under consideration. Based on the previous evaluations for the intertidal sites, controls for mass release in effluent would likely be required to limit the total mass release for the upland alternative to less than the 5-percent performance goal. As for the intertidal alternative, chemical clarification is the most effective control measure.

Surface runoff

227. The final surface of the contaminated sediments placed in an upland site could be at elevations either above or below the water table. Comparisons of dissolved and particle-associated concentrations of contaminants in surface runoff under both anaerobic and aerobic conditions with water quality criteria are presented in Appendix B. These comparisons are also valid for an upland evaluation including Smith Island.

228. Mass release of contaminants in surface runoff is directly proportional to surface area of the disposal site, since it can be assumed that rainfall occurrences would be the same for Smith Island as for the intertidal sites. Mass release was found to be negligible for the intertidal condition, and would similarly be negligible for the upland condition. As recommended for the intertidal site, placement of the contaminated material at elevations below the water table would minimize release of both surface runoff and leachate, and eventual placement of a surface cap would prevent long-term release.

Leachate

229. The leachate contaminant flux concentrations discussed in Part II and Appendix C are predictions of the concentrations of contaminants in leachate generated under anaerobic and aerobic conditions. However, the prediction of leachate impacts is a function of ground-water movement at the site under consideration. In nearshore or upland sites, various mechanisms such as

precipitation, differences in elevation, tidal pumping, etc., tend to drive ground-water movement. Movement of water from the dredged material mass into surrounding ground water can be inhibited by the presence of relatively impervious natural foundation soils, placement of surface covers to retard infiltration of precipitation, placement of liners to retard movement of leachate, etc. Even if leachate moves into surrounding ground water, the degree of impact will be determined by the degree of mixing that might occur in the ground water, adsorption of contaminants within the foundation soils, and the sensitivity and quality of surrounding ground water that may be impacted. All of the above considerations are highly site specific.

230. Depending on the site selected and site conditions, contaminated dredged material may be placed above or below the water table. If contaminated material is placed below the water table, the leachate characteristics may be estimated using anaerobic leaching test results. Leachate from material placed above the water table may be estimated using aerobic results.

231. The predicted leachate values for intertidal alternatives presented in the Disposal Alternatives report were based on preliminary anaerobic batch leach tests. Subsequent laboratory testing and evaluation yielded that the revised Cr and Pb now exceed the drinking water standards, Cd meets the drinking water standard of 0.010 mg/ℓ, and PCB has increased from 0.0002 to 0.0036 mg/ℓ. Although these values would proportionately increase their percent mass releases, the portion of mass release contributed by leachate to the total mass release was and is still negligible.

232. Since anaerobic leaching data for Pb and Cr exceeded the drinking water standards, a RAD may require some type of control to prevent any contaminant migration from material placed below the water table because of the possibility of deterioration to potential receptors. If the RAD determines that a control would be warranted, several control options are available. The site may be lined with a synthetic or natural liner. A capping system to prevent infiltrations could also be installed in concert with the liner. Leachate collection and treatment in place of lining and capping could also be considered; however, Cu and Pb concentrations from the leaching tests are increasing over time, which would necessitate long-term operation of a leachate collection and treatment system and the associated long-term expense of operation and maintenance. In situ stabilization of the sediments after disposal could also be considered as a remedial measure, should contaminant

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release increase in the future. Stabilization during disposal operations to fix the entire slurry mass or chemical admixing to contain specific contaminants are possible control options; however, any solidification/stabilization process would be expensive.

233. Aerobic leaching data indicate that Cd, Cr, and Pb exceed the drinking water standard by a much greater margin than the anaerobic test results. This may require a more extensive control measure for contaminated material placed above the water table than would be required for material placed below the water table. Again, site-specific conditions would dictate which type of control measure would be necessary. The possibility of a ground-water mixing zone to provide the necessary dilution may be possible. Also, a shallow configuration for the containment area would make the installation of a liner a more viable control option.

234. Depending on the size of the containment area, the amount of material to be dredged, and the site conditions, a practical disposal scenario would be to place the contaminated material below the water table, where the material would remain anaerobic, thereby releasing less contaminant. Cleaner material used as a surface cap could be placed above the water table.

Data Needs for Site-Specific Evaluation

235. Data requirements for site-specific evaluation of a specific confined upland disposal site are tabulated as follows:

a. Site location, area, and configuration.

b. Vegetative cover, precipitation, evaporation, and temperature data.

c. Drainage, topography, and tidal or hydrologic information.

d. Engineering and geological characteristics of foundation strata, including stratigraphy, depth to bedrock, depth to aquicludes, and depth to ground water.

e. Direction and rate of ground-water flow.

f. Foundation soil contamination.

g. Existing ground-water and/or surface water quality.

h. Typical cross sections of retaining dikes.

i. Potential receptors, sensitive ecological areas, and drinking water wells in the area.

Monitoring Requirements

236. The following monitoring requirements are recommended for upland disposal:

        a. Sediment resuspension and contaminant release during the dredging and transport operations.

        b. Effluent quality during filling operations.

        c. Surface runoff during a storm event.

        d. Ground-water quality and quality of seepage through dikes.

Monitoring plans to meet these requirements are given in Appendix I.

PART VIII: CONCLUSIONS AND FUTURE NEEDS

Conclusions

General

237. Based on the results of this study, the following general conclusions are made:

  a. Contained aquatic disposal (capping) of Everett Harbor sediments at the Deep Delta site is feasible. However, CAD at the water depth under consideration and placement of cap by hydraulic pipeline without lateral confinement has not yet been attempted.

  b. Confined disposal of Everett Harbor material at the Snohomish or a combination of the Snohomish and East Waterway sites is feasible and involves known and proven technology.

  c. Site-specific data are required for design of any of the alternatives under consideration.

238. The following conclusions are made regarding the overall sediment testing program conducted using the Everett Harbor composite sample:

  a. The physical classification of the WES composite sediment sample correlated well with independent analyses performed by others. The in situ channel density as indicated by the composite sample is within a range of values defined by independent testing.

  b. The average in-channel density of Everett Harbor sediment is not sufficiently defined for purposes of final disposal site design. Since the project involves one-time disposal, the economic site design will necessarily be such as to store only the required volume dredged. This will require that a more precise determination of the in situ density be made.

  c. The chemical composition of the composite sediment sample used for WES testing correlated well with independent analyses performed by others and was therefore considered representative of the contaminated sediments to be dredged.

  d. A series of environmental and related engineering tests were conducted on the Everett Harbor sediment, and no unusual problems in testing were encountered. Results of all tests were of sufficient reliability for use in the alternatives evaluation.

Dredging equipment

239. The following conclusions are made regarding the evaluation and selection of dredged equipment:

  a. The present proposal for both the CAD and intertidal alternatives involving use of conventional dredging equipment and

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techniques that have been successfully used in similar applications elsewhere is reasonable. Use of specialized dredging equipment due to the presence of contaminants is not considered necessary.

b. Clamshell dredging and transport in split-hull barges is considered the most compatible dredging technique for the contaminated sediments for the CAD alternative.

c. Hydraulic cutterhead dredging with direct pipelining transport is considered the best technique for dredging the uncontaminated (capping) material for the CAD alternative.

d. If a subaqueous berm is needed for the CAD alternative, clamshell dredging is considered the best technique for dredging associated with the berm construction.

e. Hydraulic cutterhead dredging with direct pipeline disposal is considered the best dredging technique for the intertidal alternatives.

f. The estimated release of contaminants in the dissolved form during dredging is negligible. Estimated mass release was considered equal to the mass sediment release, and was 2 percent for clamshell dredging and 1 percent for hydraulic cutterhead dredging.

g. Control measures during dredging to reduce sediment resuspension and contaminant release are possible options to reduce total mass release. Implementation of those control measures that involve minimal additional cost should be considered. Such measures might include use of an enclosed clamshell bucket, operational controls, and selecting dredging sequences from north to south in the waterway to the extent practicable.

Contained aquatic disposal

240. The following conclusions are made regarding evaluation of the CAD alternative:

a. The CAD alternative as now proposed involves level-bottom capping of contaminated sediments with uncontaminated sediment. This alternative is similar to conventional capping operations that have been successfully demonstrated at other locations. However, capping has not yet been attempted at the water depths proposed, nor has capping been attempted using hydraulic pipeline placement of the cap without subaqueous lateral confinement.

b. The CAD alternative should not be considered merely a variation of open-water disposal, but should be treated as an engineered structure with carefully considered design, care during construction, and monitoring to ensure that the design is adequate.

c. Capping effectiveness tests show that the Everett Harbor contaminated sediments should be capped with a minimum cap thickness of 80 cm to effectively isolate the material from the

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overlying environment. To allow for irregularity during placement, a 1-m cap thickness should be specified as an operational requirement.

d. Modeling results show that placement of a single bargeload of the contaminated sediments at the Deep Delta CAD site using surface disposal will result in an area of deposition on the bottom approximately 700 ft in diameter. Approximately 1.9 percent of the material will remain in suspension longer than 1,800 sec and was assumed to be a mass release.

e. Modeling results show that placement of the uncontaminated capping material using controlled surface discharge from a pipeline moving across the site would result in an area of deposition approximately 300 ft in width. Multiple passes of the pipeline would be required to accumulate the required cap thickness.

f. An analytical evaluation of the mound characteristics indicates that the total volume of contaminated and cap material would accumulate in a mound with bottom having a radius of approximately 2,400 ft and a final height of approximately 12 ft. Final cap thickness would be approximately 4 ft.

g. Standard elutriate testing indicated that contaminant release in dissolved form during placement of the contaminated material was below reference water concentration or criteria for most parameters. Dilution of concentrations for remaining parameters to background or criteria can be accomplished within a short distance of the placement operation. Mass release during placement was considered directly related to sediment release, and varied from 2.0 to 2.1 percent depending on the parameter. Since the performance standard for total mass release was not exceeded for the CAD alternative, no contaminant control measures are considered necessary during placement.

h. The estimated total mass release for the CAD alternative is 4.1 percent.

i. The dredging sequence as proposed shows an imbalance in the ratio of the volume of capping material to contaminated material in each phase with an 8 to 1 ratio for the first phase and a 2 to 1 ratio in the final phase. Options to correct this imbalance include dredging a larger volume of the contaminated material in the initial phase, providing some level of subaqueous confinement to reduce the impact of the imbalance, or adding additional cap material from other sources if a shortage is determined by monitoring.

j. A monitoring program for the CAD alternative should be implemented to include sediment resuspension and contaminant release during dredging and placement; configuration of the berm, mound, and cap during and after placement; and effectiveness of the cap.

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<u>k</u>. The results of the CAD evaluation in this study are generally
applicable to the deeper RAD CAD site. However, the Navy's
design needs and schedule precluded a detailed technical
evaluation of the RAD CAD site.

Intertidal disposal

241. The following conclusions are made regarding the intertidal dis-
posal alternatives:

<u>a</u>. Several options for using both the East Waterway and Snohomish
Channel sites were identified. Two options were considered
environmentally representative and were evaluated: (1) the
12.9-acre configuration for East Waterway to be used in com-
bination with the 100-acre configuration for the Snohomish
Channel site, and (2) the 155-acre configuration for the
Snohomish Channel site to be used alone.

<u>b</u>. Based on settling test results, the East Waterway site would
hold only 255,000 cu yd of the in situ contaminated material.
The remainder would be placed in the Snohomish Channel site.

<u>c</u>. Modified elutriate test results show that the dissolved con-
centrations of contaminants in the effluent discharged during
filling are below reference water concentration or criteria
for most parameters. Dilution of concentrations for remaining
parameters to background or criteria can be accomplished
within a short distance of the discharge. These results are
applicable to both sites.

<u>d</u>. Settling test and modified elutriate test results show that
the mass release in effluent varies depending on the
parameter. The maximum values were: 4.5 percent for East
Waterway, 6.6 percent for Snohomish (100 acres), and 5.4 per-
cent for the Snohomish (155 acres).

<u>e</u>. Surface runoff test results show that the dissolved concentra-
tions of contaminants in the runoff from a representative
storm event are below reference water concentration or
criteria for most parameters. Dilution of concentrations for
remaining parameters to background or criteria can be accom-
plished within a short distance of the discharge. These
results are applicable to both sites.

<u>f</u>. Surface runoff test results show that mass release of contam-
inants in runoff during a 1-year period with typical yearly
rainfall conditions is negligible. These results are appli-
cable to both sites. It is assumed that a surface cap would
be placed over the contaminated material within a year of dis-
posal to prevent long-term release from surface runoff and
uptake of contaminants by plants or animals that may colonize
the site.

<u>g</u>. Drinking water standards were exceeded in the leachate for
some parameters. Regional authority decisions regarding pos-
sible ground-water mixing zones or requirements for control

measures would necessarily depend on the final site selection and design.

h. An estimate of mass release in leachate based on modeling results and leachate test results showed that the mass release was negligible. These results are applicable to both sites.

i. The estimated total mass release for the intertidal alternatives is: 5.5 percent for the East Waterway, 7.6 percent for the Snohomish Channel (100 acres), and 6.5 percent for the Snohomish Channel (155 acres). Since the performance goal of 5 percent is exceeded, controls would be required to meet the standard. The most cost-effective controls would include reductions in sediment resuspension during cutterhead dredging and chemical clarification to reduce suspended solids and associated contaminants in the effluent during filling operations.

j. A monitoring program for intertidal disposal should include: sediment resuspension and contaminant release during dredging and transport, effluent quality during filling, surface runoff quality for a representative storm event, and ground-water quality using monitoring wells.

Upland disposal

242. The following conclusions are made regarding potential upland disposal alternatives:

a. The modified elutriate, surface runoff, and leachate testing results described in this study are directly applicable to evaluation of upland disposal alternatives. Estimates of mass release for specific sites will be dependent on site configurations and conditions.

b. A monitoring program for upland disposal should include the same elements as described for intertidal disposal.

Data Needs

243. Additional data are required for site-specific evaluations or designs. Specific data needs are summarized in the following paragraphs.

244. Data requirements for site-specific evaluation of contained aquatic disposal are as follows:

a. Site location, area, and configuration.

b. Bathymetry and water depths.

c. Temperature and salinity profiles including seasonal variations.

d. Engineering characteristics of the in situ bottom sediments.

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e. Measurements of current velocity and direction over at least one tidal cycle with predictions of seasonal maxima and minima.

245. Data requirements for site-specific evaluation of confined upland or nearshore disposal are as follows:

a. Site location, area, and configuration.

b. Vegetative cover, precipitation, evaporation, and temperature data.

c. Drainage, topography, and tidal or hydrologic information.

d. Engineering and geological characteristics of foundation strata, including stratigraphy, depth to bedrock, depth to aquicludes, and depths to ground water.

e. Direction and rate of ground-water flow.

f. Foundation soil contamination.

g. Existing ground-water and/or surface water quality.

h. Typical cross sections of retaining dikes.

i. Potential receptors, sensitive ecological areas, and drinking water wells in the area.

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APPENDIX A: ELUTRIATE TESTING

Testing Objectives

Standard elutriate

1. Standard elutriate tests were performed to estimate the dissolved contaminant release into the water column during open-water placement for the CAD alternative. These tests are designed to estimate dissolved contaminant release in dredge hoppers or pipelines, and do not consider mixing and dilution that occur during disposal operations. A second round of standard elutriate tests with analysis of total elutriate concentrations was performed to obtain qualitative data on the degree of contaminant release in the particle-associated fraction (mass release) during open-water placement for the CAD alternative. This testing procedure had not been previously performed and is not a laboratory-developed or field-verified testing procedure.

Modified elutriate

2. Modified elutriate tests were performed to estimate the contaminant concentrations in effluent discharged from confined sites during filling operations (either upland or intertidal). These tests are designed to estimate both dissolved and particle-associated contaminant concentrations in the effluent resulting from hydraulic placement of dredged material in confined sites. The modified elutriate test is considered a conservative estimate of contaminant release for material placed by mechanical means into a confined site.

Criteria

Water quality

3. The reference water and criteria for comparison of elutriate tests results were specified by the Seattle District. Test results were evaluated in terms of whether Port Gardner background concentrations were exceeded by the test results, and if so, to compare test results with Federal water quality criteria for the protection of saltwater aquatic life. The reference water and criteria are shown in Tables 1 and 2 of the main text. The background concentrations for Port Gardner were considered equal to those determined for the water sample collected during September 1985 for purposes of

A1

conducting the elutriate tests. The Port Gardner background concentrations and Federal water quality criteria are appropriately compared to the elutriate results for dissolved concentrations only. Since criteria were specified for all parameters that were detected in the elutriate tests, the evaluations from a water quality standpoint were straightforward comparisons of test results with the reference or criteria, with consideration of mixing as called for in the Decisionmaking Framework. The criteria as specified applied both to the evaluation of contaminant release in open water and effluent from confined sites (nearshore or upland).

Mass release

4. A performance goal of 5 percent for total mass release of contaminants from dredging and disposal was specified by the District. The term "mass release of contaminants" in this application refers to the total mass of in situ contaminants prior to dredging which is not placed in the disposal site or does not remain in the disposal site. A direct computation of mass release of contaminants was possible for effluent using modified elutriate results, settling data described in Appendix E, and assuming representative confined disposal conditions. Approximations of mass contaminant release during dredging and open-water placement for the CAD alternative were made based on estimates of the mass sediment release and standard elutriate data. Sediment release and contaminant release in the solids fraction are not completely equivalent, but the contaminants of concern are strongly bound to the sediment, primarily the fine-grained silt and clay fractions. Additionally, mass sediment release estimates include release of sandy material to which chemical contaminants tend not to bond. These estimates therefore allow an approximate basis of comparison for test results and disposal options.

Procedures

5. All elutriate tests were conducted in triplicate using the composite sediment sample and water samples collected as described in the main text of this report. Elutriate samples were analyzed for the trace metals, PCBs, PAHs, and 1- and 2-methylnaphthalene. These contaminants were identified as parameters of concern and roughly correspond to those detected in the bulk analysis of the composite sample.

A2

6. The standard elutriate tests were conducted using the procedure described in USEPA/USACE (1977). A schematic of this test procedure is shown in Figure A1. A second round of standard elutriate tests was conducted to compare dissolved and total concentrations in the elutriate samples. This round was conducted using the standard elutriate test procedure with the exception that the sample obtained after agitation and settling was split and analyzed for both dissolved and total concentrations.

7. The modified elutriate tests were conducted using the procedure described in Palermo (1985). A schematic of this test procedure is shown in Figure A2.

Results

8. The replicate mean results of the standard elutriate and modified elutriate tests are summarized in Tables A1 through A6 for those parameters that were detected. Results for the standard elutriate test (dissolved) reported in Table A1 are the highest replicate mean concentrations found in either of the two rounds of tests. All parameters were below detection in all the elutriate tests except for trace metals and PCB 1254.

9. The measurable contaminants were low in all the elutriate samples. At low analyte concentrations, analytical variability can mask the differences in dissolved and total concentrations that would normally be expected. Total standard elutriate concentrations were equal to or lower than filtered concentrations for copper, cadmium, chromium, mercury, and PCB 1254. The total results were elevated in comparison to dissolved results for nickel and lead. Similarly, total modified elutriate concentrations were equal to or lower than concentrations for the dissolved samples for cadmium, copper, and nickel. The total results were elevated in comparison to dissolved results for chromium and PCB 1254.
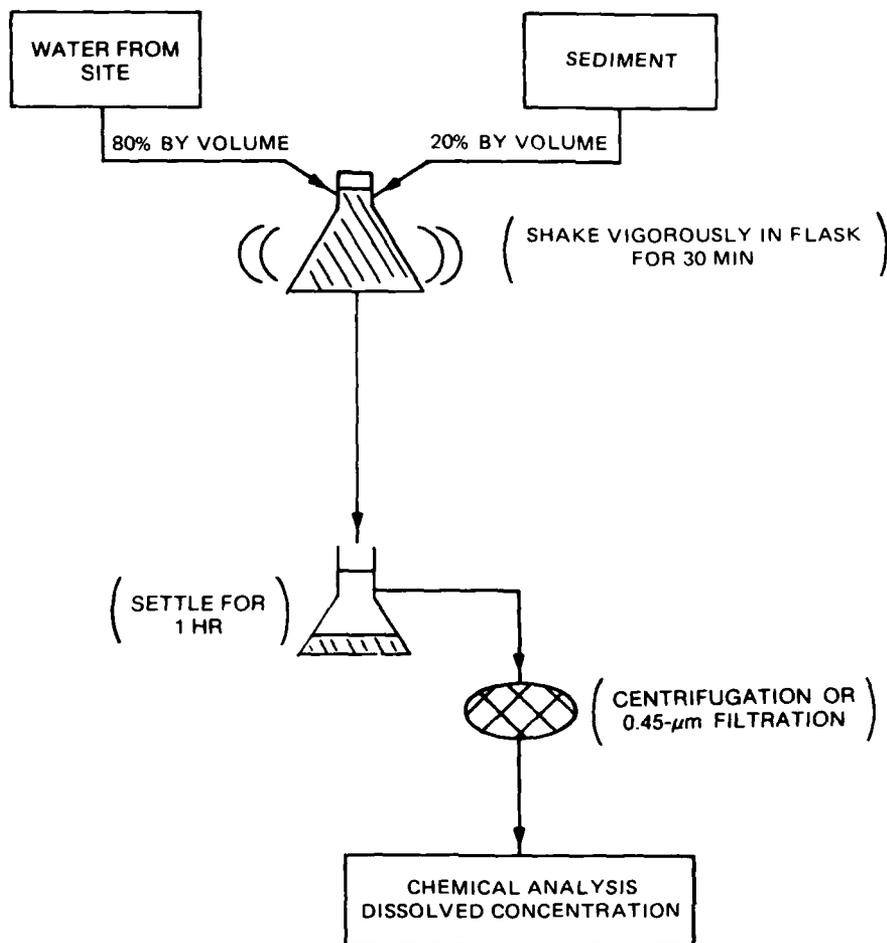
Figure A1. Schematic of standard elutriate test procedure

Interpretation and Need for Controls

Standard elutriate

10. Water quality. Standard elutriate test results (dissolved only) were compared with the background water concentrations and water quality criteria in accordance with the Decisionmaking Framework. The dissolved test results are an estimate of the dissolved release of contaminants during placement of dredged material in open water for the CAD alternative, prior to mixing and dilution. Nickel, cadmium, lead, chromium, and PCB 1254 exceeded the background concentrations. Cadmium and chromium concentrations were below both the chronic and acute exposure values give in the Federal water quality criteria. The only remaining parameters of concern, therefore, are nickel, lead, and PCB 1254.

A4
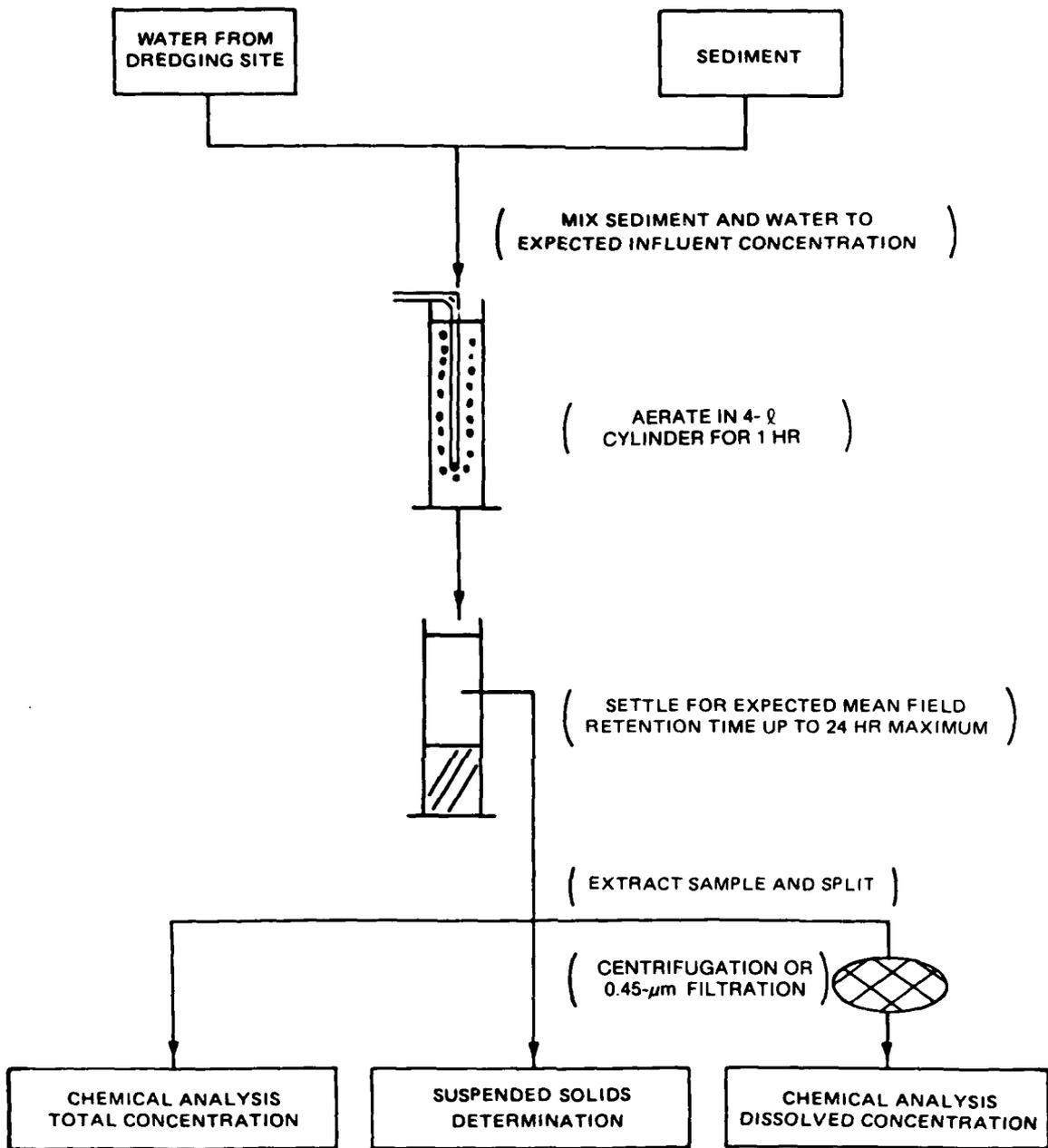
Figure A2. Schematic of modified elutriate test procedure

11. Nickel concentration was above the chronic exposure value given in the Federal water quality criteria, but was far below the acute exposure value. Since the chronic criteria for nickel is equal to the Port Gardner background concentration, the standard elutriate concentration cannot be diluted to the chronic criteria by mixing. However, the elutriate

Table A1

Summary of Dissolved Concentrations for Standard Elutriate Tests and Criteria

Parameter	Dissolved Concentration, ppm	Site Water Concentration, ppm	Federal Water Quality Criteria ppm		Remarks
			Chronic	Acute	
Copper	0.007	0.007	0.004	0.023	Test < background
Nickel	0.015	0.007	0.007	0.140	Test < acute criteria
Cadmium	0.003	0.0006	0.0045	0.059	Test < chronic criteria
Lead	0.028	<0.001	0.025	0.668	Test < background
Chromium	0.008	0.004	0.018	1.2	Test < chronic criteria
Mercury	0.0066	0.0067	0.000025	0.0037	Test < background
PCB 1254	0.0004	<0.0002	0.00003	0.00003	Dilution factor = 13

Table A2

Summary of Dissolved Concentrations for Modified Elutriate Tests and Criteria

Parameter	Dissolved Concentration, ppm	Site Water Concentration, ppm	Federal Water Quality Criteria ppm		Remarks
			Chronic	Acute	
Copper	0.006	0.007	0.004	0.023	Test < background
Nickel	0.018	0.007	0.007	0.140	Test < acute criteria
Cadmium	0.0002	0.0006	0.0045	0.059	Test < background
Chromium	0.003	0.004	0.018	1.2	Test < background
PCB 1254	0.0004	<0.0002	0.00003	0.00003	Dilution factor = 13

Table A3

Summary of Standard Elutriate Data and Mass Release

Parameter	Dissolved Concentration ppm	Total Concentration ppm	Mass Release %
Copper	0.007	0.007	--
Nickel	0.015	0.017	--
Cadmium	0.003	0.003	0.1
Lead	0.028	0.030	--
Chromium	0.008	0.005	--
Mercury	0.0066	<0.0002	0.2
PCB-1254	0.0004	0.0003	--

Table A4

Summary of Total Concentration and Mass Release
for Modified Elutriate Tests, East Waterway Site

Parameter	Bulk Sediment mg/kg	Inflow* mg/ℓ	Dissolved Modified Elutriate mg/ℓ	Total** Modified Elutriate mg/ℓ	Effluent† Concentration mg/ℓ	Mass Release %
Copper	73.4	11.01	0.006	0.003	0.006	0.05
Nickel	21.4	3.21	0.018	0.017	0.018	0.6
Cadmium	3.3	0.50	0.0002	0.0002	0.0002	0.04
Chromium	39.7	5.96	0.003	0.008	0.035	0.59
PCB 1254	0.25	0.0375	0.0004	0.0006	0.0017	4.5

* Based on an inflow concentration of 150 g/ℓ.
** Samples containing a mean suspended solids concentration of 29 mg/ℓ.
† Based on settling analysis for a 12.9-acre site, 5 cfs flow rate, resulting in an effluent suspended solids concentration of 123 mg/ℓ.

Table A5

Summary of Total Concentration and Mass Release

for Modified Elutriate Tests, Snohomish Channel (100 acres)

Parameter	Bulk Sediment mg/kg	Inflow* mg/ℓ	Dissolved Modified Elutriate mg/ℓ	Total** Modified Elutriate mg/ℓ	Effluent† Concentration mg/ℓ	Mass Release %
Copper	73.4	11.ʋ1	0.006	0.003	0.006	0.05
Nickel	21.4	3.ːi	0.018	0.017	0.018	0.6
Cadmium	3.3	0.50	0.0002	0.0002	0.0002	0.04
Chromium	39.7	5.96	0.003	0.008	0.054	0.91
PCB 1254	0.25	0.0375	0.0004	0.0006	0.0025	6.6

 * Based on an inflow concentration of 150 g/ℓ.
** Samples containing a mean suspended solids concentration of 29 mg/ℓ.
 † Based on settling analysis for a 100 acre site, 26-in. dredge, resulting in an effluent suspended solids concentration of 185 mg/ℓ.

Table A6

Summary of Total Concentration and Mass Release

for Modified Elutriate Tests, Snohomish Channel (155 acres)

Parameter	Bulk Sediment mg/kg	Inflow* mg/ℓ	Dissolved Modified Elutriate mg/ℓ	Total** Modified Elutriate mg/ℓ	Effluent† Concentration mg/ℓ	Mass Release %
Copper	73.4	11.01	0.006	0.003	0.006	0.05
Nickel	21.4	3.21	0.018	0.017	0.018	0.6
Cadmium	3.3	0.50	0.0002	0.0002	0.0002	0.04
Chromium	39.7	5.96	0.003	0.008	0.044	0.73
PCB 1254	0.25	0.0375	0.0004	0.0006	0.0020	5.4

 * Based on an inflow concentration of 150 g/ℓ.
** Samples containing a mean suspended solids concentration of 29 mg/ℓ.
 † Based on settling analysis for a 100 acre site, 26-in. dredge, resulting in an effluent suspended solids concentration of 289 mg/ℓ.

concentration is only approximately twice the chronic criteria and is well below the acute criteria.

12. Lead concentrations slightly exceeded the chronic exposure values given in the Federal water quality criteria. A dilution factor of less than 1 was calculated using procedures in the Decisionmaking Framework that would dilute the standard elutriate value to the chronic exposure value. Size and configuration of the mixing zone would depend on site-specific information not yet available. However, such a minimal mixing and dilution could easily be achieved within a short distance of the open-water disposal operation.

13. The PCB 1254 concentrations exceeded both the chronic and acute exposure values given in the Federal water quality criteria. A dilution factor of 13 was calculated using procedures in the Decisionmaking Framework to dilute the standard elutriate value to the chronic exposure value. Size and configuration of the mixing zone would depend on site-specific information not yet available. However, such a minimal mixing and dilution could easily be achieved within a short distance of the disposal operation.

14. Only 7 of 33 contaminants of concern were detected in the standard elutriate tests. Only five parameters exceeded Port Gardner background concentrations, and only three parameters (nickel, lead, and PCB 1254) exceeded the Federal water quality criteria. These parameters were all of low concentration, and dilution to background concentrations or criteria can easily be accomplished within a short distance of the disposal operation. Based on these data, there appears to be no need for controls from the standpoint of contaminant release in the dissolved form during placement of the sediments for the CAD alternative.

15. Mass release. The standard elutriate procedure with analysis of total concentrations was used to gain qualitative information only. This change in the standard procedure is not laboratory developed or field verified. The total concentrations found using this procedure were equal to or below the dissolved concentrations for copper, cadmium, chromium, mercury, and PCB 1254. Total concentrations slightly exceeded the dissolved concentrations for nickel and lead. These data were not used in the computation of mass release.

16. Approximations of mass contaminant release during dredging and open-water placement for the CAD alternative were made based on estimates of the mass sediment release and dissolved elutriate data. The modeling efforts

described in Part III of this report indicate that the sediment release (sediment remaining in the water column and not accumulating on the bottom within the disposal site) during open-water disposal would be 1.9 percent for disposal at the surface. The modeling results indicated that sediment remaining in the water column would be negligible for disposal through a vertical pipe.

17. The mass release for contaminants in the dissolved form was estimated based on the standard elutriate data, and an estimate of the total water entrained and released during open-water disposal. It was assumed that the total volume of water entrained during clamshell dredging operations would be equal to 30 percent of the total volume dredged. It was further assumed that the total entrained volume would be released with contaminant concentrations equal to the dissolved standard elutriate concentrations. The same assumptions were applied for disposal at the surface and disposal through a vertical pipe. These calculations indicated that all dissolved mass releases were negligible, except cadmium with 0.1-percent release and mercury with 0.2-percent release.

18. The total mass release for disposal at the surface will therefore range from 2.0 to 2.1 percent, depending on the parameter. For disposal through a vertical pipe, total mass release will be negligible except for cadmium with 0.1-percent release and mercury with 0.2-percent release.

Modified elutriate

19. Water quality. Modified elutriate test results (dissolved only) were compared with the background water concentrations and water quality criteria in accordance with the Decisionmaking Framework. The dissolved test results are an estimate of the dissolved concentrations of contaminants that can be expected in the effluent discharged from a confined disposal site. Only nickel and PCB 1254 exceeded the background concentrations. Nickel exceeded the chronic exposure value but was below the acute exposure value given in the water quality criteria. PCB 1254 exceeded both the chronic and acute exposure values.

20. Since the chronic exposure value for nickel is equal to the measured background concentration, the modified elutriate value is only approximately 2.5 times the chronic criteria and is well below the acute criteria.

21. The PCB 1254 concentrations exceeded both the chronic and acute exposure values given in the Federal water quality criteria. A dilution factor of 13 was calculated using procedures in the Decisionmaking Framework

A10

to dilute the modified elutriate value to the chronic exposure value. Size and configuration of the mixing zone would depend on site-specific information not yet available. However, such a minimal mixing and dilution could easily be achieved with a short distance of the effluent discharge.

22. Only 5 of 33 contaminants of concern were detected in the dissolved fraction in the modified elutriate tests. Only two parameters--nickel and PCB 1254--exceeded Port Gardner background concentrations and the Federal water quality criteria. These parameters were of low concentration, and dilution to background or criteria can easily be accomplished within a short distance of the effluent discharge. Based on these data, there appears to be no need for controls for removal of dissolved contaminants from effluents discharged from confined sites during filling operations (either upland or intertidal).

23. **Mass release.** The modified elutriate test accounts for the contaminant concentrations associated with the suspended solids discharged in the effluent. An estimate of mass release of contaminants in the effluent is therefore possible if the site characteristics are known. An estimate of mass release was made for the East Waterway and Snohomish Channel sites with site operational conditions as described in the main text. The estimates were made using procedures in Palermo (1985). Values for contaminant concentrations in the bulk sediment, inflow, effluent (dissolved and total), and mass release are tabulated in Tables A4 through A6. Mass release of all parameters was less than 0.9 percent except for PCB with a mass release of up to 6.6 percent, depending on the site configuration.

24. Calculations were made for only those parameters that were detectable in the modified elutriate tests. Inflow contaminant concentrations were estimated from bulk sediment concentrations assuming an inflow solids concentration of 150 g/ℓ. Site conditions for the East Waterway site were considered equal to an equivalent flow rate of 5 cfs and a 12.9-acre surface area with a 2-ft ponding depth. Site conditions for the Snohomish Channel site were considered equivalent to use of a 26-in. dredge and a 100-acre or 155-acre confined disposal surface area with 2 ft of ponding depth. Contaminant concentrations associated with the site water contribution to the inflow were so low by comparison that they were neglected. Effluent concentrations were calculated based on the dissolved and total modified elutriate concentrations using procedures in Palermo (1985). For those parameters where

A11

dissolved concentration exceeded total concentration, the dissolved concentration was used for the effluent concentration. Mass release was calculated by dividing the total effluent concentration by the inflow concentration and expressing as a percentage.

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APPENDIX B: SURFACE RUNOFF WATER QUALITY TESTING

Introduction

1. Sediment removed from waterways by Corps dredging projects may con-
tain potentially hazardous concentrations of contaminants such as heavy
metals, PCBs, PAHs, and pesticides. The potential for contaminated sediments
becoming hazardous depends on several factors, including the chemical form of
the contaminants and the type of disposal environment. Dredged material in
its original state is anaerobic with a pH > 7. Most contaminants are adsorbed
tightly on the sediment solids and are not bioavailable. Movement of contami-
nants in surface runoff during this period is primarily the result of sediment
transport. Erosion can result in suspended solids concentrations ranging from
5,000 to 50,000 mg/ℓ in the surface runoff. Concentrations of contaminants in
unfiltered runoff could be very high during this period, but dissolved concen-
trations in filtered runoff would be relatively low and insignificant.

2. When the material is placed in a confined upland disposal site,
physicochemical changes occur as the wet anaerobic material dries and oxi-
dizes. The extent to which these changes occur may significantly affect the
surface runoff water quality, particularly the dissolved portion. As the
sediment dries and oxidizes, it becomes more resistant to erosion with sus-
pended solids decreasing to 10 to 1,000 mg/ℓ. Unfiltered concentrations of
contaminants will be several orders of magnitude less than during the wet
stage. If high levels of sulfides are present in the sediment, oxidation may
cause the formation of sulfuric acid, lowering the sediment pH to 4.0 where
contaminants such as heavy metals become very soluble in surface runoff (Lee
and Skogerboe 1984).

3. The objective of the surface runoff testing was to determine the
quality of runoff water from precipitation events following the filling oper-
ation at confined disposal sites. The WES Rainfall Simulator-Lysimeter System
has proven to be an effective tool for conducting surface runoff water quality
tests on Corps of Engineers project sites (Westerdahl and Skogerboe 1982).
Material was collected from the proposed dredging site, brought to the WES,
and placed in lysimeters to simulate a confined upland disposal site. As the
material dried and oxidized, rainfall simulations were conducted and the run-
off water quality was monitored.

B1

Methods and Materials

4. Sediment was collected from contaminated areas in the East Waterway, brought to the WES, and placed in a lysimeter measuring 4.57 by 1.22 m. The lysimeter was loaded with ten 200-ℓ barrels of sediment to a depth of 33.02 cm. Standing water on the sediment was allowed to drain out of the lysimeter. The following day, the lysimeter was subjected to a 30-min storm event at a 6.5-cm/hr application rate. Initial sediment moisture was 35 to 40 percent, and pH was 7.9. Runoff rates were measured every minute, and 4-ℓ samples were collected for chemical analysis at 5, 15, and 25 min after runoff began to occur. Additional samples were collected for suspended solids (SS) determinations at several points along the hydrogragh. The 4-ℓ samples were combined into a composite sample and analyzed for filtered and unfiltered Cd, Cu, Zn, Pb, Hg, As, PCBs, PAHs, and pesticides.

5. The lysimeter was then moved outside the greenhouse and covered with a semitransparent top that allowed air movement over the surface of the sediment. Surface moisture and pH were monitored during the drying period. After sufficient drying and oxidation, about 6 months, storm events were conducted on the lysimeter. The depth of the sediment had decreased to 22.9 cm, the surface moisture to 5 percent, and the pH to 7.1. Surface runoff samples were collected and analyzed as in the wet stage test runs.

Results and Discussion

Water quality from
wet, anaerobic sediment

6. Predicted surface runoff water quality from the wet, anaerobic sediment was typical of sediments during this stage. The sediment pH was high, 8.1, which caused the runoff pH to also be high (Table B1). Suspended solids concentrations were very high during this stage but decreased with drying. Heavy metals were mostly in an insoluble form, and unfiltered concentrations were significantly higher than filtered concentrations. The USEPA Maximum Criteria for the Protection of Aquatic Life are provided for comparison to the filtered runoff concentration. All filtered metal concentrations were significantly less than both the USEPA Maximum Criteria and are not considered to be a problem during this stage. Concentrations of several heavy metals

B2

Lysimeter Surface Runoff Water Quality During the Early, Wet, Unoxidized Stage

Parameter	Original Sediment Concentration μg/g	Mean Unfiltered Runoff Concentration mg/ℓ	Mean Filtered Runoff Concentration mg/ℓ	USEPA Maximum Criteria	Port Gardner Reference
pH	8.1	8.0	N*	N	N
Conductivity	N	4.0 mmV/cm	N	N	N
Salinity	N	3 g/ℓ	3 g/ℓ	N	N
SS	N	6,900	N	N	N
Total PCB	<0.002	<0.0002	<0.0002	0.014	<0.0002
PAH	37.4	0.077	0.004	N	<0.005
Naphthalene	8.2	0.0085	0.0019	N	
Acenaphthene	2.1	0.005	0.0008	N	
Fluorene	2.2	0.006	<0.005	N	
Phenanthrene	5.9	0.015	0.0014	N	
Anthracene	1.5	0.0025	<0.005	N	
Fluoranthene	4.5	0.013	<0.005	N	
Pyrene	4.1	0.011	<0.005	N	
Chrysene	1.8	0.0034	<0.005	N	
Benzo(A) anthracene	2.1	0.0030	<0.005	N	
Benzo(B) fluoranthene	2.5	0.0048	<0.005	N	
Benzo(K) fluoranthene	2.5	0.0048	<0.005	N	
Heavy metals					
Cadmium	3.30	0.029	0.0002	0.0015-0.0024	0.0006
Copper	73.4	1.153	0.005	0.012-0.043	0.007
Zinc	148.5	1.78	0.034	0.180-0.570	<0.03
Lead	48.1	0.540	0.004	0.074-0.400	<0.001
Mercury	0.201	0.0025	<0.0002	0.0017	0.0067
Arsenic	5.73	0.010	<0.005	0.440	<0.005
Oil and grease		47	<7	N	N
TKN	N	38	4.35	N	N
NO3	N	8.46	11.4	N	N
NH4	167	3.78	3.11	N	N
TP	789	9.16	0.14	N	N
TOC	71.5	290	15	N	N
COD	N	3260	429	N	N

* No values available.

from Port Gardner are also provided for comparison to filtered runoff samples from East Waterway sediment. Concentrations of Cd, Cu, and Hg are less than the Port Gardner values, and concentrations of Zn and Pb are slightly greater.

7. Organic contaminant concentrations in surface runoff were also low, particularly in the filtered portions. Both filtered and unfiltered concentrations of PCBs were below detectable limits. Some PAHs were detected in the runoff but concentrations were low, and only naphthalene, acenaphthene, and phenanthrene were above detectable limits in filtered samples.

8. Problems with surface runoff water quality from wet, unoxidized sediment are associated mainly with the SS. Unfiltered concentrations of contaminants were not excessive, and can be controlled by trapping the SS before the runoff is discharged from the disposal site. Total contaminant loads discharged during the surface runoff tests are presented in Table B2 and are based on a 5-cm/hr, 30-min storm event.

Water quality during dry, oxidized stage

9. Because of the large quantity of organic material present, the East Waterway sediment did not dry and oxidize like many other dredged materials tested by the WES. The sediment did not form hard surfaces with large cracks but remained very light and fluffy. The material was highly susceptible to erosion, with SS averaging 1,000 mg/ℓ (Table B3). Other sediments that formed the hard-crusted surfaces with large cracks had much lower SS concentrations, sometimes less than 100 mg/ℓ. The East Waterway sediment pH also remained high even after 6 months of drying and oxidation, which would tend to maintain heavy metals in insoluble forms.

10. Heavy metal concentrations in filtered runoff remained significantly lower than unfiltered concentrations except for Cd. However, the high erodibility had a significant effect on the surface runoff water quality. Because of the high concentrations of suspended solids, unfiltered and filtered metal concentrations of Cd, Cu, and Cr were high. Filtered concentrations of Cd were significantly greater than the USEPA Criteria, and Cu and Zn were not significantly different than the criteria. Both unfiltered and filtered concentrations of PAHs were extremely low, and PCBs were again below detectable limits. Filtered concentrations of Cd, Cu, Zn, and Pb are also greater than the Port Gardner background values. Filtered contaminant loads being discharged during the runoff tests were calculated for the lysimeter and

B4

Table B2

Contaminant Loads in Surface Runoff from Wet, Unoxidized Sediment

During a 5-cm/hr, 30-min Storm Event (Runoff Volume = 187 ℓ)

Parameter	Filtered Concentration mg/ℓ	Load	Load/ Hectare
PAH	0.0004	0.075 mg	134 mg/ha
Cd	0.0002	0.037 mg	67.1 mg/ha
Cu	0.005	0.935 mg	1,677 mg/ha
Pb	0.004	0.748 mg	1,342 mg/ha
SS	6,900	1.29 kg	2,315 kg/ha

Table B3

Lysimeter Surface Runoff Water Quality During the Dry, Oxidized Stage

Parameter	Original Sediment Concentration µg/g	Mean Unfil- tered Runoff Concentration mg/ℓ	Mean Filtered Runoff Concentration mg/ℓ	USEPA Maximum Criteria	Port Gardner Reference
pH	7.18	7.0	7.1	N*	N
conductivity, mmV/cm	81	3.3	3.4	N	N
Salinity	180 mg/g	2 g/ℓ	2 g/ℓ	N	N
SS	N	1000	N	N	N
PAH (total)	N	0.0065	0.0002	N	<0.005
Naphthalene	N	0.0006	0.0002	N	N
Acenaphthylene	N	0.0001	<0.005	N	N
Acenaphthene	N	0.0003	<0.005	N	N
Fluorene	N	0.0001	<0.005	N	N
Phenanthrene	N	0.0020	<0.005	N	N
Fluoranthene	N	0.0020	<0.005	N	N
Pyrene	N	0.0014	<0.005	N	N
Heavy metals					
Cadmium	N	0.035	0.018 **,†	0.0015-0.0024	0.0006
Copper	N	0.217	0.007 **	0.012-0.043	0.007
Zinc	N	1.20	0.335 **	0.180-0.570	<0.001
Lead	N	0.237	0.002	0.074-0.400	<0.001
Mercury	N	0.0022	<0.0004	0.0017	0.0067
Arsenic	N	<0.025	<0.005	0.440	<0.005

 * No values available.
 ** Concentration exceeds USEPA Maximum Water Quality Criteria for Protection of Aquatic Life.
 † Filtered concentration is not statistically significantly different from unfiltered concentration.

on a per-hectare basis (Table B4). These data provided a good approximation of the total filtered contaminants that may be discharged from an upland disposal site if no control measures are implemented.

11. Because of the excessive concentrations of SS in surface runoff from the dry, oxidized sediment, control measures will be required to trap the SS. Filtered concentrations of Cd, Cu, and Zn were high, particularly Cd, and were equal to or greater than the USEPA Maximum Criteria for the Protection of Aquatic Life. Some form of restrictions on the filtered portion of the surface runoff should be required, or a mixing zone might be considered. A dilution factor of approximately 18 was calculated for the mixing zone, based on procedures presented in Peddicord et al. (in preparation). Filtered Cd from a dry, oxidized sediment was used for the calculation, and the storm event was a 5-cm/hr, 30-min storm.

12. Estimates of yearly mass release for an upland disposal site were calculated using the Universal Soil Loss Equation (USLE). This method was selected because it is simple and has widespread acceptance. The calculations for this particular application required that the soil erodibility factor (K) be calculated from the lysimeter tests, and that a rainfall erodibility factor (R) be calculated specifically for the storm event used in the tests. Because of the complex nature of dredged material and the physical chemical changes that occurred, normal methods for determining soil erodibility factors based on particle size, percent organic matter, soil structure, and permeability were inadequate. The calculated K factor was then used in the USLE along with an R factor determined for the Seattle, Wash., area (Kirkby and Morgan 1980). This method was extensively used for determination of soil erodibility nomographs for farmland and construction sites (Wischmeier, Johnson, and Cross 1971). Next, using a ratio of contaminant load to SS load calculated from the lysimeter test, a yearly mass release of filtered contaminants from dry, oxidized sediment was calculated for the Snohomish Channel and East Waterway sites (Table B5). For the purpose of these calculations, the disposal site surface areas were assumed to be 40.5 ha for the Snohomish site and 5.2 ha for the East Waterway site. Because the mass release of contaminants through surface runoff was proportional to the disposal site surface area, the East Waterway site would have significantly less mass release. The mass release of contaminants from either site, however, was very small compared to the total quantity of contaminants in the sediment.

B6

Table B4

Contaminant Loads in Surface Runoff from Dry, Oxidized Sediment During a 5-cm/hr, 30-min Storm Event (Runoff Volume = 184 ℓ)

Parameter	Filtered Concentration mg/ℓ	Load, mg	Load/Hectare
Cd	0.018	3.31	5,942 mg/ha
Cu	0.007	1.29	2,314 mg/ha
Zn	0.335	61.6	110,582 mg/ha
Pb	0.002	0.368	660 mg/ha
SS	1,000.	184,000.	330 kg/ha

Table B5

Estimates of Yearly Mass Release of Filtered Contaminants from Dry, Oxidized Sediment Placed in the Proposed East Waterway and Snohomish Channel Nearshore Disposal Sites

Parameter	Mass Release kg/ha	East Waterway (5.2 ha), kg	Shohomish Channel (40.5 ha), kg
Ss	343	1,784	13,892
Cd	6.17	0.032	0.250
Cu	2.40	0.012	0.097
Zn	115	0.598	4.658
Pb	0.686	0.004	0.028

Conclusions

13. Surface runoff water quality problems from the East Waterway sediment during the wet, anaerobic stage will be primarily in the form of high SS concentrations. This problem, which typically occurs when dredged material is first placed in upland disposal sites, is easily controlled by allowing the sediment to settle out of the runoff before discharged from the disposal site. During this period, contaminants such as heavy metals are tightly bound to the sediment, and will be removed from the runoff when the SS are removed. Concentrations of PCBs were below detectable limits, and PAHs were low and of little concern.

14. When dredged material is placed in upland sites, physicochemical changes occur which may have significant effects on surface runoff water quality. Because of a high concentration of organic material, the East Waterway sediment formed a very light fluffy surface that was highly erosive. Suspended solids concentrations in the surface runoff remained very high, causing unfiltered metal concentrations to also remain high. Solubilities of heavy metals such as Cd, Cu, and Zn increased, but Cu and Zn remained significantly lower than unfiltered concentrations. Filtered concentrations of Cd, Cu, and Zn were greater than or equal to the USEPA Maximum Criteria for the Protection of Aquatic Life and, therefore, should be of concern. Either some form of control measures or restrictions should be required to control both SS and soluble Cd in surface runoff, or before restrictions are formulated, a mixing zone outside the disposal site should be considered. If further input is required regarding the environmental impacts of surface runoff, bioassay tests utilizing simulated surface runoff may be conducted.

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APPENDIX C: LEACHATE TESTING

PART I: INTRODUCTION

Background

1. When contaminated dredged material is placed in an upland or near-
shore confined disposal facility, the potential exists to generate leachates
that may adversely impact ground waters. At present, there is no routinely
applied laboratory testing protocol capable of predicting, or even approxi-
mating, leachate quality from confined dredged material disposal sites.
Experimental testing procedures to predict leachate quality are, therefore,
being used to evaluate the confined disposal alternative for Everett Harbor
dredged material. These leaching procedures are in an early state of develop-
ment and must be interpreted with caution. If the Corps can assess leachate
quality and quantity, the potential impacts of using a confined disposal
facility (CDF) for disposal of contaminated dredged material can be
determined, therefore allowing the most cost-effective site design to be
developed.

2. The objective of this study is to evaluate and apply appropriate
testing procedures for estimating leachate contaminant levels from Everett
Harbor sediment under the CDF disposal alternative. Since the testing pro-
cedures are still developmental in nature, detailed descriptions of the pro-
cedures used are presented in this appendix.

Objectives and Approach

3. The objectives of this study were twofold. The primary objective
was to estimate leachate quality in Everett Harbor sediment. Since standard
procedures applicable to dredged material for assessing leaching potential
were not available, a supporting objective was to develop, evaluate, and apply
appropriate testing procedures for estimating leachate contaminant levels in
Everett Harbor sediment.

4. The technical approach used in this study is an integrated procedure
that involves coupling results from batch and continuous-flow column tests
with a mass transport equation (Myers, Brannon, and Griffin 1986). Comparison

of predicted and observed column effluent quality is the basis for evaluating the geochemical processes that govern contaminant leaching from Everett Harbor sediment. Description of the processes that govern the movement of pore water, site-specific hydraulics, is beyond the scope of the leachate testing.

PART II: MATERIALS AND METHODS

Sediment Preparation

5. Sediment acquisition, mixing, and transport procedures have been previously described. Upon arrival at the WES, sediment for use in the anaerobic leaching tests was refrigerated at 4° C in sealed containers until used. Sediment for use in aerobic testing was placed into 38-ℓ glass aquariums to a depth of approximately 8 cm. The aquaria were then placed in a covered enclosure open to the air and allowed to oxidize at ambient temperatures. Each week, the sediment was thoroughly stirred to expose fresh sediment to the air. When necessary, distilled, deionized water was added to the sediment to prevent drying. At the end of 6 months of aeration, the sediment was removed from the aquaria, placed into a 115-ℓ barrel, and thoroughly mixed for 2 hr. The sediment was then refrigerated at 4° C until used for all aerobic leachate testing.

Batch Testing

Salinity tests

6. Prior to testing, the effects of salinity changes in the leachate on metal releases were assessed. Triplicate 250-ml polycarbonate centrifuge tubes, fitted with a leakproof, airtight top were loaded with sufficient sediment and deoxygenated water to obtain a 4:1 water to sediment dry weight ratio for a volume of 200 ml. The 4:1 water to sediment ratio was selected for salinity and kinetic testing because this ratio had proven to be optimum during previous leaching tests. All operations were conducted in a glove box under a nitrogen atmosphere. Sufficient triplicate centrifuge tubes were loaded to allow testing at salinity levels of 0, 5, 15, and 25 ppt. Seawater of known salinity was prepared by diluting Copenhagen Standard Sea Water of known salinity with distilled, deionized water. Samples were placed upright on a mechanical shaker and shaken at 160 cycles per minute for 24 hr. The tubes were then removed from the shaker, centrifuged at 9,000 × g for 20 min, and the supernatant was filtered under a nitrogen atmosphere through 0.45-μ pore size membrane filters. The filtrate was then acidified to pH 1 with concentrated Ultrex nitric acid and stored in plastic bottles until analyzed.

C3

Kinetic tests

7. Batch testing was performed to determine shaking time necessary to achieve equilibrium or steady-state conditions for metal and organic contaminant leachate concentrations. The general experimental sequence is presented in Figure C1.

8. For testing metal releases, triplicate 250-ml polycarbonate centrifuge tubes fitted with a leakproof, airtight top were loaded with sufficient sediment and deoxygenated, distilled, deionized water to obtain a 4:1 water to sediment dry weight ratio. All operations were conducted in a glove box under a nitrogen atmosphere. Sufficient triplicate centrifuge tubes were loaded to allow sampling at 24, 48, 72, and 168 hr. Samples were placed horizontally on a mechanical shaker and shaken at 160 cycles per minute for the allotted time. Three tubes were then removed from the shaker, centrifuged at 9,000 × g for 20 min, and the supernate was filtered under a nitrogen atmosphere through 0.45-μ pore size membrane filters. The filtrate was then acidified to pH 1 with concentrated Ultrex nitric acid and stored in plastic bottles until analyzed.

9. Kinetic testing for organic contaminants was conducted in specially fabricated 450-ml stainless steel centrifuge tubes. Twenty-four acetone-rinsed centrifuge tubes were loaded with sufficient sediment and deoxygenated, distilled, deionized water to obtain a 4:1 water to sediment dry weight ratio. The total mass of sediment and water added was regulated to allow the tube to be safely centrifuged at 6,200 rpm (6,500 × g). All operations were conducted under a nitrogen atmosphere. The tubes were then laid on their sides and shaken at 160 cycles per minute for periods of 24, 48, 96, and 168 hr. At each sampling time, the samples were removed from the shaker and centrifuged for 30 min. The leachate was then recentrifuged in clean centrifuge tubes to remove remaining particulate material. The recentrifuged supernate was then filtered through a Whatman GF/D glass fiber prefilter and a Gelman AE glass fiber filter with a nominal pore size of 1.0 μ. Neither filter contained binders or detectable quantities of the organic contaminants analyzed during this study. Filtration was conducted under a nitrogen atmosphere followed by acidification with 1 ml of concentrated HCl to prevent iron precipitation and scavenging of organic contaminants from solution by iron precipitates. Samples were then stored in the dark in acetone-rinsed 2-ℓ glass bottles until analyzed.

STEP 1   PLACE SEDIMENT IN APPROPRIATE CENTRIFUGE TUBE (STAINLESS STEEL OR
          POLYCARBONATE). ADD SUFFICIENT DEOXYGENATED DISTILLED WATER TO MAIN-
          TAIN WATER TO SEDIMENT RATIO OF 4:1.

STEP 2   PLACE CENTRIFUGE TUBES HORIZONTALLY ON SHAKER AND SHAKE AT 160 CYCLES
          PER MINUTE.

STEP 3   REMOVE TUBES (ENOUGH FOR TRIPLICATE SAMPLES FOR ORGANICS AND FOR
          METALS) FROM SHAKER AT APPROPRIATE INTERVALS: 1, 2, 4, and 7 DAYS
          FOR ORGANIC CONTAMINANTS AND AT 1, 2, 3, and 7 DAYS FOR METALS.

STEP 4   CENTRIFUGE FOR 30 MIN AT 6,500 × G FOR ORGANICS AND 9000 × G FOR
          METALS. (REPETITION OF STEP 4 USING CLEAN CENTRIFUGE TUBES WAS
          NECESSARY FOR LEACHATE FOR ORGANIC ANALYSES.)

STEP 5   FILTER CENTRIFUGED LEACHATE THROUGH 0.45-μ PORE SIZE MEMBRANE FILTERS
          FOR METALS AND THROUGH A WHATMAN GF/D GLASS FIBER PREFILTER AND A
          GELMAN AE GLASS FIBER FILTER OF 1-μ NOMINAL PORE SIZE FOR ORGANICS.

STEP 6   ACIDIFY LEACHATE FOR ORGANIC ANALYSIS WITH HCl AND LEACHATE FOR
          METALS WITH ULTREX NITRIC ACID. STORE LEACHATE FOR ORGANIC ANALYSIS
          IN ACETONE-RINSED GLASS BOTTLES AND LEACHATE FOR METALS ANALYSIS IN
          PLASTIC BOTTLES.

Figure C1.   Experimental sequence for determining appropriate shaking
             times, Everett Harbor kinetic testing

Sediment-water ratio testing

10. Following determination of the shaking time necessary to obtain steady-state contaminant concentrations in the leachate, testing to determine the proper sediment to water ratio was conducted. The general test sequence is presented in Figure C2.

11. For metals, anaerobic Everett Harbor sediment was placed in acid-washed 250-ml polycarbonate centrifuge tubes in water to sediment ratios of 4:1, 8:1, 12:1, 50:1, and 100:1 using double-distilled, deionized water. The tubes were then sealed, mechanically shaken horizontally for 24 hr, then centrifuged and filtered through 0.45-μ membrane filters; the resulting supernatant was acidified and stored in plastic bottles prior to analysis as previously described. The anaerobic integrity of the samples were maintained throughout the preparation, shaking, and filtration of the sample.

12. Similar procedures were followed for organic contaminants, except that 24-hr shaking was conducted in 450-ml stainless steel centrifuge tubes. Filtration and other sample preparation procedures are as described for organic contaminants in the kinetic testing section.

Sequential batch testing

13. A 4:1 water to sediment ratio and a shaking time of 24 hr were found to be optimum for application of sequential batch leaching tests to anaerobic sediment. General test procedures for assessing steady-state leachate and sediment metal and organic contaminant concentrations are detailed in Figure C3.

14. Batch tests were designed to determine metal releases from anaerobic Everett Harbor sediment and provide sufficient leachate to challenge fresh sediment. To obtain this leachate, three 500-ml polycarbonate centrifuge bottles with leakproof caps were loaded under a nitrogen atmosphere with anaerobic Everett Harbor sediment and deoxygenated, distilled, deionized water to a 4:1 water to sediment ratio; these were mechanically shaken for 24 hr. The bottles were then centrifuged at 9000 × g for 30 min. Half of the leachate from each 500-ml centrifuge bottle was filtered through a 0.45-μ membrane filter. A portion of the unfiltered leachate was then analyzed for pH using a combination electrode and a millivolt meter, and for conductivity using a Yellow Springs Instrument Company conductivity meter and cell. Enough of the remaining unfiltered leachate was weighed into a 250-ml polycarbonate centrifuge tube containing fresh Everett Harbor sediment to obtain a 4:1 water

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STEP 1   PLACE SEDIMENT IN APPROPRIATE CENTRIFUGE TUBES; 250-ml POLYCARBONATE
          FOR METALS AND 450-ml STAINLESS STEEL FOR ORGANIC CONTAMINANTS. ADD
          WATER TO EACH TUBE TO BRING FINAL WATER TO SEDIMENT RATIO TO 4:1,
          8:1, 12:1, 50:1, and 100:1.

STEP 2   PLACE CENTRIFUGE TUBES HORIZONTALLY ON SHAKER AND SHAKE AT
          160 CYCLES PER MINUTE FOR 24 HR.

STEP 3   CENTRIFUGE FOR 30 MIN AT 6,500 × G FOR ORGANICS AND 9,000 × G FOR
          METALS. (SAMPLES FOR ORGANIC ANALYSIS REQUIRED REPETITION OF STEP 3
          USING CLEAN STAINLESS STEEL CENTRIFUGE TUBES TO REMOVE ADDITIONAL
          PARTICULATE MATTER.)

STEP 4   FILTER LEACHATE THROUGH 0.45-μ MEMBRANE FILTERS FOR METALS OR
          THROUGH A WHATMAN GD/F GLASS FIBER PREFILTER FOLLOWED BY PASSAGE
          THROUGH A GELMAN AE GLASS FIBER FILTER OF 1.0-μ NOMINAL PORE SIZE
          FOR ORGANICS.

STEP 5   ACIDIFY LEACHATE FOR ORGANIC ANALYSIS WITH HCl AND LEACHATE FOR
          METALS ANALYSIS WITH ULTREX NITRIC ACID. STORE LEACHATE FOR ORGANIC
          ANALYSIS IN ACETONE-RINSED GLASS BOTTLES AND LEACHATE FOR METALS
          ANALYSIS IN PLASTIC BOTTLES.

NOTE: THE ANAEROBIC INTEGRITY OF THE SAMPLE WAS MAINTAINED DURING SAMPLE
ADDITION TO CENTRIFUGE TUBES, SHAKING, CENTRIFUGATION, AND FILTRATION.

Figure C2. Test sequence for determining appropriate water to sediment
            ratio for use during batch testing procedures

STEP 1	LOAD SEDIMENT INTO APPROPRIATE CENTRIFUGE TUBES; 500-ml POLYCAR-BONATE FOR METALS AND 450-ml STAINLESS STEEL FOR ORGANIC CON-TAMINANTS. ADD SUFFICIENT WATER TO EACH TUBE TO BRING FINAL WATER TO SEDIMENT RATIO TO 4:1. SUFFICIENT STAINLESS STEEL TUBES MUST BE LOADED TO OBTAIN ENOUGH LEACHATE FOR ANALYSIS AND FOR USE IN LEACHING FRESH SEDIMENT.
STEP 2	GO THROUGH STEPS 2 AND 3 IN FIGURE C2.
STEP 3	FOR HALF OF THE LEACHATE FOR METALS, CARRY THROUGH STEPS 4 AND 5 OF FIGURE C2, SETTING ASIDE A SMALL AMOUNT OF LEACHATE PRIOR TO ACIDIFICATION FOR ANALYSIS OF pH AND CONDUCTIVITY. INTRODUCE THE REMAINING CENTRIFUGED LEACHATE INTO 250-ml POLYCARBONATE CENTRIFUGE TUBES FOR METALS AND 450-ml STAINLESS STEEL CENTRI-FUGE TUBES FOR ORGANIC CONTAMINANTS. CARRY THESE CENTRIFUGE TUBES THROUGH STEPS 2 THROUGH 5 OF FIGURE C2.
STEP 4	RETURN TO STEP 2 AFTER REPLACING LEACHATE REMOVED IN THE INITIAL SET OF CENTRIFUGE TUBES WITH DEOXYGENATED DISTILLED WATER. REPEAT THE ENTIRE PROCEDURE THE DESIRED NUMBER OF TIMES.

NOTE: TESTING SEQUENCE IS THE SAME FOR AEROBIC SEDIMENTS EXCEPT THAT AEROBIC SEDIMENT LEACHATE IS USED TO CHALLENGE AEROBIC SEDIMENT AND ANAEROBIC INTEG-RITY IS NOT MAINTAINED.

Figure C3. Test sequence for sequential batch leaching and challenge testing of anaerobic Everett Harbor sediment for metals and organic contaminant analysis

to sediment ratio. This procedure, whereby part of the initial leachate was set aside for analysis and the remainder used to challenge fresh anaerobic Everett Harbor sediment, was continued for 9 days. Fresh deoxygenated, distilled, deionized water was added to each 500-ml centrifuge tube to replace the leachate removed for analysis and challenging of fresh sediment. All operations were conducted under a nitrogen atmosphere. This same procedure was repeated for aerobic sediments, except that aerobic sediment leachate was used to challenge aerobic sediment.

15. Testing of Everett Harbor sediment for organic contaminants was conducted in a manner similar to that described for metals; however, 450-ml stainless steel centrifuge tubes were used for both the sequential and challenge testing and centrifugation. The filtration procedures used for organic contaminants were as previously described for the kinetic and sediment to water ratio testing, as presented in Figure C3. A subsample of filtered leachate was set aside from both the anaerobic and aerobic tests for analysis of total organic carbon. In each case, the leachate was replaced with distilled, deionized water, remixed, shaken for 24 hr, and processed as previously described for the desired number of cycles.

Interstitial water extraction

16. Interstitial water samples for metal and organic contaminant analysis were obtained by centrifugation of the Everett Harbor sediment. To obtain samples for metals from anaerobic Everett Harbor sediment, triplicate 250-ml polycarbonate centrifuge tubes fitted with a leakproof, airtight top were loaded with sediment in a glove box under a nitrogen atmosphere. The centrifuge tubes were then centrifuged at $9,000 \times g$ for 30 min, and the supernate was filtered under a nitrogen atmosphere through $0.45-\mu$ pore size membrane filters. The filtrate was then acidified to pH 1 with concentrated Ultrex grade nitric acid and stored in plastic bottles until analyzed. Procedures for obtaining interstitial water for metals analysis from aerobic Everett Harbor sediment were similar to those described for anaerobic sediment, except that all steps in the aerobic operation were conducted without the use of nitrogen.

17. Interstitial water for analysis of organic contaminants was obtained by centrifugation of anaerobic Everett Harbor sediment in 450-ml stainless steel centrifuge tubes. For interstitial water separation from anaerobic Everett Harbor sediment, six tubes were loaded with sediment, then

centrifuged for 30 min at 6,500 × g. The supernate was recentrifuged in clean centrifuge tubes to remove residual particulate matter, then filtered through a Whatman GF/D glass fiber prefilter and a Gelman AE glass fiber filter with a nominal pore size of 1.0 μ. All steps in the operation were conducted under a nitrogen atmosphere. Following filtration, the interstitial water was acidified with 1 ml of concentrated HCl, then stored in the dark in acetone-rinsed 2-ℓ glass bottles until analyzed. Aerobic interstitial water was obtained in a similar manner except that anaerobic conditions were not maintained during the operation.

Permeameter Testing

Loading and operation

18. Column leaching tests were conducted in divided-flow permeameters designed to minimize wall effects and provide for pressurized operation (Figure C4). The inner permeameter ring divides flow, separating the leachate flowing through the center of the column from that flowing down the walls, thereby minimizing wall effects on leachate quality. The applied pressure forces water through the sediment at rates sufficient to allow sample collection in a reasonable period of time.

19. Permeameter tests were run to simulate leaching of anaerobic and oxidized sediment, prepared as previously described. Permeameter effluent was analyzed for concentrations of arsenic, cadmium, chromium, lead, and zinc, and 33 organic contaminants. Separate permeameter tests were run to obtain leachate for metal and organic analysis because of the large leachate volume needed to conduct organic contaminant analyses (1 ℓ). Column tests were run in triplicate for analysis of metal and organic leachate concentrations in anaerobic and aerobic Everett Harbor sediment, a total of 12 permeameter tests.

20. Everett Harbor sediment was loaded into the permeameters in several lifts having an average thickness of 5 cm, the number of lifts added depending on the total sediment thickness desired. As each lift of water-saturated sediment was added, the permeameter was vigorously agitated on a vibrating table to remove trapped air. The weight and height of each lift was measured and recorded following vibration. Sediment height averaged 18 cm in permeameters used to obtain leachate for metal analysis and 36 cm in permeameters used to
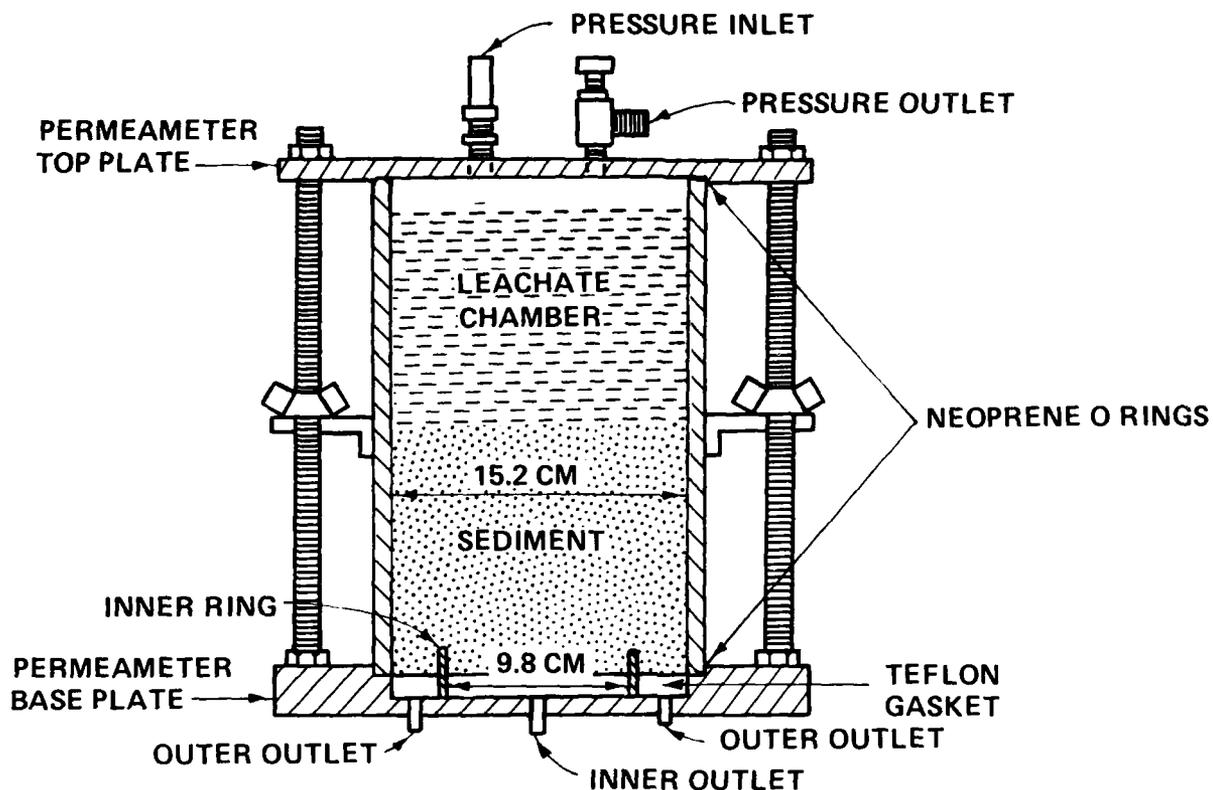
Figure C4. Divided-flow permeameter

obtain leachate for organic contaminant analysis. A greater depth of sediment
was needed in the permeameters run for organic analyses because of greater
sample volume needs for chemical analyses. Sediment pore volume in the per-
meameters was determined by measuring the weight and volume of sediment added
to the permeameter, then measuring the weight and volume of sediment samples
before and following oven drying at 105° C; weight loss upon drying was then
equated to the volume of water in the permeable voids. Next, pore volumes
were calculated for the sediment column above the inner ring of each per-
meameter. Therefore, pore volumes refer to the column of sediment above and
including the permeameter inner ring.

21. Following sediment addition, distilled, deionized water was added
to the permeameters; the apparatus was then sealed and pressurized with either
nitrogen or air depending on whether the test was conducted on anaerobic or
aerobic sediment, respectively. It was necessary to periodically add water to
the permeameters during the course of a test. Effluent from the inner and
outer permeameter rings was drained through teflon tubing into 1,000-ml

graduated cylinders. The cylinder, receiving flow from the inner outlet of each permeameter, was isolated from the atmosphere by a water trap that allowed gas used to pressurize the permeameters to escape without exposing the leachate to the atmosphere. The collection cylinder head-space was purged with nitrogen prior to testing anaerobic sediment.

22. Effluent flow from the permeameters was regulated by adjusting the operating pressure. The permeability of the sediment decreased for the first 2 weeks of operation. As permeability decreased, operating pressure was increased to maintain a constant flow. Permeameter flow generally stabilized after 2 weeks of operation. A daily record was maintained of operating pressure and flow from both the inner and outer rings of the permeameter.

Sampling

23. Permeameter effluent sampling for metals was conducted as frequently as possible as the first pore volume moved through the column (three to four samples/pore volume), then at less frequent intervals (one to two samples/pore volume) for the duration of the testing. Effluent used for metals analysis was also analyzed for dissolved organic carbon, conductivity, and pH.

24. Effluent used for organic contaminant analysis was sampled at approximately 0.5 pore volume intervals. The volume collected was analyzed for organic contaminants, except for a small amount used to analyze dissolved organic carbon concentrations.

25. Leachate samples for metals and organic contaminants from anaerobic sediment were filtered under nitrogen using procedures previously described for batch testing.

Dispersion coefficient measurement

26. The dispersion coefficient, D_p , was determined by operating a separate permeameter specifically for this purpose using anaerobic sediment and distilled, deionized water containing bromide as a tracer (constant concentration = 1,000 mg/ℓ). Effluent samples were collected periodically, filtered (0.45-μ pore size membrane filter), digested using procedures developed by Chian and DeWalle (1975) for chlorides in sanitary landfill leachate, and analyzed for bromide by silver-nitrate titration using a recording titrator with a silver-specific ion probe. From these data, the dispersion coefficient was computed using the F-curve procedure described by Levenspiel (1972). This method assumes dispersion within the column to be small, i.e.,

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$$\frac{D_p}{VL} < 0.01 \qquad\qquad (C1)$$

where

    D_p = dispersion coefficient

    V = average pore water velocity

    L = column length

Term D_p/VL is a dimensionless ratio, called the dispersion number, and is used to characterize dispersion in flow-through systems.

Chemical Analysis

27. Sediment samples and leachate from batch testing were analyzed for selected polychlorinated biphenyls (PCB congeners), polyaromatic hydrocarbons (PAHs), As, Cd, Cr, Cu, Hg, Ni, and Zn. Column leachates were analyzed for the same list of parameters with the exception of Ni and Cu. Concentrations of PCB congeners and PAH compounds in sediment samples were determined following soxhlet extraction, Florosil cleanup, and quantification in either a Hewlett Packard 5985A gas chromatograph/mass spectrophotometer (GC/MS) equipped with a flame ionization detector (PAHs) or a Hewlett Packard 5880A GC equipped with an electron capture detector (PCBs). Concentrations of PAH and PCB compounds in leachate samples following methylene chloride extraction were determined on the same equipment as for sediment samples. Sediment and leachate samples were analyzed for all metals studied except arsenic and mercury using directly coupled plasma emission spectroscopy on a Beckman Spectraspan IIIB plasma emission spectrometer or by atomic absorption spectroscopy using a Perkin-Elmer Model 5000 atomic absorption spectrometer coupled with a Perkin-Elmer Model 500 hot graphite atomizer following appropriate sample digestion procedures (Ballinger 1979). Arsenic in leachate and sediment samples was determined by hydride generation (Ballinger 1979) using a Perkin-Elmer 305 atomic absorption spectrophotometer coupled with a Perkin-Elmer Model MHC-10 hydride generator. Mercury was analyzed by the cold vapor technique (Ballinger 1979). Total organic carbon was analyzed in leachate and sediment samples using an Oceanographic International 543B organic carbon analyzer and standard procedures (Ballinger 1979).

Statistical Analysis

28. All statistical analyses were conducted using Statistical Analysis System (SAS) Institute (Barr et al. 1976) procedures. Analysis of variance procedures were used to test for differences between means. Regression analysis was used to determine the equation of the line of best fit between steady-state sediment and leachate contaminant concentrations obtained during batch testing, and to evaluate its statistical significance.

PART III: THEORETICAL BASIS FOR LEACHATE QUALITY PREDICTION

29. The purpose of this section is to present a brief overview of the equations used to predict leachate quality and their relationship to the experimental procedures described earlier. The application of these equations, for predictive purposes, to contaminated dredged material is a new approach and should be considered in the research stage of development. Development of the equations and additional discussion concerning their theoretical basis have been presented by Myers, Brannon, and Griffin (1986) and Myers, Hill, and Brannon (1988).

30. For this discussion it is assumed that water transports contaminants from the dredged material to the boundaries of a CDF. Leaching is defined as interphase transfer of contaminants from the dredged material solids to the aqueous phase as water moves past the dredged material solids. Upon contact with percolating water, contaminants associated with sediment particles can go into solution, thereby increasing contaminant levels in the leachate.

31. For contaminant leaching occurring as water percolates through porous media, the governing one-dimensional partial differential equation for steady-state flow is given below (Lapidus and Admundson 1952, Lowenbach 1978, Rao et al. 1979, Grove and Stollenwerk 1984).

$$\frac{\partial C}{\partial t} + \frac{p}{\theta}\left(\frac{\partial q}{\partial t}\right) = D_p\left(\frac{\partial C}{\partial z}\right) - V\left(\frac{\partial C}{\partial z}\right) \tag{C2}$$

where

C = aqueous phase contaminant concentration, mg/ℓ

t = time, sec

p = bulk density, kg/ℓ

θ = porosity, dimensionless

q = solid phase contaminant concentration, mg/kg

D_p = bulk dispersion coefficient, cm^2/sec

z = direction, cm

V = average pore water velocity, cm/sec

Equation C2 is sometimes referred to as the permeant-porous media equation. The derivation of this equation is based on balancing the mass flux into and out of any arbitrary volume within a column of dredged material. The first term on the right-hand side represents dispersive transport of contaminant; the second represents convective transport (bulk flow). The first term on the left side, sometimes referred to as the accumulation term, represents the resulting change in aqueous phase contaminant concentration with time; the second term on the left side, sometimes referred to as the source or reactive term, represents interphase transfer of contaminant from the sediment solids to the aqueous phase.

32. The first step in applying Equation C2 is the development of a mathematical formulation for the source term. In this study a linear equilibrium source term was used, resulting in Equation C3.

$$\left(\frac{\partial C}{\partial t}\right) + \left(\frac{pK_d}{\Theta}\right) \left(\frac{\partial C}{\partial t}\right) = D_p \left(\frac{\partial C}{\partial z}\right) - V \left(\frac{\partial C}{\partial z}\right) \tag{C3}$$

In this equation K_d is referred to as the distribution coefficient and has units of $1/kg$. The leach tests described in this report were conducted to test the hypothesis that contaminant leaching from Everett Harbor sediment is described by Equation C3, i.e., the source term can be described as equilibrium-controlled, linear desorption.

33. An equilibrium relationship between sediment and aqueous phase contaminant concentrations in a batch system can be written as follows (Myers, Brannon, and Griffin 1986):

$$q = K_d C \tag{C4}$$

In this equation, q refers to the reversibly sorbed component of the sediment contaminant. However, if q is defined as the bulk sediment contaminant concentration, the nonreversible component must be added to Equation C4 as follows:

$$q = K_d C + q_r \tag{C5}$$

where q_r is the nonreversible component resistant to leaching. Equation C5 is a general relationship that applies to a batch system at steady state. In a continuous-flow system, q and C at any point do not remain constant over time but change as percolating water leaches contaminants. Application of Equation C5 to a continuous-flow system requires

$$\frac{\partial q}{\partial t} = K_d \left(\frac{\partial C}{\partial t}\right)$$ (C6)

Equation C6 describes a local, linear equilibrium condition at the sediment solids/water interface in a continuous-flow system. Substitution of Equation C6 into Equation C2 yields Equation C3.

34. Equation C3 is the basis of design for the sequential batch leaching tests, described earlier. By sequentially leaching a portion of sediment with successive aliquots of clean water, a table of C and corresponding q values can be generated and plotted. Such a plot is called a desorption isotherm with slope K_d and intercept q_r . If the desorption isotherm goes through the origin, then q_r is equal to zero. Thus, the intercept value can be interpreted as the contaminant fraction resistant to leaching. Ideal desorption isotherms illustrating the important theoretical features of isotherm analysis are shown in Figure C5.

35. The previous discussion presents the basic theory behind the development and use of the sequential batch leach tests for Everett Bay sediment. It is clear that sequential batch leach tests, designed to evaluate K_d and q_r , do not provide a complete picture of how the contaminant concentration varies with time and position in a continuous-flow system. According to the permeant-porous media equation, as water percolates through a column of dredged material, the temporal variation in leachate contaminant concentration at any point is determined not only by the source term but also by the effects of advection and dispersion.

36. As previously stated, the integrated approach consists of using results from batch leach tests, column leach tests, and Equation C2 to test the hypothesis that contaminant leaching from Everett Harbor sediment can be described as equilibrium-controlled, linear desorption. Application of the integrated approach is illustrated in Figure C6.

q_o = INITIAL SEDIMENT CONTAMINANT CONCENTRATION

q_L = LEACHABLE SEDIMENT CONTAMINANT CONCENTRATION

q_r = SEDIMENT CONTAMINANT CONCENTRATION RESISTANT TO LEACHING

● DENOTES EXPERIMENTAL DATA

$K_d = \Delta q/\Delta c$: SLOPE-DERIVED DISTRIBUTION COEFFICIENT

$K_d = q/c$: SINGLE-POINT DISTRIBUTION COEFFICIENT
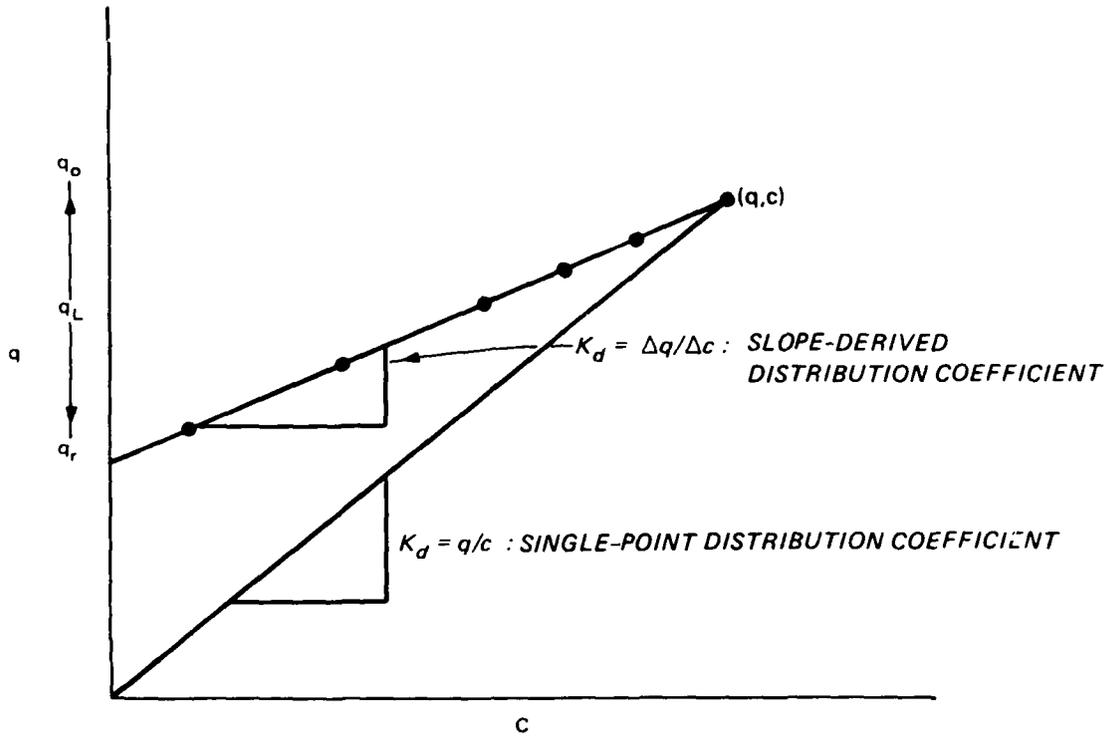
Figure C5. Ideal desorption isotherms: slope and
single-point distribution coefficients

37. Once the information needed to solve Equation C7 is obtained, col-
umn and batch leaching data can be combined using the permeant-porous media
equation to provide an integrated picture of leachate quality as a function of
time or pore volumes passing through the dredged material. An analytical
solution to this equation for equilibrium-controlled, linear desorption is
presented below (Ogata and Banks 1961).

$$C(z,t) = C_I + (C_o - C_I)\ 0.5\ \text{erfc}\ \frac{Rz - Vt}{2(DRt)^{0.5}}$$

$$+ 0.5\ \exp \frac{Vz}{D}\ \text{erfc}\ \frac{Rz + Vt}{2(DRt)^{0.5}} \tag{C7}$$

C18

INTERGRATED APPROACH FOR EXAMINING THE SOURCE TERM

CONDUCT
CONTINUOUS-FLOW COLUMN
LEACH TESTS

OBTAIN TRACE OF
CONTAMINANT CONCENTRATION
VERSUS VOLUME OF
LEACHATE PRODUCED

DETERMINE
D_p AND V
FOR THE
COLUMN

CONDUCT
STANDARD SOILS
TESTS

DETERMINE
ρ
AND
θ

CONDUCT
SEQUENTIAL BATCH LEACH
TESTS

DETERMINE
K_d FOR EACH
CONTAMINANT
OF INTEREST

PREDICT EFFLUENT CONCENTRATIONS
FROM CONTINUOUS-FLOW COLUMN BY
SUBSTITUTING ρ, θ, K_d, D_p, AND V
INTO A MASS TRANSFER EQUATION

COMPARE PREDICTED
CURVE TO OBSERVED
CURVE

Figure C6. Schematic of integrated approach

C19

where

z = distance from top of sediment column, cm

C_I = initial contaminant concentration in the interstitial water, mg/ℓ

C_o = contaminant concentration in the water entering the sediment, mg/ℓ, equal to zero for the test procedures used in this study

$R = \dfrac{1 + K_d}{\Theta}$ = retardation coefficient, dimensionless

D = longitudinal dispersivity = $\dfrac{D_p}{V}$, cm

The initial and boundary conditions used to obtain Equation C7 are as follows:

$$C(z,0) = C_I$$

$$C(0,t) = C_o$$

$$\frac{\partial C}{\partial z}(\infty,t) = 0$$

38. If test procedures are free from error, the solution obtained from Equation C7 should agree with observed effluent concentrations from the permeameters. Thus, the integrated approach can be used to verify the mathematical form of an assumed source term.

PART IV: RESULTS AND DISCUSSION

Sediment Chemical Concentrations

39. Contaminant concentrations in Everett Harbor anaerobic sediment and interstitial water are presented in Table C1. Sediment solids contained low concentrations of PCB congeners, PAH compounds, and mercury, but relatively high concentrations of Cu, Pb, and Zn. Interstitial water concentrations of PAH compounds and PCB congeners were below detection limits as were concentrations of arsenic and mercury. Concentrations of other metals in the interstitial water were low.

40. Contaminant concentrations in aerobic Everett Harbor sediment and metal concentrations in the interstitial water are also presented in Table C2. Organic contaminants were not determined in the aerobic interstitial water because of the low total concentrations of organic contaminants in the aerobic sediment, the lack of detectable organic contaminants in the anaerobic interstitial water, and the small amounts of interstitial water extractable from aerobic sediment. Of particular notice were the high concentrations of Cd, Cu, Ni, and Zn in the aerobic interstitial water, a result of the lower pH in the aerobic sediment (3.9) compared to the anaerobic sediment (7.0).

41. In this report, organic contaminants are referred to by number because of the complexity of compound names and the number of organic contaminants analyzed. The key to organic compound identification is presented as Table C3. Specific PCB congeners were analyzed and reported instead of PCB Aroclors in order to achieve the enhanced limits of detection in water for congeners (0.01 µg/ℓ) compared to Aroclors (0.10 µg/ℓ). Only PCB Arochlor 1254 was detectable (0.25 mg/kg) in Everett Harbor sediment. Sediment detection limits for PCB congeners were 0.002 µg/g.

Salinity Testing

42. Leaching with water of varying salinity was conducted to determine if salinity would significantly impact metal concentrations in Everett Harbor leachate. Test data are presented in Table C4. These data show that increasing salinity had no apparent impact on release of heavy metals from Everett Harbor sediment solids into the leachate. The salinity of the water

Table C1

Contaminant Concentration in Anaerobic Everett Harbor
Sediment and Interstitial Water

Parameter	Sediment Concentration, µg/g	Interstitial Water Concentration, mg/ℓ
Metals		
As	5.7	<0.005
Cd	3.3	0.0014(0.0001)*
Cr	39.7	0.014(0.003)
Cu	73.4	0.004(0.001)
Pb	48.1	0.056(0.006)
Hg	0.2	<0.002
Ni	21.4	0.01(0.0003)
Zn	148.5	0.049(0.006)
Organics**		
1	8.2	<0.005
2		<0.005
3		<0.005
4	<1	<0.005
5	2.0	<0.005
6	2.2	<0.005
7	5.7	<0.005
8	1.5	<0.005
9	4.5	<0.005
10	4.0	<0.005
11	1.8	<0.005
12	2.1	<0.005
13	2.5	<0.005
14	2.5	<0.005
15	1.4	<0.005
16	<1	<0.005
17	<1	<0.005
18	<1	<0.005
19	<0.002	<0.00001
20	0.0087	<0.00001
21	<0.002	<0.00001
22	<0.002	<0.00001
23	<0.002	<0.00001
24	<0.002	<0.00001
25	<0.002	<0.00001
26	0.0079	<0.00001
27	<0.002	<0.00001
28	0.0087	<0.00001

(Continued)

* Standard error is given in parentheses.
** Key to organic contaminants is presented as Table C3.

Table C1 (Concluded)

Parameter	Sediment Concentration, μg/g	Interstitial Water Concentration, mg/ℓ
Organics (Cont.)		
29	0.0036	<0.00001
30	0.042	<0.00001
31	<0.002	<0.00001
32	0.01	<0.00001
33	<0.002	<0.00001
34	0.0809	

Table C2

Contaminant Concentration in Aerobic Everett Harbor
Sediment and Interstitial Water

Parameter	Sediment Concentration, µg/g	Interstitial Water Concentration, mg/ℓ
Metals		
As	5.7	<0.005
Cd	3.3	0.52(0.01)*
Cr	39.7	0.02(0.0007)
Cu	73.4	0.48(0.01)
Pb	48.1	0.09(0.003)
Hg	0.2	<0.0008
Ni	21.4	2.94(0.03)
Zn	148.5	37.5(0.015)
Organics**		
1	4.2	NS[†]
2		NS
3		NS
4	0.17	NS
5	1.3	NS
6	1.4	NS
7	5.0	NS
8	0.65	NS
9	5.3	NS
10	3.6	NS
11	1.4	NS
12	2.5	NS
13	2.5	NS
14	2.5	NS
15	1.1	NS
16	0.53	NS
17	<0.63	NS
18	0.38	NS
19	<0.002	NS
20	0.0093	NS
21	0.0061	NS
22	<0.002	NS
23	<0.002	NS
24	<0.002	NS
25	0.0061	NS
26	0.0079	NS
27	<0.002	NS
28	0.012	NS

(Continued)

* Standard error is given in parentheses.
** Key to organic contaminants is presented as Table C3.
[†] NS = not sampled.

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Table C2 (Concluded)

Parameter	Sediment Concentration, μg/g	Interstitial Water Concentration, mg/ℓ
Organics (Cont.)		
29	0.047	NS
30	<0.002	NS
31	<0.002	NS
32	0.021	NS
33	0.042	NS
34	0.151	NS

Table C3

Organic Compound Identification Key Used in This Report

1.	Naphthalene	18.	Benzo(g h i)perylene
2.	1-methylnaphthalene	19.	2,4-dichlorobiphenyl
3.	2-methylnaphthalene	20.	2,4'-dichlorobiphenyl
4.	Acenaphthalene	21.	2,4,4'-trichlorobiphenyl
5.	Acenapthene	22.	2,3',4',5-tetrachlorobiphenyl
6.	Fluorene	23.	2,2',4,5'-tetrachlorobiphenyl
7.	Phenathrene	24.	2,2',5,5'-tetrachlorobiphenyl
8.	Anthracene	25.	2,2',4,6-tetrachlorobiphenyl
9.	Fluoranthene	26.	2,2',3',4,5-pentachlorobiphenyl
10.	Pyrene	27.	2,2',4,5,5'-pentachlorobiphenyl
11.	Chrysene	28.	2,2',3,4,5'-pentachlorobiphenyl
12.	Benzo(a)anthracene	29.	2,2',3,4,4',5'-hexachlorobiphenyl
13.	Benzo(b)fluoranthene	30.	2,2',4,4',5,5'-hexachlorobiphenyl
14.	Benzo(k)fluoranthene	31.	2,2',3,3',6,6'-hexachlorobiphenyl
15.	Benzo(a)pyrene	32.	2,2',3,4,5,6'-hexachlorobiphenyl
16.	Indeno(1 2 3-c d)pyrene	33.	2,2',3,4,4',5,5'-heptachlorobiphenyl
17.	Dibenzo (a h) anthracene	34.	Total Arochlor 1254 congeners

Table C4

Heavy Metal Leachate Concentration as a

Function of Leachate Salinity

Parameter	Salinity, ppt			
	0	5	15	25
As	0.009 (0.0006)	0.009 (0.002)	0.008 (0.0025)	0.008 (0.0005)
Cd	0.002 (0.001)	0.0006 (0.0001)	0.0003 (0.0001)	0.0004 (0.0001)
Cr	0.003 (0.0006)	0.002 (0.0003)	0.002 (0.0000)	0.006 (0.002)
Cu	0.003 (0.0006)	0.003 (0.0003)	0.003 (0.0007)	0.009 (0.006)
Pb	0.020 (0.007)	0.004 (0.0000)	0.004 (0.0005)	0.003 (0.0006)
Hg	<0.002	<0.002	<0.002	<0.002
Ni	0.007 (0.0015)	0.006 (0.0006)	0.0095 (0.002)	0.01 (0.002)
Zn	0.048 (0.011)	0.050 (0.003)	0.044 (0.002)	0.053 (0.006)

Note: Concentrations are given in milligrams per litre (standard error).

used in the testing should, therefore, exert little influence on leachate results.

Kinetic Testing

43. Kinetic testing was performed to determine shaking time necessary to reach steady-state leachate contaminant concentrations. Test results for metals are presented in Table C5. Results show that leachate metal concentrations following 1 day of shaking did not significantly differ ($p < 0.05$) from leachate metal concentrations following 2, 3 or 7 days of shaking. It was therefore determined that a 24-hr shaking time was sufficient for metal concentrations to reach steady-state conditions. No release of Hg was observed, but testing for this parameter was continued.

44. Organic contaminant leachate results as a function of shaking time are presented in Table C6. Data showed that shake time did not alter leachate concentrations of the three PAH compounds detected. However, concentrations of these compounds were near the detection limit and were only detected because the GC/MS signal is particularly strong for these compounds. In this test, PCB congeners were not run since, during early testing of this sediment, all PCB Arochlor concentrations were below detection limits, and testing for PCB congeners had not yet begun. Previous work on Indiana Harbor sediment has shown, however, that PCB congeners and PAH compounds behave similarly during kinetic testing. Therefore a 24-hr shaking time was considered appropriate for batch testing of organic contaminants as well as metals.

Selection of Water to Sediment Ratio

45. Batch leaching tests were also conducted to determine the water to sediment ratio that would approximate contaminant distributions found in settled dredged material placed in a CDF. When dredged material is first added to a site, this would approximate a 1:1 ratio. However, the water to sediment ratio must also be large enough to allow generation of sufficient leachate for organic contaminant analyses (approximately 1 ℓ/sample). The effect of varying the water to sediment ratio on leachate metal concentrations from anaerobic Everett Harbor sediment is presented in Table C7. Concentrations at water to sediment ratios of 4:1 were either higher than (As) or

Table C5

Release of Metals into Leachate as a Function of Shaking Time

Parameter	Time of Shaking, days			
	1	2	3	7
As	9 (0.6)	9 (1)	5.6 (0.7)	5 (0.0)
Cd	2.2 (1)	1 (0.4)	0.6 (0.2)	0.2 (0.1)
Cr	1 (0.0)	3 (0.0)	3.3 (1.3)	3.7 (0.9)
Cu	3 (0.6)	4.7 (0.9)	5.0 (1.7)	6.0 (0.6)
Pb	20 (6.6)	13 (0.9)	15 (0.7)	9 (0.7)
Hg	<2	<2	<2	<2
Ni	7 (1.5)	20 (9)	13 (3)	5 (0.9)
Zn	48 (11)	187 (64)	64 (21)	45 (9)

Note: Concentrations are given in micrograms per litre (standard error).

Table C6

Release of Organic Contaminants into Leachate as a Function
of Time of Shaking

Parameter	Time of Shaking, days			
	1	2	3	4
5	0.0023 (0.0012)	0.0026 (0.0014)	0.002 (0.0004)	0.0021 (0.0003)
9	0.0016 (0.0006)	0.0036 (0.0007)	0.0022 (0.0044)	0.0033 (0.0002)
10	0.002 (0.0005)	0.C03 (0.0006)	0.002 (0.0004)	0.0026 (0.0001)

Note: Concentrations are given in milligrams per litre (standard error).
All other organic contaminants tested were below detection limits.

Table C7

Release of Metals into Leachate from Anaerobic Everett Harbor
Sediment as a Function of Water to Sediment Ratio

Parameter	Water to Sediment Ratio				
	4:1	8:1	12:1	50:1	100:1
As	0.024 (0.001)	<0.005	<0.005	<0.005	<0.005
Cd	0.0014 (0.0003)	0.001 (0.0001)	0.0008 (0.00003)	0.0011 (0.000)	0.0007 (0.00007)
Cr	0.002 (0.0006)	0.003 (0.0009)	0.003 (0.0009)	0.007 (0.002)	0.006 (0.001)
Cu	0.004 (0.0003)	0.004 (0.0003)	0.006 (0.003)	0.003 (0.000)	0.004 (0.0007)
Pb	0.006 (0.001)	0.002 (0.000)	0.002 (0.0003)	0.002 (0.0007)	0.002 (0.0003)
Hg	<0.002	<0.002	<0.002	<0.002	<0.002
Ni	0.016 (0.004)	0.013 (0.002)	0.009 (0.004)	0.007 (0.002)	0.004 (0.0007)
Zn	0.050 (0.010)	0.030 (0.003)	0.045 (0.004)	0.042 (0.0042)	0.035 (0.0035)

Note: Concentrations are given in milligrams per litre (standard error).

statistically the same as ($p < 0.05$) leachate metal concentrations measured at higher water to sediment ratios. Comparison of anaerobic interstitial water metal concentrations (Table C1) with anaerobic leachate results in Table C7 showed general agreement with the exception of As, which was lower in the interstitial water, and Pb, which was slightly higher. Therefore, use of a 4:1 water to sediment ratio should yield contaminant distributions that reasonably estimate the distribution at a liquid-solids ratio of 1:1.

46. Aerobic Everett Harbor sediment leachate possessed a low pH, which can strongly impact metal mobility. As a result, an additional water to sediment ratio test was conducted with the aerobic sediment to determine if results observed for metals with anaerobic sediment held for the aerobic sediment. Results are presented in Table C8 and show that leachate metal

concentrations at water to sediment ratios of 4:1 were either higher or statistically the same (p < 0.05) as leachate metal concentrations at higher water to sediment ratios. Therefore, a 4:1 water to sediment ratio was also considered appropriate for aerobic Everett Harbor sediment despite its low pH. Leachate pH during this test averaged 4.3 with a standard error of 0.03.

47. The effect of the water to sediment ratio on leachate concentrations of organic contaminants in anaerobic Everett Harbor sediment is presented in Table C9. Leachate concentrations in the 4:1 water to sediment ratio test were either higher than or equal to leachate concentrations at higher water to sediment ratios. Organic contaminants were not detected in the Everett Harbor interstitial water (Table C1); thus, leachate concentrations in the 4:1 water to sediment ratio provided a possible worst-case estimate.

Sequential Batch Leaching

General leachate quality

48. Leachate conductivity, pH, and total organic carbon (TOC) concentrations for the batch leaching tests are summarized in Tables C10, C11 and C12, respectively. For all tests conducted, leachate conductivity gradually decreased. Leachate pH from anaerobic sediment was 7.3 during the first two leaching sequences, then increased steadily to a peak of 8.8 as leaching continued, a pH rise of 1.5 units. Similar trends were observed in the anaerobic challenge tests although the rise in pH was not as high and occurred two leach sequences later. Anaerobic leachate TOC concentrations peaked in the fourth step of sequential batch testing, coincident with the rise in leachate pH. Similar trends were observed in the anaerobic challenge testing. Total organic carbon in the aerobic batch tests did not show the trends observed during anaerobic testing, but exhibited a generally steady decrease from initial values. There was no difference in initial TOC concentrations between anaerobic and aerobic tests despite the large difference between anaerobic (7.15 percent) and aerobic (3.11 percent) sediment TOC concentrations. A marked difference in leaching conditions was therefore experienced during the course of the anaerobic leaching procedure. The change in anaerobic conductivity should not cause changes in metal release characteristics based on results of the salinity tests. The same cannot be said for the change in

Table C8

Release of Metals into Aerobic Leachate as a Function of Liquid to Solid Ratio

Parameter	Water to Sediment Ratio					
	4:1	10:1	20:1	30:1	40:1	50:1
As	0.004(0.001)	0.005(0.002)	0.008(0.001)	0.008(0.003)	<0.005	<0.005
Cd	0.11(0.006)	0.07(0.005)	0.009(0.000)	0.004(0.003)	0.009(0.000)	0.006(0.000)
Cr	0.008(0.001)	0.007(0.002)	0.25(0.25)	0.003(0.0004)	0.003(0.0003)	0.002(0.000)
Cu	0.008(0.002)	0.03(0.006)	0.02(0.003)	0.03(0.008)	0.01(0.001)	0.008(0.0004)
Pb	0.10(0.008)	0.04(0.003)	0.01(0.001)	0.01(0.001)	0.005(0.001)	0.008(0.003)
Hg	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Ni	0.93(0.02)	0.47(0.008)	0.20(0.008)	0.14(0.004)	0.12(0.002)	0.08(0.004)
Zn	10.36(0.2)	5.10(0.10)	1.79(0.07)	1.27(0.03)	1.02(0.04)	0.69(0.03)

Note: Concentrations are given in milligrams per litre (standard error).

Table C9

Release of PAH Compounds into Leachate from Anaerobic Everett
Harbor Sediment as a Function of Liquid to Solid Ratio

Water to Sediment Ratio	Parameter			
	5	7	9	10
4:1	0.0012(0.0002)	0.0036(0.0002)	0.0023(0.0001)	0.0023(0.00007)
8:1	0.0013(0.00003)	0.0003(0.0005)	0.0017(0.0001)	0.0015(0.0001)
12:1	0.0015(0.0003)	0.001(0.0005)	0.001(0.0005)	0.001(0.0004)
50:1	0.0007(0.0003)	0.0015(0.0008)	<0.001	<0.001
100:1	0.0005(0.0002)	<0.001	<0.001	<0.001

Note: Concentrations are given in milligrams per litre (standard error).

leachate pH over the course of the anaerobic leaching procedure. Such a pro-
nounced change would be expected to have a marked impact on anaerobic metal
release.

49. Aerobic Everett Harbor sediment leachate pH was much lower than the
values observed for anaerobic sediment (Table C11). Challenging aerobic sed-
iment with aerobic leachate resulted in even lower pH values. Leachate pH
during the initial aerobic testing exceeded the value of 4.3 observed in the
water to sediment ratio testing; this occurred even though only 1 week passed
between the two tests and the aerobic sediment was refrigerated at 4° C
between tests. These pH differences were apparently due to reduction
processes in the stored sediment. The redox potential of stored aerobic sedi-
ment that gave a leachate pH of 4.8 was +200 mV. When this sediment was
placed into glass aquaria and allowed to oxidize for 2 weeks using the same
procedure employed during the initial oxidation, redox potential of the sedi-
ment rose to +550 mV and pH dropped to 4.3. Because of the pH rise during
storage, aerobic challenge testing results most closely match leaching

Table C10

Conductivity in Everett Harbor Serial Batch Leachate

Test	Sequential Leach Number								
	1	2	3	4	5	6	7	8	9
Anaerobic	8.23(0.6)	3.0(0.02)	1.2(0.04)	2.4(0.01)	0.3(0.02)	0.2(0.01)	0.2(0.01)	0.2(0.01)	0.1(0.01)
Anaerobic challenge	9.9(0.2)	4.5(0.1)	2.5(0.05)	1.2(0.02)	0.7(0.03)	0.5(0.01)	NC	0.4(0.02)	0.4(0.03)
Aerobic	6.5(0.21)	2.9(0.11)	1.4(0.13)	0.6(0.05)	0.3(0.03)	0.3(0.00)	0.2(0.01)	0.2(0.01)	0.2(0.004)
Aerobic challenge	14.3(0.13)	5.2(0.2)	2.2(0.04)	1.3(0.05)	0.7(0.02)	0.5(0.01)	0.4(0.01)	0.4(0.01)	NC

Notes: Values are given in millisiemens (standard error).
NC = not conducted.

Table C11

Everett Harbor Serial Batch Leachate pH

Test	Sequential Leach Number								
	1	2	3	4	5	6	7	8	9
Anaerobic	7.3(0.005)	7.3(0.25)	8.2(0.01)	8.5(0.02)	8.7(0.02)	8.8(0.02)	8.8(0.01)	8.8(0.01)	8.7(0.14)
Anaerobic challenge	7.1(0.002)	7.4(0.015)	7.2(0.00)	7.5(0.01)	8.4(0.04)	8.5(0.04)	NC	8.1(0.12)	7.8(0.15)
Aerobic	4.8(0.03)	4.7(0.02)	4.9(0.04)	4.9(0.03)	4.8(0.03)	4.8(0.03)	4.6(0.04)	4.7(0.03)	4.6(0.03)
Aerobic challenge	3.9(0.05)	3.5(0.07)	3.6(0.08)	3.6(0.07)	3.9(0.07)	3.9(0.06)	3.8(0.07)	3.7(0.07)	NC

Notes: Standard error given in parentheses.
NC = not conducted.

Table C12

Total Organic Carbon Concentration in Everett Harbor Leachate

Time days	Anaerobic Testing		Aerobic Testing	
	Sequential	Challenge	Sequential	Challenge
1	84(10)	75(6)	54(5)	77(9)
2	94(25)	86(4)	28(2)	52(12)
3	130(37)	125(32)	22(60)	26(1)
4	181(28)	152(63)	39(8)	25(1)
5	85(8)	168(86)	37(11)	34(2)
6	67(8)	127(32)	42(7)	21(2)
7	56(10)	NT*	31(3)	NT

Notes: Values are expressed in (milligrams/litre (standard error)).
       NT = not tested.

conditions for fully oxidized Everett Harbor sediment. In the future, only freshly oxidized, unstored sediment should be used for aerobic testing.

Metal releases

50. Steady-state metal concentrations in sediment (q) and leachate (C) obtained from the sequential batch leaching tests for anaerobic Everett Harbor sediment are presented in Tables C13 and C14, respectively. Steady-state q and C concentrations obtained from the challenge testing for anaerobic Everett Harbor sediment are presented in Tables C15 and C16, respectively. Changes in releases of metals in anaerobic leachate can be seen in Figure C7, which presents changes in leachate concentration of As and Ni as a function of sequential leach number. These data show that As and Ni leachate concentrations were low initially, peaked at either the third or fourth leach step, then declined. That is, initially the isotherms for these elements exhibited an inverse relationship (C increases as q decreases). However, after the third or fourth leaching step, the relationship between q and C changed to a direct one (C decreases as q decreases).

Table C13

Steady-State Sediment Metal Concentrations for Everett Harbor

Sediment Following Anaerobic Leaching

Metal	Sequential Leach Number								
	1	2	3	4	5	6	7	8	9
As	5.73(0.00)	5.73(0.00)	5.67(0.01)	5.55(0.03)	5.45(0.03)	5.39(0.03)	5.35(0.03)	5.33(0.03)	5.31(0.03)
Cd	3.30(0.00)	3.29(0.00)	3.29(0.01)	3.27(0.01)	3.23(0.02)	3.22(0.02)	3.21(0.02)	3.20(0.02)	3.19(0.02)
Cr	39.7(0.00)	39.7(0.00)	39.6(0.03)	39.6(0.03)	39.5(0.03)	39.5(0.03)	39.4(0.03)	39.4(0.03)	39.3(0.04)
Cu	73.3(0.00)	73.3(0.01)	73.2(0.03)	72.9(0.04)	72.6(0.05)	72.4(0.05)	72.2(0.05)	72.0(0.06)	71.7(0.06)
Pb	48.0(0.00)	48.0(0.01)	47.9(0.01)	47.8(0.08)	47.6(0.09)	47.4(0.09)	47.2(0.09)	47.1(0.08)	46.9(0.08)
Ni	21.3(0.00)	21.3(0.01)	21.1(0.03)	21.0(0.01)	20.9(0.02)	20.8(0.02)	20.8(0.02)	20.7(0.02)	20.6(0.01)
Zn	148.4(0.01)	148.0(0.11)	147.7(0.10)	147.3(0.19)	146.9(0.20)	146.5(0.19)	146.0(0.19)	145.8(0.24)	145.5(0.23)

Note: Concentrations are given in micrograms per gram (standard error).

C35

Table C14

Heavy Metal Concentrations in Leachate from Anaerobic Everett Harbor Sediment

Compound	Sequential Leach Number								
	1	2	3	4	5	6	7	8	9
As	<5	<5	19.5(3.7)	37.8(9.1)	34.5(1.0)	18.5(1.0)	10.8(0.5)	8.3(0.8)	6.3(0.8)
Cd	0.7(0.1)	1.0(0.1)	3.3(0.3)	6.1(3.2)	10.7(7.5)	3.1(0.3)	4.3(0.1)	3.1(0.1)	4.6(0.2)
Cr	6.3(1.1)	8.8(0.6)	24.5(10.5)	20.5(3.6)	20.5(4.9)	13.0(1.1)	13.0(0.7)	11.0(0.6)	12.5(1.7)
Cu	4.5(1.2)	16.5(3.3)	44.0(11.4)	96.5(20.7)	81.3(9.2)	49.0(2.9)	82.0(3.7)	71.5(5.0)	95.5(2.9)
Pb	29.5(1.0)	10.3(1.8)	14.5(1.9)	56.0(26.9)	47.8(5.2)	61.0(2.4)	60.8(1.1)	52.8(3.0)	57.0(1.7)
Ni	10.0(1.4)	20.8(4.2)	52.0(8.6)	33.0(3.0)	31.5(4.3)	21.5(2.0)	25.8(3.1)	14.5(1.0)	22.3(2.9)
Zn	42.8(5.0)	131(40)	82.3(15.7)	149(31.7)	127(13.8)	135(13.4)	133(2.3)	71.5(21.8)	97.8(3.6)

Note: Concentrations are given in micrograms per litre (standard error).

C36

Table C15

Steady-State Sediment Metal Concentrations in Anaerobic

Everett Harbor Sediment Challenged with Leachate from

Anaerobic Everett Harbor Sediment

Parameter	Sequential Leach Number								
	1	2	3	4	5	6	7	8	9
As	5.66(0.06)	5.45(0.11)	5.34(0.11)	5.22(0.10)	5.18(0.09)	5.15(0.09)	5.13(0.08)	5.12(0.08)	5.12(0.08)
Cd	3.30(0.002)	3.25(0.05)	3.24(0.05)	3.21(0.04)	3.20(0.04)	3.12(0.07)	3.11(0.07)	3.09(0.07)	3.08(0.07)
Cr	39.7(0.002)	39.6(0.12)	39.6(0.12)	38.7(0.68)	38.6(0.67)	38.5(0.7)	38.5(0.7)	38.4(0.7)	38.4(0.7)
Cu	73.3(0.004)	72.4(0.79)	72.4(0.79)	72.0(0.75)	71.6(0.73)	71.0(0.7)	70.2(0.7)	69.8(0.7)	69.6(0.6)
Pb	48.0(0.04)	47.6(0.41)	47.5(0.41)	47.4(0.39)	47.2(0.37)	46.8(0.32)	46.5(0.3)	46.4(0.3)	46.3(0.3)
Ni	21.3(0.02)	20.9(0.26)	20.9(0.25)	20.1(0.33)	19.9(0.31)	19.8(0.31)	19.7(0.3)	19.6(0.3)	19.6(0.3)
Zn	148.2(0.19)	147.1(0.88)	146.9(0.87)	145.5(1.23)	144.9(1.15)	144.3(1.11)	143.8(1.09)	143.4(1.06)	143.1(1.04)

Note: Concentrations are given in micrograms per gram (standard error).

C37

Table C16

Steady-State Leachate Metal Concentrations in Anaerobic Everett Harbor Sediment

Challenged with Leachate from Anaerobic Everett Harbor Sediment

Parameter	Sequential Leach Number								
	1	2	3	4	5	6	7	8	9
As	23.7(18.2)	31.0(1.6)	35.3(3.1)	34.0(4.9)	15.0(2.5)	8.5(0.6)	7.8(0.8)	1.3(1.3)	BDL
Cd	1.6(0.6)	1.3(0.2)	1.2(0.1)	8.7(6.4)	3.9(0.4)	24.3(11.8)	5.4(0.3)	3.9(0.3)	3.0(0.5)
Cr	7.3(0.9)	7.8(0.9)	9.8(0.8)	270(223)	26.3(4.0)	21.0(3.1)	19.3(2.1)	11.3(1.5)	8.5(1.6)
Cu	10.0(1.7)	23.0(9.5)	21.8(3.6)	105(46)	120(6.1)	21.0(10.3)	233(13.3)	115(10.2)	65(15.3)
Pb	25.3(10.8)	18.3(3.7)	12.5(2.3)	44.3(16.6)	57(3.9)	118(18)	81(1.5)	37(8.4)	19(5.7)
Ni	33.0(8.5)	20(7.0)	21(3.3)	239(122)	43(6.1)	37(1.2)	36(1.3)	18(2.7)	11(1.6)
Zn	108(63)	43(18.3)	53(4.2)	444(319)	183(21.4)	179(9.3)	169(8.9)	126(14.3)	76(9.5)

Note: Concentrations are given in micrograms per litre (standard error).
BDL = below detection limit.
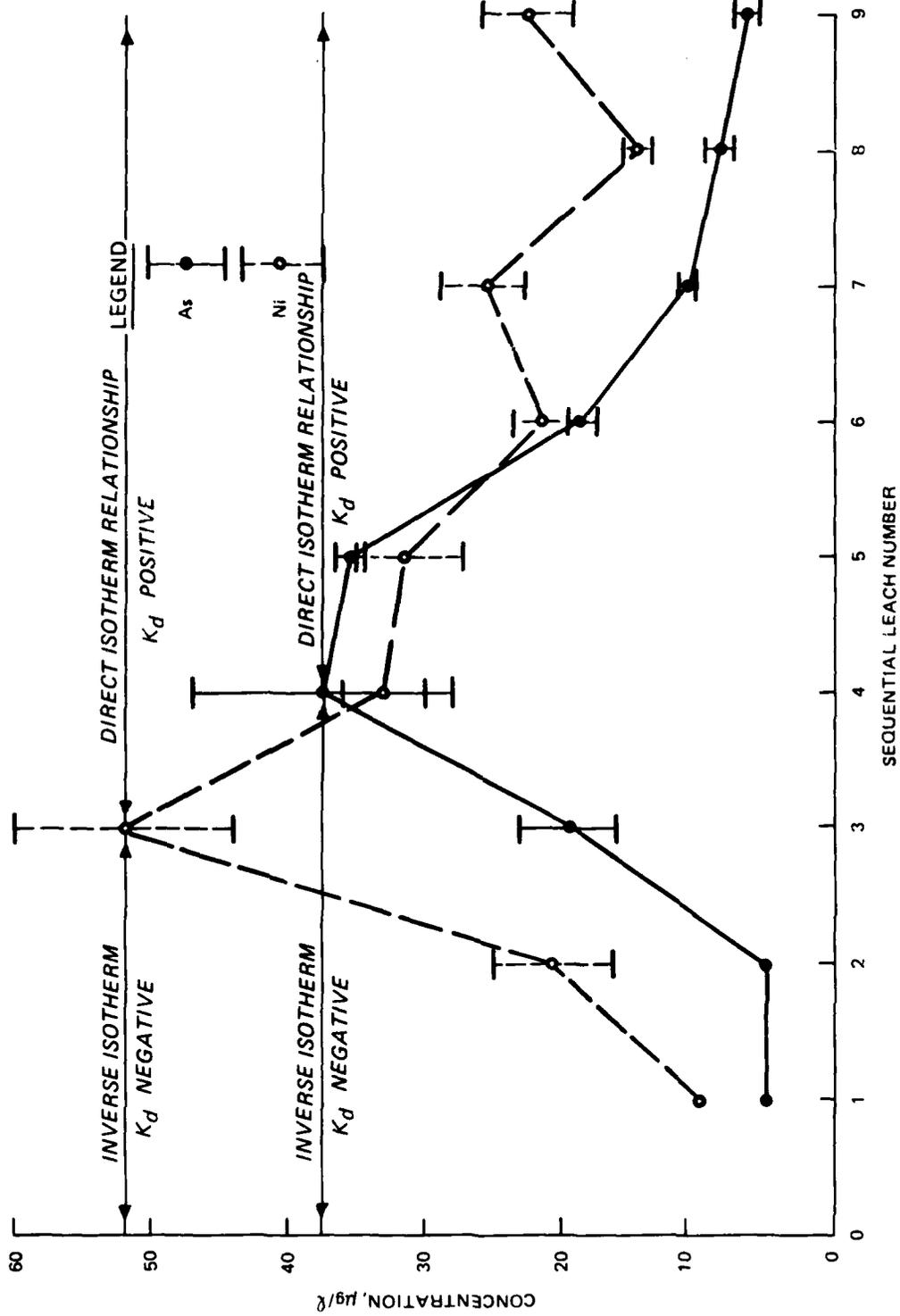
Figure C7. Arsenic and nickel concentrations in Everett Harbor leachate

51. Desorption isotherms for the anaerobic metal data are provided in Figures C8-C14. As shown in these figures, release of metals from anaerobic sediment did not follow the ideal desorption isotherms presented in Figure C5. Two of the desorption isotherms are double-valued (Figures C8 and C13), and two, although linear, had reverse slopes (Figures C11 and C12). The turning point for the As and Ni desorption isotherms (Figures C8 and C13) is coincident with establishment of steady leachate pH (Table C11). Reverse and double-valued desorption isotherms are indicative of nonconstant sediment chemistry, probably variable pH, that affects metal mobility.

52. If all the steps in the sequential leach procedure are considered, there is no significant ($p < 0.05$) linear relationship between steady-state sediment and leachate As or Ni concentrations. However, if only data following the peak are considered, there is a strong linear relationship between steady-state sediment and leachate concentrations for As and Ni. Thus, after pH became constant, distribution of As and Ni between sediment solids and leachate behaved like an ideal desorption isotherm. Distribution coefficients for As and Ni and the associated standard errors for the ideal portion of the desorption isotherm were 5.36(0.56) and 8.56(1.49), respectively. The data in Tables C13 and C14 and Figures C8-C14 show that the remainder of the metals analyzed did not exhibit the leaching trends of As and Ni. Copper and lead showed significant inverse linear relationships ($p < 0.05$) between steady-state sediment and leachate concentrations yielding distribution coefficients (standard error) of -13.9(0.58) and -15.7(0.84), respectively. The nonideal desorption isotherms for Cu and Pb (reverse isotherms) are also probably a pH effect, although a turning point was not observed. Theoretically and practically, a turning point must exist; otherwise the desorption isotherm will intersect the abscissa, a physical impossibility. Mercury was not detected in any of these leachates. The remainder of the metals, Cd, Cr, and Zn, displayed no well-defined relationship between steady-state sediment and leachate concentrations.

53. Many of the same trends observed in the anaerobic sequential testing were also observed in the anaerobic sequential challenge testing (Tables C15 and C16). Leachate concentrations of Ni and As showed similar trends to those presented in Figure C7, although peak leachate concentrations for both parameters occurred during the fourth leach cycle. Distribution coefficients (standard error) in the challenge tests derived for As and Ni in
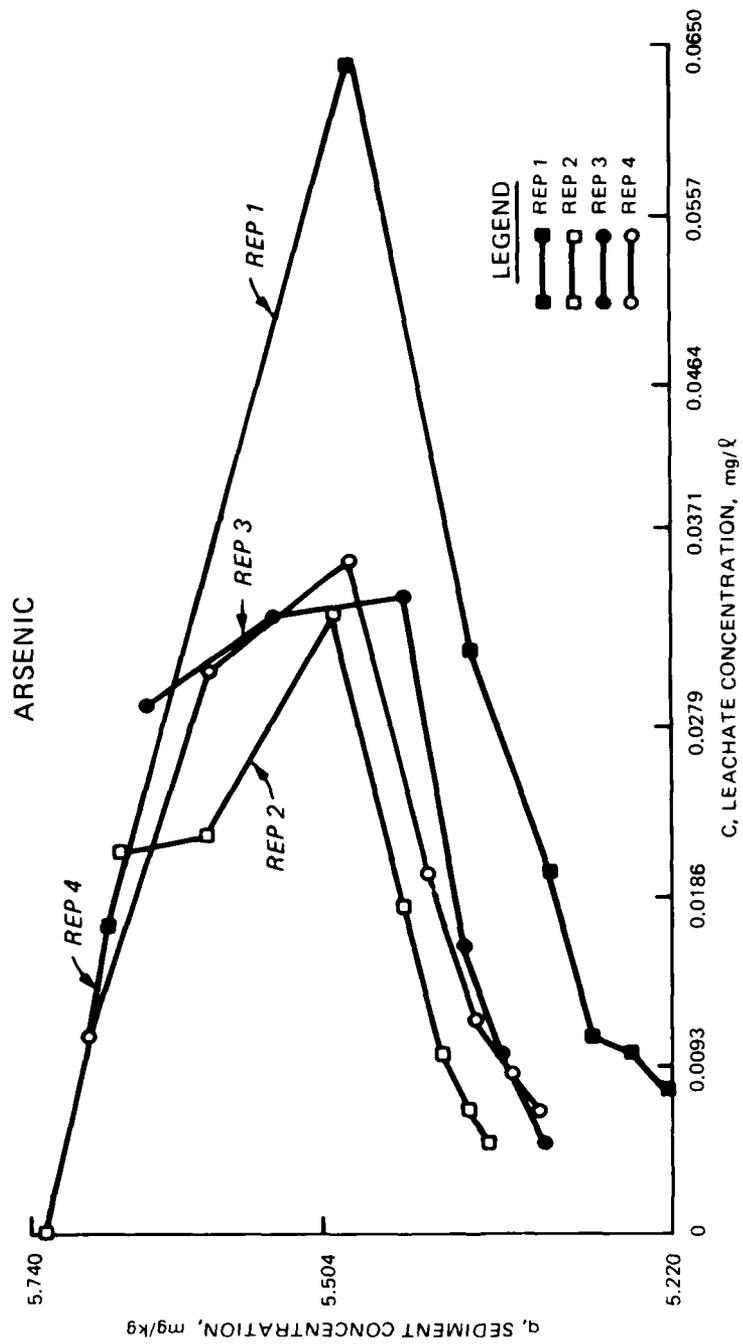
Figure C8. Arsenic desorption isotherm, anaerobic sediment
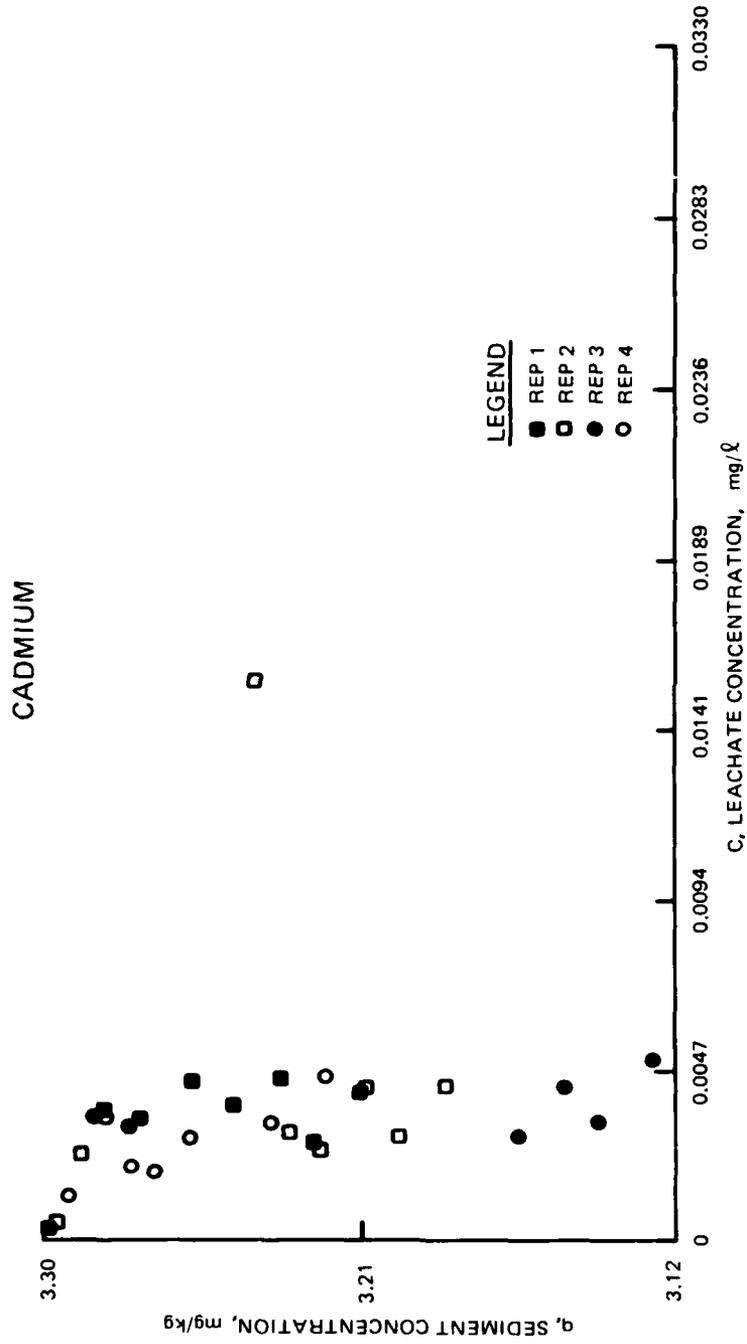
Figure C9. Cadmium desorption isotherm, anaerobic sediment
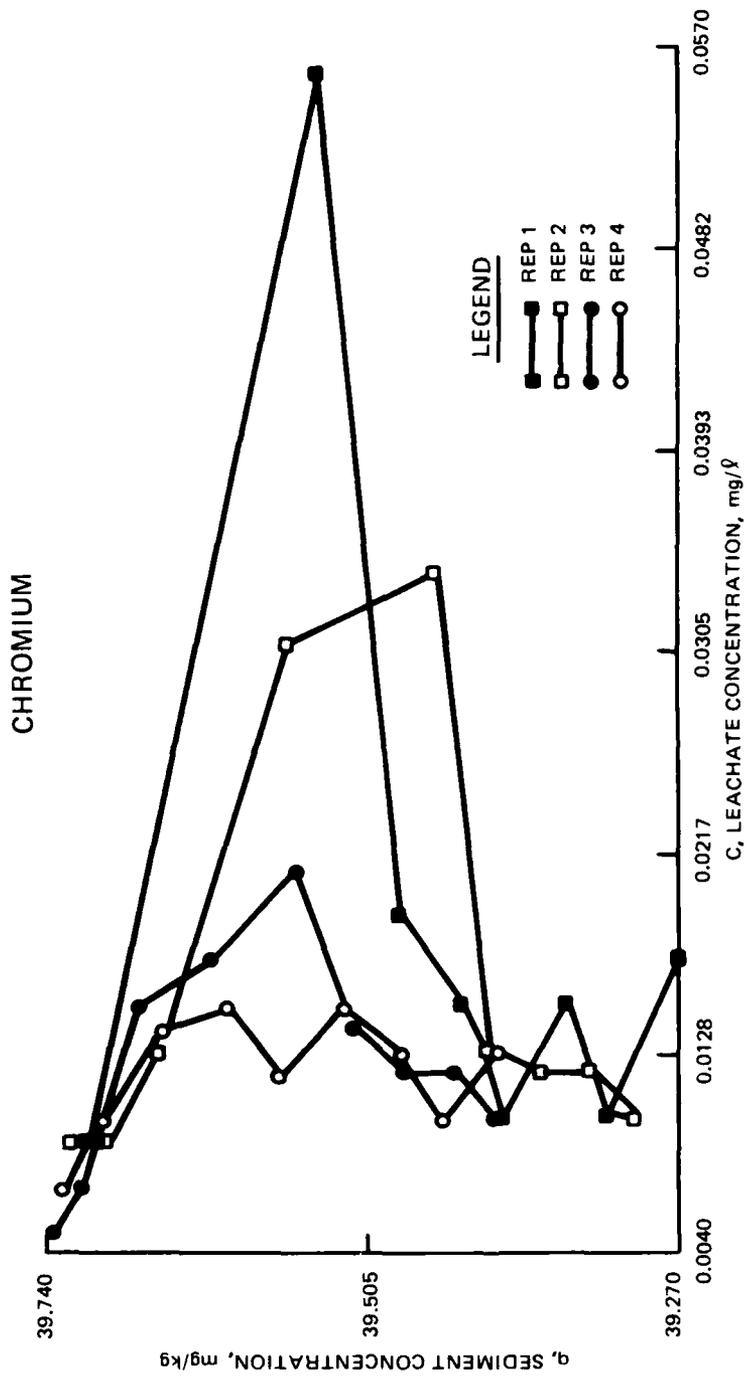
Figure C10. Chromium desorption isotherm, anaerobic sediment
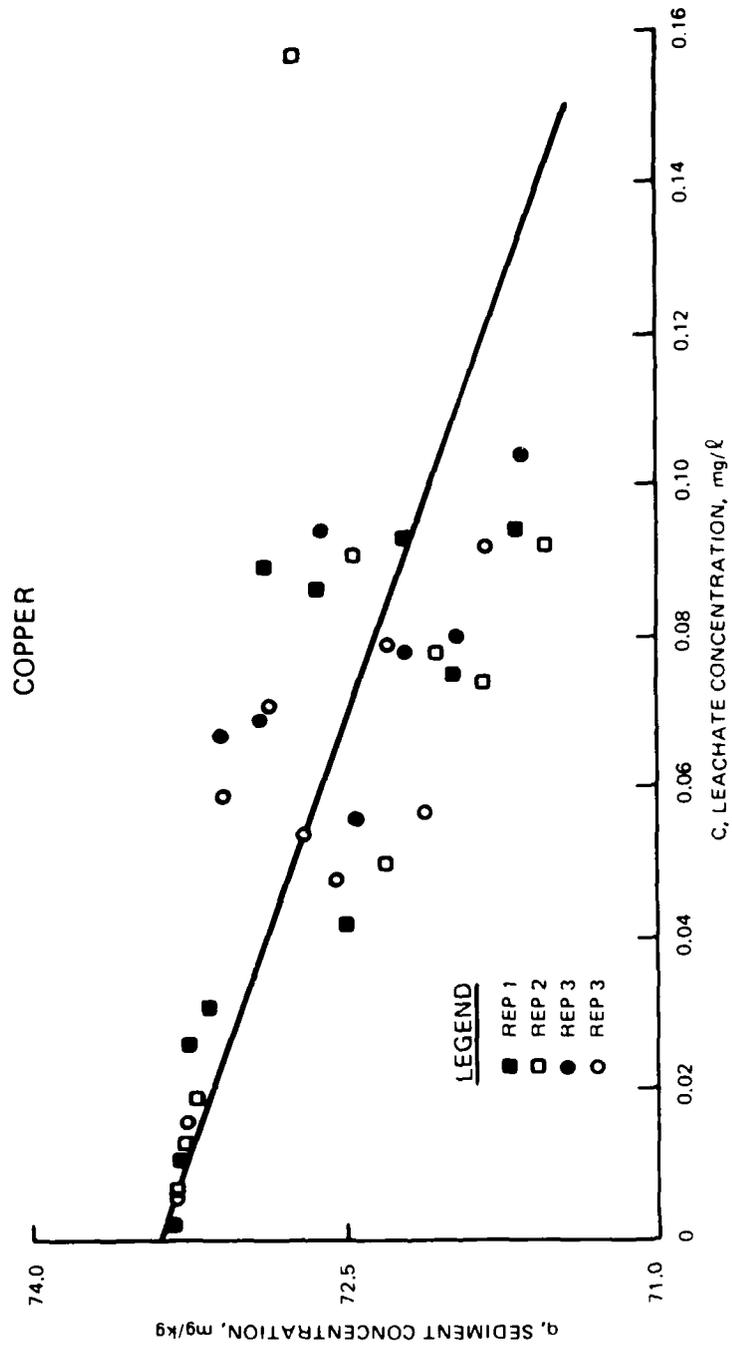
Figure C11. Copper desorption isotherm, anaerobic sediment
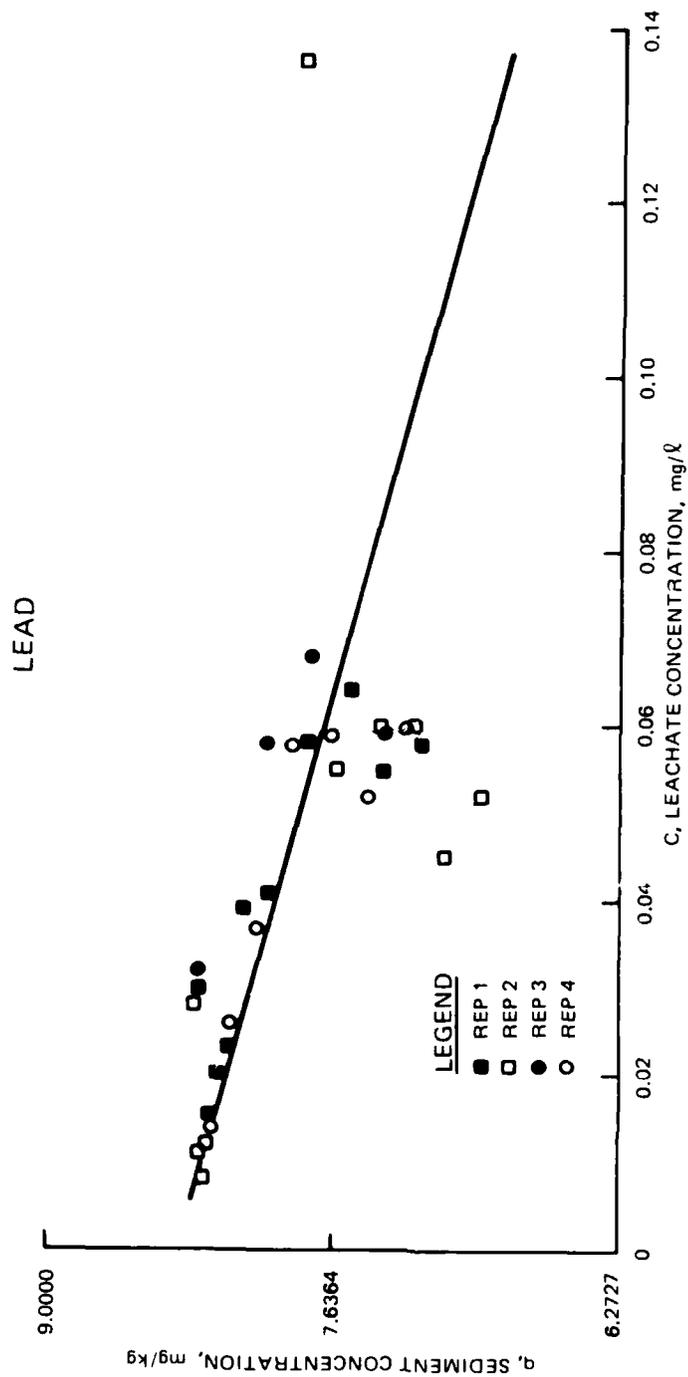
Figure C12. Lead desorption isotherm, anaerobic sediment
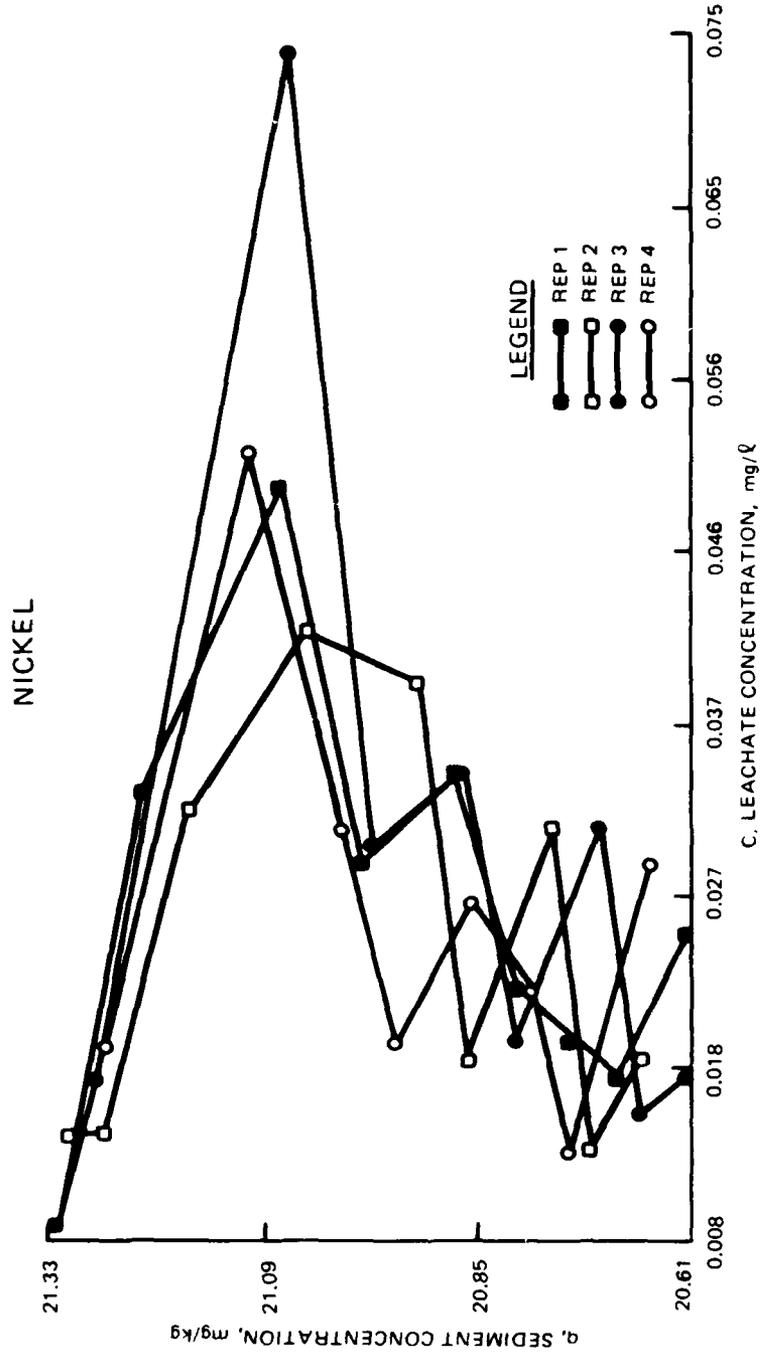
Figure C13. Nickel desorption isotherm, anaerobic sediment

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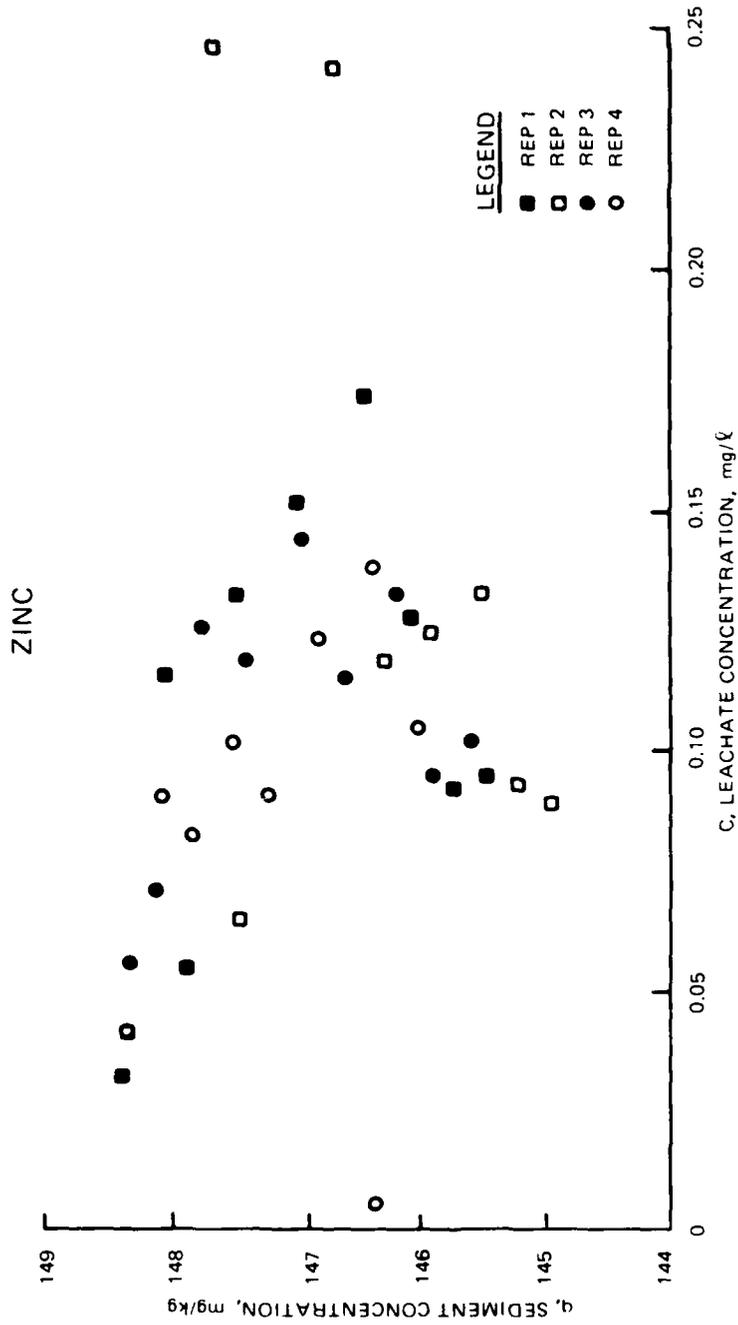
Figure C14. Zinc desorption isotherm, anaerobic sediment

the same manner as for the sequential batch tests following peak concentra-
tions were 3.75(0.44) and 4.11(1.65), respectively. The remainder of the
metals displayed no well-defined relationship between q and C .

54. Steady-state q and C metal concentrations obtained from the
sequential batch leaching tests under aerobic conditions are presented in
Tables C17 and C18, respectively. Steady-state q and C metal concentra-
tions obtained from the challenge sequential batch leaching tests under
aerobic conditions are presented in Tables C19 and C20, respectively. Mercury
data are not presented because all values were below the detection limit of
0.002 mg/ℓ. Arsenic and chromium displayed no linear relationship between
concentrations for either sequential or challenge batch testing, as did Cd,
Cu, and Pb in the sequential batch testing. Distribution coefficients for
aerobic Everett Harbor sequential and challenge batch leaching for which a
statistically significant (p < 0.05) linear relationship exists are summarized
in Table C21.

55. Development of aerobic conditions in Everett Harbor sediment
resulted in substantial releases of heavy metals into batch test leachate.
Metal losses observed during this study under anaerobic and aerobic leaching
conditions are summarized in Table C22. As can be seen, release of over
85 percent of sediment-bound Zn occurred during the course of aerobic chal-
lenge testing.

Organic contaminant releases

56. Steady-state organic contaminant concentrations in leachate and
sediment of anaerobic Everett Harbor sediment are listed in Tables C23 and
C24, respectively. Of particular note is that only 8 of 33 compounds moni-
tored were detected in the leachate. Compounds that were detected were
present in very low concentrations, generally below the stated detection
limits of 5 μg/ℓ for PAH compounds analyzed using GC/MS. They were detected
only because they have a strong, stable molecular ion that does not readily
fragment, resulting in a strong signal at the detector. Concentrations of PCB
congeners were very low, as would be expected based on the low concentrations
in the sediment. Similar results were obtained in the sequential challenge
testing for organic contaminants in anaerobic sediment (Tables C25 and C26).
Changes in steady-state sediment concentrations for both sequential and chal-
lenge batch testing were small; 0.124 μg/g was the the highest concentration

Table C17

Steady-State Sediment Metal Concentration for Aerobic Everett Harbor Sediment

Metal	Sequential Leach Number								
	1	2	3	4	5	6	7	8	9
As	5.7(0.000)	5.7(0.000)	5.7(0.000)	5.7(0.000)	5.68(0.006)	5.68(0.000)	5.68(0.000)	5.68(0.000)	5.65(0.01)
Cd	3.27(0.009)	3.03(0.08)	2.92(0.032)	2.91(0.003)	2.89(0.002)	2.88(0.001)	2.86(0.003)	2.84(0.005)	2.80(0.006)
Cr	39.7(0.003)	39.7(0.004)	39.7(0.004)	39.5(0.04)	39.3(0.03)	39.2(0.03)	39.0(0.02)	38.9(0.02)	38.7(0.02)
Cu	73.4(0.003)	73.3(0.01)	73.3(0.007)	73.1(0.06)	72.8(0.06)	72.5(0.04)	72.3(0.02)	72.1(0.02)	72.0(0.02)
Pb	48.1(0.009)	47.9(0.06)	47.8(0.003)	47.7(0.01)	47.6(0.01)	47.6(0.01)	47.5(0.007)	47.5(0.004)	47.5(0.008)
Ni	19.6(0.41)	16.6(0.74)	15.0(0.10)	14.6(0.04)	14.4(0.03)	14.2(0.04)	13.9(0.05)	13.5(0.06)	13.1(0.08)
Zn	140.9(1.55)	127.4(3.6)	117.9(0.88)	114.6(0.37)	113.0(0.24)	110.9(0.36)	107.9(0.58)	103.8(0.69)	99.4(0.76)

Note: Concentrations are given in micrograms per gram (standard error).

Table C18

Steady-State Leachate Metal Concentrations for Aerobic Everett Harbor Sediment

Metal	Sequential Leach Number								
	1	2	3	4	5	6	7	8	9
As	<0.005	<0.005	<0.005	<0.005	0.002(0.002)	<0.005	<0.005	<0.005	0.003(0.002)
Cd	0.007(0.002)	0.020(0.019)	0.002(0.001)	0.001(0.0001)	0.001(0.000)	0.0007(0.0002)	0.001(0.0001)	0.002(0.0002)	0.002(0.0001)
Cr	0.002(0.0006)	0.002(0.0003)	0.001(0.0005)	0.011(0.003)	0.013(0.001)	0.010(0.0005)	0.009(0.001)	0.008(0.0008)	0.008(0.0005)
Cu	0.005(0.0008)	0.004(0.0006)	0.003(0.0003)	0.019(0.007)	0.023(0.002)	0.015(0.0009)	0.011(0.002)	0.007(0.0005)	0.006(0.0006)
Pb	0.010(0.002)	0.017(0.013)	0.002(0.0006)	0.004(0.001)	0.005(0.0004)	0.004(0.0003)	0.003(0.0005)	0.002(0.0003)	0.003(0.0004)
Ni	0.449(0.102)	0.249(0.102)	0.036(0.002)	0.013(0.0006)	0.012(0.001)	0.015(0.0003)	0.020(0.002)	0.026(0.001)	0.023(0.002)
Zn	1.88(0.387)	1.20(0.424)	0.312(0.051)	0.138(0.014)	0.091(0.026)	0.141(0.003)	0.220(0.01)	0.269(0.008)	0.289(0.015)

Note: Concentrations are given in milligrams per litre (standard error).

C50

Table C19

Steady-State Sediment Metal Concentration in Aerobic Everett Harbor Sediment

Challenged with Leachate from Aerobic Everett Harbor Sediment

Metal	Sequential Leach Number							
	1	2	3	4	5	6	7	8
As	5.67(0.005)	5.64(0.02)	5.65(0.01)	5.65(0.01)	5.65(0.01)	5.64(0.01)	5.64(0.01)	5.64(0.01)
Cd	2.68(0.04)	2.18(0.05)	1.99(0.05)	1.89(0.05)	1.84(0.05)	1.78(0.05)	1.75(0.05)	1.68(0.05)
Cr	39.7(0.001)	39.6(0.005)	39.6(0.01)	39.6(0.01)	39.6(0.01)	39.6(0.01)	39.6(0.02)	39.5(0.02)
Cu	73.4(0.004)	73.2(0.02)	73.0(0.06)	72.8(0.08)	72.6(0.11)	72.4(0.14)	72.1(0.18)	71.8(0.22)
Pb	47.4(0.09)	46.9(0.14)	46.8(0.20)	46.7(0.15)	46.7(0.15)	46.7(0.15)	46.6(0.16)	46.6(0.16)
Ni	15.9(0.12)	12.6(0.24)	11.3(0.28)	10.7(0.31)	10.4(0.32)	10.1(0.33)	9.7(0.37)	9.3(0.40)
Zn	93.3(1.78)	58.1(3.08)	43.6(3.93)	37.5(4.53)	33.5(5.06)	30.2(5.59)	26.2(6.23)	21.4(6.96)

Note: Concentrations are given in micrograms per litre (standard error).

C51

Table C20

Steady-State Leachate Metal Concentration in Aerobic Everett Harbor Sediment

Challenged with Leachate from Aerobic Everett Harbor Sediment

Metal	Sequential Leach Number							
	1	2	3	4	5	6	7	8
As	0.009(0.001)	0.004(0.003)	<0.005	<0.005	<0.005	0.002(0.002)	<0.005	<0.005
Cd	0.155(0.01)	0.126(0.003)	0.047(0.008)	0.024(0.002)	0.014(0.001)	0.017(0.004)	0.006(0.002)	0.018(0.003)
Cr	0.007(0.00003)	0.006(0.001)	0.006(0.002)	0.004(0.0005)	0.004(0.001)	0.004(0.001)	0.005(0.001)	0.005(0.001)
Cu	0.006(0.001)	0.016(0.004)	0.075(0.01)	0.06(0.006)	0.05(0.007)	0.05(0.015)	0.07(0.009)	0.09(0.01)
Pb	0.187(0.022)	0.107(0.015)	0.039(0.004)	0.014(0.002)	0.009(0.001)	0.009(0.002)	0.008(0.001)	0.009(0.003)
Ni	1.37(0.03)	0.837(0.029)	0.330(0.01)	0.143(0.007)	0.082(0.005)	0.073(0.008)	0.097(0.01)	0.11(0.009)
Zn	13.8(0.45)	8.81(0.36)	3.60(0.25)	1.53(0.16)	1.00(0.14)	0.904(0.14)	1.00(0.16)	1.20(0.19)

Note: Concentrations are given in milligrams per litre (standard error).

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Table C21

Distribution Coefficients for Sequential and
Challenge Batch Leaching of Metals from Aerobic
Everett Harbor Sediment

Metal	Sequential Testing	Challenge testing
As	NLR	NLR
Cd	NLR	5.38(0.62)
Cr	NLR	NLR
Cu	NLR	-14.3(1.6)
Pb	NLR	3.73(0.21)
Ni	1.6(0.16)	4.4(0.11)
Zn	3.03(0.15)	4.7(0.28)

Note: NLR = no linear relationship.

Table C22

Summary of Metal Losses from Sediment Following Sequential
and Challenge Leaching of Anaerobic and
Aerobic Everett Harbor sediment

Metal	Anaerobic Leaching				Aerobic Leaching			
	Sequential		Challenge		Sequential		Challenge	
	µg/g	%	µg/g	%	µg/g	%	µg/g	%
As	0.042	7.3	0.58	10.2	0.02	0.4	0.06	1.1
Cd	0.11	3.3	0.22	6.7	0.15	4.5	1.62	49.1
Cr	0.40	1.0	1.3	3.3	0.26	0.7	0.16	0.4
Cu	1.67	2.3	3.8	5.2	0.36	0.5	1.62	2.2
Pb	1.12	2.3	1.8	3.7	0.20	0.4	1.52	3.2
Ni	0.68	3.2	1.8	8.4	3.39	15.8	12.13	56.7
Zn	2.85	1.9	5.4	3.6	18.2	12.3	127.1	85.6

Note: Values given are in micrograms per gram dry weight and percentage of
total sediment concentration.

Table C23

Steady-State Organic Contaminant Leachate Concentrations for

Anaerobic Everett Harbor Sediment Following Anaerobic Leaching

Compound	Sequential Leach Number						
	1	2	3	4	5	6	7
5	1.3(0.2)	1.0(0.5)	2.1(0.4)	1.9(0.2)	1.0(0.5)	1.4(0.0)	0.8(0.4)
7	2.9(0.5)	2.0(1.2)	0.9(0.9)	3.6(0.5)	0.7(0.7)	0.6(0.6)	0.5(0.5)
9	2.0(0.2)	1.7(0.4)	4.3(0.9)	2.0(1.0)	0.7(0.7)	1.5(0.1)	0.3(0.3)
10	2.0(0.2)	1.4(0.7)	3.1(2.0)	3.0(0.5)	1.2(0.6)	0.9(0.5)	0.5(0.3)
28	0.01(0.01)	0.01(0.01)	0.01(0.01)	0.01(0.01)	0.01(0.01)	0.01(0.01)	0.03(0.00)
29	0.06(0.03)	0.06(0.03)	0.09(0.05)	0.03(0.03)	0.03(0.03)	0.07(0.03)	0.03(0.03)
30	0.04(0.04)	0.04(0.01)	0.08(0.03)	0.03(0.03)	0.04(0.03)	0.04(0.01)	0.12(0.01)
32	0.03(0.03)	0.08(0.01)	0.06(0.03)	0.09(0.01)	0.07(0.01)	0.06(0.03)	0.07(0.04)

Notes: Compounds for which no data are presented were not detected in the leachate. Concentrations are given in micrograms per litre (standard error).

Table C24

Steady-State Sediment Organic Contaminant Concentrations for

Everett Harbor Sediment Following Anaerobic Leaching

Compound	Sequential Leach Number						
	1	2	3	4	5	6	7
5	1.995(0.001)	1.985(0.002)	1.963(0.004)	1.941(0.005)	1.923(0.002)	1.910(0.003)	1.899(0.003)
7	5.689(0.002)	5.674(0.002)	5.665(0.004)	5.628(0.009)	5.612(0.003)	5.604(0.002)	5.597(0.002)
9	4.492(0.001)	4.479(0.003)	4.437(0.009)	4.401(0.004)	4.388(0.000)	4.377(0.004)	4.368(0.001)
10	3.992(0.001)	3.982(0.002)	3.943(0.003)	3.918(0.008)	3.982(0.002)	3.881(0.002)	3.875(0.002)
28	0.008(0.000)	0.008(0.000)	0.008(0.000)	0.008(0.000)	0.007(0.000)	0.007(0.000)	0.007(0.000)
29	0.0026(0.000)	0.0025(0.000)	0.0024(0.000)	0.0024(0.000)	0.0023(0.000)	0.0022(0.000)	0.0022(0.000)
30	0.041(0.000)	0.040(0.000)	0.040(0.000)	0.039(0.000)	0.038(0.000)	0.038(0.000)	0.037(0.000)
32	0.018(0.000)	0.017(0.000)	0.016(0.000)	0.015(0.000)	0.014(0.000)	0.014(0.000)	0.013(0.000)

Note: Data are not presented for compounds that were not detected in the leachate.
Concentrations are given in micrograms per gram dry weight (standard error).

Table C25

Steady-State Organic Contaminant Leachate Concentrations for Anaerobic Everett Harbor
Sediment Challenged with Leachate from Anaerobic Everett Harbor Sediment

| Compound | Sequential Leach Number | | | | | |
	1	2	3	4	5	6
5	0.23(0.23)	0.83(0.28)	2.05(0.65)	1.6(1.6)	0.95(0.35)	1.9(1.9)
7	ND	ND	0.65(0.65)	2.3(2.3)	1.1(1.0)	2.5(2.5)
9	ND	ND	0.75(0.75)	2.9(0.75)	1.1(1.1)	2.6(2.6)
10	ND	0.57(0.57)	ND	2.8(0.8)	1.9(0.00)	2.2(2.2)
28	0.013(0.013)	0.03(0.007)	0.03(0.01)	0.02(0.02)	0.02(0.02)	0.02(0.02)
29	0.013(0.013)	0.09(0.05)	0.11(0.01)	0.14(0.04)	0.05(0.05)	0.05(0.05)
30	0.02(0.02)	0.09(0.05)	0.12(0.01)	0.05(0.05)	0.06(0.06)	0.05(0.05)
32	0.07(0.03)	0.06(0.04)	0.08(0.04)	0.07(0.005)	0.08(0.02)	0.1(0.02)

Notes. Concentrations are given in micrograms per litre (standard error).
ND = not detected.

Table C26

Steady-State Sediment Organic Contaminant Concentrations for Anaerobic Everett Harbor Sediment Challenged with Leachate from Anaerobic Everett Harbor Sediment

Compound	Sequential Leach Number					
	1	2	3	4	5	6
5	2.00(0.001)	1.99(0.001)	1.97(0.003)	1.96(0.000)	1.95(0.0012)	1.94(0.000)
7	5.70(0.000)	5.70(0.000)	5.69(0.003)	5.68(0.000)	5.67(0.0003)	5.65(0.000)
9	4.50(0.000)	4.50(0.000)	4.49(0.003)	4.47(0.004)	4.46(0.000)	4.44(0.000)
10	4.00(0.000)	3.99(0.000)	3.99(0.000)	3.97(0.004)	3.96(0.004)	3.94(0.000)
28	0.0078(0.00005)	0.0075(0.00009)	0.0072(0.00008)	0.0070(0.00006)	0.007(0.00006)	0.007(0.00006)
29	0.0026(0.00005)	0.0025(0.0003)	0.0024(0.0002)	0.0023(0.0002)	0.0023(0.0002)	0.0022(0.0002)
30	0.041(0.00008)	0.040(0.0003)	0.039(0.0002)	0.038(0.0002)	0.038(0.0002)	0.038(0.0002)
32	0.018(0.0001)	0.017(0.0002)	0.016(0.0002)	0.016(0.0001)	0.015(0.0002)	0.014(0.0002)

Note: Concentrations are given in milligrams per gram (standard error).

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of any organic contaminant and 0.005 µg/g the highest concentration of any PCB congener released during the sequential leaching process (Table C27).

57. Organic contaminant concentrations present in steady-state leachate and sediment of aerobic Everett Harbor sediment are given in Tables C28 and C29, respectively. Only seven compounds were detected in the leachate, although they differed somewhat from those detected during anaerobic testing. Analysis of first-day leachate from sequential challenge batch testing for organic contaminants showed that only five of seven compounds found in the aerobic batch test were detected. Concentrations of these compounds were similar to those measured in the batch testing. For reasons given in the following paragraphs, it was not necessary to analyze further aerobic challenge samples to obtain a valid single-point organic challenge distribution coefficient.

58. Statistical analysis of the organic contaminant data revealed that no significant ($p < 0.05$) linear relationship existed between steady-state sediment and leachate organic contaminant concentrations from either the anaerobic sequential or challenge batch leaching and the aerobic sequential batch leaching. This type of behavior is expected if the distribution coefficient is very large and the resulting changes in steady-state contaminant concentration are small. It is reasonable to assume that, unlike metals, all of the organic contaminants associated with a sediment are potentially leachable. The lack of complete reversibility observed in numerous experiments is probably due to kinetics, i.e., the presence of a slowly desorbing sediment contaminant component (Di Toro 1985). This is not the case for metals because of the known association of metals with immobile sediment phases (Brannon et al. 1976; Brannon, Plumb, and Smith 1980). Using this assumption, single-point organic contaminant distribution coefficients were calculated for the sequential and challenge batch testing using the average steady-state leachate and sediment concentrations for each of the three replicate tests conducted. These data are presented in Table C30. Distribution coefficients for both the anaerobic sequential and challenge testing were high; K_d values for PAH compounds did not fall below 1,000 ℓ/mg. Distribution coefficients for PCB congeners were somewhat lower than those measured for PAH compounds. Distribution coefficients for aerobic testing were generally comparable to those noted under anaerobic conditions when the same compounds were released under both conditions.

Table C27

Summary of Organic Contaminant Losses from Everett Harbor

Sediment and Percent of Total Sediment Concentration

Lost During Sequential and Challenge Testing

Organic Parameter	Sequential		Challenge	
	μg/g	Percent	μg/g	Percent
5	0.096	4.8	0.06	3.0
7	0.092	1.8	0.05	0.9
9	0.124	2.9	0.06	1.3
10	0.117	3.1	0.06	1.5
28	0.008	10.3	0.0011	0.1
29	0.0004	15.4	0.0004	15.4
30	0.004	9.8	0.005	11.9
32	0.005	27.9	0.004	22.2

Note: Concentrations are given in micrograms per gram dry weight.

Table C28

Steady-State Organic Contaminant Leachate Concentrations

for Aerobic Everett Harbor Sediment

Compound	Sequential Leach Number		
	1	2	3
20	0.003(0.003)	0.003(0.003)	0.007(0.007)
21	0.007(0.003)	ND	0.013(0.007)
25	0.007(0.003)	ND	0.013(0.007)
26	0.007(0.007)	0.023(0.023)	0.057(0.029)
28	0.030(0.015)	0.013(0.013)	0.037(0.018)
32	0.020(0.020)	0.033(0.033)	0.014(0.02)
33	0.003(0.003)	0.063(0.018)	ND

Notes: Concentrations are given in micrograms per litre (standard error).
ND = not detected.

Table C29

Steady-State Sediment Contaminant Concentrations for

Everett Harbor Sediment Following Aerobic Leaching

Compound	Sequential Leach Number		
	1	2	3
20	0.0093(0.00001)	0.0093(0.00003)	0.0092(0.00005)
21	0.0061(0.00001)	0.0061(0.00001)	0.0060(0.00004)
25	0.0061(0.00001)	0.0061(0.00001)	0.0060(0.00004)
26	0.0079(0.00003)	0.0078(0.00008)	0.0076(0.0002)
28	0.0119(0.00006)	0.0118(0.0001)	0.0117(0.0001)
32	0.0209(0.00008)	0.0208(0.0001)	0.0202(0.0002)
33	0.0420(0.00001)	0.0417(0.0006)	0.0417(0.00006)

Note: Concentrations are given in micrograms per gram dry weight (standard error).

Table C30

Single-Point Distribution Coefficients for Organic

Contaminants in Everett Harbor Leachate

Parameter	Anaerobic Testing		Aerobic Testing	
	Sequential	Challenge	Sequential	Challenge
5	1473(141)	3574(2879)	NMR	NMR
7	3774(629)	5981(7969)	NMR	NMR
9	3045(2453)	5460(2453)	NMR	NMR
10	2579(653)	4359(1876)	NMR	NMR
20	614(413)	NMR	3220(467)	NMR
21	NMR	NMR	682(229)	454(153)
25	NMR	NMR	682(229)	454(153)
26	NMR	NMR	549(394)	109(0)
28	1835(3)	561(304)	525(182)	167(0)
29	553(133)	378(64)	NMR	NMR
30	929(261)	935(458)	NMR	NMR
32	266(12)	227(23)	605(260)	NMR
33	NMR	NMR	2335(533)	NMR
34	483(116)	480(138)	1173(440)	2855(2369)

Note: Values are expressed in litres per kilogram (standard error).
NMR = no measurable release.

Permeameter Testing

59. Continuous flow column leaching tests were conducted using divided flow permeameters, as previously described, with both anaerobic and aerobic Everett Harbor sediment. Approximately 3 pore volumes passed through the anaerobic columns and 3.5 pore volumes through the aerobic columns before testing ended.

Metals and DOC

60. Effluent metal concentrations and corresponding pore volumes are summarized in Tables C31 and C32 for anaerobic and aerobic columns, respectively. In general, samples from the anaerobic columns had relatively low concentrations, usually within a factor of 10 of the detection limit. Dissolved organic carbon (DOC) increased from around 50 mg/ℓ to 225 mg/ℓ. This is consistent with results obtained during batch testing which showed DOC concentrations peaking at the fourth step (181 mg/ℓ). Leachate pH increased from 7.3 to 8.4 during column operation, again consistent with the increase observed in the anaerobic sequential batch tests.

61. Metal concentrations measured in the effluent from aerobic columns were generally higher by an order of magnitude than corresponding samples from the anaerobic columns. Chromium and zinc leachate concentrations were more variable than other metals between columns. Average DOC concentrations ranged from 64 to 85 mg/ℓ, showing no washout or significant increase. Batch DOC concentrations were generally constant around 40 mg/ℓ, also showing no washout or significant increase. Initially the pH of the aerobic column leachate was low, around 3.5. However, pH increased to 7.0 by the conclusion of column operation. This is contrary to results obtained in the sequential batch leach tests (Table C11). The difference between batch and column leachate pH is probably due to differences in oxidation-reduction potential. In the column tests the sediment is in a flooded condition. Due to sediment oxygen demand, the system rapidly becomes anaerobic, resulting in a decrease in redox potential and a rise in pH. In the aerobic batch tests, oxygen is continually replenished by turbulence, redox potential remains high, and the pH remains low. Consequently, the leaching conditions are not comparable, and contaminant mobility will not be the same.

Table C31

Metal and Dissolved Organic Carbon Concentration in Permeameter

Effluent from Anaerobic Everett Harbor Sediment

Pore Volume	Parameter					
	As	Cd	Cr	Pb	Zn	DOC
0.085	<0.005 -	0.0022 (0.0001)	0.009 (0.005)	0.009 (0.005)	<0.03 -	48 (0.1)
0.22	<0.005 -	0.0016 (0.0001)	0.009 (0.004)	0.010 (0.003)	<0.03 -	49 (1.0)
0.38	<0.005 -	0.0007 (0.0003)	0.009 (0.001)	0.005 (0.009)	<0.03 -	44 (3.2)
0.56	<0.005 -	0.0008 (0.0001)	0.008 (0.003)	0.001 (0.001)	<0.03 -	37 (1.5)
0.78	<0.005 -	0.0034 (0.0008)	0.012 (0.002)	0.015 (0.001)	<0.03 -	42 (0.1)
1.00	<0.005 -	0.0036 (0.0001)	0.033 (0.002)	0.015 (0.001)	<0.03 -	46 (0.1)
1.22	<0.005 -	0.0026 (0.0001)	0.016 (0.002)	0.043 (0.011)	<0.03 -	59 (2.3)
1.43	<0.005 -	<0.0001 -	0.017 (0.003)	0.004 (0.001)	<0.03 -	88 (2.6)
2.29	<0.005 -	<0.0001 -	0.079 (0.003)	0.003 (0.001)	0.03 (0.01)	361 (16)
3.00	0.006 (0.001)	0.0002 (0.0001)	0.074 (0.003)	0.005 (0.007)	0.052 (0.013)	259 (43)
3.45	0.005 (0.001)	0.0008 (0.0006)	0.067 (0.005)	0.005 (0.002)	0.029 (0.015)	224 (17)
3.51	0.005 (0.001)	0.0001 (0.0001)	0.063 (0.001)	0.004 (0.001)	0.051 (0.008)	256 (11)

Note: Concentrations are given in milligrams per litre (standard error).

Table C32

Metal and Dissolved Organic Carbon Concentrations in
Permeameter Effluent from Aerobic Everett Harbor Sediment

Pore Volume	Parameter					
	As	Cd	Cr	Pb	Zn	DOC
0.14	<0.005	0.0343	0.068	0.210	3.65	64
	-	(0.0110	(0.045)	(0.063)	(0.20)	(2)
0.51	<0.005	0.0018	2.25	0.050	2.13	66
	-	(0.0012)	(2.20)	(0.002)	(0.38)	(1)
1.56	<0.005	0.0017	0.472	0.090	0.217	68
	-	(0.0016)	(0.469)	(0.089)	(0.201)	(7)
2.07	<0.005	0.0002	0.136	0.002	0.060	72
	-	(0.0001)	(0.126)	(0.001)	(0.042)	(3)
2.76	<0.005	0.0042	0.058	0.004	0.030	89
	-	(0.0038)	(0.042)	(0.007)	(0.016)	(13)
3.42	<0.005	0.0002	0.018	0.012	0.097	85
	-	(0.0001)	(0.009)	(0.009)	(0.049)	(9)

Note: Concentrations are given in milligrams per litre (standard error).

Organics and DOC

62.   No PAH compounds were detected in the effluent from either aerobically or anaerobically operated columns. Concentrations of each PCB congener and DOC are provided in Tables C33 and C34 for aerobic and anaerobic columns, respectively. Variation in pH, conductivity, and DOC during batch and column studies is summarized in Table 35. Total Arochlor 1254 congener concentration varied from 0.00001 to 0.00036 mg/ℓ in leachate from the anaerobic columns. Five samples from aerobic columns have been analyzed; total congener concentrations range from 0.00001 to 0.00176 mg/ℓ. The DOC values from the anaerobic columns increased from around 50 mg/ℓ to 250 mg/ℓ,

Table C33

Everett Harbor Aerobic Permeameter Organics and DOC

Sample Number	Cumu-lative Pore Volume	Average Pore Volume	DOC	Ave. DOC	Compound Number Concentration, mg/l															Total PCB Conc.	Ave. Total PCB Conc.
					19	20	21	22	23	24	25	26	27	28	29	30	31	32	33		
1	0.066		63		--NO DATA--															----	----
	0.065	0.06	62	62																	
	0.054		61																		
2	0.405		117		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00000	
	0.418	0.40	115	117	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00034	0.00001	0.00001	0.00001	0.00034	0.00011
	0.380		118		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00000	
3	0.940		65		0.00001	0.00014	0.00230	0.00001	0.00001	0.00005	0.00230	0.00001	0.00001	0.00002	0.00002	0.00002	0.00001	0.00009	0.00001	0.00499	
	0.975	0.95	54	56	0.00001	0.00009	0.00001	0.00001	0.00001	0.00005	0.00001	0.00002	0.00001	0.00002	0.00003	0.00001	0.00001	0.00008	0.00001	0.00030	0.00176
	0.936		48		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00000	
4	1.47		135		0.00001	0.00001	0.00001	0.00001	0.00001	0.00004	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00009	0.00001	0.00013	
	1.53	1.50	118	124	0.00001	0.00001	0.00001	0.00007	0.00001	0.00007	0.00001	0.00001	0.00001	0.00007	0.00001	0.00001	0.00001	0.00000	0.00001	0.00031	0.00019
	1.49		119		0.00001	0.00001	0.00001	0.00001	0.00001	0.00005	0.00001	0.00001	0.00001	0.00001	0.00001	0.00008	0.00001	0.00001	0.00001	0.00013	
5	2.03		202		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	
	2.09	2.05	213	215	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001
	2.03		230		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	

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Table C34

Everett Harbor Anaerobic Organics and DOC

Sample Number	Cumulative Pore Volume	Average Pore Volume	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	Total PCB Conc.	Ave. Total PCB Conc.	DOC	Avg. DOC
											Compound Number Concentration, mg/l										
1	0.330		0.00001	0.00005	0.00001	0.00002	0.00001	0.00001	0.00001	0.00002	0.00001	0.00002	0.00004	0.00005	0.00001	0.00005	0.00001	0.00028		56	
	0.312	0.33	0.00001	0.00005	0.00001	0.00002	0.00004	0.00001	0.00001	0.00001	0.00001	0.00001	0.00005	0.00006	0.00001	0.00005	0.00001	0.00031		54	
	0.340		0.00001	0.00005	0.00001	0.00002	0.00001	0.00001	0.00001	0.00003	<0.00001	0.00004	0.00012	0.00012	0.00001	0.00006	0.00001	0.00048	0.00036	52	54
2	1.01		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00002	0.00001	0.00001	0.00001	0.00005	0.00001	0.00009		56	
	0.94	0.99	0.00004	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00002	0.00001	0.00002	0.00004	0.00011		64	
	1.01		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00017	0.00001	0.00001	0.00001	0.00017	0.00012	70	63
3	1.63		0.00001	0.00001	0.00001	0.00019	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00019		306	
	1.53	1.61	0.00001	0.00001	0.00001	0.00004	0.00001	0.00001	0.00001	0.00005	0.00001	0.00006	0.00001	0.00001	0.00001	0.00001	0.00001	0.00017		305	
	1.67		0.00001	0.00001	0.00008	0.00009	0.00001	0.00001	0.00008	0.00001	0.00001	0.00009	0.00001	0.00001	0.00001	0.00017	0.00001	0.00051	0.00029	289	300
4	2.25		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00002	0.00001	0.00001	0.00001	0.00002		255	
	2.14	2.23	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001		187	
	2.29		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00000	0.00001	214	219
5	2.89		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00007	0.00001	0.00001	0.00007		241	
	2.81	2.88	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00005	0.00001	0.00001	0.00005		289	
	2.95		0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00000	0.00006	258	263

C65

Table C35

Summary of pH, Conductivity, and DOC Trends During Batch

and Column Leach Testing

Test*	pH	Conductivity	DOC
Anaerobic batch	Increased (7.3 to >8.7)	Decreased	Peaked (84 to >181 to >56)
Anaerobic column M+	Increased (7.3 to >8.5)	Decreased	Increased (47 to >250)
Anaerobic column Or	ND	ND	Increased (50 to >250)
Aerobic batch	Static (3.8)	Decreased	Static (40)
Aerobic column M+	Increased (3.5 to >7.5)	Decreased	Increased (64 to >85)
Aerobic column Or	ND	ND	Increased (62 to 215)

* M+ = metals leaching column; Or = organics leaching column; ND = no data.

behavior similar to that observed for anaerobic metals. Aerobic DOC concentrations increased from 60 mg/ℓ to around 200 mg/ℓ.

63. As described earlier and shown in Table C30, an average, single-point distribution coefficient was computed for each congener measured and for total Aroclor 1254 congeners using anaerobic batch leaching data. Using Equation C2 and the appropriate value of K_d in Table C30 an approximate equilibrium concentration for each congener detected and for total Aroclor 1254 congeners was computed. These values are provided in Table C36, along with the average measured concentration for each sample. Measured and computed equilibrium concentrations were generally similar.

Table C36

Predicted and Observed Values of PCB Compounds

from Anaerobic Everett Harbor Sediment

Compound*	Pore Volume	Average Concentration mg/ℓ	Computed Equilibrium Concentration mg/ℓ
28	0.33	0.00002	
	0.99	0.00001	
	1.61	0.00005	
	2.23	<0.00001	
	Average	0.00002	<0.00001
29	0.33	0.00007	
	0.99	<0.00001	
	1.61	<0.00001	
	2.23	<0.00001	
	Average	0.00002	0.00001
30	0.33	0.00008	
	0.99	0.00006	
	1.61	<0.00001	
	2.23	<0.00001	
	Average	0.00004	0.000005
32	0.33	0.00005	
	0.99	0.00002	
	1.61	0.00005	
	2.23	<0.00001	
	Average	0.00003	0.00004
34	0.33	0.00036	
	0.99	0.00012	
	1.61	0.00029	
	2.23	0.00001	
	Average	0.0002	0.00002

* See Table C3 for organic compound identification key.

Integrated Approach

Anaerobic metals

64. The contaminant transport equation, Equation C7 previously presented in this appendix, assumes that sequential batch leach data will provide ideal desorption isotherms (Figure C5) for contaminants of interest. For an ideal desorption isotherm, K_d is a constant greater than zero. As

previously discussed, the desorption isotherms for Everett Harbor anaerobic metals were generally nonideal. The plots for Zn, Cd, and Cr did not exhibit statistically valid linear relationships between q and C, thus K_d could not be determined as the isotherm slope for these metals. Isotherm plots for Cu and Pb exhibited an inverse relationship between q and C, that is, C increased as q decreased, as illustrated in Figures C11 and C12. Desorption isotherms for As and Ni initially exhibited an inverse relationship but changed orientation to a ideal relationship (C decreased as q decreased) at the third and fourth steps, respectively, of the sequential leaching procedure, as shown in Figures C7 and C8. Because the contaminant transport equation requires constant values of K_d, it is not possible to predict permeameter leachate concentrations using this equation. The effort required to develop a numerical solution to Equation C2 for variable distribution coefficients was not within the scope of this study.

65. A simplified alternative method that roughly approximates Equation C2 was therefore developed. Houle and Long (1980) recognized that a continuously leached column is equivalent to running a series of discrete batch leach tests. If the physical-chemical processes in a series of batch leach tests are the same as those occurring in a continuous flow column, it should be possible to predict the general shape of a column elution curve using desorption isotherm analysis. Further, each step in the sequential leach test can be related to a pore volume of water through a continuous flow, allowing a direct comparison of batch leachate concentration and column leachate concentration to be made.

66. If dispersion is neglected, column leachate concentrations can be predicted by relating the leachate concentrations in each step of the sequential batch test to an equivalent pore volume through the columns. This is done on the basis of equivalent liquid-solids ratios. A liquid-solids ratio for an operating column is defined as the weight of the accumulated volume passed through the column divided by the weight of the sediment in the column. For Everett Bay sediment the initial water content (W_w/W_s) in the columns was 1.81, while that in each step of the sequential leaching process is 4:1. Because the weight of water contacting the solids in the column increases with increasing throughput, the column liquid-solids ratio will reach 4:1 when 2.2 (4/1.8) pore volumes have passed through the column. Thus, each step in the batch leaching procedure is equivalent to the passage of 2.2 pore volumes

through the column. The leachate concentration obtained during each step in the batch procedure represents the average concentration over the corresponding pore volume increment. Thus, the concentration measured during the first step in the sequential batch leach test is an estimate of the column leachate concentration at 1.1 (0- to 2.2-pore volume interval) pore volumes. Cumulative pore volumes, equivalent liquid-solids ratios, and the corresponding batch test step number are listed in Table C37.

67. As noted above, the desorption isotherm data for Cu and Pb produced desorption isotherms with inverse slopes. An "inverse isotherm" predicts that column contaminant concentrations should continuously increase with time (pore volumes). The desorption isotherms for As and Ni were double-valued, changing slopes from inverse to direct (ideal). An isotherm that changes direction (inverse to direct) implies that column concentrations should increase to a peak, then decrease. Thus, the sequential batch leach data can be used to indicate the general shape of the column elution curves for Cu, Pb, As, and Ni. However, as with anything that is simple and direct, there are limitations. Since the direct comparison procedure does not include advection and dispersion, the procedure cannot predict shifting and spreading of peaks caused by advection and dispersion.

68. Using the direct comparison procedure described above, predicted column concentrations and corresponding pore volumes are plotted for As, Cd, Cr, Pb, and Zn in Figures C15-C19, respectively. In the same figures are plotted the observed column concentrations. The predicted concentrations of Ni and Cu are plotted in Figure C20. Several metals showed concentration peaks between 6 and 10 pore volumes. With the exception of a single observed Cr value, both predicted and observed values were relatively low for all metals.

69. Overlap of batch and column data for the direct comparison method began at 1.1 pore volumes. Operation of the columns was terminated at approximately 3.5 pore volumes. In the region where observed and predicted results can be compared (1.1 < pore volume < 3.0), agreement is reasonably good for As, Cd, and Pb. Substantial disagreement occurred for As and Cr. Because predicted and observed data agree reasonably well for As, Cd, and Pb, it seems reasonable that extrapolation of the direct comparison method to the field is valid, at least for indicating the overall pattern of contaminant release.

Table C37

Batch Sequence Number and Equivalent Pore
Volume Through Everett Harbor Permeameters*

Batch Sequence Number	Cumulative Batch L-S Ratio	Cumulative L-S for Batch Leachate Concentration	Equivalent Pore Volume Through Permeameters
1	0 to 4:1	2:1	1.1
2	4:1 to 8:1	6:1	3.3
3	8:1 to 12:1	10:1	5.6
4	12:1 to 16:1	14:1	7.8
5	16:1 to 20:1	18:1	10.0

* Batch conducted at liquid to solids ratio (L-S) of 4:1; L-S in permeameters = 1.8:1.

ELEMENT: ARSENIC
DETECTION LIMIT: 5 $\mu g/\ell$

LEGEND

SEQUENTIAL BATCH DATA
PERMEAMETER DATA
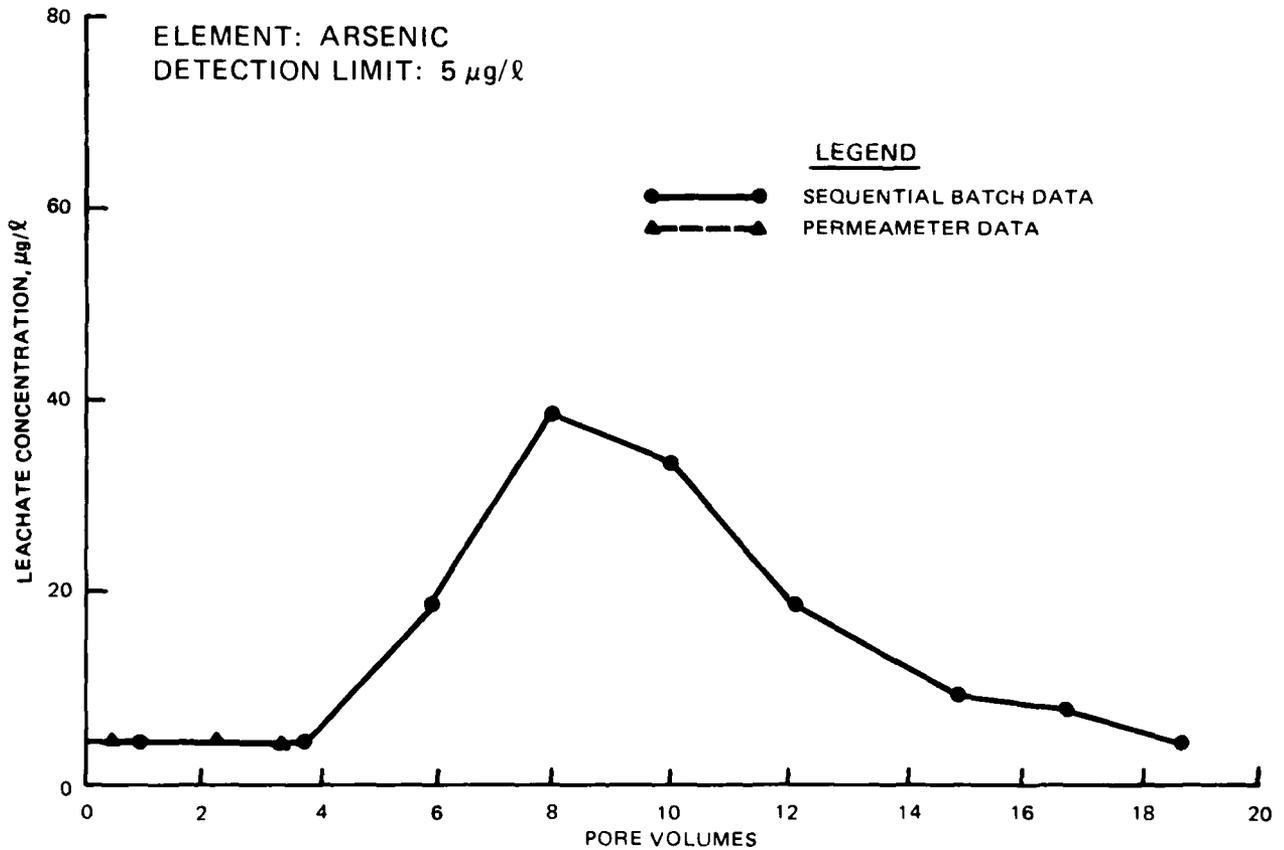
Figure C15. Comparison of observed and predicted arsenic concentrations in leachate from anaerobic permeameters

C70
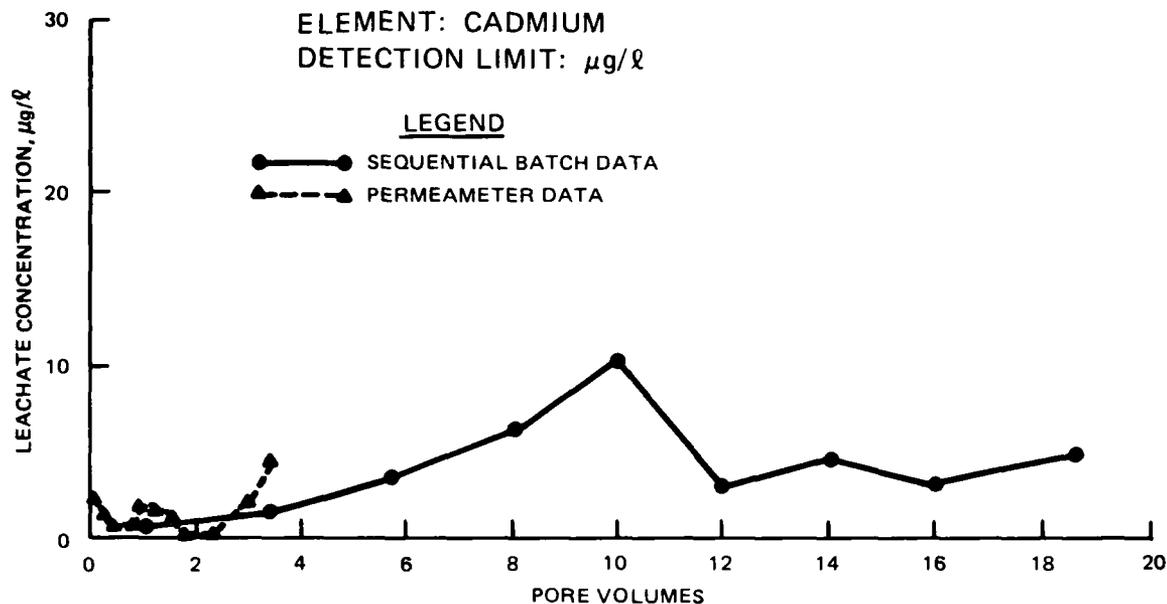
Figure C16. Comparison of observed and predicted cadmium concentrations in leachate from anaerobic permeameters

Anaerobic organics

70. Previous work (Myers, Brannon, and Griffin 1986) has demonstrated that when the desorption coefficient, K_d , is large, as is the case for PCB or PAH compounds, the source term in the one-dimensional contaminant transport equation is dominant. Predicted contaminant concentrations will therefore remain at or near initial equilibrium pore water levels (Figure C21). As a result, application of the integrated approach to PCB and PAH compounds in sediment involves comparing the equilibrium concentrations predicted using batch test data to those in the column effluent to verify the value of K_d used. Initial equilibrium concentrations are computed using the following equation:

$$C = \frac{q_o}{(K_d + L\text{-}S)} \qquad (C8)$$

where

q_o = initial bulk contaminant concentration

K_d = desorption coefficient determined from batch testing

L-S = liquid-solids ratio

C71
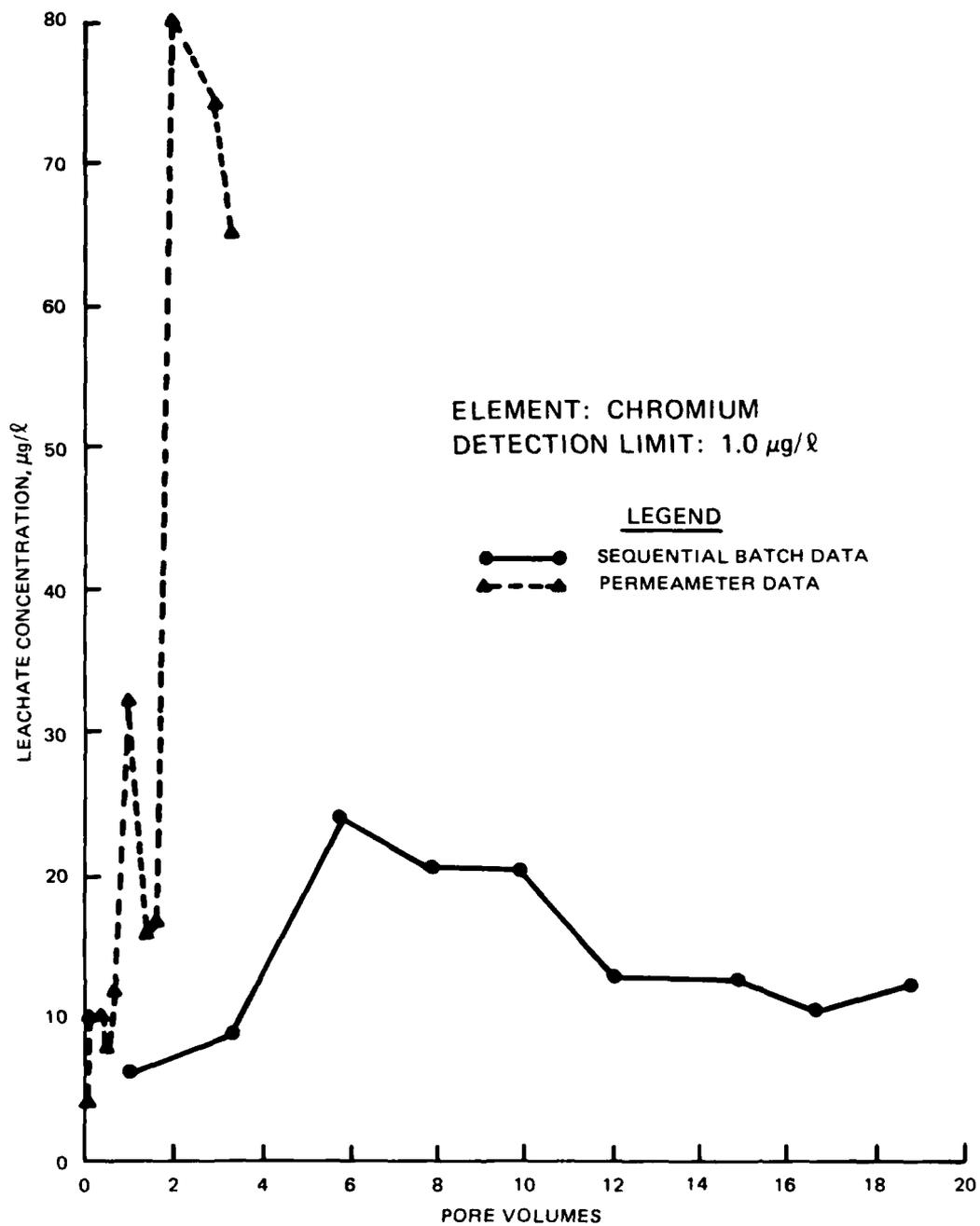
Figure C17. Comparison of observed and predicted chromium concentrations in leachate from anaerobic permeameters
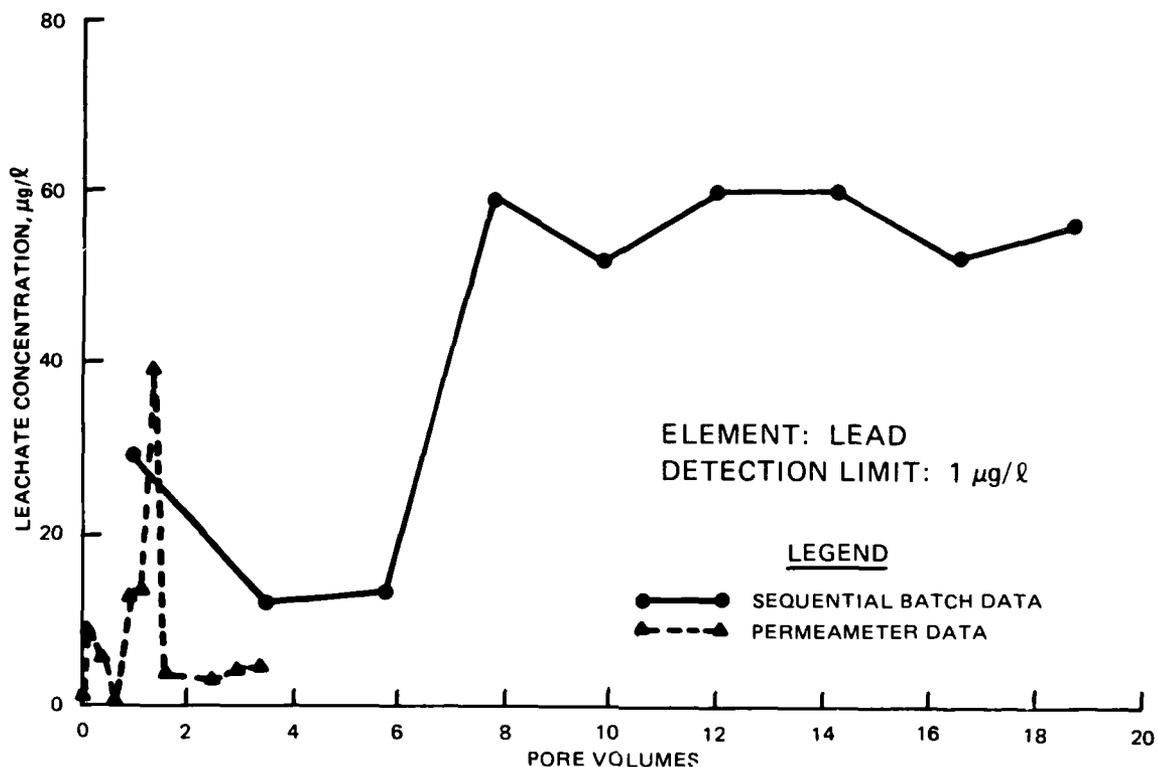
Figure C18. Comparison of observed and predicted lead concentrations in leachate from anaerobic permeameters

Since the L-S ratio in the column tests is 1.8 and the distribution coefficients are greater than 100 ℓ/kg, L-S can be neglected.

71. The data in Table C36 were used to compare predicted equilibrium congener concentrations with observed values for all PCB compounds for which a value of K_d is available (compound numbers 28, 29, 30, and 32) as well as total PCB congener concentration. The average congener and total congener concentration of each of the four column samples collected varies around their respective predicted equilibrium values. Given the complexity of the sequential procedure and column operation, such variation is not unexpected. Conservative estimates of contaminant flux are assured if the maximum observed average column concentration is used in each case.

72. To illustrate application of Equation C7, computed and predicted concentrations of total Arochlor 1254 congeners are compared in Figure C22. Predicted concentrations were computed using Equation C7. This figure clearly shows the effect of a large distribution coefficient (K_d = 483) on resulting
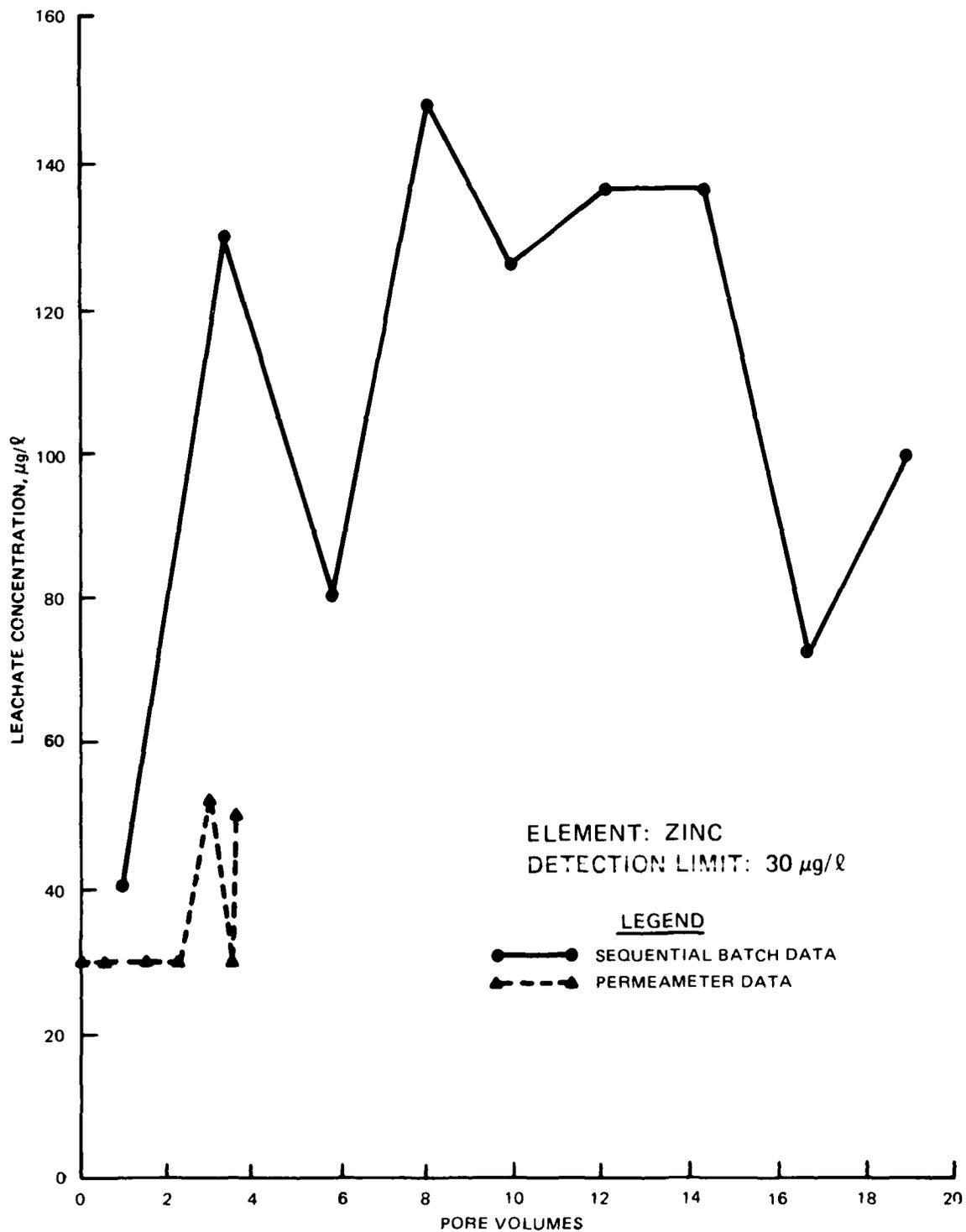
Figure C19. Comparison of observed and predicted zinc concentrations in leachate from anaerobic permeameters
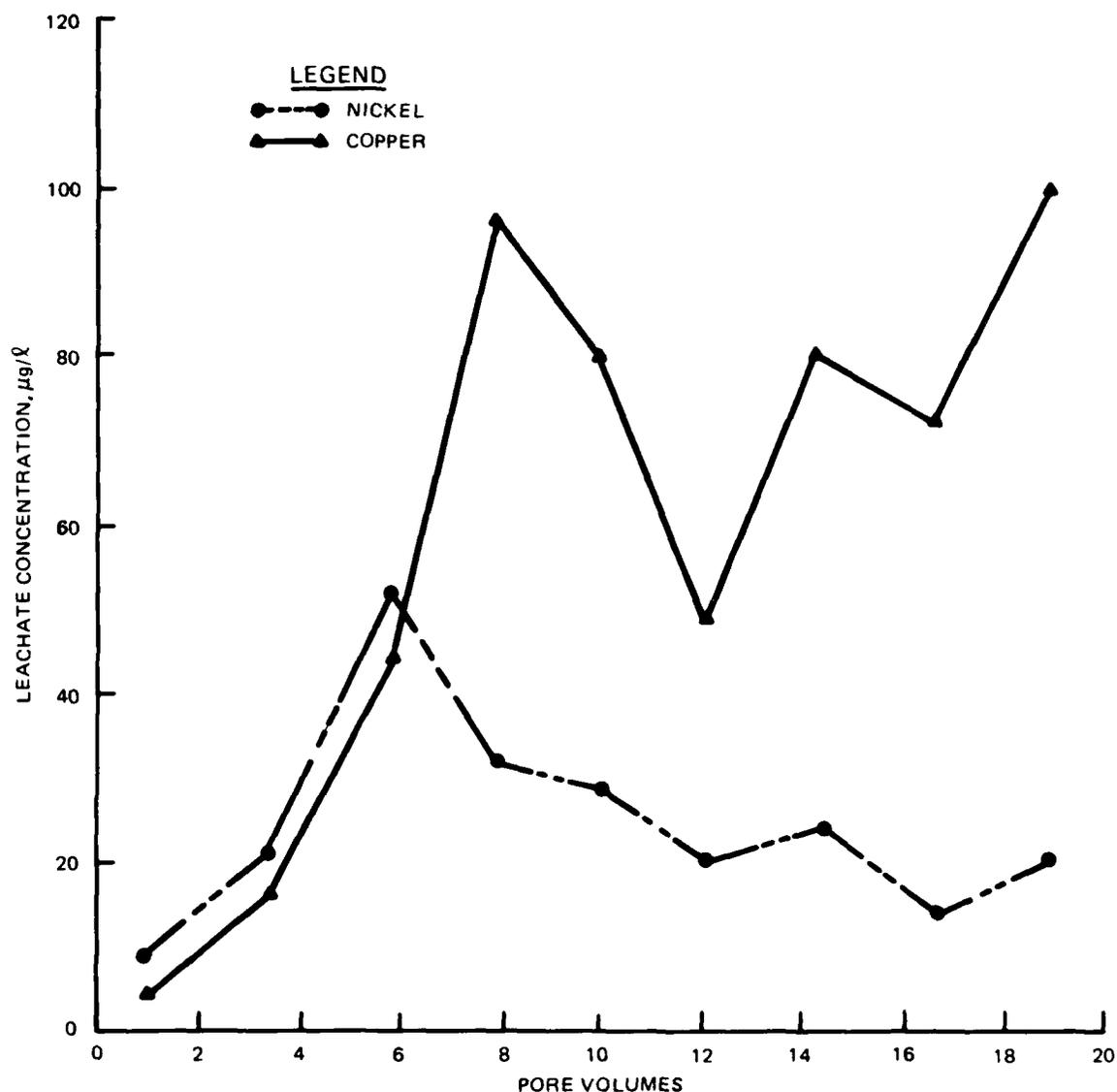
Figure C20. Predicted permeameter leachate
concentrations for copper and nickel

contaminant concentrations. Varying K_d between 367 and 599 (K_d plus or minus 1 standard error) had no effect on computed concentrations, which remained at the initial value of 0.0002 mg/ℓ. Since individual PCB congeners detected are characterized by distribution coefficients ranging from 266 to 1,835 ℓ/kg, similar behavior would be expected.

73. The batch data suggest that two PAH compounds, compounds 7 and 9, should have been detected in the column leachates. At present, the absence of
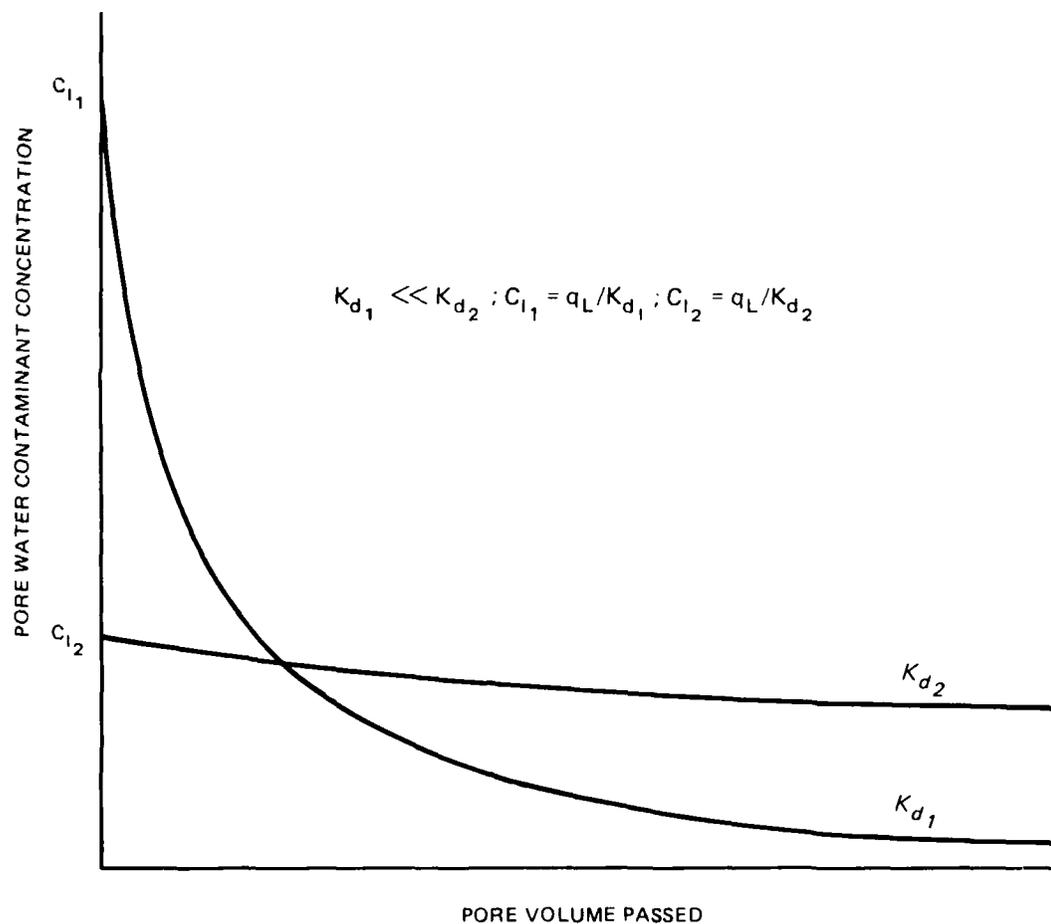
Figure C21. Schematic showing the effect of a large K_d
on pore water concentration

detectable concentrations of these two contaminants in column leachates cannot
be explained.

Aerobic metals and organics

74. Previous work (Environmental Laboratory 1987) has shown that the
use of batch desorption coefficients determined under aerobic conditions to
predict contaminant concentrations from columns initially filled with aerobic
sediment is inappropriate. Even sediment placed in an oxidizing environment
for 6 months retains enough oxygen demand to become anaerobic once it is
placed in a column and flooded. This change in the oxidation-reduction poten-
tial of the sediment affects its desorptive properties. The differences
between aerobic column and aerobic batch leachate data are illustrated in Fig-
ures C23-C26 for Cr, Cd, Zn, and Pb. Unlike anaerobic column results where
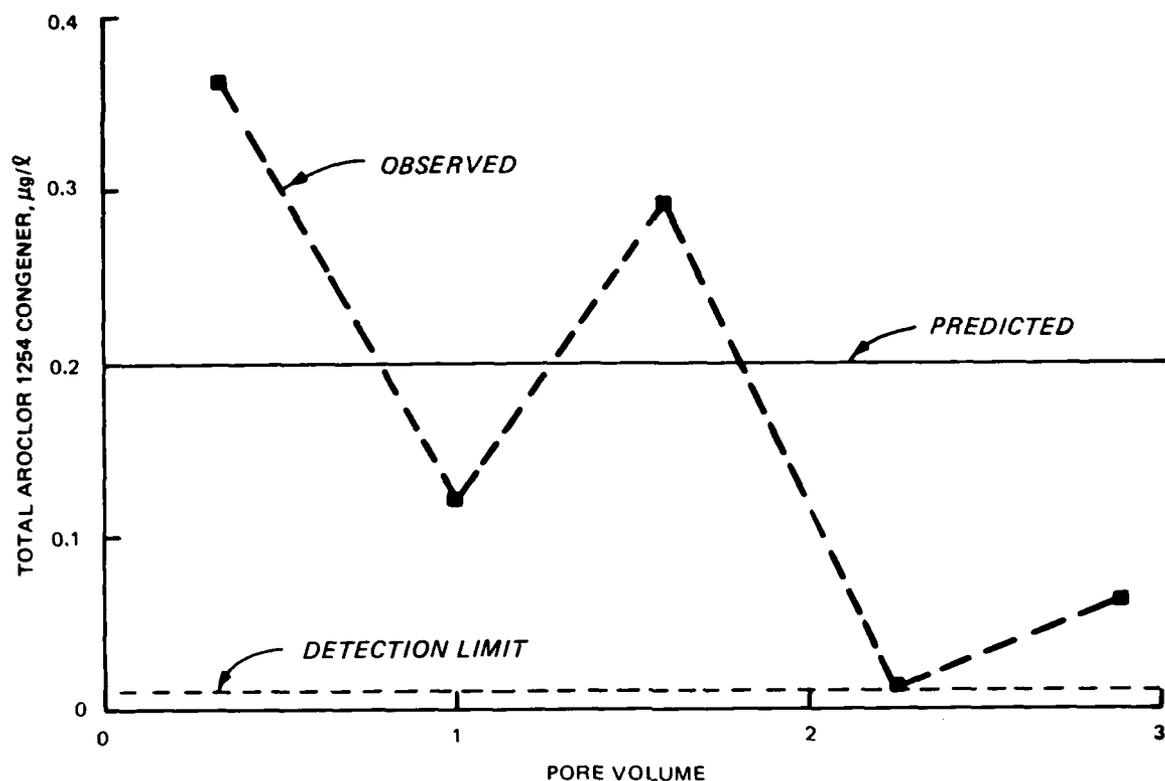
Figure C22. Comparison of observed and predicted total Aroclor
1254 congener concentrations in anaerobic permeameter leachate

agreement between observed and predicted concentrations was usually reason-
able, the initial concentrations from the "aerobic" columns were much higher
than obtained during batch testing. The physical chemical basis for these
differences has not yet been fully explained. However, the pH variation
during the anaerobic column test matched that in the anaerobic batch test
quite closely. In the aerobic batch test the pH dropped, while in the aerobic
column study the pH rose substantially. Because of the pH differences between
aerobic batch and column tests, application of the integrated approach to par-
tially oxidized sediment is of limited value because the assumption of equiv-
alent leaching environments is not fully satisfied.

Summary

75. Releases of metals during anaerobic testing were relatively low.
Two elements (Cu and Pb) were characterized by inverse desorption isotherms
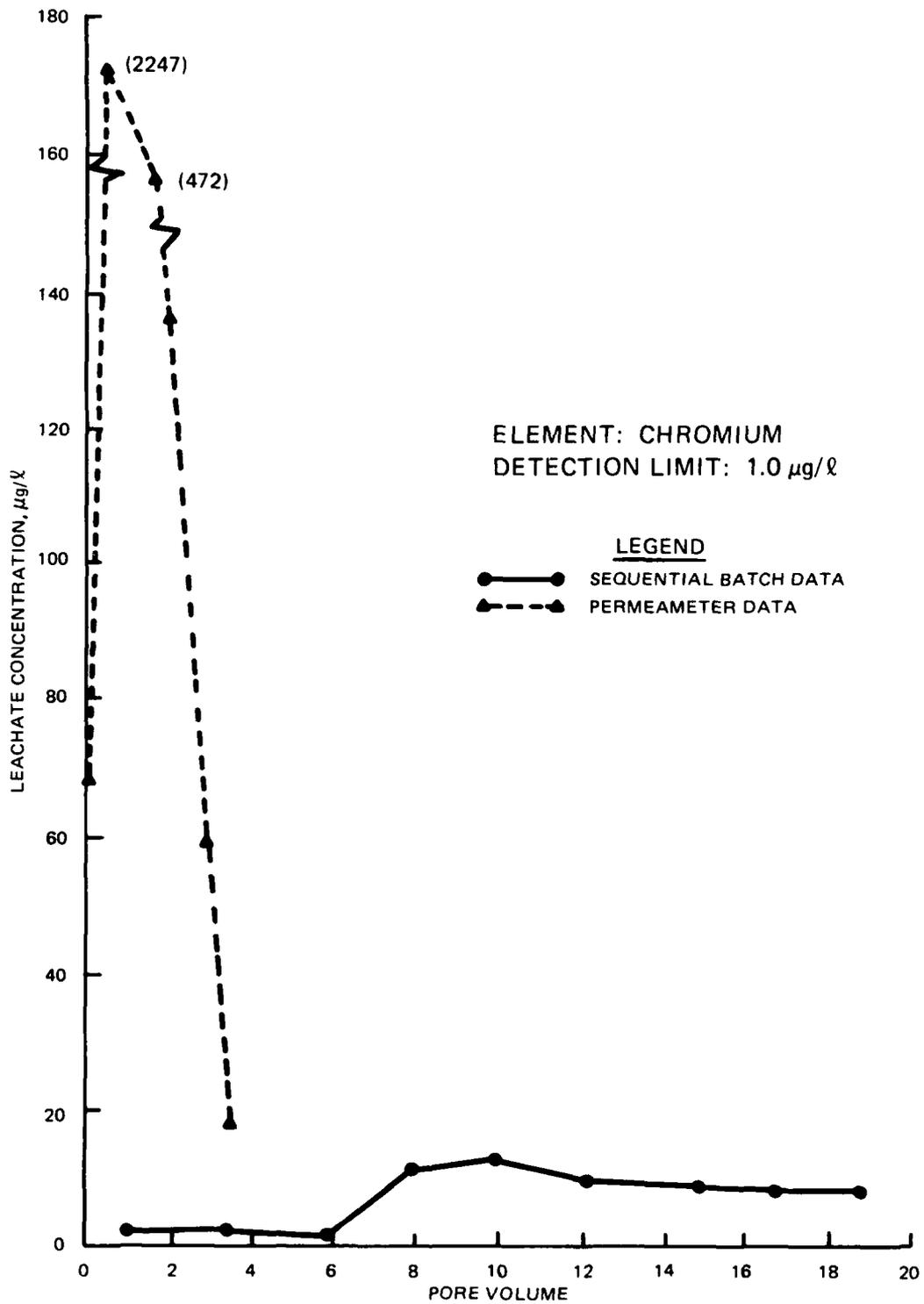
Figure C23. Comparison of permeameter and sequential batch
leachate chromium concentrations during aerobic leaching
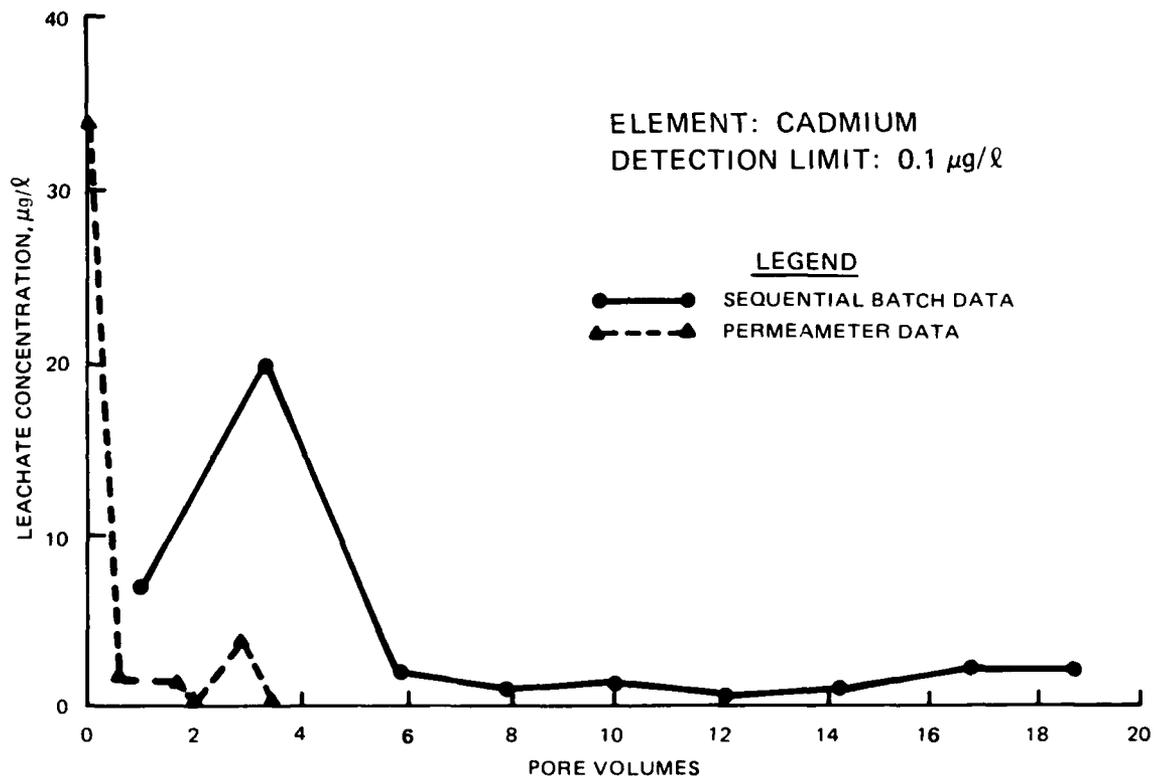
Figure C24. Comparison of permeameter and sequential batch leachate cadmium concentrations during aerobic leaching
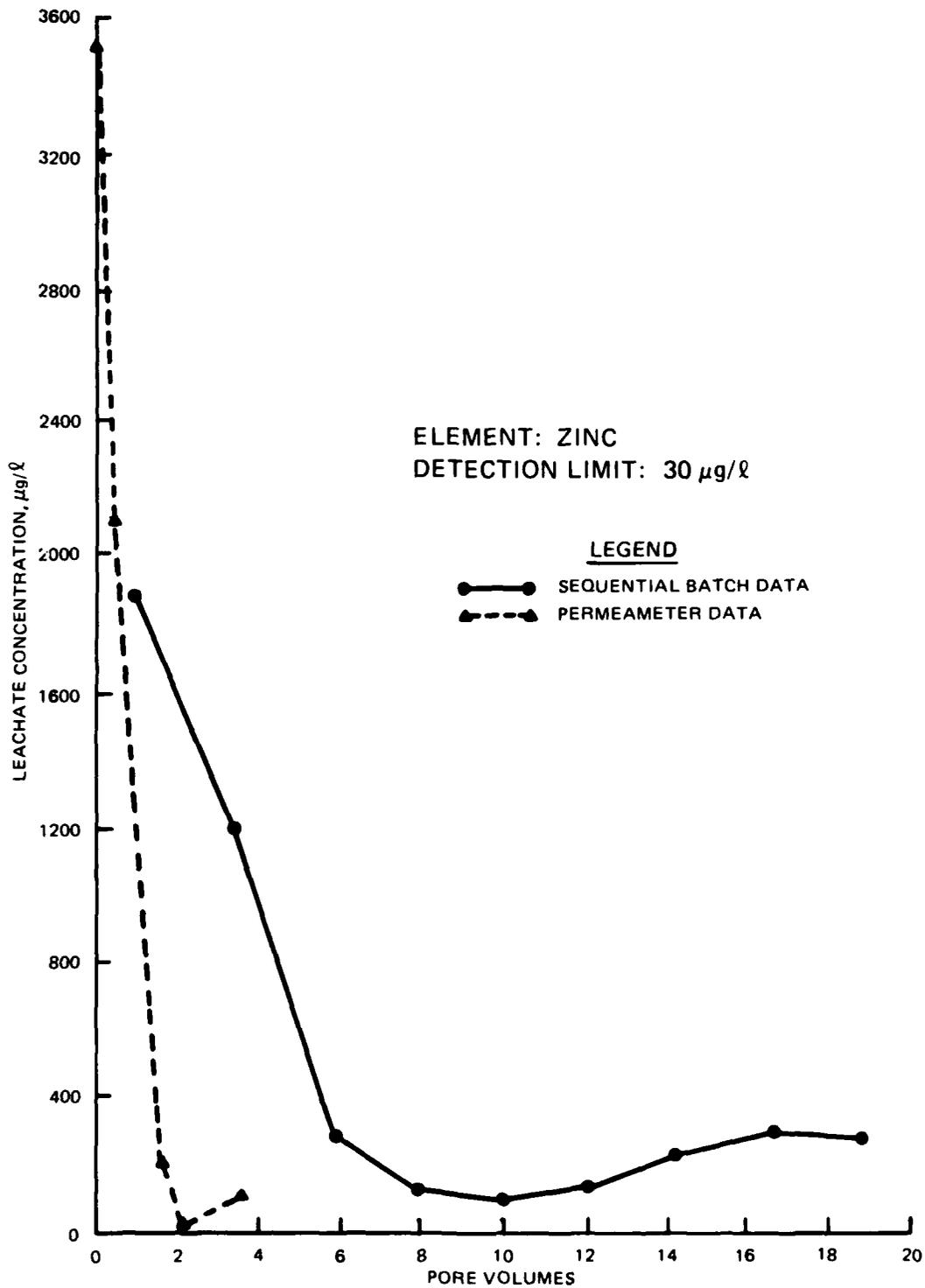
Figure C25. Comparison of permeameter and sequential batch leachate zinc concentrations during aerobic leaching

C80
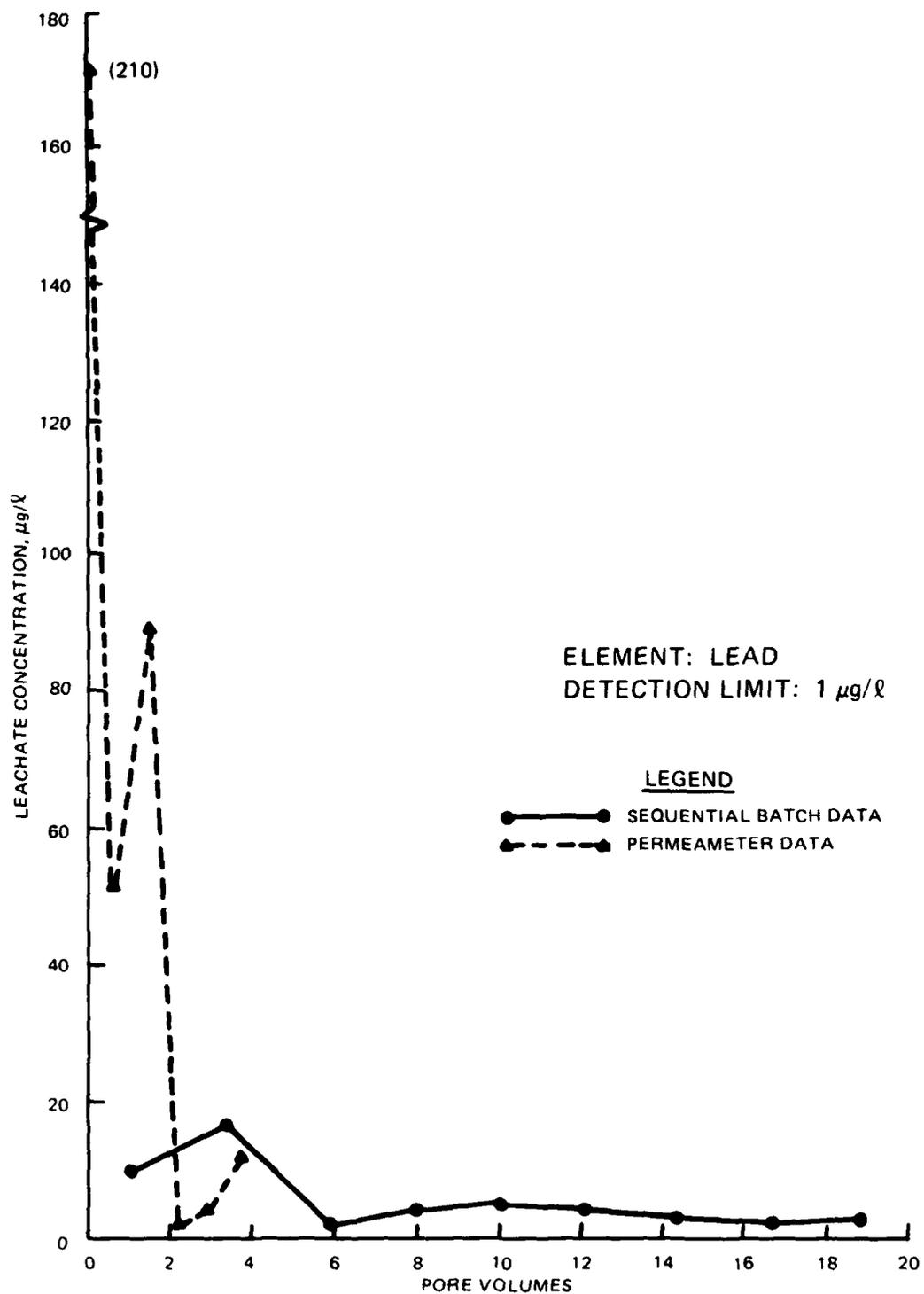
Figure C26. Comparison of permeameter and sequential batch leachate lead concentrations during aerobic leaching

and two others (As and Ni) by double-valued desorption isotherms. The remainder (As, Cd, and Cr) produced clustered desorption isotherms for which well-defined relationships were not evident. This is believed to be the first time inverse and double-valued desorption isotherms have been reported in sediment leaching studies. As previously discussed, the inverse and double-valued isotherms are indicative of nonconstant geochemistry during the sequential leaching. Figure C27 shows how changing sediment chemistry can produce inverse desorption isotherms and the upper limb of double-valued desorption isotherms. The changes in sediment chemistry between steps in the sequential leach procedure increase contaminant mobility (decrease in K_d). The concept presented in Figure C27 is tentative, and further testing and verification are required before this explanation of inverse and double-valued desorption isotherms can be accepted.

76. Using a simplified integrated approach, direct comparison of anaerobic batch and column data was possible. For those metals analyzed during both anaerobic batch and column studies (As, Cd, Cr, Pb, and Zn), column behavior was well predicted for As, Cd, and Zn. Less agreement was observed for Pb and Cr.

77. Aerobic test results were characterized by large metal losses during batch testing. Thus, the potential for contaminant release is higher in a CDF that allows the dredged material to become oxidized than in a CDF that maintains anaerobic leaching condition. In most CDFs, partially oxidized sediment will constitute a relatively thin surface crust making up a small part of the total sediment mass. Even though the contaminant release from the crust may be significantly higher than from underlying material, contaminant flux through foundation soils or through dikes probably will not be affected unless a significant portion of the CDF reaches a partially oxidized state. The disposal alternative for which oxidization of the dredged material is most likely to be important is the upland alternative.

78. Average concentrations of specific PCB congeners (compounds 28, 29, 30, and 32) as well as total PCB congeners were about the same in anaerobic batch and column tests. Average anaerobic column concentrations agreed well with equilibrium concentrations computed using single-point estimates of K_d .

79. Worst-case contaminant flux calculations can be made using the maximum concentration observed in either the batch or column testing. For example, the maximum anaerobic concentration for Cr was observed in column
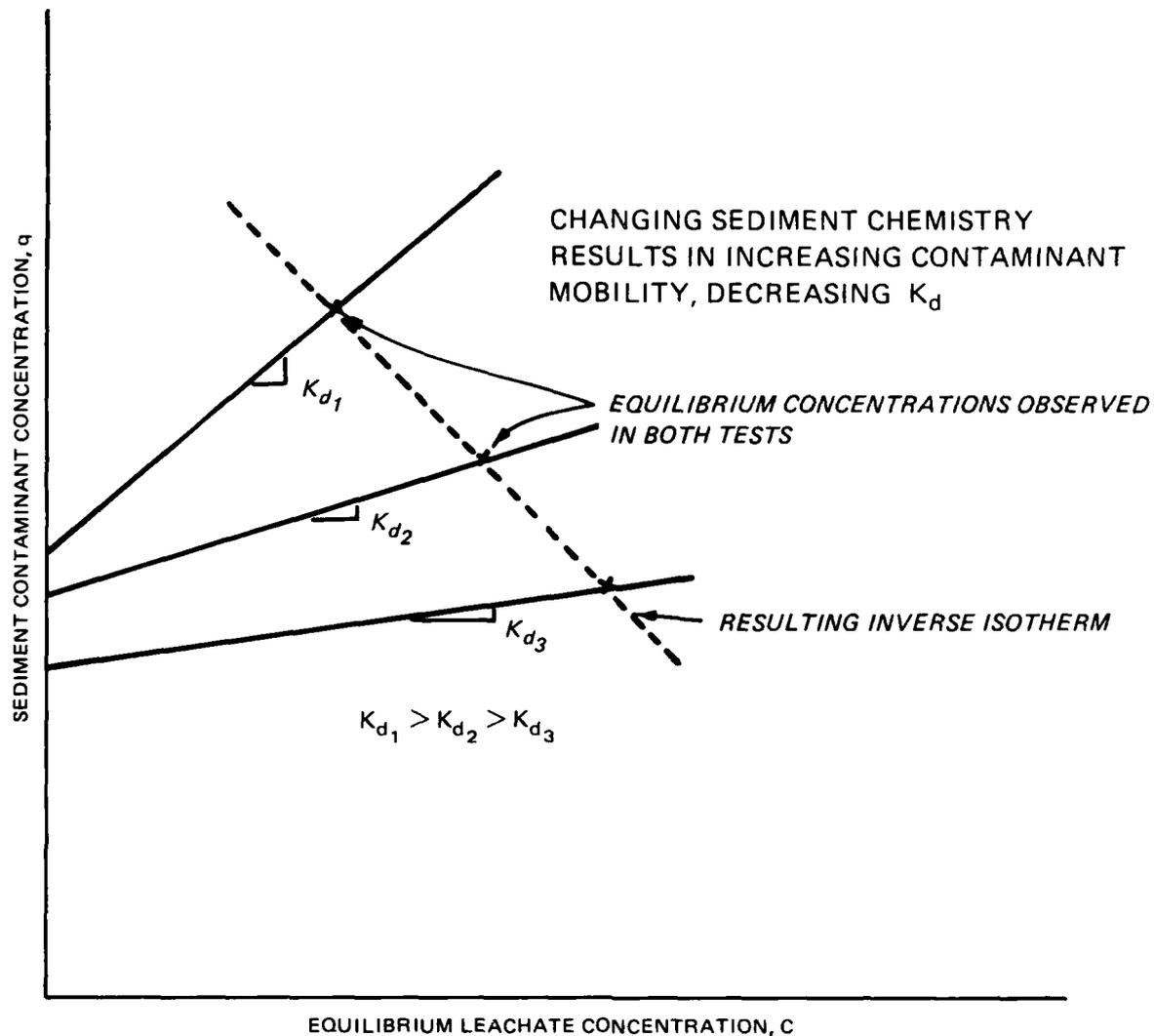
Figure C27. Effect of changing sediment chemistry on contaminant
distribution between sediment solids and leachate

tests while that for Zn was observed in batch tests. In the case of Ni and
Cu, column data are not available and maximum batch values must be used. Con-
taminant concentrations recommended for contaminant flux calculations are
listed in Table 5 in the main text. Because the peak concentration values
used in this table do not occur until several pore volumes have passed, the
peak contaminant flux may not occur until a CDF has been in operation for some
time. Further, maximum flux for all metals is not expected to occur
simultaneously.

C83

PART V: CONCLUSIONS

80. An integrated laboratory approach was used to investigate contaminant leaching from Everett Harbor sediment. The integrated approach appears to provide a useful theoretical framework within which to describe leaching phenomena. The results presented in this appendix, in part, provide the basis for performing contaminant flux analysis for proposed confined disposal facilities. Specific conclusions are provided below.

<u>a</u>. A contaminant transfer equation based on the assumption of equilibrium-controlled linear desorption reasonably predicted anaerobic column leachate concentrations for PCBs.

<u>b</u>. Overall, Everett Harbor results indicate that anaerobic column behavior could be predicted using batch data, although the basis for direct comparison using an approximate method was limited. Results for the anaerobic column data and application of the direct comparing method are presented in Figures C15-C19 and C22.

<u>c</u>. Approximate methods for applying the integrated approach can be used. However, methods that do not use a contaminant transport equation will require significantly longer column operation.

<u>d</u>. A contaminant transport equation with variable coefficients is needed to couple interphase transfer of contaminants from sediment solids to leachate with the advective and dispersive flux in continuous-flow systems. To apply a more sophisticated equation, functional relationships between distribution coefficients and pore-volume throughput will be required. The effort required to develop reliable input needed for a complicated model was not within the scope of this study.

<u>e</u>. Higher contaminant release to the environment from Everett Harbor sediment will occur in instances where the sediment is allowed to oxidize. The potential significance of this result is dependent on the operating scenario of the CDF and is therefore highly site specific.

<u>f</u>. The anaerobic sequential batch leach tests for Everett Harbor sediment exhibited nonconstant geochemistry (variable pH) that resulted in two types of nonideal desorption isotherms for metals, inverse and double-valued. This is believed to be the first time inverse and double-valued desorption isotherms have been reported for sediment.

<u>g</u>. An understanding of the diversity of chemical interactions and sediment geochemistry is required to interpret data from batch leach tests. Data reduction and analysis by statistical procedures alone can be seriously misleading. The integrated approach used in this study provides a technical basis for interpretating batch leach data.

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APPENDIX D: CAPPING EFFECTIVENESS TESTING

Objective

1. The objective of the capping effectiveness testing was to determine
the minimum cap depth required to chemically isolate contaminated Everett Har-
bor sediment from the overlying water column, using native sediment as a cap-
ping material in a small-scale laboratory test. Large reactor units were then
used to verify the effects of cap thickness on the efficiency of capping in
preventing impacts to aquatic biota as well as the overlying water column.
Results of this study will assist the US Navy in designing an environmentally
acceptable plan for the disposal of contaminated sediment from the East Water-
way of Everett Harbor.

Small-Scale Predictive Tests

2. The small-scale predictive tests are procedures being developed in
the Long-Term Effects of Dredging Operations (LEDO) Program. The ability of
the capping material to chemically seal contaminated dredged material contain-
ing relatively mobile and oxygen-demanding constituents was determined in
22.6-ℓ cylindrical Plexiglas small-scale units. The design and sediment-
loading arrangement of an individual unit are shown in Figure D1. This exper-
iment was conducted in a controlled-environment chamber where the temperature
was regulated at 20 ± 0.50° C. A 10-cm-deep layer of Everett Harbor sediment
was placed into the bottom of the small-scale unit, followed by 2 to 30 cm of
native sediment and 10 ℓ of water. Uncapped Everett sediment alone and cap
materials alone were used as controls. Ten litres of artificial seawater at
20 ppt, prepared from TRI S artificial sea salts, was added as gently as
possible to each column.

3. All treatments were initially aerated for 3 days to ensure dissolved
oxygen saturation by slowly bubbling air through the water column. Then, the
aeration apparatus was removed and a Plexiglas stirring plunger was suspended
between the sediment and the surface of the water column. To prevent exchange
of dissolved oxygen with the atmosphere, a layer of mineral oil (4-cm depth)
was added to seal the surface of the water column from the atmosphere. To
ensure a homogeneous sample, the overlying water was manually mixed daily with

D1
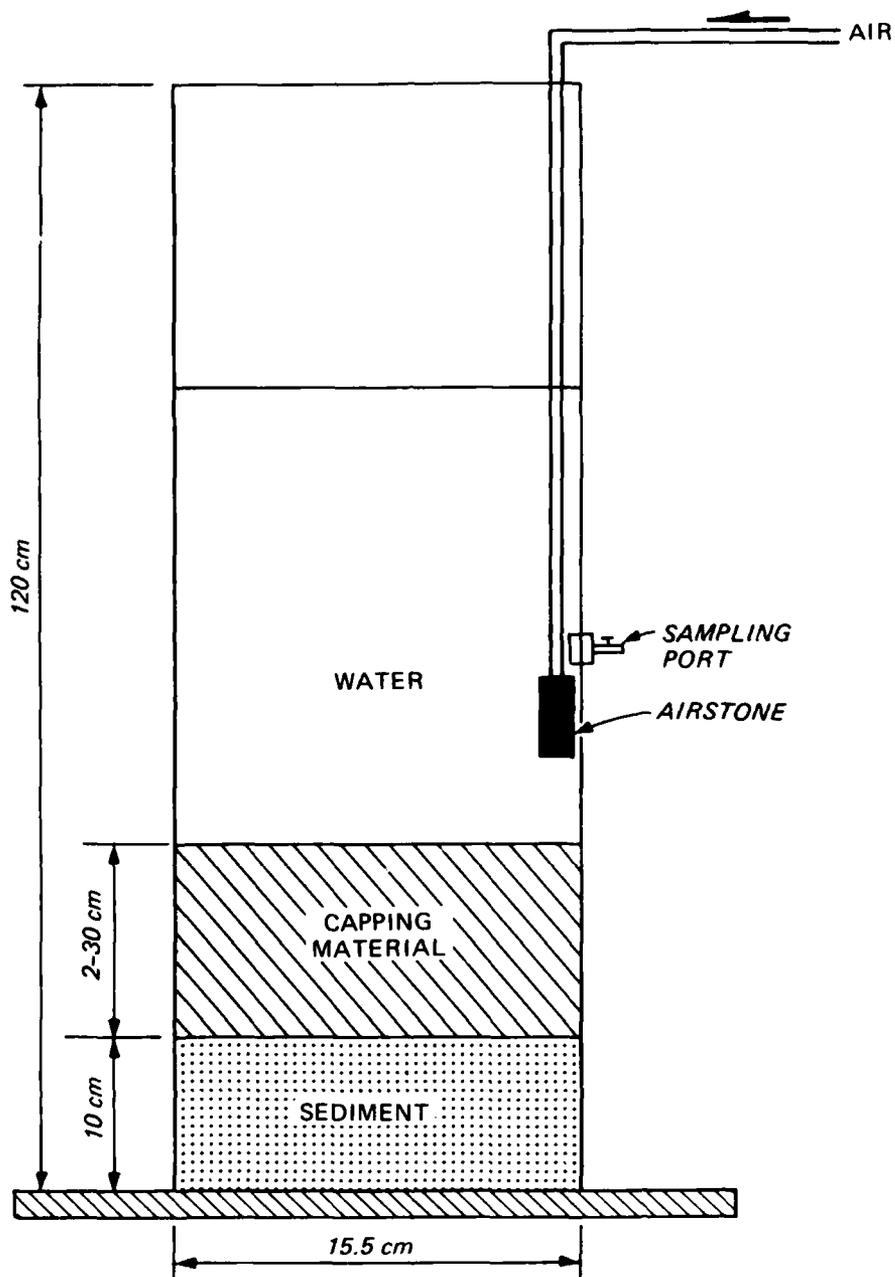
Figure D1. Design of small-scale unit

the Plexiglas plunger. All experiments were conducted in triplicate. Water samples were taken initially and at regular intervals for 30 days or until the dissolved oxygen was depleted.

4. Dissolved oxygen was measured in samples by permitting water to flow gently from a long tube attached to the small-scale unit sampling port into a standard biological oxygen demand (BOD) bottle. Dissolved oxygen was

determined with the azide modification of the Winkler Method as described in Standard Methods (APHA 1980).

5. Water samples to be analyzed for ammonium-nitrogen and orthophosphate-phosphorus (relatively mobile ionic chemical species that are released under anaerobic conditions) were cleared of particulate matter by passage through a 0.45-μ membrane filter under a nitrogen atmosphere and then preserved by acidification with concentrated HCl to pH 2, followed by immediate freezing and storage at 4° C. Ammonium-nitrogen and orthophospate-phosphorus were determined using a Technicon Autoanalyzer II in accordance with procedures recommended by Ballinger (1979).

Large Reactor Unit Experiments

6. Laboratory studies to assess the medium-term (40 days) effectiveness of native sediment in isolating Everett Harbor sediments were conducted in a controlled-environment chamber maintained at 20 ± 0.5° C, using modified 250-ℓ flow-through large reactor units (Figure D2), described in detail by Gunnison et al. (1987). These large reactor units are 121 cm in height and measure 46 cm on a side. Modifications included sealing of sampling ports with Plexiglas, removal of the mixing pump from the system, and provision for constant aeration of the water column. First, 17 cm of Everett Harbor sediment was placed on the bottom of each reactor unit. This sediment was then capped with 50 cm of native sediment. (This depth cap was based on results of the small-scale tests and the depth to which polychaetes burrow.) Then, 60 ℓ of artificial seawater at 20 ppt salinity was added and allowed to equilibrate with aeration for 14 days. A 14-day equilibration time was selected to allow initial compaction to occur and material suspended during water addition to settle. At the end of this equilibration/consolidation period, flow-through of artificial seawater was initiated at a rate of 1.2 ℓ/hr. At this flow rate, 50 percent of the overlying water was replaced every 36 hr (Sprague 1969). The water column in each large reactor unit was continuously aerated from the bottom to ensure a well-mixed aerobic water column.

7. The results of previous capping studies have demonstrated the importance of using at least two organisms to assess the effectiveness of capping in preventing movements of contaminants into the biota (see Brannon et al. 1986). One organism should be representative of the benthic surface
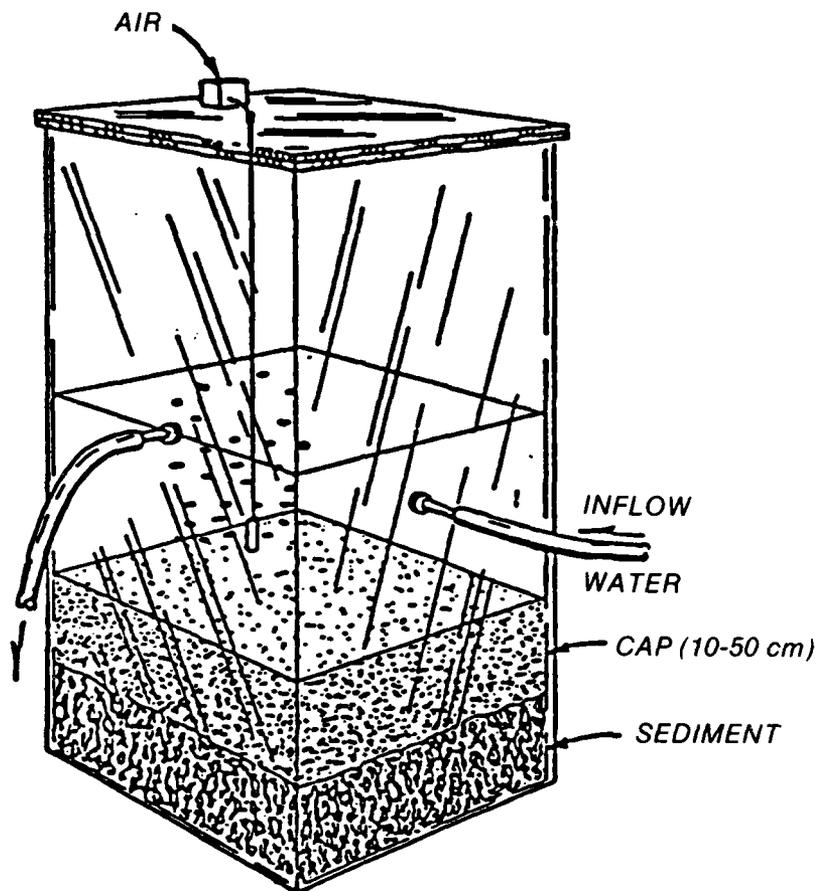
Figure D2. Large reactor units

community; the second should be a variety of mussel that can be suspended in the overlying water column. Through personal communications with John Malek of the Seattle District, three organisms were selected: polychaete (*Nereis virens*), clam (*Macoma nasuta*),and mussel (*Mytilus edulis*). The clam and poly-chaete were used to assess the effect of capping on contaminant bioaccumula-tion in benthic and infaunal organisms and to provide a source of bioturbation, while mussels were used to determine whether contaminants were moving through the cap and into the water column. The polychaetes were obtained from the Maine Bait Company, New Castle, Maine; mussels and clams, from John Brezina, Dillon Beach, California. All animals were acclimated to test conditions in the laboratory for at least 3 weeks prior to being added to the large reactor units.

8. After 4 days of flow-through operation in the large reactor units, polychaetes, clams, and mussels were added to the various units as shown in Table D1.

Table D1

Experimental Setup for Everett Harbor Sediment with Native Sediment Cap

| | Animals in Reactor Unit | | |
Treatment	Polychaetes	Clams	Suspended Mussels
Control (native sediment)	x	x	x
Everett Harbor	x	x	x
50-cm cap	x	x	x

9. Fifty mussels were suspended in a basket (mesh size, 2.5 mm^2) in the water column of each large reactor unit, approximately 5 cm above the sediment surface. A total of 30 clams and 35 polychaetes were added to the surface of the sediment in each large reactor unit. Concurrent with the addition of animals to the large reactor units, samples of each animal were removed from the holding tanks for initial chemical characterization. Samples of mussels suspended in the water column were removed at 10- and 40-day intervals. Samples of polychaetes and clams were taken initially and at the conclusion of the experiment. The polychaetes were depurated for 24 hr prior to analyses; this was done to remove sediment and food from the gut. Mussels and clams in each large reactor unit were fed 5 ml of marine invertebrate diet per day. Polychaetes were fed 1 g of ground Tetramin per day. Each experimental combination was conducted in triplicate.

10. At the conclusion of the experiment, water samples were obtained from the water column for chemical analysis for key contaminants identified in the sediment by bulk chemical analysis. Samples used for PCB and PAH analyses were placed in hexane-washed, heated (105° C for 24 hr), 3.8-ℓ glass jars. Samples for metal analyses were filtered through 0.45-μ pore sized membrane filters. The first 100 ml of filtrate was discarded; this was done to remove final residues washed from the filter. The subsequent filtrate was acidified to pH 1 with concentrated nitric acid. Water samples were analyzed for As, Cd, Cr, Cu, Pb, Hg, Ni, and Zn using a Perkin Elmer Model 2100 heated graphite atomizer and a Perkin-Elmer Model 503 atomic absorption spectrometer. Mercury

was determined using a Perkin-Elmer Model 503 atomic adsorption unit coupled to a Perkin-Elmer MHS-10 hydride generator.

11. Water, tissue, and sediment samples were analyzed for seven PCB isomers (total monochlorobiphenyls through total heptachlorobiphenyls). Isomer concentrations were determined following soxhlet extraction, sulfuric acid cleanup, and quantification in an electron capture detector gas chromatograph. Sixteen compounds, comprising the family of compounds collectively referred to as PAHs, were also determined in water, tissue, and sediment (Table D2). Samples were soxhlet extracted overnight with benzene:methanol. The aromatic hydrocarbon fraction was then separated using silical gel chromatography, concentrated, and subjected to capillary gas chromatographic analyses on a Hewlett Packard 5840A gas chromatograph equipped with a flame ionization detector. Individual compounds were quantified using analytical standards and an internal standard.

12. Total organic carbon (TOC) in sediment samples was determined by dry combustion (Allison 1965). Sediment particle size distribution was determined using the method of Patrick (1958).

Results

13. Means and standard errors were determined for each parameter within a treatment. The Duncan Multiple Range Test was used to determine the statistical significance of differences between treatments. Statements of significance made in the text refer to the 5-percent level ($p < 0.05$) or less.

Sediment characterization

14. The native sediment had a higher concentration of heavy metals than Everett Harbor sediment, with the exception of Zn, Pb, and Hg (Table D3). The native sediments were also higher in total organic carbon; however, there was no significant difference in texture (Table D3).

15. Native sediment PCB concentrations were below detection limits (Table D4). Total PCB concentration in Everett Harbor sediment was 0.0132 µg/g. Total trichlorobiphenyl, total hexachlorobiphenyl, and total heptachlorobiphenyl constituted the largest fraction of PCBs in Everett Harbor sediment (Table D4).

16. Everett Harbor sediment contained much higher levels of PAH compounds than did the native (cap) sediment (Table D5), except for six-ring

Table D2

PAHs Determined in Sediment, Water, and Tissue Samples

Number of Rings	Name of Compound
Two-ring compounds	Napthalene
Three-ring compounds	Acenaphthalene
	Acenapthene
	Phenanthrene
	Anthracene
	Fluorene
	Fluoranthene
Four-ring compounds	Pyrene
	Chrysene
	Benzo(a)anthracene
Five-ring compounds	Benzo(a)pyrene
	Benzo(b)fluoranthene
	Benzo(k)fluoranthene
	Benzo(a,h)anthracene
Six-ring compounds	Indeno (1,2,3-c,d)pyrene
	Benzo(g,h,i)perylene

compounds. Since the PAHs were generally higher in the Everett Harbor sediment, they served as tracers in this study.

Contaminant release and uptake

17. The concentrations of selected contaminants were determined in the water column, mussels, clams, and polychaetes to assess the ability of 50-cm native sediment to isolate contaminated dredged material. The mortality rate for each group of organisms (mussel, clams, and polychaetes) was very low in the large reactor units; 90 percent of the animals in each group survived.

18. Heavy metals. Heavy metals concentrations in the water column above capped sediments did not differ from their respective concentrations in the control (native sediment only) unit water columns (Table D6).

19. PAHs. Replicated samples for PAHs in the water column were composited to obtain lower detection limits by increasing the volume of water available for extraction. Even using these techniques, PAH concentrations were below the detection limits of 0.005 µg/ℓ in all treatments.

Table D3

Heavy Metal Concentrations and Selected Sediment Physical Characteristics

| Sediment | Metal Concentration* | | | | | | | | | | Texture, Percent Sand:Silt:Clay |
	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Mn	TOC	
Everett Harbor	5.73	3.30	39.75	73.35	48.10	0.20	21.35	148.50	237.00	71.50	20:25:55
Native sediment (capping)	8.19	3.43	57.20	80.10	34.60	0.10	59.30	110.50	334.80	18,192.00	18:24:58

* Given in micrograms per gram sediment, dry weight.

Table D4

Sediment PCB Concentrations*

| | Sediment | |
	Everett Harbor	Native
Total monochlorobiphenyls	<0.0002	<0.0002
Total dichlorobiphenyls	<0.0002	<0.0002
Total trichlorobiphenyls	0.0015	<0.0002
Total tetrachlorobiphenyls	<0.0002	<0.0002
Total pentachlorobiphenyls	<0.0002	<0.0002
Total hexachlorobiphenyls	0.0053	<0.0002
Total heptachlorobiphenyls	0.0064	<0.0002
Total	0.0132	

* Given in micrograms per gram sediment, dry weight.

Table D5

Sediment PAH Concentration*

Number of Rings	Everett Harbor	Native
Two-ring compounds	7.70	1.00
Three-ring compounds	12.30	0.94
Four-ring compounds	7.70	0.80
Five-ring compounds	6.40	0.84
Six-ring compounds	<1.00	0.20
Total PAHs	34.10	3.78

* Given in micrograms per gram sediment, dry weight.

Table D6

Water Column Heavy Metal Concentration* in the Everett

Harbor Capping Study Following 40 Days of Incubation

Treatment	As	Cd	Cr	Cu	Pb	Hg	N	Zn
Native sediment	0.005 (±0.0000)	0.002 (±0.0009)	0.005 (±0.0003)	0.022 (±0.0021)	0.009 (±0.0040)	0.002 (±0.0000)	0.048 (±0.0050)	0.071 (±0.0200)
Everett Harbor	0.005 (±0.0000)	0.002 (±0.0001)	0.004 (±0.0003)	0.004 (±0.0003)	0.009 (±0.0110)	0.002 (±0.0003)	0.035 (±0.0010)	0.03 (±0.0000)
50-cm cap**	0.005 (±0.0000)	0.002 (±0.0003)	0.005 (±0.0000)	0.027 (±0.0060)	0.009 (±0.0005)	0.002 (±0.0000)	0.040 (±0.0020)	0.034 (±0.0040)
Inflow water	0.005 (±0.0000)	0.002 (±0.0003)	0.006 (±0.0000)	0.027 (±0.0060)	0.009 (±0.0005)	0.002 (±0.0000)	0.050 (±0.0030)	0.034 (±0.0040)

* Given in micrograms per litre (± standard error).
** Polychaete present.

20. **PCBs.** Samples analyzed for the presence of PCB isomer groups in the water column revealed PCB concentrations below the detection limits of 0.00001 µg/ℓ in all water samples tested (Table D7).

21. Heavy metals, PAH, and PCB concentrations were below detection in the inflow water.

Heavy metals

22. **Mussel.** Concentrations of heavy metals in mussel tissue did not significantly ($p < 0.05$) exceed those in mussels exposed to the cap material alone (control) in any of the treatments following 10 and 40 days of exposure (Table D8).

23. **Clam.** Concentrations of heavy metals in clam tissue from the control generally exceeded concentrations in clam tissue from the Everett Harbor sediment treatment after 40 days (Table D9). While there appeared to be a difference in Cr and Cu tissue concentrations between the control and the Everett Harbor sediment, this difference was not significant ($p < 0.05$).

24. **Worm.** Significant Cd bioaccumulation by polychaetes compared to the control was noted in the uncapped Everett Harbor treatment after 40 days of exposure (Table D10). However, there was no significant difference ($p < 0.05$) in uptake of cadmium between the control (cap material alone) and the 50-cm cap treatment. This would indicate that the 50-cm cap effectively isolated Everett Harbor sediment.

PCBs and PAHs

25. **Mussel.**

    <u>a</u>. **PAHs.** After 10 and 40 days of incubation, mussel tissue PAH concentrations in all treatments were below the detection limit of 0.40 µg/g wet weight.

    <u>b</u>. **PCBs.** All PCB concentrations in the mussel tissue in the control were below the detection limit of 0.001 µg/g wet weight. Mussel tissue PCB concentrations in the uncapped Everett Harbor and 50-cm cap treatments were below the detection limit, except total pentachlorobiphenyl and total hexachlorobiphenyl; those values were 0.002 and 0.004 µg/g wet weight, respectively. However, after 40 days of exposure there was no significant difference in uptake between the uncapped Everett Harbor and the 50-cm cap.

26. **Worm.**

    <u>a</u>. **PAHs.** There was no significant difference in PAH accumulation by worm tissue in uncapped Everett Harbor or capped Everett Harbor treatments compared to controls. Concentrations of PAH

Table D7

Water Column PCB Concentration* Following 40 Days of Incubation

Parameters	Inflow Water	Native Cap	Everett Harbor	50-cm Cap**
Total monochlorobiphenyl	<0.00001	<0.00001	<0.00001	<0.00001
Total dichlorobiphenyl	<0.00001	<0.00001	<0.00001	<0.00001
Total trichlorobiphenyl	<0.00001	<0.00001	<0.00001	<0.00001
Total tetrachlorobiphenyl	<0.00001	<0.00001	<0.00001	<0.00001
Total pentachlorobiphenyl	<0.00001	<0.00001	<0.00001	<0.00001
Total hexachlorobiphenyl	<0.00001	<0.00001	<0.00001	<0.00001
Total heptachlorobiphenyl	<0.00001	<0.00001	<0.00001	<0.00001

 * Given in micrograms per litre.
** Polychaete present.

in all treatments were below the detection limit (0.002 µg/g wet weight).

b. **PCBs.** Total hexachlorobiphenyl concentration in worm tissue in the control treatment was higher after 40 days (0.007 µg/g wet weight) than the uncapped Everett Harbor or the capped Everett Harbor treatments. All other PCB concentrations in all treatments were below the detection limit (0.001 µg/g wet weight).

27. **Clam.**

a. **PAHs.** All PAH clam tissue concentrations were below the detection limit in all treatments (0.002 µg/g wet weight).

b. **PCBs.** PCB concentrations in clam tissue were below the detection in all treatments, except total hexachlorobiphenyl. The values for total hexachlorobiphenyl in the control, uncapped Everett and 50-cm cap were 0.012, 0.10 and 0.10 µg/g wet weight, respectively. There was no significant difference ($p < 0.05$) between these values.

Small-scale tests

28. **Dissolved oxygen depletion rates.** Small-scale tests were conducted to determine the thickness of cap necessary to chemically isolate a contaminated sediment from the water column. Dissolved oxygen depletion in the water column would not normally be expected to be a problem in an open-water disposal environment because of mixing and reaeration. Dissolved oxygen

D12

Table D8

Heavy Metal Concentration* in Mussel (*Mytilus edulis*)

at 10 and 40 Days

	Treatment					
	Native Sediment Only (Control)		Everett Harbor Only		Everett Harbor with 50-cm Cap	
Metal	10 days	40 days	10 days	40 days	10 days	40 days
As	1.10 (±0.012)	1.22 (±0.068)	0.88 (±0.446)	0.33 (±0.330)	1.22 (±0.076)	0.36 (±0.360)
Cd	5.15 (±0.337)	7.20 (±0.527)	5.20 (±0.527)	5.14 (±0.140)	5.32 (±0.620)	7.35 (±0.990)
Cr	2.32 (±0.236)	3.09 (±0.450)	3.49 (±0.949)	2.36 (±0.170)	3.07 (±0.180)	3.48 (±0.430)
Cu	9.07 (±0.299)	9.61 (±0.290)	10.42 (±0.740)	9.28 (±0.230)	11.37 (±1.110)	10.47 (±0.970)
Pb	1.93 (±0.202)	2.63 (±0.070)	2.24 (±0.164)	2.22 (±0.022)	2.52 (±0.120)	2.38 (±0.590)
Hg	0.08 (±0.038)	0.05 (±0.050)	0.17 (±0.014)	0.20 (±0.024)	0.16 (±0.053)	0.17 (±0.008)
Ni	10.30 (±0.300)	26.07 (±1.040)	9.72 (±0.691)	20.40 (±0.450)	12.13 (±2.232)	26.95 (±2.730)
Zn	197.67 (±11.667)	245.00 (±15.500)	149.00 (±18.730)	198.00 (±10.790)	159.00 (±12.680)	224.00 (±23.640)

* Concentration given in micrograms per gram, wet weight (± standard error).

Table D9

Heavy Metal Concentration* in Clam (*Macoma nasuta*) Tissue
Following 40 Days of Exposure

Metals	Native Sediment Only (Control)		Everett Harbor Sediment Only		Everett Harbor with 50-cm Cap**	
As	5.83	(±0.555)	3.97	(±0.343)	6.73	(±0.384)
Cd	1.22	(±0.900)	1.03	(±0.061)	0.84	(±0.032)
Cr	12.30	(±0.500)	23.85	(±9.021)	0.38	(±0.738)
Cu	24.40	(±0.650)	25.67	(±4.065)	24.18	(±0.794)
Pb	8.57	(±0.365)	6.44	(±-.609)	7.11	(±0.180)
Hg	0.23	(±0.011)	<0.10	(±0.000)	0.21	(±0.037)
Ni	32.25	(±9.987)	31.10	(±10.553)	27.43	(±0.260)
Zn	227.00	(±56.000)	212.67	(±43.97)	229.00	(±40.011)

* Concentration given in micrograms per gram, wet weight (± standard error).
** Polychaetes present.

Table D10

Heavy Metal Concentrations* in *Nereis virens*
Following 40 Days of Exposure

Metal	Native Sediment		Everett Harbor		Everett Harbor	
As	3.57	(±0.340)	3.50	(±0.068)	3.81	(±0.362)
Cd	0.58	(±0.215)	4.19	(±0.817)	3.22	(±0.216)
Cr	2.18	(±0.615)	3.29	(±0.849)	1.27	(±0.592)
Cu	12.45	(±0.550)	12.37	(±0.441)	12.57	(±0.549)
Pb	1.60	(±0.520)	1.47	(±0.349)	1.09	(±0.214)
Hg	0.12	(±0.018)	0.07	(±0.037)	<0.10	(±0.000)
Ni	12.20	(±1.000)	10.90	(±0.893)	11.57	(±0.612)
Zn	182.50	(±11.500)	158.30	(±13.169)	142.67	(±17.910)

* Concentration given in micrograms per gram, wet weight (± standard error).

depletion, however, can be used as a tracer for determining how effectively a cap can isolate the underlying dredged material having an oxygen demand that is higher than the proposed capping material.

29. The dissolved oxygen depletion rates of the native sediment (635 ± 40 $mg/m^2/day$) were not significantly different ($p < 0.05$) from those of the Everett Harbor sediment (638 ± 15 $mg/m^2/day$). This precluded the use of dissolved oxygen depletion rates as a means of evaluating the effectiveness of capping.

30. <u>Nutrient release rates.</u> Ammonium-N (NH_4^+-N) release rates to the overlying water, derived by performing linear regression analysis of mass release per unit area (mg/m^2) versus time, are presented as a function of cap depth in Figure D3. Rates plotted are the means and standard deviations for three replicates. The 5-cm cap depth reduced the NH_4^+-N release rates by 32 percent from those observed with uncapped Everett sediment. The NH_4^+-N release rates decreased linearly ($r = 0.95$, $p < 0.05$) until a cap depth of 30 cm was reached. At this point, NH_4^+-N release rates of the capped Everett sediment were not significantly different ($p < 0.05$) from those of the native sediment.

31. Orthophosphate-phosphorus release rates to the overlying water, derived in the same manner as for NH_4^+-N, are shown in Figure D4. Based on the data, a cap thickness of 30 cm resulted in a 95-percent reduction in orthophosphate-phosphorus release rate.

Summary and Discussion

32. The small-scale predictive tests indicate that clean native sediment is effective in isolating contaminated Everett Harbor sediment from the water column. Increasing the cap thickness retarded the release of ammonium-nitrogen and orthophophate-phosphorus from the sediment to the overlying water. The ability to significantly reduce the movement of these reduced chemical constituents is used as an indicator of cap effectiveness because these species are much more mobile than most chemical contaminants associated with sediment. Data from the small-scale study showed that a minimum cap depth of 30 cm was an effective chemical seal. However, 50 cm was tested in the large reactor units because large polychaetes (450 mm in length) used in the study were shown through visual observation to burrow to a depth of 50 cm
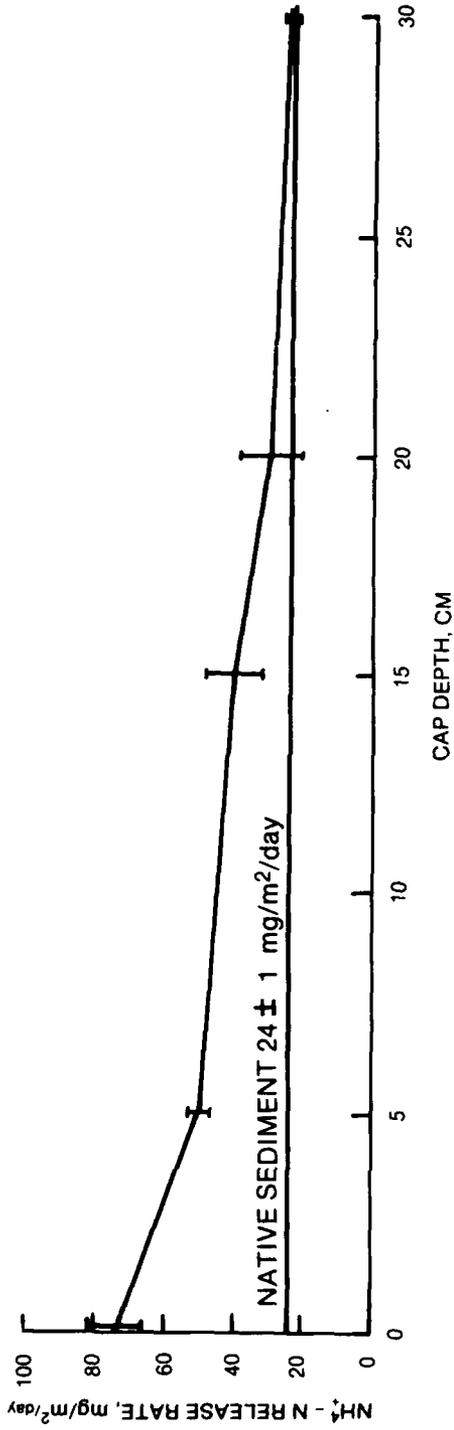
Figure D3. Effect of native sediment cap depth on ammonium-N release rates
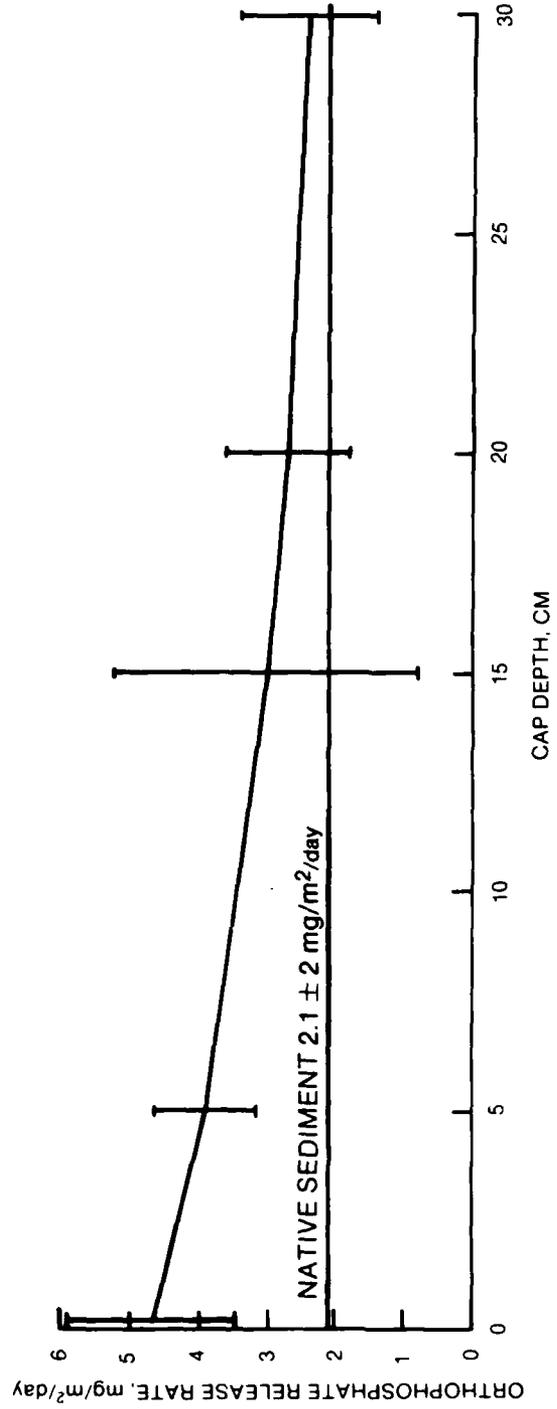
Figure D4. Effect of native sediment cap depth on orthophosphate release rates

D16

(Brannon et al. 1985). If the polychaetes were able to breech the cap, this could result in movement of contaminants into the overlying water column and biota.

33. The large reactor units confirmed the cap thickness required to obtain chemical sealing of the sediment from the overlying water column as determined by the small-scale predictive test, i.e., 30 cm plus an additional 20-cm depth to allow for polychaete burrowing. Results demonstrated that a 50-cm cap of native sediment overlying Everett Harbor sediment was effective in preventing the transfer of heavy metal, PAHs, and PCBs from the contaminated sediment into the overlying water and biota, even with bioturbation. To prevent exposure of burrowing benthic organisms to contaminated sediment, it is recommended that a safety margin be added to the thickness required to achieve a chemical seal. This safety margin is determined by assessing the depth reached by the deepest burrowing benthic organism within the region. Based on the experience of Burton Hamner and John Malek of the Seattle District (personal communication, August 1985), geoduck, a benthic organism found in the Puget Sound, burrows to a depth of approximately 0.5 m. Taking this depth into account, the thickness required to chemically and biologically isolate contaminated sediment from the overlying water column and aquatic biota is 80 cm, i.e., 30 cm plus the additional 50 cm to account for burrowing by the Puget Sound organism. This thickness does not take into account any additional material needed to allow for erosion and consolidation.

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APPENDIX E: SEDIMENTATION TESTING

Testing Objectives

1. The objective of this testing was to determine the settling behavior
of the sediments when placed hydraulically in confined disposal sites (inter-
tidal or upland). The zone, compression, and flocculent settling tests were
run in an 8-in.-diam* column to obtain required technical data regarding the
behavior of the dredged material.

Experimental Procedure

2. The settling tests were run using procedures found in WES Technical
Report DS-78-10 (Palermo, Montgomery, and Poindexter 1978) and EEDP Technical
Notes EEDP-02-1 through 4 (Palermo 1985). The tests generally involved mixing
a sediment and water slurry and then observing each of several types of
sedimentation behavior. The slurry was pumped from a 55-gal drum with a
positive displacement pump into an 8-in.-diam column (see Figure E1).

3. Salinity was measured on the supernatant in the drum before the
contents were mixed and the total solids concentration determined. The
salinity and total solids concentration were 25 ppt and 310.8 g/ℓ, respec-
tively. Everett Bay material contained a significant amount of wood chips.
The wood chips were removed by straining the slurry through a 3/8-in. sieve.

4. Removal of the wood chips was necessary because of operational
problems they posed, primarily in pumping.

Pilot test

5. The initial concentration of the slurry was reduced to 81.9 g/ℓ to
run a pilot test. The pilot test was performed to determine if flocculent or
zone processes will govern the initial settling. An interface was visible
after only a few minutes of settling, indicating that zone settling processes
would govern. The depth to interface was measured over time and plotted to
determine transition concentration. The transition concentration was
153.1 g/ℓ.

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* A table of factors for converting non-SI units of measurement to SI
   (metric) units is presented on page 14 of the main text.
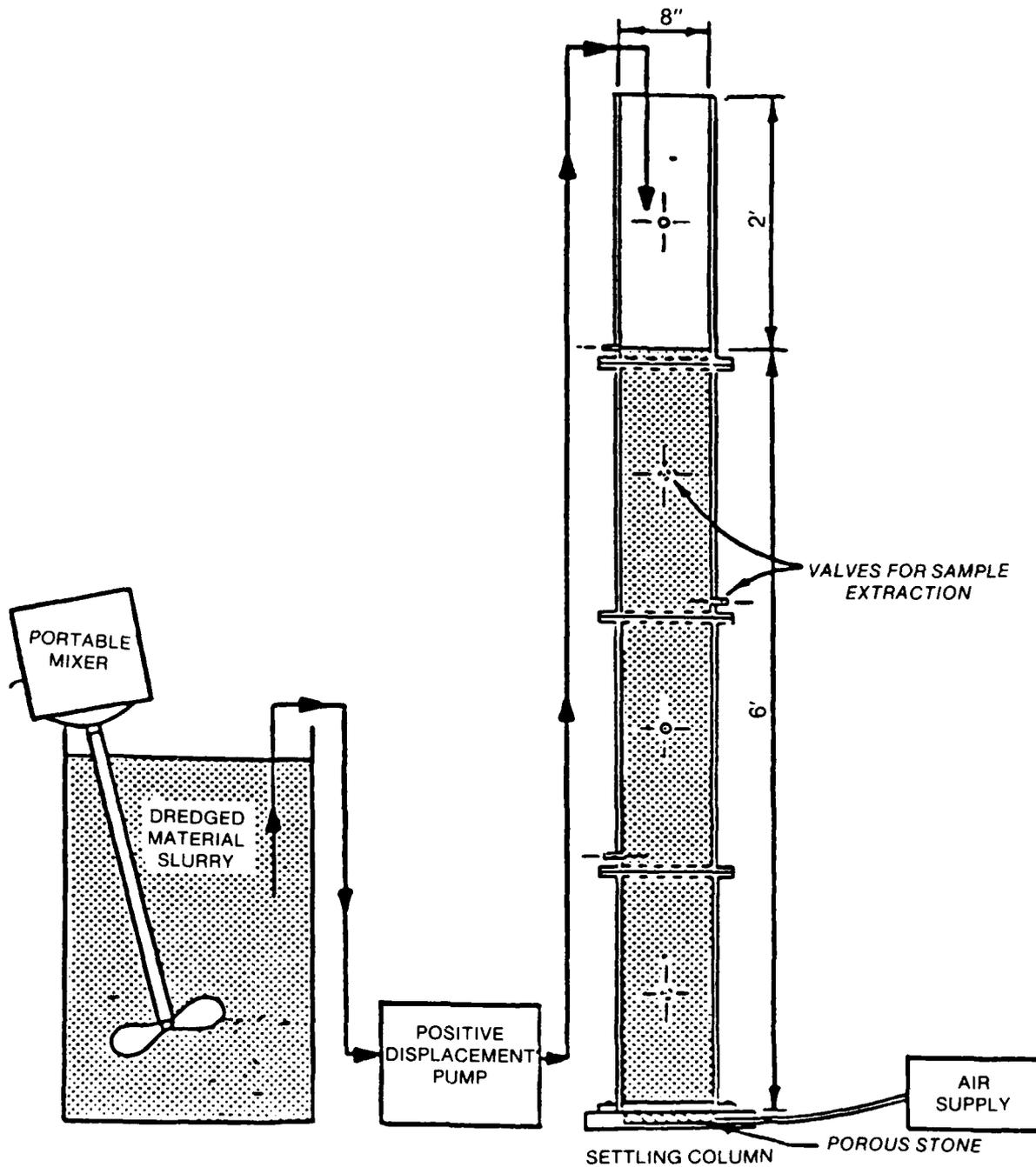
Figure E1.   Schematic of dredged material settling column

Compression test

6. The target slurry concentration was 150 g/ℓ to run the compression test. After the slurry was thoroughly mixed and pumped into the column, six samples for total solids were extracted from ports starting at the 5.5-ft level. The total solids concentration was determined to be 133.6 g/ℓ. The depth to the interface was measured every 15 min for the first 13 hr and every day thereafter for approximately 15 days. Results are plotted in Figure E2.

Zone test

7. A series of zone settling tests were run at concentrations ranging from 43.5 to 157.1 g/ℓ. The depth to the interface was read every 15 min after loading the column. The total solids concentration was determined from six samples extracted from the ports of the column immediately after loading. Each zone test ran for approximately 5 or 6 hr. Two additional zone tests were run, without removing the chips, at 117.8- and 140.3-g/ℓ concentrations to determine the effect of removing the wood chips. The resulting settling velocities were similar to the zone tests of like concentrations without the wood chips. From the plots of the depth to interface (feet) versus time (hours), zone settling velocities were determined by the slope of the best-fit line through the data. These data are plotted in Figure E3. A solids loading curve was then determined as shown in Figure E4.

Flocculent test

8. The flocculent test concentration was run at 133.6 g/ℓ. Samples of the supernatant were extracted through each port above the interface with a syringe at different time intervals. Suspended solids concentrations were then determined on the extracted supernatant. Plots of the concentration profiles are shown in Figure E5. Plots of suspended solids remaining in the column as a function of retention time are shown in Figure E6.

Data Analysis and Results

Analysis technique

9. The behavior of Everett Harbor sediments at slurry concentrations equal to those expected for inflow to a confined site was governed by zone settling processes. The sediments exhibited a clear interface between settled material and clarified supernatant water as expected for saltwater conditions.

E3
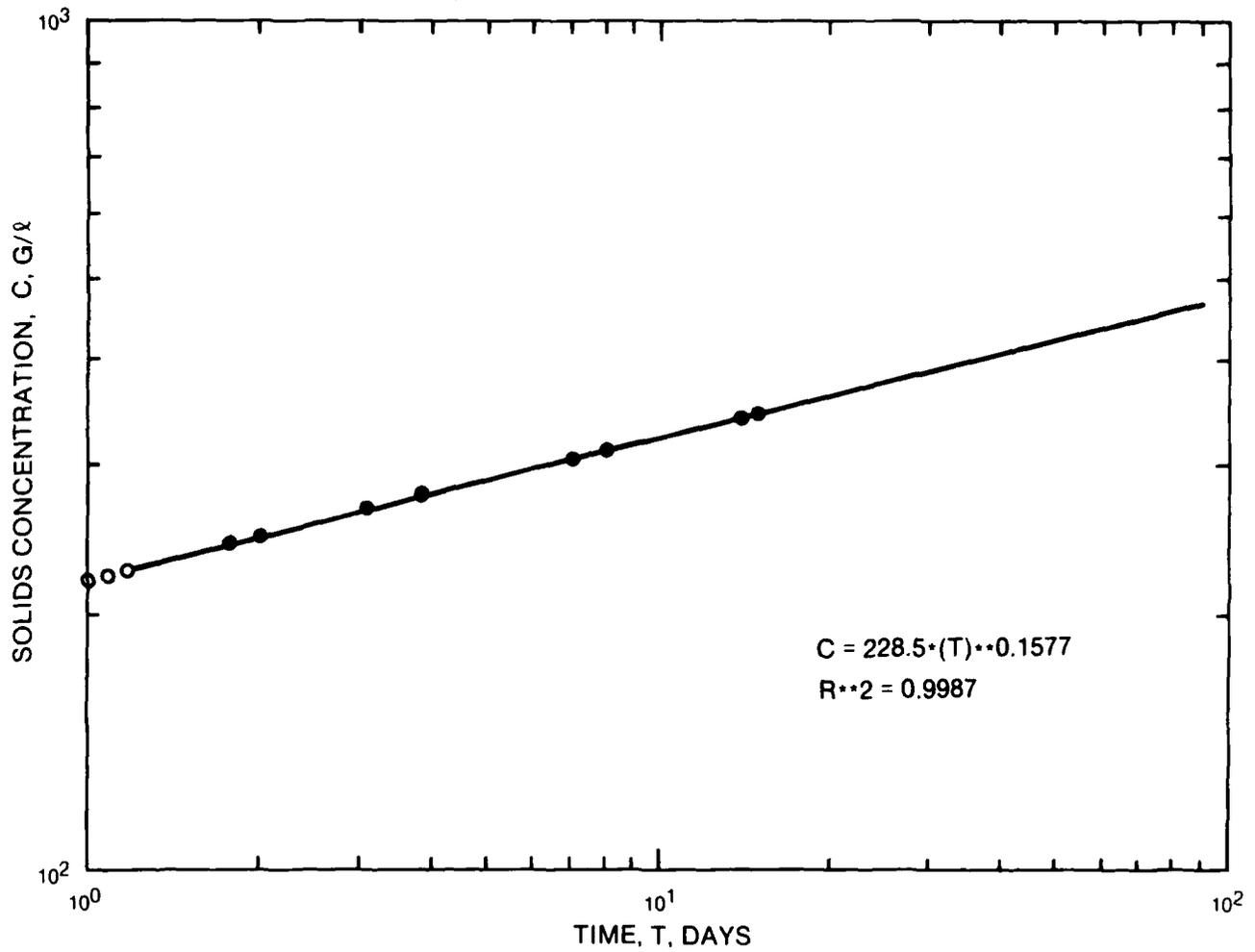
Figure E2. Compression settling test results

The settling test data were entered into the ADDAMS system (Hayes et al., in preparation). This system is a collection of computer programs to assist in the planning, design, and operation of dredging and dredged material disposal projects. ADDAMS was used to define confined disposal site geometry necessary for effective sedimentation for representative dredging conditions as described in Part V of the main text.
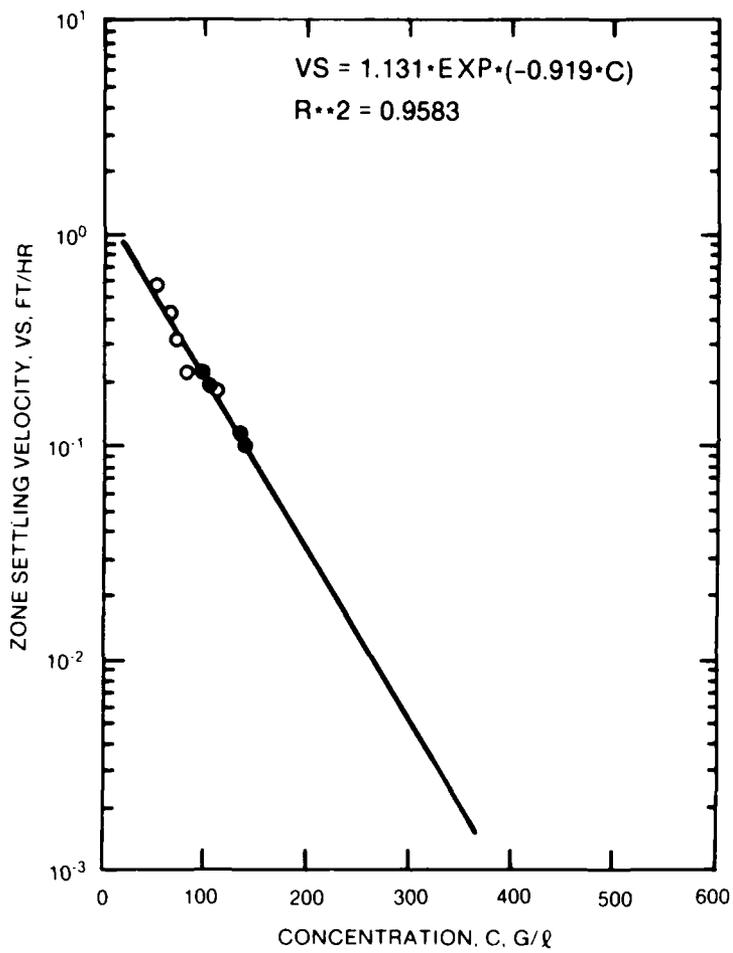
Figure E3. Zone settling test results
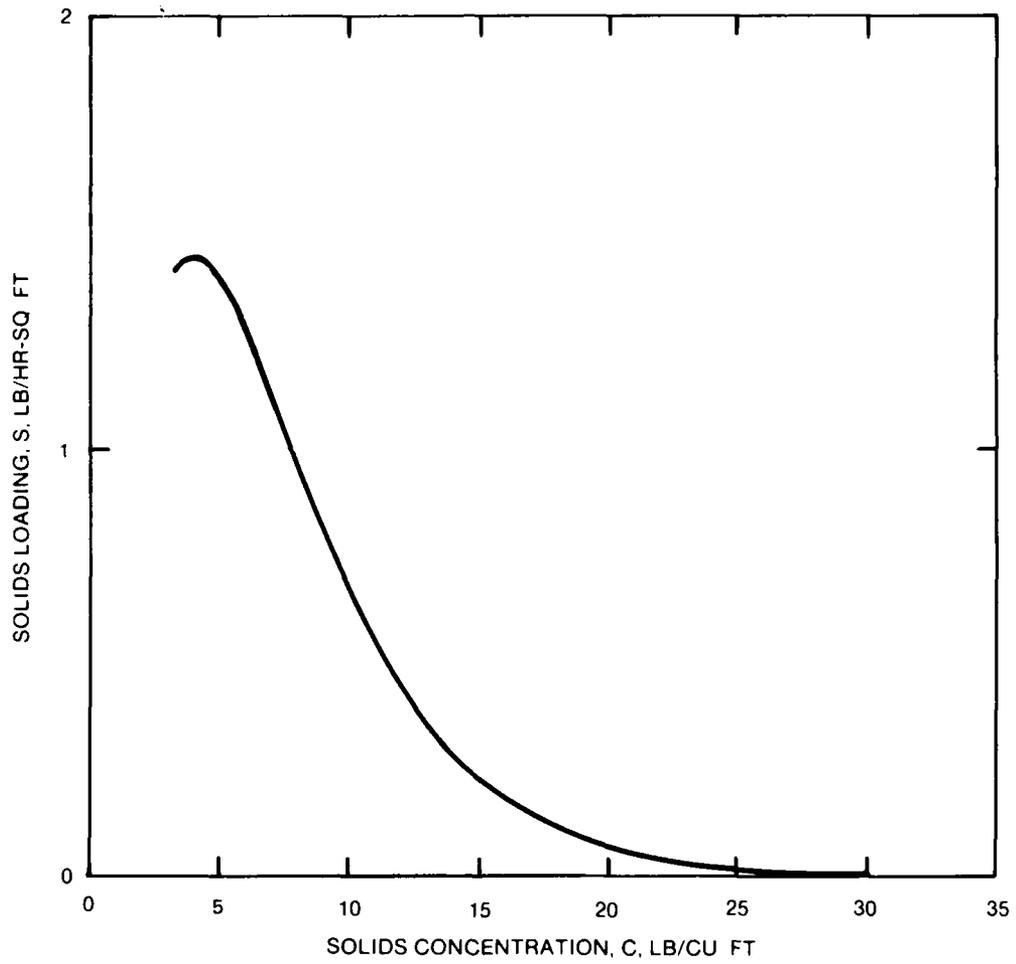
Figure E4. Solids loading curve for zone settling
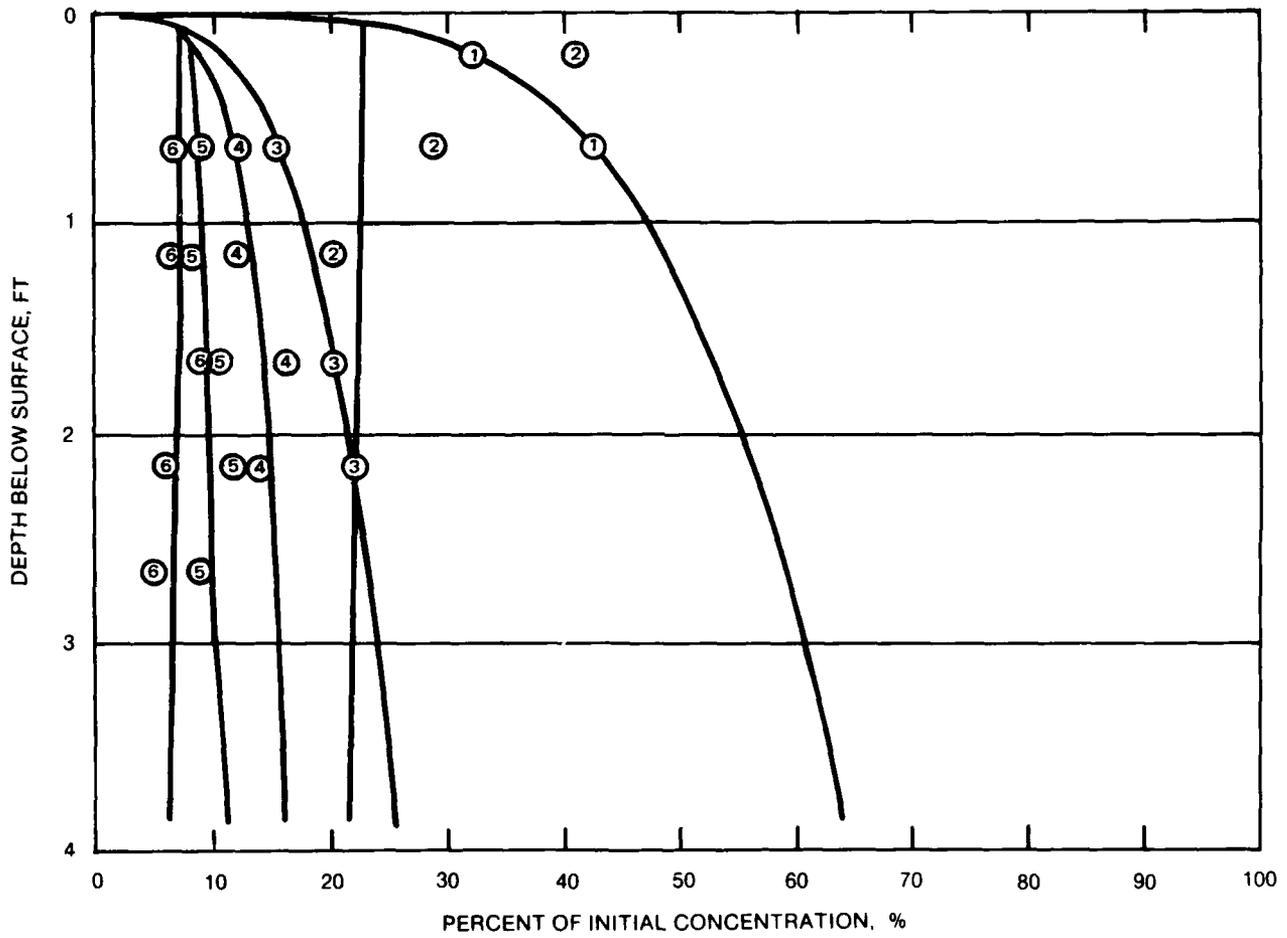
Figure E5. Flocculent settling test results
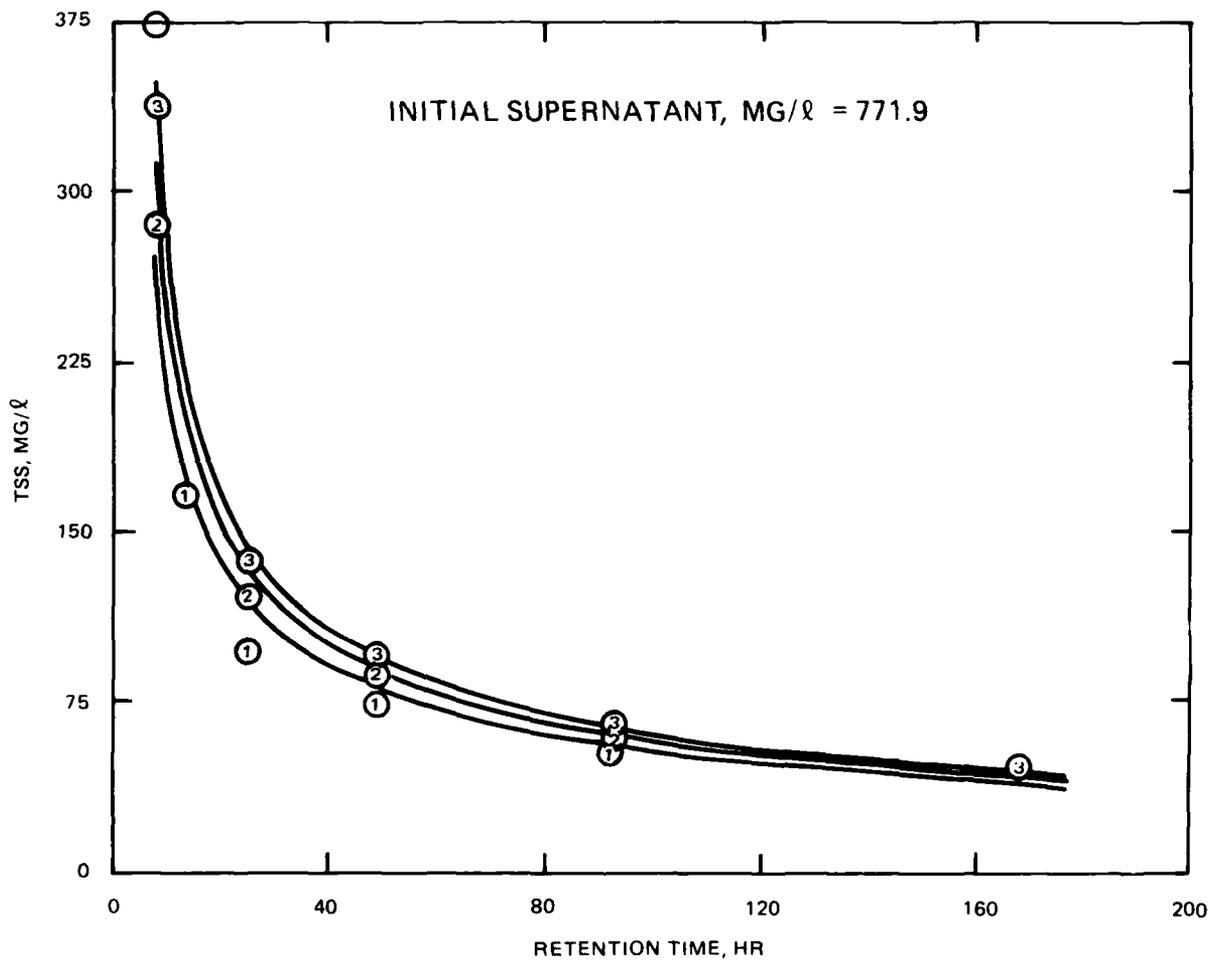
Figure E6. Suspended solids versus retention time for column test

References

Hayes, D. L. et al. "ADDAMS: Automated Dredging and Disposal Alternatives Management System," Instruction Report in preparation, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Palermo, M. R. 1985. "Interim Guidance for Predicting Quality of Effluent Discharged from Confined Dredged Material Disposal Areas," Environmental Effects of Dredging Programs Technical Notes EEDP-02-1 through 4, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

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

APPENDIX F: CHEMICAL CLARIFICATION TESTING

Testing Objectives

1. The objective of this testing was to screen selected polymers to determine their effectiveness in removing suspended solids from effluent or surface runoff waters generated by disposal of contaminated dredged material from the Everett Bay homeport project. The test data also can be used to predict the effect of chemical clarification on the effluent quality.

Experimental Procedures

Testing procedures

2. The testing was accomplished using procedures given in WES Technical Report D-83-2.* The procedures include those for screening polymers and for determining optimum dosage and mixing requirements for effective clarification.

Sample preparation

3. A dredged material slurry was prepared using the composite sediment sample obtained from the Everett Bay East Waterway. The total suspended solids concentration of the sediment sample was 426 g/ℓ. The slurry concentration was reduced by the addition of salt water obtained at the proposed dredging site. After thorough mixing, the total suspended solids of this diluted slurry was 125 g/ℓ. The slurry was allowed to settle overnight. The supernatant was then collected and stored in a 35-gal** drum. The supernatant was agitated at least 15 min before samples were extracted into 1,000-ml beakers for running the jar tests. The initial suspended solids of the resulting supernatant ranged between 313 and 1,200 mg/ℓ. The range of suspended solids concentrations was large because another batch of supernatant had to be generated due to the number of polymers tested.

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  * P. R. Schroeder. 1983. "Chemical Clarification Methods for Confined Dredged Material Disposal," Technical Report D-83-2, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
 ** A table of factors for converting non-SI units of measurement to SI (metric) units is presented on page 14 of the main text.

Preparation and screening of polymers

4. Three forms of polymers were evaluated: liquid, emulsion, and dry. Polymers were obtained from technical sales representatives of various polymer manufacturers and prepared for testing using their recommended procedures. Table F1 lists the polymers that were evaluated during this study.

Polymer screening

5. Polymers were selected for initial screening based on the recommendations of technical sales representatives from the respective polymer manufacturers. Two manufacturer's representatives (Allied and American Cyanamid) performed laboratory screening at WES using Everett Bay dredged materials. Others simply made recommendations based on their past experience. Those representatives performing laboratory screening also screened combinations of polymers; however, combinations of polymers proved ineffective for suspended solids removal. The technical sales representatives also provided cost information on each polymer.

Optimization of polymer dosage

6. As a result of the initial screening process, 13 cationic and three anionic polymers were selected for further evaluation in accordance with the testing procedures.* Suspended solids versus polymer dosage graphs were plotted for each polymer. Figures F1-F3 illustrate the typical form of these curves. The optimum dosage was calculated as the polymer dosage resulting in the minimum value for supernatant suspended solids.

Discussion of Results

Observations

7. Both qualitative and quantitative observations were made during each test procedure. Qualitative observations included floc size and capture of fines. Quantitative observations included initial suspended solids, polymer dosage, and final suspended solids. A summary of the more important observations is presented in Table F2.

Polymer selection

8. Based on the results of the optimization testing, an appropriate polymer was selected. The primary selection factors were effectiveness and

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* P. R. Schroeder, op. cit.

Table F1

Polymers Used for Optimization

Manufacturer	Product Code	Product Form	Type	Selection*
Allied Corp.	Clarifloc A-210	Emulsion	Anionic	
Allied Corp.	Clarifloc C-1020	Liquid	Cationic	
Allied Corp.	Clarifloc C-2020	Liquid	Cationic	x
Betz	1167L	Emulsion	Cationic	x
Betz	1165L	Emulsion	Cationic	x
Betz	1192	Liquid	Cationic	x
Betz	1160	Dry	Cationic	x
Calgon	Cat Floc T-2	Liquid	Cationic	x
Calgon	R-300	Dry	Anionic	x
Calgon	Cat Floc L	Liquid	Cationic	x
Calgon	WT-7736	Emulsion	Anionic	x
Cyanamid	Magnifloc 581	Liquid	Cationic	x
Cyanamid	Magnifloc 1223	Liquid	Cationic	x
Hercules	1018	Dry	Anionic	x
Hercules	815DE	Dry	Cationic	x
NALCO	7135	Liquid	Cationic	x
NALCO	7109	Liquid	Cationic	x
NALCO	603	Liquid	Cationic	x

* Polymers selected for optimization.
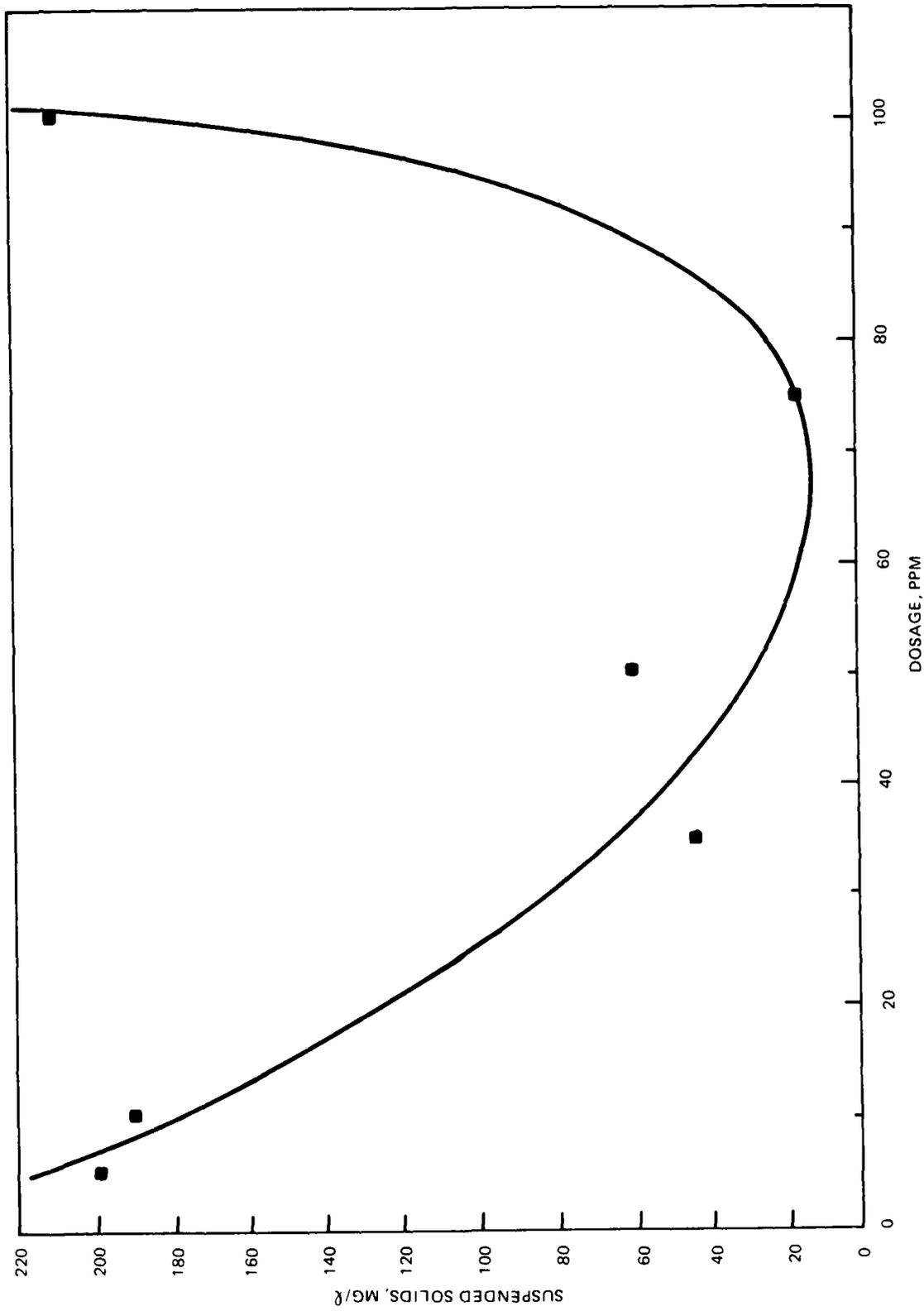
Figure F1. Plot of suspended solids versus dosage of Clarifloc C-2020

F4
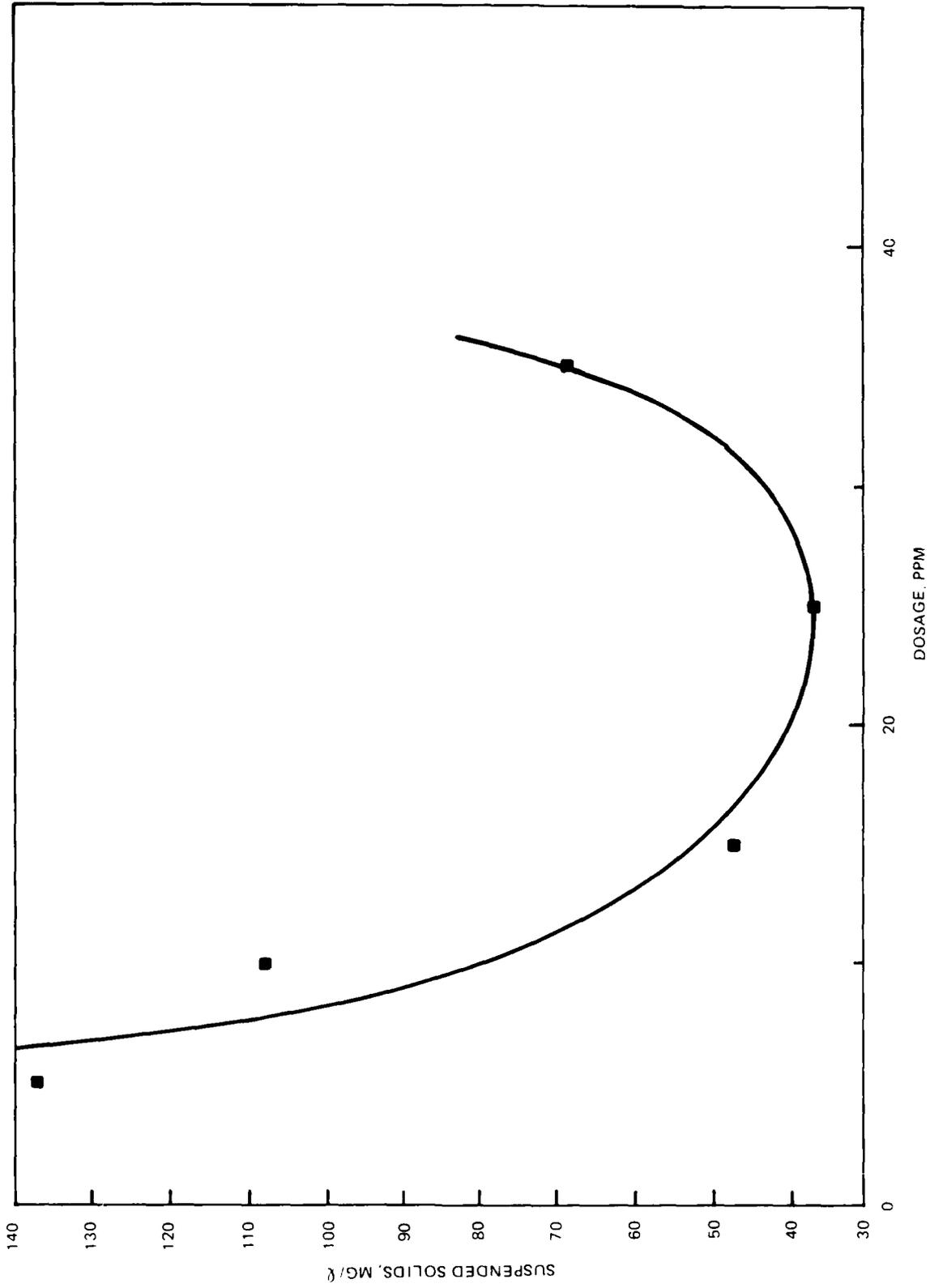
Figure F2. Plot of suspended solids versus dosage of Magnifloc 581
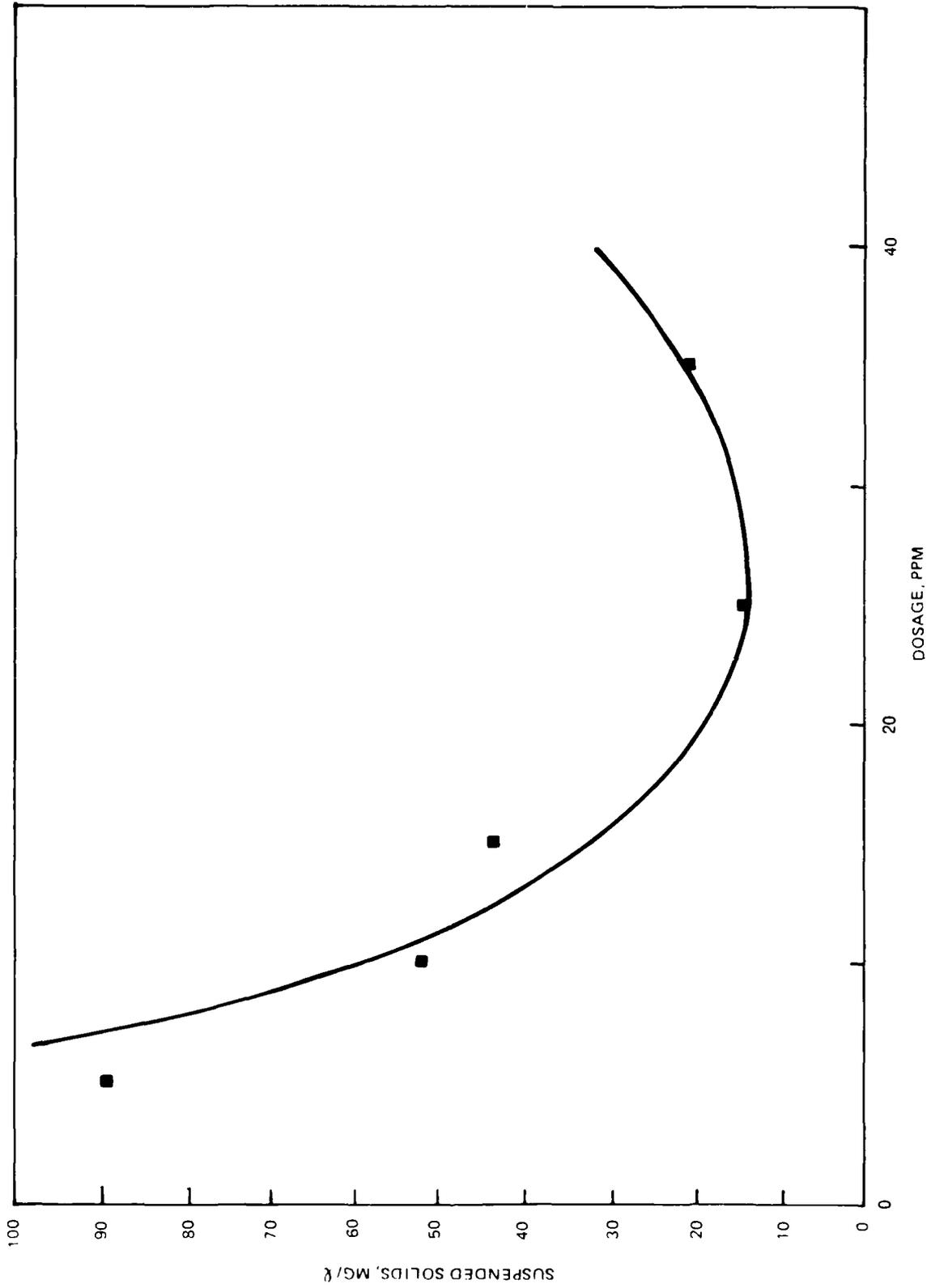
Figure F3. Plot of suspended solids versus dosage of Nalco 603

Table F2

Optimization Results*

Product	Optimum Dosage ppm	Observed Floc Size*	Observed Capture of Fines	Final Suspended Solids mg/ℓ	Percent Removal	Pounds of Polymer per Dry Ton Solids Removed	Cost ($)/ Dry Ton
BETZ 1192	50	P	Poor	43.5	87.3	335.05	169.20
CLAR. C-2020	75	M-	Fair	15.4	95.5	459.37	124.03
CAT FLOC L	50	P	Poor	13.2	96.8	249.17	236.71
CAT FLOC T-2	25	P+	Poor	42.3	89.8	134.30	202.79
MAGNIF. 581	25	M-	Poor	36.5	91.2	132.24	131.76
NALCO 7135	35	M	V. Fair	31.4	90.0	248.44	231.05
NALCO 7109	100	S+	Fair	22.7	92.7	689.32	158.54
NALCO 603	25	M+	V. Fair	14.8	96.1	138.61	97.02
BETZ 1160	3	M	Good	62.3	91.3	9.20	25.12
CALG. R-300	3	S	Good	46.0	93.5	8.98	24.26
HERCO. 1018	1	M+	V. Fair	59.6	95.1	1.74	2.44
HERCO. 815DE	3	M	V. Fair	50.0	95.9	5.19	7.30

* Symbols are defined as follows:
   P = pin-size floc.
   P+ = slightly larger than P.
   S = small floc.
   S+ = slightly larger than S.
   M = medium floc.
   M- = slightly smaller than M.
   M+ = slightly larger than M.
** Based on 100-percent removal.

costs. The emulsion polymers were found to be ineffective when applied to this material, and were eliminated from further consideration. Polymers that produced less than 85-percent removal of suspended solids were also eliminated from further consideration.

9. Following the initial technical effectiveness evaluation, the cost-effectiveness of the remaining polymers was evaluated. This was accomplished by calculating the estimated cost per ton of solids removed for each polymer. The results of this analysis are also summarized in Table F2.

10. The dry polymers had the lowest optimum dosage and cost. However, because of the complexity of dry polymer handling equipment, the liquid polymers are preferred. Therefore, polymers NALCO 603, Clarifloc C-2020, and Magnifloc 581 were selected as having the greatest potential for application to the Everett Bay Homeport Project.

Conclusions and Recommendations

Conclusions

11. Based on the results of this study, it is concluded that:

a. Chemical clarification using polymer addition is an effective method for improving the removal of suspended solids from site effluents generated by disposal of Everett Bay dredged material.

b. The dry polymer Hercofloc 1018 was found to be the most effective at low dosage rates; however, to obtain an adequate mixing of this polymer at the site may be very difficult and will require more handling equipment to be installed. Therefore, Hercofloc 1018 was not recommended.

c. Low-viscosity, highly cationic liquid polymers were found to be the most effective and the simplest to use for simulated Everett Bay site effluent.

d. Based on the analysis of cost per dry ton solids removed, NALCO 603 liquid cationic polymer appeared to be the most cost effective. The optimum dosage rate for NALCO 603 was determined to be approximately 25 mg/ℓ.

e. Magnifloc 581 and Clarifloc C-2020 can be used as alternate polymers should NALCO 603 be unavailable.

F8

Recommendations

12. Based on the results of this study, it is recommended that:

    a. If chemical clarification is required, NALCO 603 liquid polymer should be used as the coagulant.

    b. If NALCO 603 is not available, then Magnifloc 581 or Clarifloc C-2020 should be used.

    c. The overall cost of handling the Hercofloc 1018 should be compared to the NALCO 603. If the cost of Hercofloc 1018 is very low compared to the cost of NALCO 603, the complexibility of handling this polymer may be justified.

APPENDIX G: CONSOLIDATION TESTING

This appendix presents the results of a consolidation test conducted using the composite sample of Everett Harbor contaminated sediment. The test provides data for evaluation of filling and settlement rates for confined sites. The test results are applicable for evaluation of both intertidal and upland sites. The tests were conducted using standard odometers and procedures developed specially for soft sediments (see K. W. Cargill, 1983, "Procedures for Prediction of Consolidation in Soft, Fine-Grained Dredged Material," Technical Report D-83-1, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.).
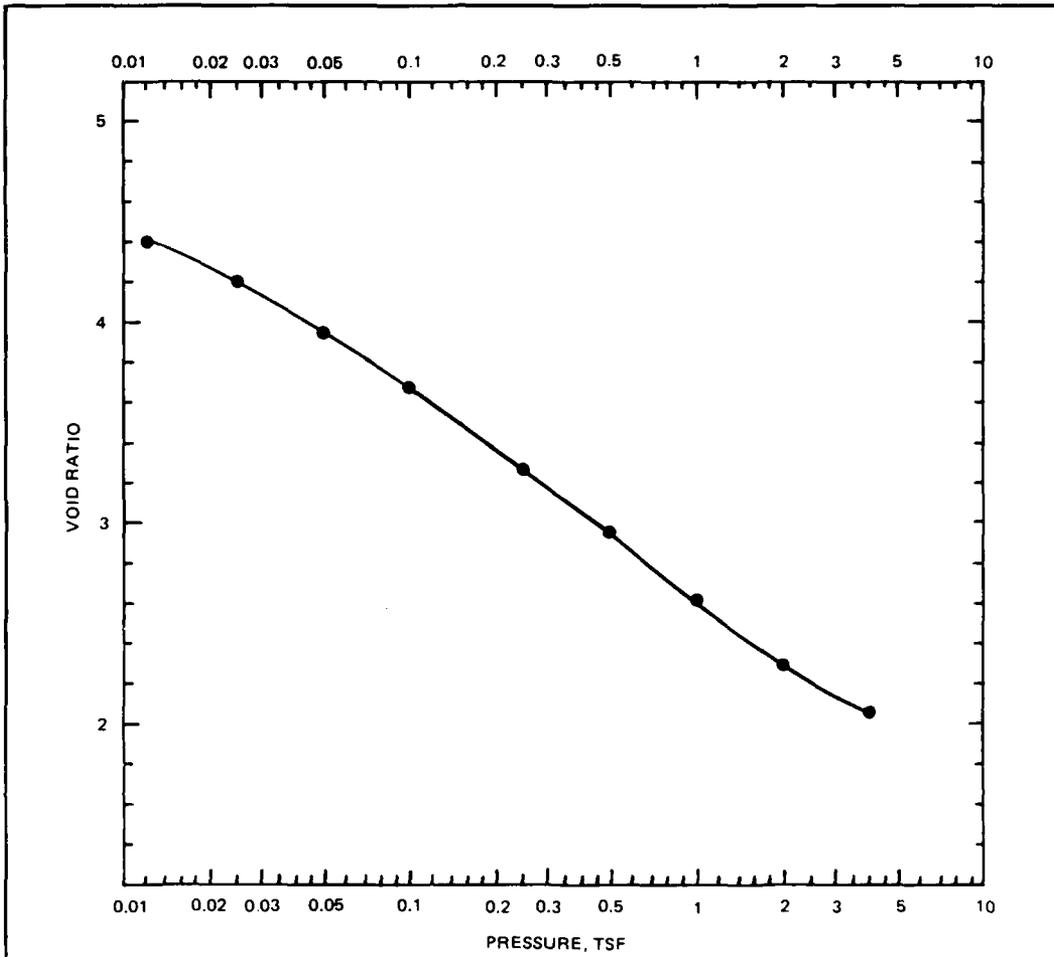
PRESSURE, TSF

				BEFORE TEST	AFTER TEST
OVERBURDEN PRESSURE, TSF			WATER CONTENT, %	165.1	86.2
PRECONSOL. PRESSURE, TSF			DRY DENSITY, PCF	30.6	56.1
COMPRESSION INDEX			SATURATION, %	98.7	100*
TYPE SPECIMEN	UNDISTURBED		VOID RATIO	4.517	2.003
DIA. IN 4.44	HT. IN 1.235		BACK PRESSURE, TSF		
CLASSIFICATION ORGANIC SILT (OH), GRAYISH BLACK					
LL 116	PL 57	PI 59	PROJECT EVERETT BAY, WA		
GS 2.70 (EST)	D_{10}				
REMARKS			BORING NO. -		SAMPLE NO. -
			DEPTH/ELEV -		DATE 11 FEB 86
			CONSOLIDATION TEST REPORT		

SHEET   OF

0 012 TSF
± SPECIMEN INUNDATED AT 30 MIN

0 025 TSF

0 05 TSF

TIME MINUTES

G3

1 TSF

2 - TSF

4 TSF

TIME MINUTES

G5
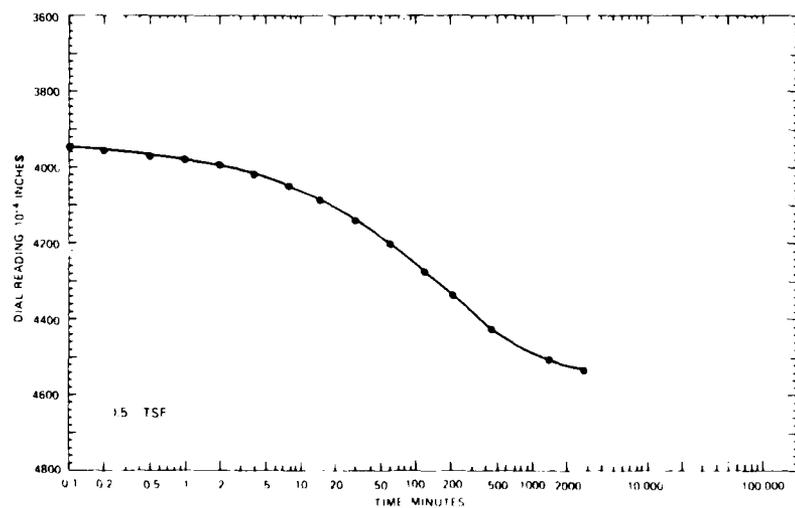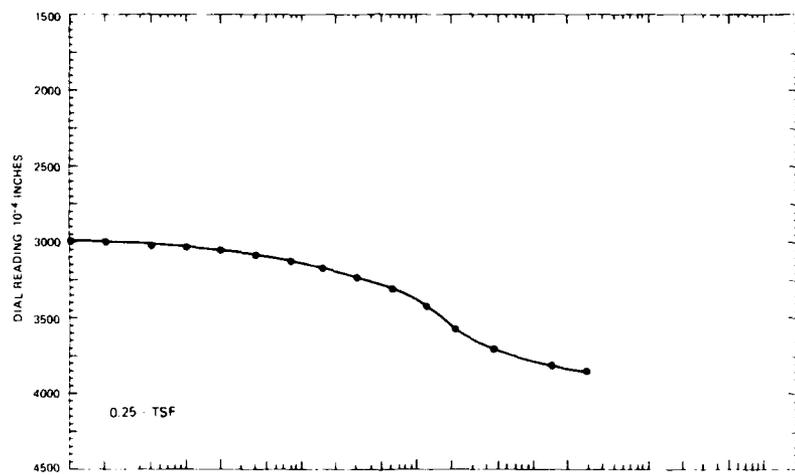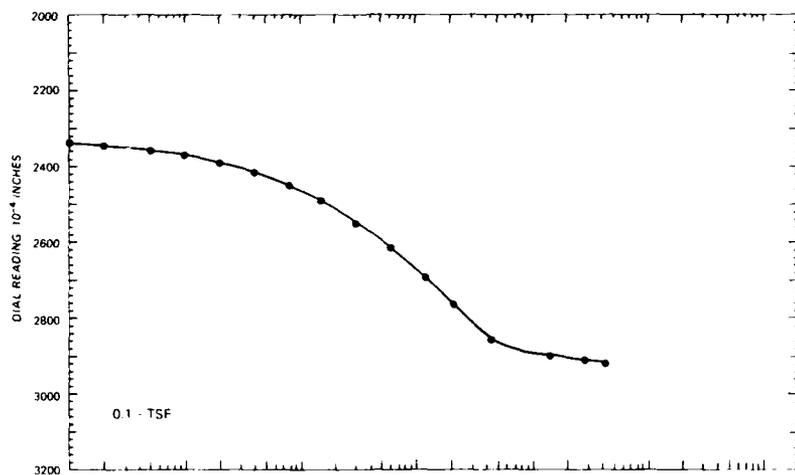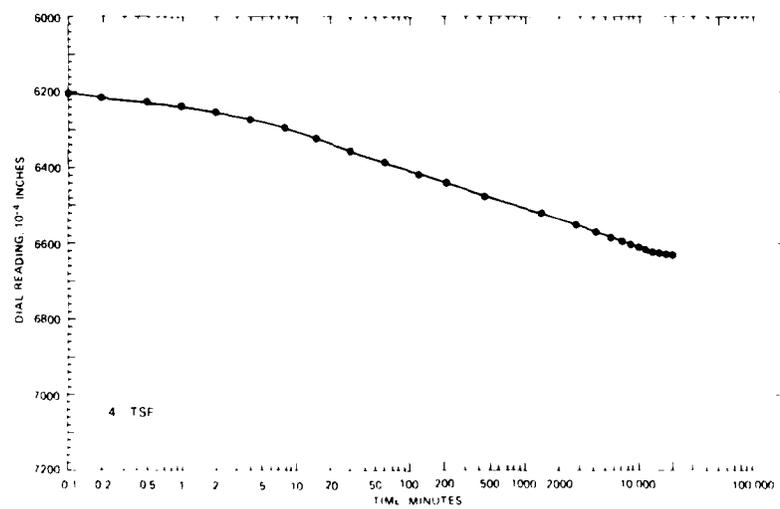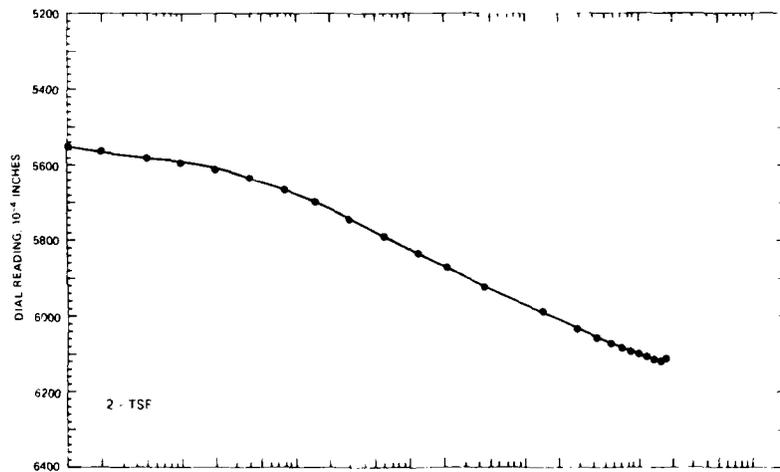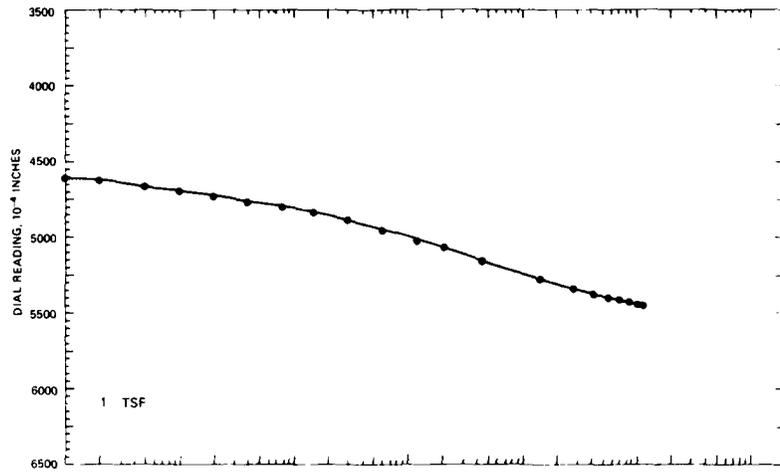
APPENDIX H: PROCEDURES FOR EVALUATION OF SOLIDIFICATION/STABILIZATION TECHNOLOGY

Introduction

1. When contaminated dredged material with a potential for leaching is disposed in an upland site, the site must be planned to prevent ground-water pollution. Current strategies for minimizing ground-water pollution include proper site selection, dewatering to minimize leachate production, lining of bottom and sides to prevent leakage and seepage, capping to minimize infiltration and thereby leachate production, and leachate collection and treatment. Economic considerations and tough environmental constraints for disposal are providing initiative for developing innovative approaches to upland disposal of contaminated dredged material. With proper development, new strategies such as solidification/stabilization of dredged material to prevent or retard leaching and the use of clean dredged material to adsorb contaminants in leachate draining from solidified/stabilized dredged material could provide the disposal technology needed to contain and immobilize contaminants in an upland site.

2. Solidification/stabilization is a state-of-the-art technology for the treatment and disposal of contaminated materials. The technology has been applied in Japan to bottom sediments containing toxic substances (Kita and Kubo 1983, Nakamura 1983, Otsuki and Shima 1984) and in the United States to industrial wastes (Pojasek 1979; Malone, Jones, and Larson 1980). Tittlebaum et al. (1985) reviewed the current technology and its potential application to wastes high in organic contaminants. Because of sediment contamination in parts of Everett Bay, innovative contaminant immobilization techniques may be needed to satisfy site-specific environmental constraints for disposal. Experiences in Japan with bottom sediments and in the United States with industrial sludges indicate that solidification/stabilization is a promising contaminant immobilization technology for materials that show a potential for leaching.

3. Solidification is the process of eliminating the free water in a semisolid by hydration with a setting agent(s). Typical setting agents include portland cement, lime, fly ash, kiln dust, slag, and combinations of these materials. Stabilization can be both physical and chemical. Physical

stabilization refers to improved engineering properties such as bearing capacity and trafficability. Chemical stabilization is the alteration of the chemical form of the contaminants to make them less soluble and/or less leachable. Solidification is a physical stabilization process that usually, but not always, provides some chemical stabilization.

4. Since physical stabilization and solidification are equivalent in terms of the end products, the terms are often used interchangeably, with solidification being the more commonly used term. The literature also uses the terms "chemical stabilization" and "stabilization" interchangeably, albeit not without some confusion.

5. Solidification (physical stabilization) immobilizes contaminants through alteration of the physical character of the material. The development of structure immobilizes contaminated solids (i.e., the solid mass is dimensionally stable), and the solids do not move. Since most of the contaminants in dredged material are tightly bound to the sediment phase, solidification is an important immobilizing mechanism (Kita and Kubo 1983). Solidification also reduces the accessibility of water to the contaminated solids within the cemented matrix. Water accessibility to the contaminated solids is an important factor because it partially determines the rate at which contaminants are leached.

6. Solidification/stabilization processes are usually formulated to minimize the solubility of metals by controlling pH and alkalinity. Additional metal immobilization can be obtained by modifying the process to include chemisorption (Myers 1985). Because anions are typically more difficult to bind in insoluble compounds, most solidification/stabilization processes rely on microencapsulation to immobilize anions. Some vendors of solidification/stabilization technology claim to be able to immobilize organic contaminants. There is as yet, however, no scientific evidence that stabilization of organic contaminants against aqueous leaching occurs using cement- and pozzolan-based systems (Tittlebaum et al. 1985). Practically no published information exists on the aqueous leaching of organic contaminants from solidified/stabilized materials. Further, the state of the art for process design is primarily empirical. Thus, a process formulation cannot be designed on the basis of chemical characterization of the material to be solidified/ stabilized alone. It is, therefore, necessary to conduct laboratory leach tests to evaluate chemical stabilization effectiveness. Although chemical

stabilization has to be evaluated on a case-by-case basis, isolation of contaminated dredged material solids in a cemented matrix appears to be a promising technology for significantly reducing or eliminating the release of contaminants, particularly metals, from dredged material.

7. The technical feasibility of reducing contaminant mobility in Everett Bay sediment by solidification/stabilization was investigated in a series of laboratory-scale applications of selected solidification/ stabilization processes. The processes evaluated were portland cement, portland cement with Firmix (a proprietary additive), Firmix, and lime with fly ash. All of these processes are commercially available.

Materials and Methods

Materials

8. Sediment acquisition, mixing, and transportation procedures have been previously described. The sediment was stored at 4° C until used. Prior to use, the contents of the sediment container (55-gal* drum) were mixed and sieved through a 1/4-in. sieve to remove large wood chips that were present in the sediment. No other processing (e.g., dewatering) was applied prior to applying the various solidification/stabilization processes. Type I portland cement was used in the processes involving portland cement, class C fly ash was used in the processes involving fly ash, and hydrated lime was used in the lime with fly ash process. The proprietary additive, Firmix, is a solidification agent that is commercially available. Firmix was obtained from Trident Engineering, Baltimore, Md.

Laboratory processing

9. The process additives were mixed with sediment in a Hobart C-100 mixer (2.5-gal capacity) for 5 min per additive. After mixing, the freshly prepared solidified sediment was cast in 2-in. cube molds for unconfined compressive strength testing and standard compaction molds for chemical leach testing. The samples were stored at 98-percent relative humidity and 23° C until tested. A standard cure time of 28 days was used in all of the testing unless otherwise noted.

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* A table of factors for converting non-SI units of measurement to SI (metric) units is presented on page 14 of the main text.

Experimental design

10. Each process was applied in three formulations. The formulations for each process differed in respect to the dosage of setting agent used, not the types of agents used. By testing different processes in varying formulations, data were obtained for making comparisons among processes and process formulations.

11. Unconfined compressive strength was the key test for physical stabilization; the serial, graded batch leach test was the key test for chemical stabilization. Leach tests and unconfined compressive strength tests were conducted on each process formulation.

Physical properties tests

12. Unconfined compressive strength (UCS) was determined according to the ASTM Compressive Strength of Hydraulic Cement Mortars (C-109) procedure. Three replicates were run for each determination at 7-, 14-, 21-, and 28-day cure time intervals. In addition, unconfined compressive strength at 60- and 90-day cure times was determined for some formulations.

Serial, graded batch leach tests

13. **Background.** The serial, graded batch leach test is a simplification of the sequential batch leach test described in Appendix C. In the serial, graded procedure, a sample is leached one time at several liquid-solids ratios (Houle and Long 1980). A table of solid phase and aqueous phase concentrations is developed from analyses of the leachates produced. These data are plotted to produce a desorption isotherm. This procedure is simpler than the sequential leach procedure because the mass of solids being leached has to be measured and handled only once.

14. From the desorption isotherm, contaminant-specific coefficients can be obtained that describe the interphase transfer of contaminant from the solid phase to the aqueous phase. The interpretation of data from serial, graded batch leach tests is similar to the interpretation previously described in Appendix C for data from sequential batch leach tests. Of particular importance is Equation C5 (Equation H1 below) and Figure C5 of Appendix C.

$$q_t = K_d C + q_r \qquad (H1)$$

Equation H1 assumes that a fraction of the solid phase contaminant concentration is resistant to leaching and the solid to liquid phase transfer of the leachable fraction is governed by a reversible process. In this model, the relationship between the solid phase concentration, q, and the aqueous phase concentration, C, is linear. Two parameters describe the relationship, a distribution coefficient, K_d, that relates leachable solid phase concentration to aqueous phase concentration and the solid phase concentration resistant to leaching, q_r. Similar models have been used in various studies on contaminant mobility in sediments (Di Toro and Horzempa 1982, Jaffe and Ferrara 1983). If the desorption isotherms obtained from leach tests are not linear or do not provide a well-defined relationship between solid and aqueous phase concentrations, other models and approaches to interpreting the data may be necessary.

15. The serial, graded batch leach procedure assumes that the liquid-solids ratio does not affect the chemistry of the leaching process, i.e., the distribution coefficient is not dependent on liquid-solids ratio. The literature indicates that this assumption is probably not correct for untreated sediment although the reason for this is not entirely clear (Voice, Rice, and Weber 1983; Di Toro et al. 1986). For solidified/stabilized sediment, changes in the chemistry of the aqueous phase with varying liquid-solids ratio probably have a more profound effect on interphase contaminant transfer than changes in the concentration of solids. Specifically, if pH varies significantly, the solubility of metals will vary. The excess alkalinity of the solidification reagents, however, tends to stabilize pH.

16. <u>Chemical leach tests.</u> Serial, graded batch leach tests were run on samples taken from the center of the 4-in.-diam specimens cast in compaction molds. The 4-in. specimens were broken apart to obtain the samples for chemical leach testing. The samples were ground on a Brinkman centrifugal grinding mill to pass a 0.5-mm screen before leach testing. The leach procedure consisted of contacting solidified sediment samples with distilled-deionized water on a mechanical shaker for 24 hr in liquid-solids ratios as follows: 100 ml:50 g, 100 ml:20 g, 100 ml:10 g, 100 ml:5 g, and 100 ml:1 g. The extractions were run in triplicate in 250-ml polyethylene bottles laid in the horizontal position. After shaking, the mixtures were filtered through 0.45-μ membrane filters and analyzed for arsenic, cadmium, chromium, lead,

zinc, and organic carbon. Blanks were prepared by carrying deionized-
distilled water through the same shaking and filtration procedures. Chemical
analysis procedures are described in Appendix C.

17. The chemical leach data were reduced to tables of solid and aqueous
phase concentrations using the calculations described below. The solid phase
contaminant concentration after leaching is given by:

Solidified sediment contaminant concentration after leaching	=	Solidified sediment contaminant concentration before leaching	−	Mass of contaminant leached / Mass solidified sediment leached

or

$$q = q_o - C(V/M) \qquad (H2)$$

where

q = total contaminant concentration in the solid phase after
leaching, mg/kg

q_o = initial contaminant concentration in the solid phase, mg/kg

C = contaminant concentration in the leachate, mg/ℓ

V = volume of aqueous phase (leachate), ℓ

M = mass of solidified sediment leached, kg

Equation H2 relates to a single contaminant. Since the liquid-solids ratio
(L/S) is given by V/M, Equation H2 can be written as

$$q = q_o - C(L/S)$$

Equation H2 was used to calculate the solid phase concentration, q, corre-
sponding to the aqueous phase concentration determined by chemical analysis
for the L/S used. Since all the tests used 100 ml of distilled-deionized
water, the L/S is 100 ml divided by the mass of solidified/stabilized sediment
leached in grams.

18. The initial solid phase concentration, q_o, for each contaminant
is given by the following equation

$$q_o = \frac{S_x}{(1 + w)(1 + R)} \tag{H3}$$

where

S_x = contaminant concentration in the sediment before solidification, mg/kg (dry weight basis)

w = moisture content of the wet sediment, kg water/kg sediment solids

R = dosage of solidification/stabilization reagents, kg reagents/kg wet sediment processed

The moisture content of the sediment was 1.572 kg/kg, and values for S_x are given in Table C1, Appendix C, of this report.

Results

Physical properties

19. The UCS for the portland cement, portland cement with Firmix, Firmix, and lime with fly ash processes was measured at cure times of 7, 14, 21, and 28 days. These data are presented in Tables H1 through H4, and are plotted in Figures H1 through H4. The points in the figures are averages of three replicates.

20. The UCS data showed, as expected, that the higher the additive dosage, the higher the strength of the solidified product. For example, the 28-day UCS for the 0.05 portland cement:1 sediment weight ratio was 35 psi; for the 0.1:1 weight ratio of portland cement to sediment the 28-day UCS was 71 psi, and the 28-day UCS for the formulation using a 0.2:1 weight ratio of portland cement to sediment was 226 psi. The gain in UCS with cure time for the various portland cement formulations is shown in Figure H1. For the portland cement with Firmix process, the optimum formulation for strength development was the formulation using equal proportions of portland cement and Firmix. This is shown in Figure H2. As shown in Figure H3, a higher dosage of fly ash in the fly ash with lime process formulation produced a stronger product. The 28-day value for the 0.5 fly ash:0.1 lime:1.0 sediment formulation in Figure H3 is questionable. One of the three replicates for this point is in agreement with the data for the other points on the strength versus cure

H7

Table H1

Comparison of Unconfined Compressive Strengths for Various
Portland Cement/Sediment Formulations

Formulation*	Unconfined Compressive Strength, psi, by Cure Time, days				
	7	14	21	28	60
0.05/1.0	16	29	32*	35	24
0.1/1.0	44	64	70	71	74
0.2/1.0	150	179	188	226	210

* Portland cement/sediment.

Table H2

Comparison of Unconfined Compressive Strengths for Various
Portland Cement/Firmix/Sediment Formulations

Formulation*	Unconfined Compressive Strength, psi, by Cure Time, days				
	7	14	21	28	60
0.1/0.2/1.0	225	359	484	507	536
0.15/0.15/1.0	361	472	562	605	711
0.2/0.1/1.0	242	341	**	385	485

 * Portland cement/Firmix/sediment.
** No data.

Table H3

Comparison of Unconfined Compressive Strengths for Various

Firmix/Sediment Formulations

Formulation*	Unconfined Compressive Strength, psi, by Cure Time, days					
	7	14	21	28	60	90
0.4/1.0	4	5	3	7	93	565
0.5/1.0	5	21	28	53	111	560
0.6/1.0	7	22	38	274	1,153	1,176

* Firmix/sediment.

Table H4

Comparison of Unconfined Compressive Strengths for Various

Type C Fly Ash/Lime/Sediment Formulations

Formulation*	Unconfined Compressive Strength, psi, by Cure Time, days				
	7	14	21	28	60
0.3/0.1/1.0	13	17	16	26	49
0.4/0.1/1.0	23	38	36	51	72
0.5/0.1/1.0	35	48	57	199	75

* Fly ash/lime/sediment.
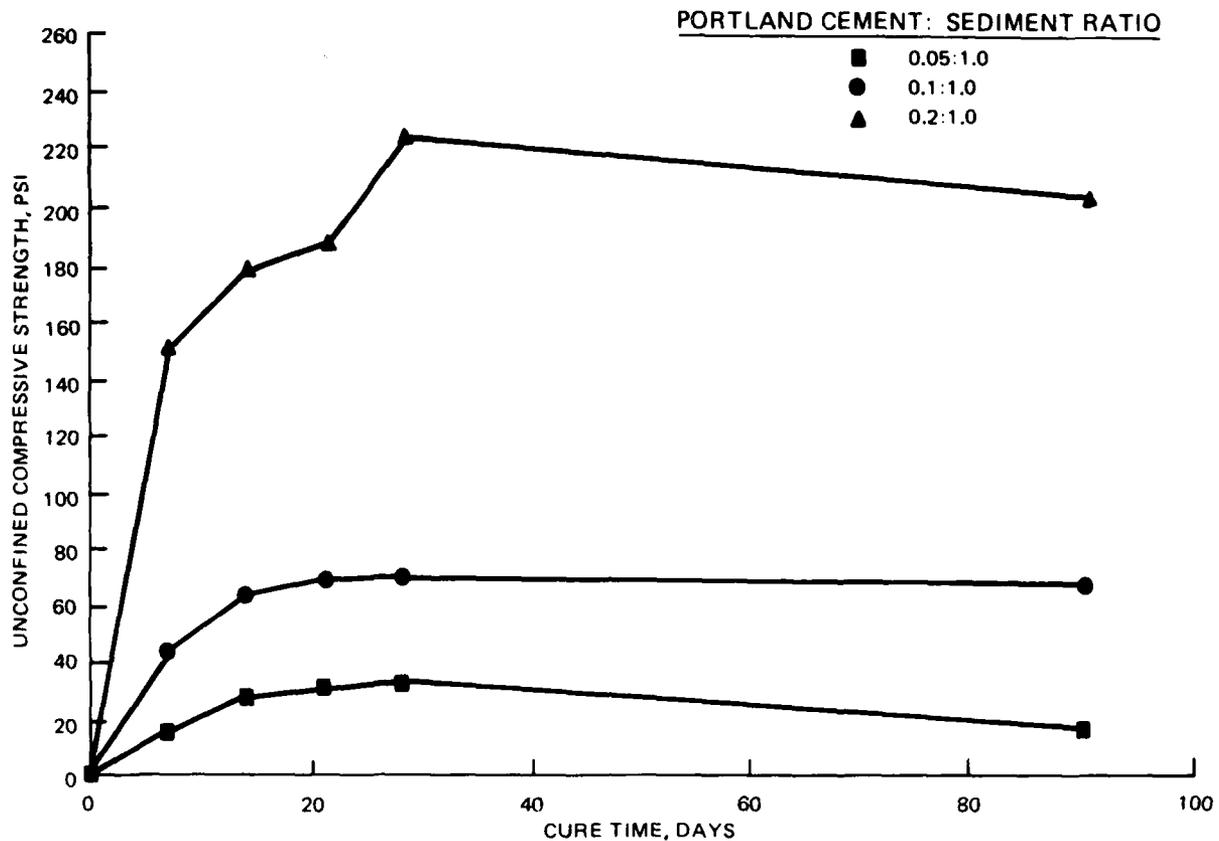
Figure H1.   Unconfined compressive strength, portland cement process

time curve.   The other two replicates were extremely high relative to the
other data for the lime with fly ash process, possibly due to an instrument
malfunction during UCS testing.

21.   The fly ash/lime process produced the product with the lowest UCS
at 28 days, and the portland cement with Firmix process produced the product
with the highest 28-day UCS.   The Firmix process produced the highest 90-day
strength of all the processes tested (1,176 psi).

22.   The steady gain in strength with cure time recorded for all of the
process formulations, Figures H1-H4, showed that the sediment solidified
despite the potential for interference from the various contaminants in the
sediment.   If the setting reactions responsible for solidification were not
occurring, the products would not gain strength as they cured.   This is a
significant finding in light of what is known about the potential for inter-
ference (Jones et al. 1985).

23.   There is, however, evidence of retardation in set time for the
Firmix formulations.   The strength versus cure time curves in Figure H4 showed
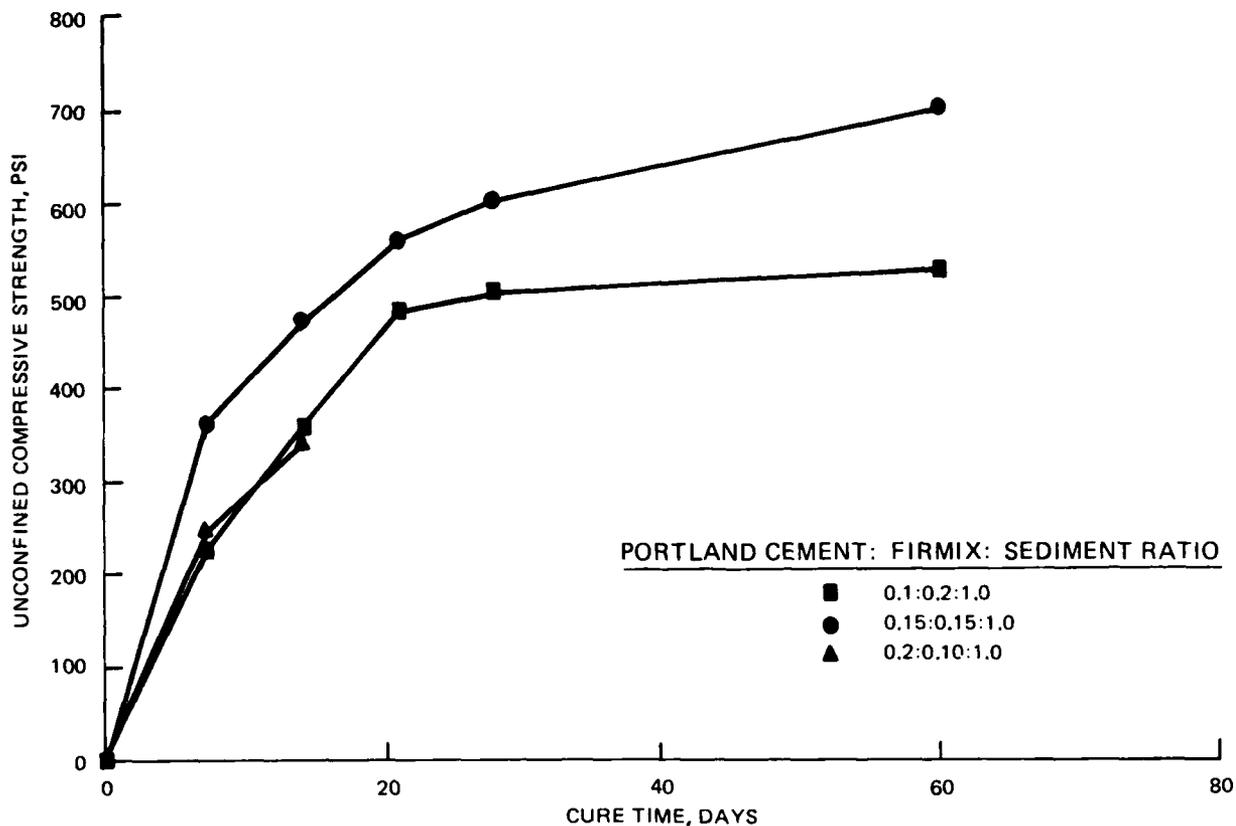
Figure H2. Unconfined compressive strength, portand cement/Firmix process

that strength is continuing to develop beyond 28 days. Firmix usually reaches maximum strength in about 30 days with clean sediments.*

24. The range in product strengths, 35 to 1,176 psi, is indicative of the versatility and flexibility of solidification as a treatment process for immobilizing the contaminated solids in Everett Bay sediments. For comparison, the unconfined compressive strengths of concrete clays of various consistency and solidified industrial sludge are shown in Table H5. Solidified/ stabilized Everett Bay sediments had strengths that were above the range normally associated with hard clay and solidified industrial sludge and below the range normally associated with low-strength concrete.

Chemical leach data

25. Analysis of the blanks. Analysis of the blanks analyzed during the chemical leach tests is summarized in Table H6, which lists the detection

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* Personal Communication, 1986, Mitchell Kaplan, Trident Engineering, Baltimore, Md.
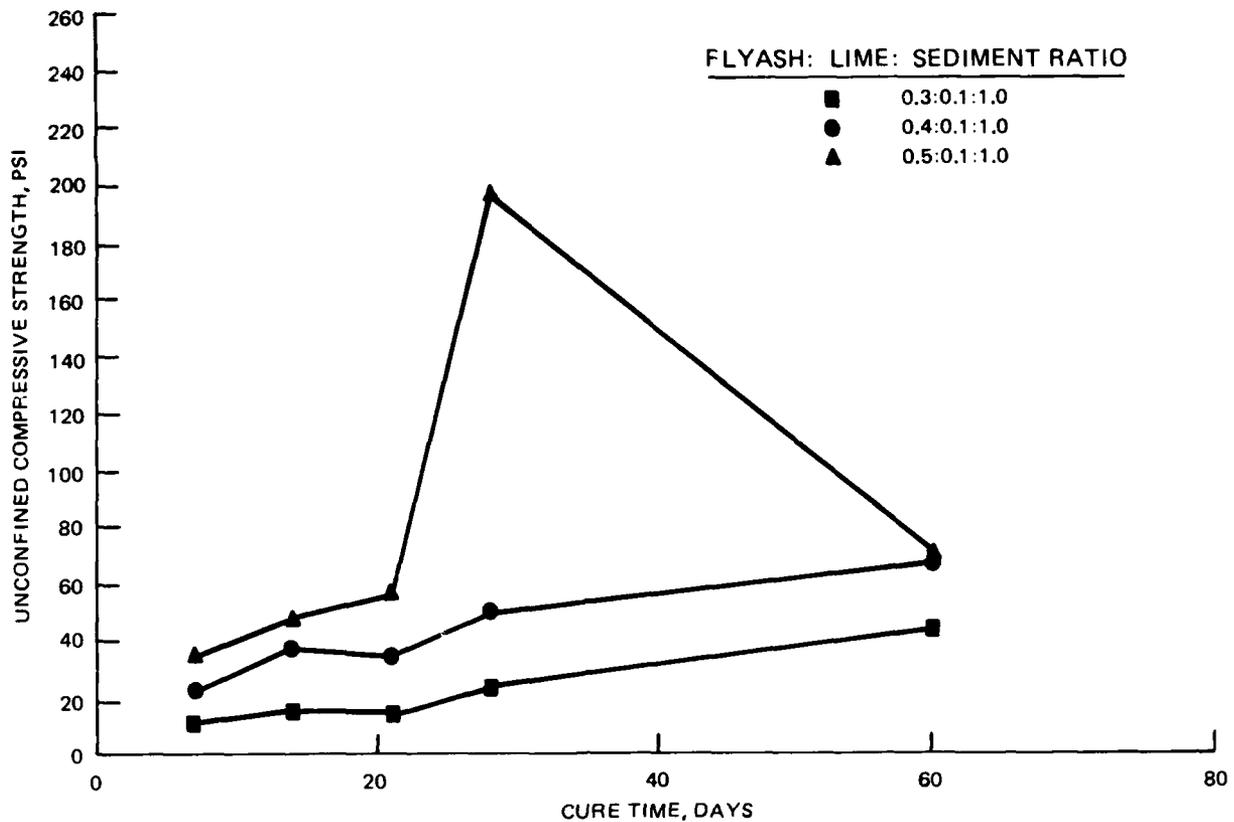
Figure H3. Unconfined compressive strength, Firmix process

limits, range, mean, standard deviation, and 95-percent confidence interval for arsenic, cadmium, chromium, lead, zinc, and dissolved organic carbon (DOC). The blanks were generally near or below the chemical analytical detection limits. Arsenic, zinc, and DOC were below the detection limit for all the blanks. Cadmium, chromium, and lead were above the detection limits in the majority of the blanks. Leachate samples with contaminant concentrations within the 95-percent confidence interval or concentrations below the detection limits were considered not distinguishable from the blanks and were assigned contaminant concentrations equal to the value for the 95-percent confidence interval. Chromium had two blank concentrations that were extremely high, thus driving the value for the 95-percent confidence interval up. The high values were 0.014 and 0.021 mg/ℓ, and could be considered outliers. They were not discarded from the data set, however, because an explanation for these high values could not be reconstructed from an examination of the laboratory notebooks. The 95-percent confidence interval values for cadmium and lead were not affected by data that could be outliers. When
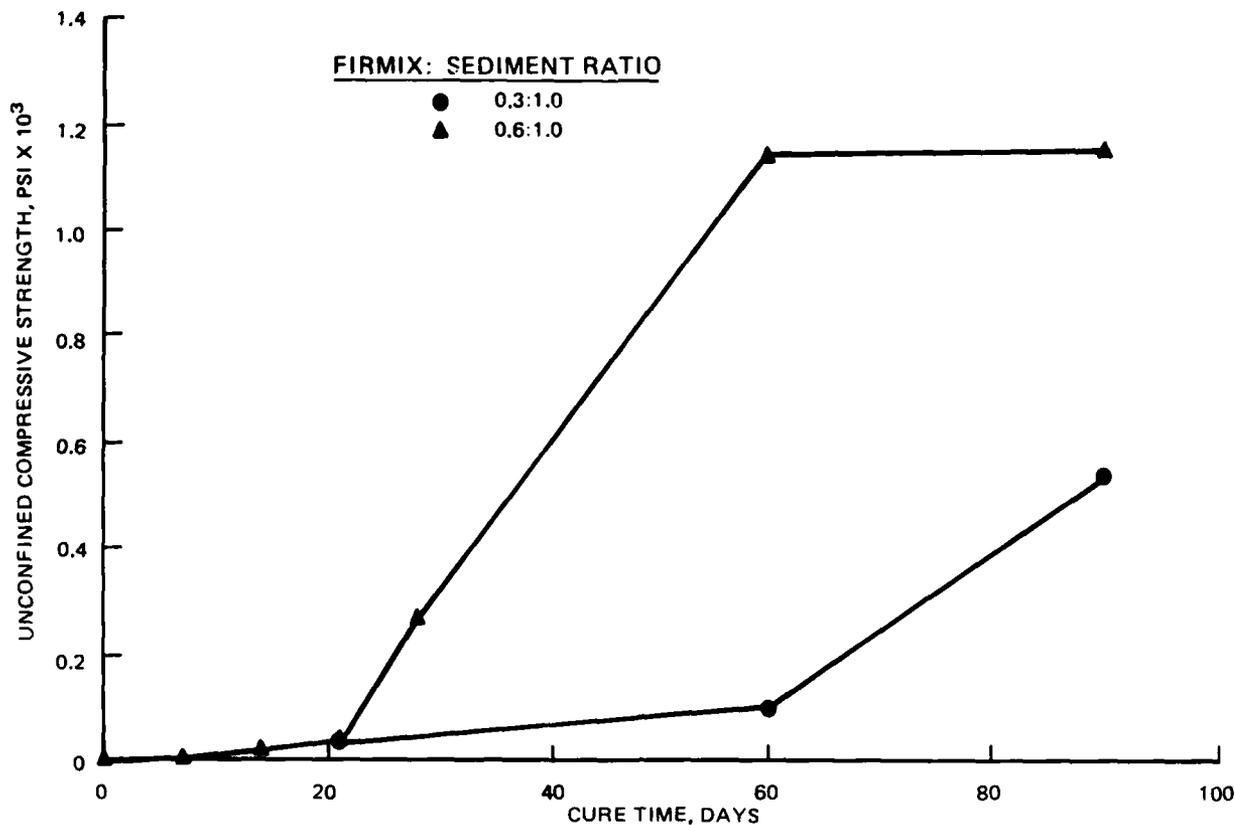
Figure H4. Unconfined compressive strength, lime/fly ash process

determining the various statistical parameters, concentration values less than
the detection limit were given a value equal to one half the detection limit.

26. Desorption isotherm data. The results from the serial, graded
batch leach tests conducted on portland cement, lime with fly ash, Firmix, and
portland cement with Firmix solidified/stabilized Everett Bay sediments are
presented in Tables H7 through H18. The tables are organized by process and
process formulation. Each table contains data for one process formulation.
The first column in each table lists the nominal liquid-solids ratio. The
mass of solidified sediment leached with 100 ml of water is presented in the
second column. The remaining entries in each table list aqueous phase con-
taminant concentration, C , and the corresponding solid phase concentration,
q , for five metals and organic carbon. Differences in solid phase concen-
tration for identical aqueous phase concentrations at the same liquid-solids
ratio reflect slight differences in the amount of solids weighed for leach

H13

Table H5

Unconfined Compressive Strengths of Various Materials

Material	Type	Unconfined Compressive Strength psi
Clay	Very soft	<3.5
	Soft	3.5-7
	Medium	7-14
	Stiff	14-28
	Hard	28-56
	Very hard	56
Concrete	Low strength	2,000
	Medium strength	5,000
Soil-like	FGD sludge	23-43
solidified waste	Electroplating sludge	32
(Bartos and	NI/CAD battery sludge	8
Palermo 1977)	Brine sludge	22
	CA fluoride sludge	25

testing. The aqueous phase concentration, C , refers to the contaminant concentration in the filtered (0.45-μ) leachate.

27. Desorption isotherms were plotted for the data in Tables H7-H18. Representative desorption isotherms are presented in Figures H5-H8. The isotherms in this set of figures illustrate the important features of the different types of isotherms that were obtained, as discussed below.

28. **Classification of desorption isotherms.** A classification scheme was developed to provide a convenient framework for interpreting the desorption data. The data collected from the serial, graded batch leaching tests fall into four general classifications: no-release, low-release, clustered, and curvilinear isotherms. The characteristics of these desorption isotherm classifications are discussed below. Table H19 lists the processes by formulation and the respective desorption isotherm classification for each process formulation and contaminant.

29. For some of the desorption isotherm data, the leachate concentrations were within the 95-percent confidence interval for the blanks for all of the liquid-solids ratios used in the series of leach tests. The tests in

Table H6

Statistical Analysis of Everett Bay Solidification Blanks

Parameter	As	Cd	Cr	Pb	Zn	DOC
Detection limits, mg/ℓ	0.005	0.0001	0.001	0.001	0.03	1.0
Number of blanks	12	12	12	12	12	10
Number of blanks below detection limits	12	1	4	3	12	10
Range	--	<0.0001 to 0.0007	<0.001 to 0.021	<0.001 to 0.005	--	--
Mean	<0.005	0.00028	0.00533	0.00246	<.03	<1
Standard deviation	--	0.00019	0.00624	0.00171	--	--
95-percent confidence interval	≤0.005	≤0.0004	≤0.01	≤0.004	≤0.03	≤1.0

which contaminant release was not measurable at any of the liquid-solid ratios are termed "no-release isotherms." All of the arsenic and zinc desorption isotherms tested for solidified/stabilized Everett Bay sediment were classified as no-release isotherms. Most of the cadmium and some of the chromium and lead isotherms could be classified as no-release isotherms. Since the contaminant is resistant to leaching, Equation C5 does not apply to contaminants characterized by no-release isotherms. The solid phase concentration is constant ($q = q_r = q_o$), and the leachate concentration is either below the detection limit or within the 95-percent confidence interval for the blanks.

30. There was one exception to the rule for classification as a no-release isotherm. The lead desorption isotherm for the portland cement with Firmix process at an additive to sediment formulation of 0.1 portland

Table H7

Desorption Isotherm Data for Portland Cement Process

(0.05:1 Portland Cement:Sediment)

Liquid/Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.003	<0.005	>1.61	<0.0004	>1.182	<0.010	>13.59	0.009	16.91	<0.030	>51.4	31	23400.1
100:1	1.002	<0.005	>1.61	<0.0004	>1.182	<0.010	>13.59	0.005	17.31	<0.030	>51.4	<31	23397.0
100:1	1.002	<0.005	>1.61	<0.0004	>1.182	<0.010	>13.59	0.007	17.11	<0.030	>51.4	<29	23596.6
20:1	5.005	<0.005	>2.01	<0.0004	>1.214	<0.010	>14.39	0.085	16.11	<0.030	>53.8	<135	23793.5
20:1	5.004	<0.005	>2.01	<0.0004	>1.214	<0.010	>14.39	0.041	16.99	<0.030	>53.8	<125	23992.8
20:1	5.003	<0.005	>2.01	<0.0004	>1.214	<0.010	>14.39	0.041	16.99	<0.030	>53.8	<127	23952.3
10:1	10.000	<0.005	>2.06	<0.0004	>1.218	<0.010	>14.49	0.082	16.99	<0.030	>54.1	<238	24110.8
10:1	10.001	<0.005	>2.06	<0.0004	>1.218	<0.010	>14.49	0.075	17.06	<0.030	>54.1	<250	23991.0
10:1	10.007	<0.005	>2.06	<0.0004	>1.218	<0.010	>14.49	0.073	17.08	<0.030	>54.1	<244	24052.5
5:1	20.002	<0.005	>2.08	0.0005	1.219	<0.010	>14.54	0.183	16.90	<0.030	>54.3	<423	24376.0
5:1	20.006	<0.005	>2.08	<0.0004	>1.220	<0.010	>14.54	0.231	16.66	<0.030	>54.3	<413	24426.4
5:1	20.002	<0.005	>2.08	0.0007	1.218	<0.010	>14.54	0.262	16.50	<0.030	>54.3	<450	24241.0
2:1	50.000	<0.005	>2.10	0.0022	1.218	<0.010	>14.57	0.009	17.79	<0.030	>54.4	<796	24898.8
2:1	50.000	<0.005	>2.10	0.0019	1.218	<0.010	>14.57	0.018	17.77	0.031	>54.4	<775	24940.8
2:1	50.000	<0.005	>2.10	0.0019	1.218	<0.010	>14.57	0.011	17.79	<0.030	>54.4	<766	24958.8

* Mass of solidified sediment leached with 100 ml of water.

H16

Table H8

Desorption Isotherm Data for Portland Cement Process

(0.1:1.0 Portland Cement:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.003	<0.005	>1.51	<0.0004	>1.127	<0.010	>12.93	0.006	16.40	<0.030	>49.0	38	21498.0
100:1	1.000	<0.005	>1.51	<0.0004	>1.126	<0.010	>12.93	0.008	16.20	<0.030	>49.0	34	21886.7
100:1	1.000	<0.005	>1.51	<0.0004	>1.126	<0.010	>12.93	0.007	16.30	<0.030	>49.0	35	21786.7
20:1	5.001	<0.005	>1.91	<0.0007	>1.152	<0.010	>13.73	0.006	16.88	<0.030	>51.4	139	22507.2
20:1	5.001	<0.005	>1.91	<0.0004	>1.158	<0.010	>13.73	0.039	16.22	<0.030	>51.4	142	22447.2
20:1	5.003	<0.005	>1.91	<0.0004	>1.158	<0.010	>13.73	<0.004	>16.92	<0.030	>51.4	140	22488.3
10:1	10.001	<0.005	>1.96	<0.0004	>1.162	0.024	13.69	<0.004	>16.96	<0.030	>51.7	257	22716.9
10:1	10.001	<0.005	>1.96	<0.0004	>1.162	0.011	13.82	<0.004	>16.96	<0.030	>51.7	253	22756.9
10:1	10.002	<0.005	>1.96	<0.0004	>1.162	0.011	13.82	<0.004	>16.96	<0.030	>51.7	250	22787.2
5:1	20.000	<0.005	>1.99	<0.0004	>1.164	0.024	13.81	<0.004	>16.98	<0.030	>51.8	430	23136.7
5:1	20.001	<0.005	>1.99	<0.0004	>1.164	0.029	13.78	0.006	16.97	<0.030	>51.8	427	23151.8
5:1	20.003	<0.005	>1.99	0.0023	1.155	0.028	13.79	0.005	16.98	<0.030	>51.8	416	23207.0
2:1	50.002	<0.005	>2.00	0.0024	1.162	0.022	13.88	0.005	16.99	<0.030	>51.9	792	23702.7
2:1	50.000	<0.005	>2.00	0.0027	1.161	0.019	13.89	0.013	16.98	<0.030	>51.9	791	23704.7
2:1	50.000	<0.005	>2.00	0.0005	1.165	<0.010	>13.91	0.014	16.97	<0.030	>51.9	796	23694.7

* Mass of solidified sediment leached with 100 ml of water.

Table H9

Desorption Isotherm Data for Portland Cement Process

(0.2:1 Portland Cement:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.006	<0.005	>1.35	<0.0004	>1.029	<0.010	>11.77	0.005	15.09	<0.030	>44.6	23	20893.2
100:1	1.005	<0.005	>1.35	<0.0004	>1.029	<0.010	>11.77	0.011	14.49	<0.030	>44.6	23	20890.9
100:1	1.008	<0.005	>1.35	<0.0004	>1.030	<0.010	>11.77	0.011	14.49	<0.030	>44.7	23	20897.7
20:1	5.002	<0.005	>1.74	<0.0004	>1.061	<0.010	>12.57	0.010	15.38	<0.030	>47.0	77	21640.0
20:1	5.003	<0.005	>1.74	<0.0004	>1.061	<0.010	>12.57	0.097	13.65	<0.030	>47.0	78	21620.4
20:1	5.003	<0.005	>1.74	<0.0004	>1.061	<0.010	>12.57	0.051	14.57	<0.030	>47.0	78	21620.4
10:1	10.005	<0.005	>1.79	<0.0004	>1.065	<0.010	>12.67	0.120	14.39	<0.030	>47.3	158	21600.2
10:1	10.001	<0.005	>1.79	<0.0004	>1.065	<0.010	>12.67	0.056	15.02	<0.030	>47.3	159	21589.6
10:1	10.001	<0.005	>1.79	<0.0004	>1.065	<0.010	>12.67	0.057	15.01	<0.030	>47.3	158	21599.6
5:1	20.005	<0.005	>1.82	<0.0004	>1.067	0.013	12.70	0.013	15.52	<0.030	>47.5	261	21874.8
5:1	20.005	<0.005	>1.82	<0.0004	>1.067	0.013	12.70	<0.004	>15.56	<0.030	>47.5	282	21769.8
5:1	20.009	<0.005	>1.82	<0.0004	>1.067	<0.010	>12.72	0.005	15.56	<0.030	>47.5	284	21760.1
2:1	50.000	<0.005	>1.83	<0.0004	>1.068	0.043	12.68	0.019	15.55	<0.030	>47.6	538	22103.4
2:1	50.000	<0.005	>1.83	<0.0004	>1.068	0.050	12.67	0.024	15.54	<0.030	>47.6	548	22083.4
2:1	50.000	<0.005	>1.83	<0.0030	>1.063	0.018	>12.73	0.008	15.57	<0.030	>47.6	551	22077.4

* Mass of solidified sediment leached with 100 ml of water.

H18

Table H10

Desorption Isotherm Data for Firmix Process

(0.4:1.0 Firmix: Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/l	q mg/kg	C mg/l	q mg/kg	C mg/l	q mg/kg	C mg/l	q mg/kg	C mg/l	q mg/kg	C mg/l	q mg/kg
100:1	1.000	<0.005	>1.08	<0.0004	>0.88	<0.010	>9.94	<0.004	>12.96	<0.03	>37.82	12	18,668
100:1	1.003	<0.005	>1.08	<0.0004	>0.88	<0.010	>9.95	<0.004	>12.96	<0.03	>37.83	11	18,771
100:1	1.002	<0.005	>1.08	<0.0004	>0.88	<0.010	>9.94	<0.004	>12.96	<0.03	>37.83	12	18,670
20:1	5.004	<0.005	>1.48	<0.0004	>0.91	<0.010	>10.74	<0.004	>13.28	<0.03	>40.22	48	18,909
20:1	5.002	<0.005	>1.48	<0.0004	>0.91	<0.010	>10.74	<0.004	>13.28	<0.03	>40.22	42	19,028
20:1	5.002	<0.005	>1.48	<0.0004	>0.91	<0.010	>10.74	<0.004	>13.28	<0.03	>40.22	40	19,068
10:1	10.001	<0.005	>1.53	<0.0004	>0.91	<0.010	>10.84	<0.004	>13.32	<0.03	>40.52	72	19,148
10:1	10.004	<0.005	>1.53	<0.0004	>0.91	<0.010	>10.84	<0.004	>13.32	<0.03	>40.52	69	19,178
10:1	10.003	<0.005	>1.53	<0.0004	>0.91	<0.010	>10.84	<0.004	>13.32	<0.03	>40.52	71	19,158
5:1	20.004	<0.005	>1.56	<0.0004	>0.91	<0.011	>10.89	<0.004	>13.34	<0.03	>40.67	111	19,313
5:1	20.002	<0.005	>1.56	<0.0004	>0.91	<0.010	>10.89	<0.004	>13.34	<0.03	>40.67	117	19,283
5:1	20.008	<0.005	>1.56	<0.0004	>0.91	<0.010	>10.89	<0.004	>13.34	<0.03	>40.67	110	19,318
2:1	50.000	<0.005	>1.57	<0.0004	>0.92	<0.010	>10.92	<0.004	>13.35	<0.03	>40.76	263	19,342
2:1	50.000	<0.005	>1.57	<0.0004	>0.92	<0.011	>10.92	0.008	13.34	<0.03	>40.76	262	19,344
2:1	50.000	<0.005	>1.57	<0.0004	>0.92	<0.011	>10.92	0.006	13.35	<0.03	>40.76	260	19,348

Table H11

Desorption Isotherm Data for Firmix Process

(0.5:1.0 Firmix:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.005	<0.005	>0.98	<0.0004	>0.8156	0.011	9.118	<0.004	>12.070	<0.03	>35.118	13	17250
100:1	1.004	<0.005	>0.98	<0.0004	>0.8155	<0.010	> 9.217	<0.004	>12.069	<0.03	>35.115	12	17348
100:1	1.002	<0.005	>0.98	<0.0004	>0.8154	<0.010	> 9.215	<0.004	>12.068	<0.03	>35.109	13	17246
20:1	5.005	<0.005	>1.37	<0.0004	>0.8474	<0.010	>10.013	<0.004	>12.388	<0.03	>37.503	51	17525
20:1	5.004	<0.005	>1.37	<0.0004	>0.8474	<0.010	>10.013	<0.004	>12.388	<0.03	>37.503	51	17524
20:1	5.008	<0.005	>1.38	<0.0004	>0.8474	0.016	9.893	<0.004	>12.388	<0.03	>37.504	52	17505
10:1	10.002	<0.005	>1.42	<0.0004	>0.8514	0.015	10.063	<0.004	>12.428	<0.03	>37.803	84	17704
10:1	10.002	<0.005	>1.42	<0.0004	>0.8514	0.023	9.983	<0.004	>12.428	<0.03	>37.803	85	17694
10:1	10.003	<0.005	>1.42	<0.0004	>0.8514	0.033	9.883	<0.004	>12.428	<0.03	>37.803	85	17694
5:1	20.004	<0.005	>1.45	<0.0004	>0.8534	0.022	10.103	<0.004	>12.448	<0.03	>37.953	165	17719
5:1	20.005	<0.005	>1.45	<0.0004	>0.8534	0.026	10.083	<0.004	>12.448	<0.03	>37.953	161	17739
5:1	20.009	<0.005	>1.45	<0.0004	>0.8534	0.036	10.033	<0.004	>12.448	<0.03	>37.953	163	17729
2:1	50.000	<0.005	>1.46	<0.0004	>0.8546	0.044	10.125	0.006	12.456	<0.03	>38.043	342	17860
2:1	50.000	<0.005	>1.46	<0.0004	>0.8546	0.039	10.135	0.005	12.458	<0.03	>38.043	347	17850
2:1	50.000	<0.005	>1.46	<0.0030	>0.8546	0.037	>10.139	0.005	12.458	<0.03	>38.043	352	17840

* Mass of solidified sediment leached with 100 ml of water.

H20

Table H12

Desorption Isotherm Data for Firmix Process

(0.6:1.0 Firmix:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.001	<0.005	>0.88	<0.0004	>0.7619	<0.010	>8.58	0.006	11.09	<0.03	>32.72	10	16386
100:1	1.000	<0.005	>0.88	0.0012	0.6819	<0.010	>8.57	0.004	11.29	<0.03	>32.72	10	16385
100:1	1.000	<0.005	>0.88	<0.0004	>0.7619	<0.010	>8.57	0.005	11.19	<0.03	>32.72	10	16385
20:1	5.009	<0.005	>1.28	0.0005	0.7919	<0.010	>9.37	<0.004	>11.61	<0.03	>35.12	44	16506
20:1	5.002	<0.005	>1.28	<0.0004	>0.7939	0.028	9.01	0.028	11.13	<0.03	>35.12	47	16445
20:1	5.001	<0.005	>1.28	<0.0004	>0.7939	<0.010	>9.37	0.005	11.59	<0.03	>35.12	47	16445
10:1	10.004	<0.005	>1.33	0.0007	0.7949	<0.010	>9.47	0.008	11.61	<0.03	>35.42	72	16665
10:1	10.004	<0.005	>1.33	0.0006	0.7959	<0.010	>9.47	<0.004	>11.65	<0.03	>35.42	74	16645
10:1	10.002	<0.005	>1.33	<0.0004	>0.7979	<0.010	>9.47	<0.004	>11.65	<0.03	>35.42	72	16665
5:1	20.004	<0.005	>1.36	<0.0004	>0.7999	<0.010	>9.52	<0.004	>11.67	<0.03	>35.57	109	16840
5:1	20.005	<0.005	>1.36	<0.0004	>0.7999	<0.010	>9.52	0.005	11.66	<0.03	>35.57	111	16830
5:1	20.004	<0.005	>1.36	<0.0004	>0.7999	<0.010	>9.52	0.005	11.66	<0.03	>35.57	111	16830
2:1	50.000	<0.005	>1.37	0.0006	0.8007	0.011	9.55	0.024	11.64	<0.03	>35.66	264	16857
2:1	50.000	<0.005	>1.37	0.0006	0.8007	<0.010	>9.55	0.022	11.64	<0.03	>35.66	270	16845
2:1	50.000	<0.005	>1.37	0.0005	0.8009	0.011	9.55	0.023	11.64	<0.03	>35.66	261	16863

* Mass of solidified sediment leached with 100 ml of water.

H21

Table H13

Desorption Isotherm Data for Lime/Fly Ash Process

(0.1:0.3:1.0 Lime:Fly Ash:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.002	<0.005	>1.08	<0.0004	>0.877	0.020	8.95	<0.004	>12.96	<0.030	>37.8	21	17772.3
100:1	1.005	<0.005	>1.08	<0.0004	>0.877	0.017	9.25	<0.004	>12.96	<0.030	>37.8	21	17778.5
100:1	1.000	<0.005	>1.08	<0.0004	>0.876	0.014	9.54	<0.004	>12.96	<0.030	>37.8	22	17668.1
20:1	5.007	<0.005	>1.48	<0.0004	>0.908	0.023	10.48	0.015	13.06	<0.030	>40.2	69	18490.0
20:1	5.006	<0.005	>1.48	<0.0004	>0.908	0.025	10.44	0.010	13.16	<0.030	>40.2	70	18469.8
20:1	5.005	<0.005	>1.48	<0.0004	>0.908	0.020	10.54	0.009	13.18	<0.030	>40.2	68	18509.4
10:1	10.001	<0.005	>1.53	<0.0004	>0.912	0.024	10.70	0.021	13.15	<0.030	>40.5	125	18618.2
10:1	10.003	<0.005	>1.53	<0.0004	>0.912	0.026	10.68	0.028	13.08	<0.030	>40.5	129	18578.5
10:1	10.006	<0.005	>1.53	<0.0004	>0.912	0.021	10.73	0.026	13.10	<0.030	>40.5	120	18668.8
5:1	20.002	<0.005	>1.56	<0.0004	>0.914	0.029	10.80	0.051	13.10	<0.030	>40.7	226	18738.2
5:1	20.005	<0.005	>1.56	<0.0004	>0.914	0.023	10.83	0.055	13.08	<0.030	>40.7	213	18803.4
5:1	20.002	<0.005	>1.56	<0.0005	>0.914	0.028	10.80	0.058	13.07	<0.030	>40.7	219	18773.2
2:1	50.000	<0.005	>1.57	<0.0005	>0.915	<0.010	>10.92	0.119	13.12	<0.030	>40.8	483	18902.1
2:1	50.000	<0.005	>1.57	<0.0006	>0.915	<0.010	>10.92	0.128	13.10	<0.030	>40.8	483	18902.1
2:1	50.000	<0.005	>1.57	<0.0004	>0.916	<0.010	>10.92	0.016	13.33	<0.030	>40.8	473	18922.1

* Mass of solidified sediment leached with 100 ml of water.

Table H14

Desorption Isotherm Data for Lime/Fly Ash Process

(0.1:0.4:1.0 Lime:Fly Ash:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.004	<0.005	>0.98	<0.0004	>0.816	0.014	8.82	<0.004	>12.07	<0.030	>35.1	20	16551.5
100:1	1.003	<0.005	>0.98	<0.0004	>0.815	0.014	8.82	0.006	11.87	<0.030	>35.1	21	16449.8
100:1	1.004	<0.005	>0.98	<0.0004	>0.816	0.013	8.92	<0.004	>12.07	<0.030	>35.1	23	16252.7
20:1	5.005	<0.005	>1.37	0.0072	0.712	0.012	9.97	0.011	12.25	<0.030	>37.5	77	17005.1
20:1	5.003	<0.005	>1.37	<0.0004	>0.847	<0.010	>10.01	0.012	12.23	<0.030	>37.5	71	17124.4
20:1	5.001	<0.005	>1.37	<0.0004	>0.847	0.013	9.95	0.007	12.33	<0.030	>37.5	71	17123.8
10:1	10.005	<0.005	>1.42	<0.0004	>0.851	<0.010	>10.11	0.020	12.27	<0.030	>37.8	126	17284.2
10:1	10.007	<0.005	>1.42	<0.0004	>0.851	0.011	10.10	0.016	12.31	<0.030	>37.8	125	17294.4
10:1	10.009	<0.005	>1.42	<0.0004	>0.851	<0.010	>10.11	0.015	12.32	<0.030	>37.8	126	17284.7
5:1	20.003	<0.005	>1.45	<0.0004	>0.853	0.018	10.12	0.132	11.81	<0.030	>38.0	243	17328.7
5:1	20.007	<0.005	>1.45	0.0005	0.853	0.012	10.15	0.140	11.77	<0.030	>38.0	237	17359.0
5:1	20.008	<0.005	>1.45	0.0005	0.853	0.017	10.13	0.053	12.20	<0.030	>38.0	232	17384.0
2:1	50.000	<0.005	>1.46	<0.0004	>0.855	0.017	10.18	0.105	12.26	<0.030	>38.0	427	17689.5
2:1	50.000	<0.005	>1.46	0.0026	0.850	<0.010	>10.19	0.121	12.23	<0.030	>38.0	428	17687.5
2:1	50.000	<0.005	>1.46	0.0018	0.852	0.014	10.18	0.094	12.28	<0.030	>38.0	426	17691.5

* Mass of solidified sediment leached with 100 ml of water.

H23

Table H15

Desorption Isotherm Data for Lime/Fly Ash Process

(0.1:0.5:1 Lime:Fly Ash:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.009	<0.005	>0.89	<0.0004	>0.762	0.021	7.49	<0.004	>11.29	<0.030	>32.7	23	15105.1
100:1	1.004	<0.005	>0.88	<0.0004	>0.762	0.023	7.28	0.007	10.99	<0.030	>32.7	21	15292.9
100:1	1.005	<0.005	>0.89	<0.0004	>0.762	0.020	7.58	0.007	10.99	<0.030	>32.7	20	15394.5
20:1	5.003	<0.005	>1.28	<0.0004	>0.794	0.018	9.21	0.013	11.43	<0.030	>35.1	70	15985.4
20:1	5.006	<0.005	>1.28	<0.0004	>0.794	0.017	9.23	0.011	11.47	<0.030	>35.1	70	15986.3
20:1	5.004	<0.005	>1.28	<0.0004	>0.794	0.019	9.19	0.012	11.45	<0.030	>35.1	68	16025.7
10:1	10.008	<0.005	>1.33	<0.0004	>0.798	0.011	9.46	0.019	11.50	<0.030	>35.4	127	16115.6
10:1	10.002	<0.005	>1.33	<0.0004	>0.798	0.013	9.44	0.089	10.80	<0.030	>35.4	122	16164.8
10:1	10.001	<0.005	>1.33	<0.0004	>0.798	0.013	9.44	0.016	11.53	<0.030	>35.4	122	16164.7
5:1	20.007	<0.005	>1.36	<0.0004	>0.800	0.013	9.51	0.026	11.56	<0.030	>35.6	304	15865.1
5:1	20.004	<0.005	>1.36	<0.0004	>0.800	<0.010	>9.52	0.022	11.58	<0.030	>35.6	231	16229.8
5:1	20.002	<0.005	>1.36	<0.0004	>0.800	0.012	9.51	0.003	11.67	<0.030	>35.6	237	16199.7
2:1	50.000	<0.005	>1.37	<0.0004	>0.801	<0.010	>9.55	0.077	11.53	<0.030	>35.7	453	16478.6
2:1	50.000	<0.005	>1.37	<0.0004	>0.801	0.012	9.55	0.100	11.49	<0.030	>35.7	443	16498.6
2:1	50.000	<0.005	>1.37	<0.0004	>0.801	0.015	9.54	<0.004	>11.68	<0.030	>35.7	464	16456.6

* Mass of solidified sediment leached with 100 ml of water.

H24

Table H16

Desorption Isotherm Data for Portland Cement/Firmix Process

(0.10:0.20:1.0 Portland Cement:Firmix:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/l	q mg/kg	C mg/l	q mg/kg	C mg/l	q mg/kg	C mg/l	q mg/kg	C mg/l	q mg/kg	C mg/l	q mg/kg
100:1	1.002	<0.005	>1.203	<0.0004	>0.947	<0.010	>10.786	<0.004	>13.986	0.03	40.971	19	19500
100:1	1.006	<0.005	>1.205	<0.0004	>0.947	<0.010	>10.790	<0.004	>13.986	<0.03	>40.982	19	19508
100:1	1.002	<0.005	>1.203	<0.0004	>0.947	<0.010	>10.786	<0.004	>13.986	<0.03	>40.971	19	19500
20:1	5.005	<0.005	>1.602	<0.0004	>0.979	0.016	11.464	<0.004	>14.306	<0.03	>43.365	63	20138
20:1	5.003	<0.005	>1.602	<0.0004	>0.979	<0.010	>11.584	<0.004	>14.306	<0.03	>43.365	65	20097
20:1	5.002	<0.005	>1.602	<0.0004	>0.979	0.012	11.544	0.008	14.226	<0.03	>43.365	63	20137
10:1	10.004	<0.005	>1.652	<0.0004	>0.983	0.012	11.661	<0.004	>14.346	<0.03	>43.665	106	20337
10:1	10.004	<0.005	>1.652	<0.0004	>0.983	0.017	11.614	<0.004	>14.346	<0.03	>43.665	105	20347
10:1	10.001	<0.005	>1.652	<0.0004	>0.983	0.015	11.634	<0.004	>14.346	<0.03	>43.665	102	20377
5:1	20.005	<0.005	>1.677	<0.0004	>0.985	0.018	11.694	<0.004	>14.366	<0.03	>43.815	179	20502
5:1	20.006	<0.005	>1.677	<0.0004	>0.985	0.017	11.699	<0.004	>14.366	<0.03	>43.815	184	20477
5:1	20.002	<0.005	>1.677	<0.0004	>0.985	0.020	11.684	<0.004	>14.366	<0.03	>43.815	181	20491
2:1	50.000	<0.005	>1.692	0.0012	0.985	0.030	11.724	<0.004	>14.378	<0.03	>43.905	410	20576
2:1	50.000	<0.005	>1.692	<0.0004	>0.986	0.027	11.730	<0.004	>14.378	<0.03	>43.905	399	20598
2:1	50.000	<0.005	>1.692	<0.0004	>0.986	0.031	11.722	<0.004	>14.378	<0.03	>43.905	407	20582

* Mass of solidified sediment leached with 100 ml of water.

H25

Table H17

Desorption Isotherm Data for Portland Cement/Firmix Process

(0.20:0.10:1.0 Portland Cement:Firmix:Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.004	<0.005	>1.204	<0.0004	>0.947	<0.010	>10.788	<0.004	>13.987	0.03	40.98	18	19604
100:1	1.006	<0.005	>1.205	<0.0004	>0.947	0.012	10.591	<0.004	>13.988	<0.03	>40.98	19	19508
100:1	1.004	<0.005	>1.204	<0.0004	>0.947	<0.010	>10.788	<0.004	>13.987	<0.03	>40.98	19	19504
20:1	5.004	<0.005	>1.602	<0.0004	>0.979	0.016	11.464	<0.004	>14.306	<0.03	>43.37	70	19998
20:1	5.006	<0.005	>1.602	<0.0004	>0.979	0.012	11.544	<0.004	>14.306	<0.03	>43.37	70	19998
20:1	5.009	<0.005	>1.602	<0.0004	>0.979	0.015	11.484	<0.004	>14.306	<0.03	>43.37	71	19979
10:1	10.006	<0.005	>1.652	<0.0004	>0.983	0.012	11.664	<0.004	>14.346	<0.03	>43.66	120	20197
10:1	10.007	<0.005	>1.652	<0.0004	>0.983	0.018	11.604	<0.004	>14.346	<0.03	>43.66	124	20157
10:1	10.007	<0.005	>1.652	<0.0004	>0.983	0.016	11.624	<0.004	>14.346	<0.03	>43.66	119	20207
5:1	20.005	<0.005	>1.677	<0.0004	>0.985	0.023	11.669	<0.004	>14.366	<0.03	>43.81	315	19822
5:1	20.005	<0.005	>1.677	<0.0004	>0.985	0.017	11.699	<0.004	>14.366	<0.03	>43.81	297	19912
5:1	20.006	<0.005	>1.677	<0.0004	>0.985	0.023	11.669	<0.004	>14.366	<0.03	>43.81	291	19942
2:1	50.000	<0.005	>1.692	<0.0004	>0.986	0.035	11.714	0.015	14.356	<0.03	>43.90	555	20286
2:1	50.000	<0.005	>1.692	<0.0004	>0.986	0.031	11.722	<0.004	>14.378	<0.03	>43.90	526	20344
2:1	50.000	<0.005	>1.692	<0.0004	>0.986	0.029	11.726	0.016	14.354	<0.03	>43.90	531	20334

* Mass of solidified sediment leached with 100 ml of water.

Table H18

Desorption Isotherm Data for Portland Cement/Firmix Process

(0.15:0.15:1.0 Portland Cement:Firmix::Sediment)

Liquid/ Solid Ratio	M(s)* g	Arsenic		Cadmium		Chromium		Lead		Zinc		TOC	
		C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg	C mg/ℓ	q mg/kg
100:1	1.007	<0.005	>1.207	<0.0004	>0.948	<0.010	>10.80	<0.004	>14.00	0.030	41.02	23	19129
100:1	1.003	<0.005	>1.205	<0.0004	>0.948	<0.010	>10.80	<0.004	>14.00	<0.030	>41.01	20	19419
100:1	1.000	<0.005	>1.203	<0.0004	>0.948	<0.010	>10.79	<0.004	>14.00	<0.030	>41.00	20	19413
20:1	5.006	<0.005	>1.603	<0.0004	>0.980	0.015	11.49	<0.004	>14.32	<0.030	>43.40	66	20095
20:1	5.003	<0.005	>1.603	0.0008	0.972	<0.010	>11.59	<0.004	>14.32	<0.030	>43.40	73	19954
20:1	5.005	<0.005	>1.603	<0.0004	>0.980	<0.010	>11.59	<0.004	>14.32	<0.030	>43.40	70	20014
10:1	10.004	<0.005	>1.653	<0.0004	>0.984	0.012	11.67	<0.004	>14.36	<0.030	>43.70	126	20154
10:1	10.002	<0.005	>1.653	<0.0004	>0.984	0.011	11.68	<0.004	>14.36	<0.030	>43.70	120	20213
10:1	10.001	<0.005	>1.653	<0.0004	>0.984	<0.010	>11.69	<0.004	>14.36	<0.030	>43.70	126	20153
5:1	20.003	<0.005	>1.678	0.0005	0.985	0.014	11.72	<0.004	>14.38	<0.030	>43.85	211	20358
5:1	20.008	<0.005	>1.678	<0.0004	>0.986	0.011	11.74	<0.004	>14.38	<0.030	>43.85	211	20358
5:1	20.004	<0.005	>1.678	<0.0004	>0.986	0.012	11.73	<0.004	>14.38	<0.030	>43.85	210	20363
2:1	50.000	<0.005	>1.693	<0.0004	>0.987	<0.010	>11.77	<0.004	>14.39	<0.030	>43.94	433	20547
2:1	50.000	<0.005	>1.693	<0.0004	>0.987	0.011	11.77	0.007	14.38	<0.030	>43.94	444	20525
2:1	50.000	<0.005	>1.693	<0.0004	>0.987	<0.010	>11.77	0.019	14.36	<0.030	>43.94	440	20533

* Mass of solidified sediment leached with 100 ml of water.
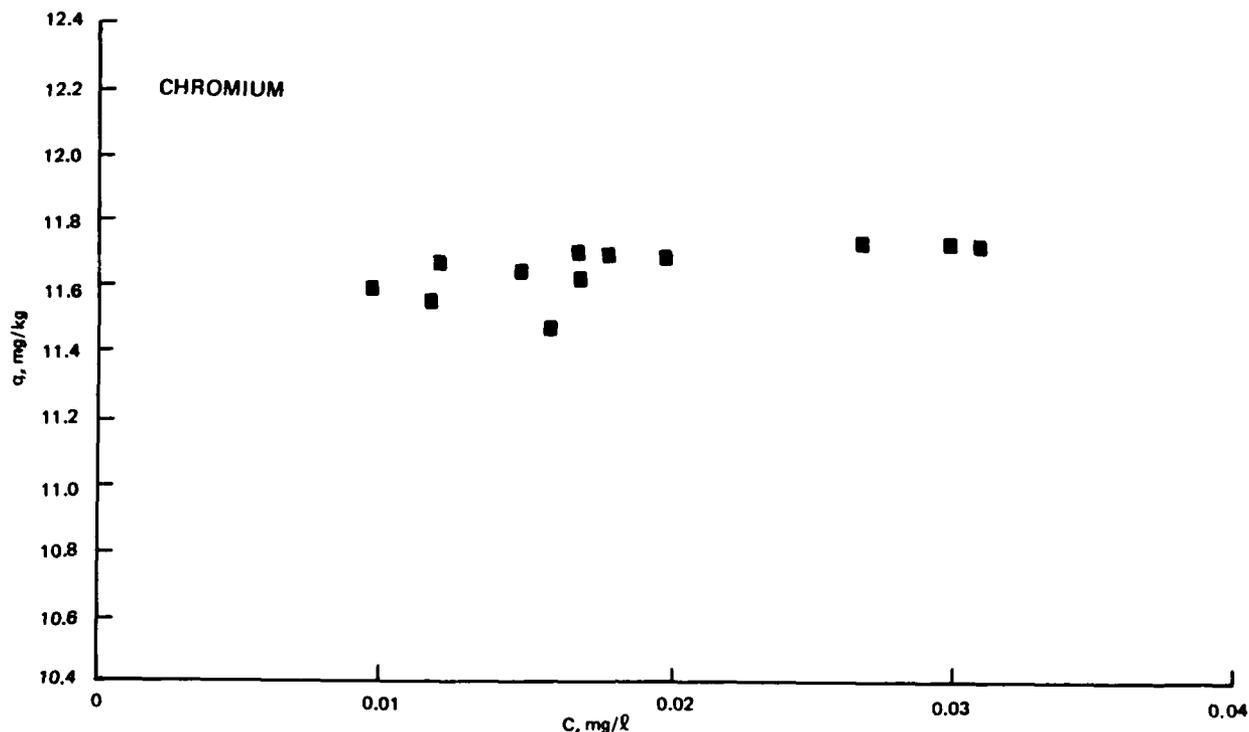
Figure H5. Chromium desorption isotherm, 0.1:0.2:1.0
portland/Firmix process

cement:0.2 Firmix:1 sediment was classified as a no-release isotherm. Lead
was detected in one sample out of 15 samples included in the series of leach
tests for this process formulation (Table H16). The concentration in this one
sample was relatively low (0.008 mg/ℓ). This isotherm was therefore clas-
sified as a no-release isotherm.

31. For some of the desorption data, the amount of contaminant released
was below the detection limit for all but one or two of the liquid-solids
ratios in the series. When the contaminant was detected, it was usually
detected in the tests conducted at the lowest liquid-solids ratios used in the
series, i.e., 2:1 and 5:1. Desorption isotherms characterized by aqueous
phase contaminant concentrations below the detection limit for liquid-solids
ratios greater than 5:1 are termed "low-release isotherms." Several of the
cadmium, chromium, and lead desorption isotherms were classified as low-
release isotherms. These are listed in Table H19. Low-release isotherms do
not provide enough points above the detection limit to determine if

H28
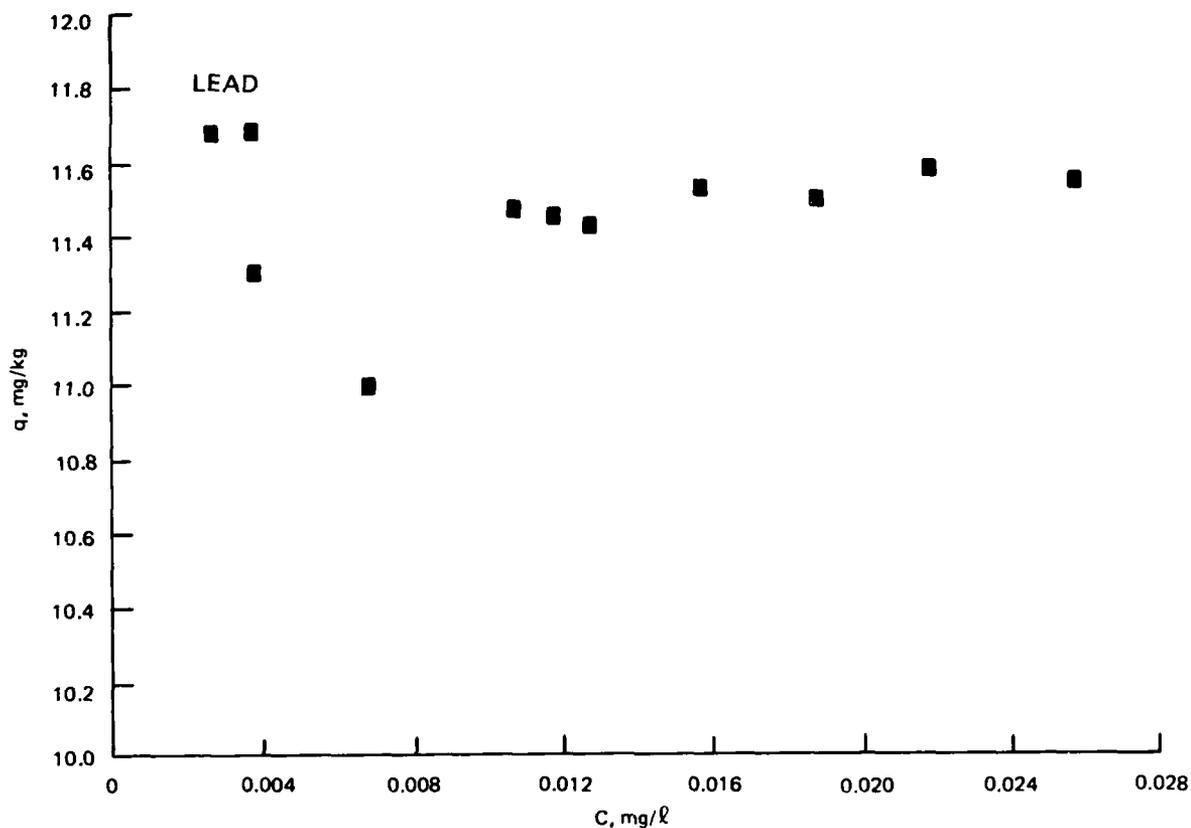
Figure H6. Lead desorption isotherm, 0.1:0.5:1.0
lime/fly ash process

Equation C5 models contaminant release. Since low-release isotherms charac-
terize contaminants that leach near the detection limit, low-release isotherms
are indicative of solidified/stabilized sediment that does not have a signif-
icant leaching potential.

32. The desorption isotherm plots for some of the leachate data were
clustered. Plots that produced clusters are termed "clustered isotherms."
Clustered desorption isotherms indicate that there is not a well-defined
relationship between solid and aqueous phase concentrations, and Equation C5
does not therefore model the data. Most of the serial, graded batch leach
tests for chromium and lead produced clustered isotherms with horizontal
orientations. Examples of clustered isotherms with horizontal orientations
are shown in Figures H5 and H6.

33. A clustered isotherm with a horizontal orientation indicates that
the distribution coefficient, K_d , is zero. Theoretically, when K_d is
equal to zero, the q versus C plot should be a horizontal line that
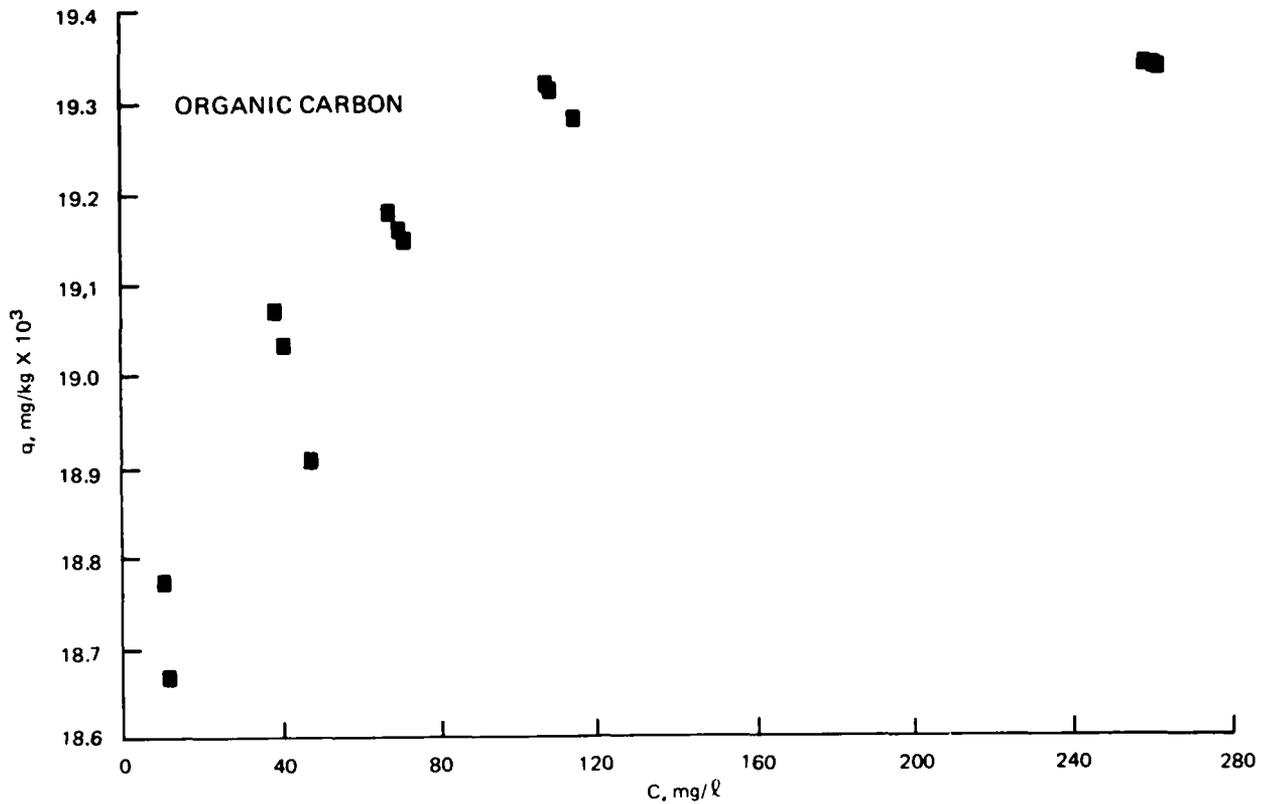
Figure H7. Organic carbon isotherm, 0.4:1.0 Firmix process

intercepts the ordinate at q_r . If K_d is zero, all of the leachable contaminant concentration in the solidified/stabilized sediment is released in each leach test in the graded series. Thus, the solid phase concentration at the end of each test approaches the concentration that is resistant to leaching, q_r . Since the solid phase concentration of leachable contaminant is constant and neither reversible exchange or sorption occurs, the aqueous phase concentration, C , depends only on the dilution provided by the various liquid-solids ratios used in the series. The aqueous phase concentration, therefore, decreases by dilution with increasing liquid-solids ratio. The isotherms shown in Figures H5 and H6 closely approximate the theoretical result for K_d equal to zero. For horizontally oriented clustered isotherms, Equation C5 becomes

$$q = q_r$$

H30
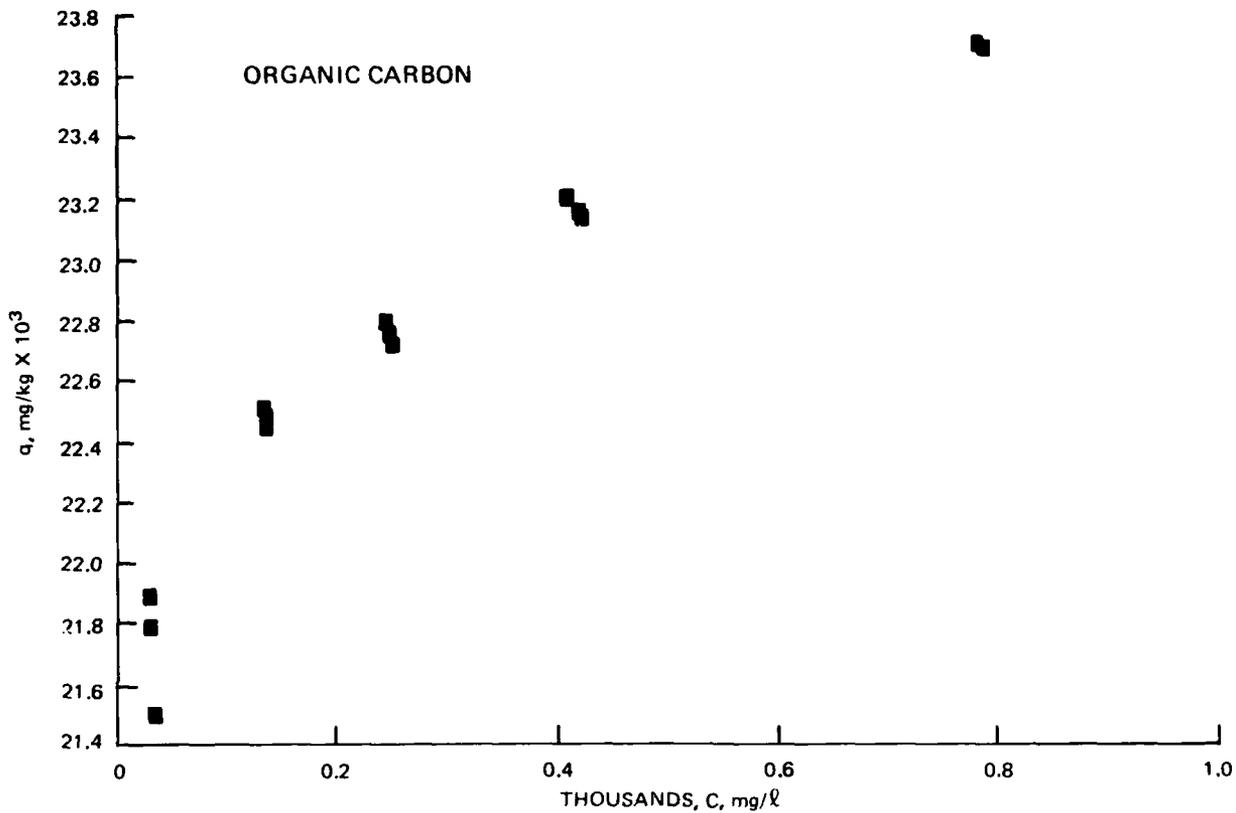
Figure H8. Organic carbon isotherm, 0.1:1.0 portland cement process

and Equation H2 becomes

$$q_r = q_o - C(L/S)$$

34. The desorption isotherms for organic carbon (OC) indicated a curvilinear relationship between solid and aqueous phase OC concentrations. Some examples of these isotherms are shown in Figures H7 and H8. Curvilinear plots usually occur in adsorption studies involving organic chemicals. Three adsorption isotherms are well known, the BET, Freundlich, and Langmuir isotherms (Weber 1972).

35. The Langmuir equation was chosen for application to the OC desorption isotherm data. The Langmuir equation is given below.

$$q = \frac{QbC}{(1 + bC)} \tag{H4}$$

Table H19

Comparison of Process Isotherm Types*

Process	As	Cd	Cr	Pb	Zn	OC**
Portland cement						
0.05:1.0	NRI	LRI	NRI	CI	NRI	CLI
0.10:1.0	NRI	LRI	CI	CI	NRI	CLI
0.20:1.0	NRI	NRI	LRI	CI	NRI	CLI
Firmix						
0.40:1.0	NRI	NRI	NRI	LRI	NRI	CLI
0.50:1.0	NRI	NRI	CI	LRI	NRI	CLI
0.60:1.0	NRI	CI	CI	CI	NRI	CLI
Lime:fly ash						
0.1:0.3:1.0	NRI	LRI	CI	CI	NRI	CLI
0.1:0.4:1.0	NRI	CI	CI	CI	NRI	CLI
0.1:0.5:1.0	NRI	NRI	CI	CI	NRI	CLI
Portland:Firmix						
0.10:0.20:1.0	NRI	LRI	CI	NRI	NRI	CLI
0.20:0.10:1.0	NRI	NRI	CI	LRI	NRI	CLI
0.15:0.15:1.0	NRI	CI	CI	LRI	NRI	CLI

*   NRI = no-release isotherm.
     LRI = low-release isotherm.
      CI = clustered isotherm.
     CLI = curvilinear isotherm.
**   OC = organic carbon.

where

> q = solid phase contaminant concentration, mg/kg
>
> Q = monolayer sorption capacity of the solid phase, mg/kg
>
> b = Langmuir constant related to the energy of adsorption, 1/mg
>
> C = aqueous phase concentration, mg/ℓ

Equation H4 models a contaminant that is totally leachable, i.e., q_r is equal to zero.

36. By fitting the data to the linearized form of the Langmuir equation given below, the Langmuir coefficients, Q and b, can be obtained.

$$\frac{C}{q} = \frac{1}{Qb} + \frac{1C}{Q} \tag{H5}$$

37. The Langmuir coefficients determined by regression of Equation H5 onto the OC desorption data are presented in Table H20. The coefficients of determination, r^2, values, and normalized sorption capacities, Q_n, are also presented in Table H20. Normalized sorption capacities are discussed later.

38. The r^2 values indicate that the fit of the nonlinear desorption model provided by the Langmuir equation was good for all of the OC data. However, since fitting Equation H5 to experimental data involves regressing C against itself, the r^2 values have limited meaning. An inspection of the OC desorption isotherms showed the nonlinearity of the process controlling OC desorption to be unmistakable. Thus, a nonlinear model, such as the Langmuir equation, is appropriate.

39. <u>Process effectiveness for contaminant immobilization.</u> If a process provides complete immobilization for each contaminant, all of the contaminant desorption isotherms will be no-release isotherms. None of the processes investigated completely immobilized all of the contaminants in Everett Bay sediment. On the basis of the number of no-release isotherms (Table H19), the Firmix process had the best metals immobilization potential, with nine no-release isotherms. The portland cement and portland cement with Firmix processes each had eight no-release isotherms, and the lime with fly ash had seven no-release isotherms.

Table H20

Comparison of Langmuir Coefficients and Normalized Sorption Capacity

Process	r^2	b	Q, mg/kg	Q_n, mg/kg
Portland cement				
0.05:1.0	0.999	0.1551	25,016.4	67,599.3
0.10:1.0	0.999	0.1188	23,853.7	61,965.2
0.20:1.0	0.999	0.3756	22,149.2	58,107.1
Firmix				
0.40:1.0	0.999	1.4787	19,393.9	51,876.4
0.50:1.0	0.999	1.1200	17,822.3	48,292.9
0.6:1.0	0.999	1.3234	16,901.5	46,078.9
Lime:fly ash				
0.1:0.3:1.0	0.999	0.5153	18,974.0	50,753.2
0.1:0.4:1.0	0.999	0.3242	17,759.6	52,529.3
0.1:0.5:1.0	0.999	0.3743	16,488.3	44,952.4
Portland:Firmix				
0.10:0.20:1.0	0.999	0.6964	20,655.5	69,063.7
0.20:0.10:1.0	0.999	0.6404	20,293.6	67,853.7
0.15:0.15:1.0	0.999	0.4723	20,615.6	68,930.3

40. As discussed previously, the leach data for metals produced no-release, low-release, and horizontally oriented clustered isotherms. Since all of the arsenic and zinc leach data produced no-release isotherms, the fraction of arsenic and zinc that is resistant to leaching, q_r, is near or equal to the initial metal concentration in the solidified/stabilized sediment, q_o. Thus, solidified/stabilized Everett Bay sediment does not appear to have a significant leaching potential for arsenic and zinc.

41. For low-release and clustered isotherms, the contaminant concentration resistant to leaching, q_r, was determined by averaging the solid phase concentrations corresponding to leachate concentrations above the detection

limit. The bar graphs in Figures H9 through H12 show the fraction of cadmium,
chromium, and lead resistant to leaching, q_r/q_o , in the solidified/
stabilized products. Figure H9 shows that greater than 98 percent of the
metals in the Firmix products was resistant to leaching. As indicated in
Figure H10, greater than 95 percent of the metals in the portland cement prod-
ucts was resistant to leaching. Figure H11 shows that greater than 97 percent
of the metals in the portland cement with Firmix products was resistant to
leaching. Figure H12 shows that the fraction leachable from the lime with fly
ash products was generally greater than 93 percent of q_o . Thus, depending
on the process formulation and the metal of interest, 93 percent or more of
the contaminant was resistant to leaching.

42. Contaminant-specific methodologies for comparing process effective-
ness are outlined below for metals and organic carbon. The methodology for
metals is based on the normalized leachable metal concentration in the
solidified/stabilized sediment, and the methodology for organic carbon is
based on normalized Langmuir curves.

43. The leachable contaminant concentration in the solidified sediment,
q_L , is given by

$$q_L = q_o - q_r \qquad (H6)$$

where

q_L = leachable contaminant concentration in the solidified/stabilized
     sediment, mg/kg

q_o = initial contaminant concentration in the solidified/stabilized
     sediment before leaching, mg/kg

q_r = contaminant concentration in the solidified/stabilized sediment
     that is resistant to leaching, mg/kg

The leachable contaminant concentration, q_L , in the solidified sediment is
an important index of contaminant mobility since this quantity is the mass of
contaminant available for release to the aqueous phase.

44. As previously discussed, no-release isotherms indicate that q_r is
approximately equal to q_o . The leachable concentration, q_L , in this case
is zero. The fraction resistant to leaching, q_r/q_o , for desorption iso-
therms classified as no-release and clustered isotherms was also discussed
earlier. Leachable metal concentrations as defined by Equation H6 were cal-
culated using the same data used to prepare Figures H9-H12.

H35
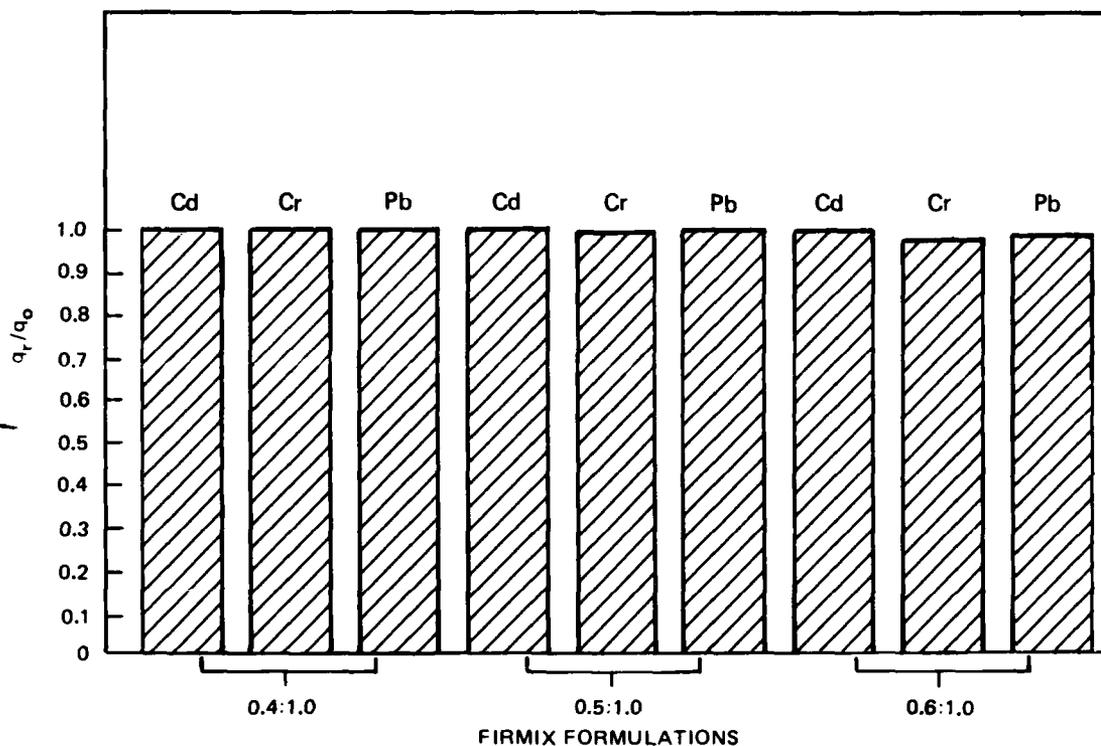
Figure H9. Fraction of contaminant resistant to leaching, Firmix process

Leachable metal concentrations of cadmium, lead, and chromium for each process are presented in Tables H21, H22, and H23, respectively. To compare processes with different additive dosages and to compare solidified/stabilized sediment with untreated sediment, the leachable contaminant concentration in the solidified sediment was normalized with respect to the mass of wet sediment that was processed for solidification/stabilization. The leachable contaminant concentration normalized with respect to the mass of sediment that was processed is given by

$$q_{nL} = q_L (1 + R)(1 + w) \qquad (H7)$$

where

q_{nL} = leachable contaminant with respect to the mass of the sediment processed by solidification, mg/kg

q_L = leachable contaminant concentration with respect to the mass of solidified sediment, mg/kg

R = dosage of solidification/stabilization reagents, kg reagents/kg wet sediment processed
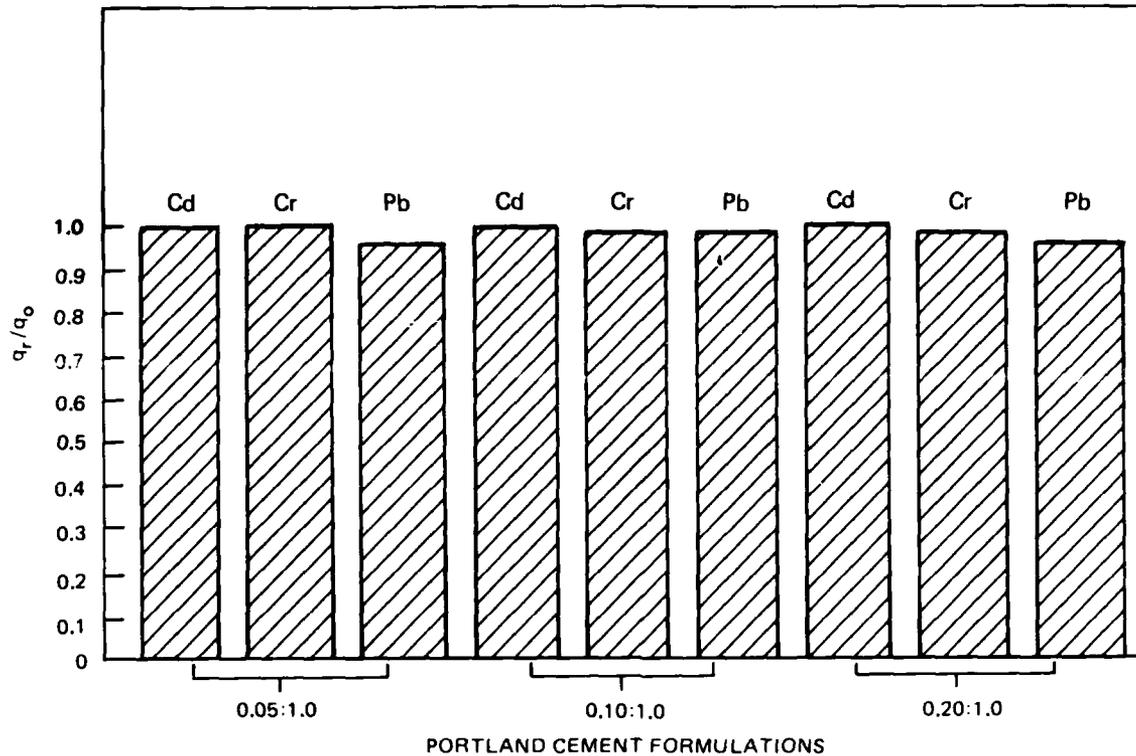
Figure H10. Fraction of contaminant resistant to leaching,
portland cement process

w = moisture content of the wet sediment, kg water/kg sediment
solids

45. Tables H21, H22, and H23 also list for each process the normalized
leachable concentrations for cadmium, lead, and chromium, respectively. The
leachable metal concentrations in untreated anaerobic sediment (Table C11) are
also presented in each table for comparison.

46. As shown in Table H21, solidification/stabilization reduced the
mass of leachable cadmium in the sediment. The Firmix process was particu-
larly effective in reducing the normalized leachable cadmium concentration.
The order of decreasing effectiveness was Firmix > portland cement > portland
cement with Firmix > lime with fly ash.

47. Table H22 lists the leachable and normalized leachable concentra-
tions for lead. The Firmix and portland cement with Firmix processes reduced
the mass of leachable lead in the sediment. The portland cement and lime with
fly ash processes showed increased q_{nL} . The order of decreasing effective-
ness was portland cement with Firmix > Firmix > lime with fly ash > portland
cement.
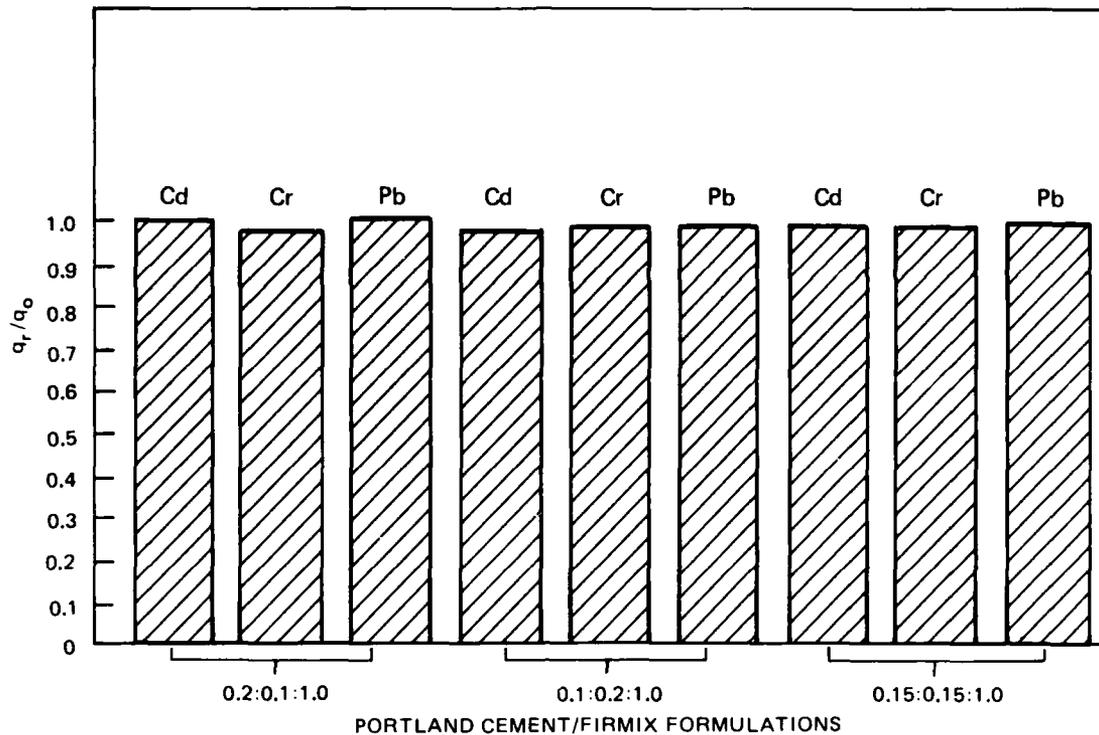
Figure H11. Fraction of contaminant release to
leaching, portland cement/Firmix process

48. The data for chromium indicated that solidification/stabilization
increased the leachable chromium in the sediment. This is shown by the nor-
malized leachable chromium concentrations presented in Table H23. The port-
land cement process increased q_{nL} the least. It is possible that the
increases were due to contamination in the process setting agents. However,
it is not likely that all of the setting agents would be contaminated. In
previous work with the same processes and another sediment, the results for
chromium were inconsistent, i.e., no process consistently showed increased or
reduced q_{nL} for all additive dosages. Another explanation for increases in
q_{nL} is that solidification/stabilization increased the leachability of the
chromium in the sediment. It is difficult, however, to reconcile such an
explanation with the published literature on solidification/stabilization
technology. Chromium mobilization by solidification/stabilization has not
been previously reported. It is also possible that the increases are
"apparent increases" that result when the leachable concentration, q_L , is
normalized. If the leachate concentrations are controlled or influenced by
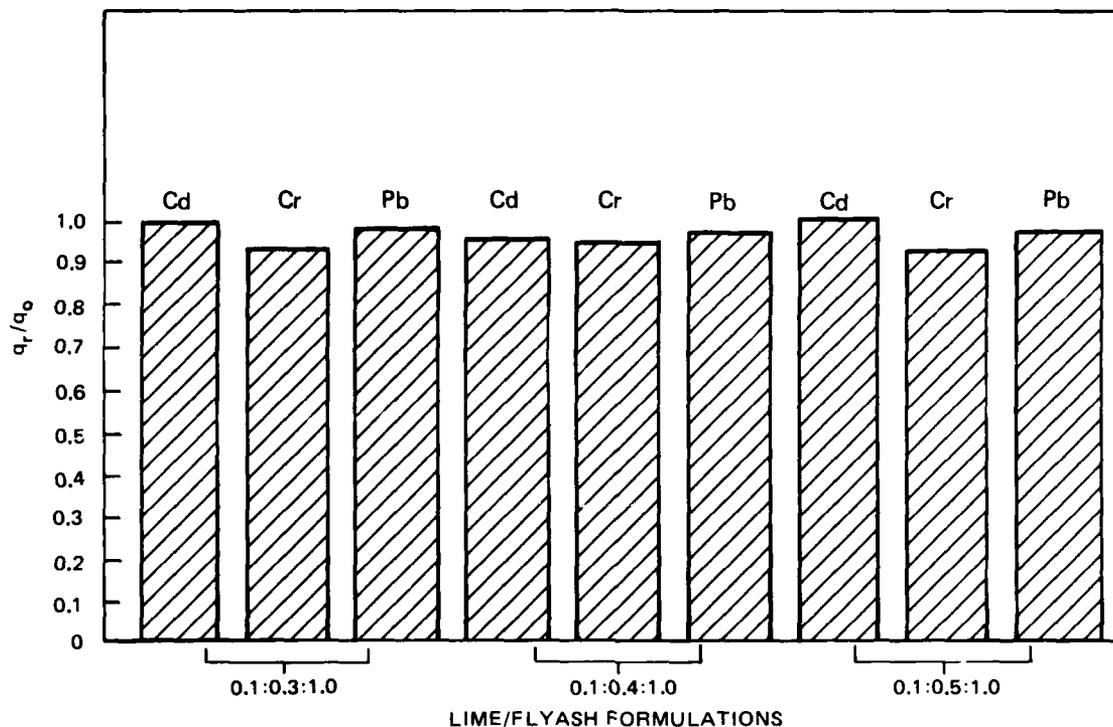
Figure H12. Fraction of contaminant resistant to
leaching, lime/fly ash process

random variability associated with testing near the detection limit, the
multiplication factors for dilution by setting agents and moisture in the
normalizing equation could produce "apparent increases." The available data
do not provide a basis for determining which of the three explanations pro-
posed above, alone or in combination, accounts for the increases in q_{nL} .
However, since the leachate concentrations were relatively low (0.01 to
0.05 mg/), there does not appear to be a significant potential for release of
chromium from solidified/stabilized sediment.

49. As previously discussed, all the organic carbon desorption iso-
therms for solidified/stabilized Everett Bay sediments were curvilinear. The
curvilinear relationship between q and C was adequately modeled by the
Langmuir equation. Since the organic carbon analysis consisted of determining
total organic carbon in filtered leachate, the analysis included naturally
occurring organic compounds such as humic and fluvic acids that are normally
found in high concentrations in sediments. Hence, the organic carbon desorp-
tion isotherms may reflect primarily the desorption characteristics of these
substances.

Table H21

Summary of Leaching Indices for Solidified/Stabilized

Sediment, Cadmium Data

Process	q_L, mg/kg	q_{nL}, mg/kg
Untreated anaerobic sediment	0.11	0.11
Portland cement/sediment		
0.05:1.0	0.0018	0.0048
0.10:1.0	0.00925	0.026
0.20:1.0	NRI	NRI
Firmix/sediment		
0.4:1.0	NRI	NRI
0.5:1.0	NRI	NRI
0.6:1.0	0.014	0.0057
Lime/fly ash/sediment		
0.1:0.3:1.0	0.005	0.018
0.1:0.4:1.0	0.036	0.14
0.1:0.5:1.0	NRI	NRI
Portland cement/Firmix/sediment		
0.2:0.1:1.0	NRI	NRI
0.1:0.2:1.0	0.029	0.097
0.15:0.15:1.0	0.011	0.037

Table H22

Summary of Leaching Indices for Solidified/Stabilized

Sediment, Lead Data

Process	q_L, mg/kg	q_{nL}, mg/kg
Untreated anaerobic sediment	1.12	1.12
Portland cement/sediment		
0.05:1.0	0.76	2.04
0.10:1.0	0.32	2.57
0.20:1.0	0.61	1.89
Firmix/sediment		
0.4:1.0	0.025	0.09
0.5:1.0	0.023	0.09
0.6:1.0	0.275	0.88
Portland cement/firmix/sediment		
0.2:0.1:1.0	NRI	NRI
0.1:0.2:1.0	NRI	NRI
0.15:0.15:1.0	0.03	0.1
Lime/fly ash/sediment		
0.1:0.3:1.0	0.24	0.86
0.1:0.4:1.0	0.32	1.23
0.1:0.5:1.0	0.32	1.32
Portland cement/Firmix/sediment		
0.2:0.1:1.0	NRI	NRI
0.1:0.2:1.0	NRI	NRI
0.15:0.15:1.0	0.03	0.1

Table H23

Summary of Leaching Indices for Solidified/
Stabilized Sediment, Chromium Data

Process	q_L, mg/kg	q_{nL}, mg/kg
Untreated anaerobic sediment	0.4	0.4
Portland cement/sediment		
0.05:1.0	NRI	NRI
0.10:1.0	0.22	0.62
0.20:1.0	0.17	0.52
Firmix/sediment		
0.4:1.0	NRI	NRI
0.5:1.0	0.26	1
0.6:1.0	0.28	1.2
Lime/fly ash/sediment		
0.1:0.3:1.0	0.72	2.4
0.1:0.4:1.0	0.54	2.1
0.1:0.5:1.0	0.69	2.8
Portland cement/Firmix/sediment		
0.2:0.1:1.0	0.33	1.1
0.1:0.2:1.0	0.23	0.77
0.15:0.15:1.0	0.19	0.63

50. The sorption capacities of the solidified/stabilized sediment (Table H20) were normalized with respect to the mass of the wet sediment solidified using the same approach previously described for normalized leachable metal concentrations. The normalized sorption capacity, Q_n, represents the maximum organic carbon concentration that the solid phase can sorb. Hence, the higher Q_n, the greater the capacity of the solids for organic carbon.

51. All of the normalized sorption capacities for the solidified/stabilized sediment were slightly less than the organic carbon concentration of the untreated anaerobic sediment (71,500 mg/kg). Normalized sorption capacities less than the original bulk sediment organic carbon concentrations were expected since setting agents probably compete with sorbed contaminants and organic matter for reactive sites on the sediment solids. Apparently, the setting agents add little or no sorption capacity.

52. Process effectiveness can be compared using normalized sorption capacities. However, this approach can be misleading if Q_n is large and the Langmuir sorption constant, b, is low. The product, $Q_n b$, represents the slope of the isotherm in the linear region at the lower end of the isotherm. The steeper the slope, the better the immobilization of organic carbon. A better approach to comparing the relative effectiveness of the processes is to graphically compare normalized desorption isotherms. Figure H13 shows the normalized organic carbon desorption isotherms for each process formulation. From this figure, it is evident that the portland cement with Firmix process provided the best control for leaching of organic carbon.

Limitations of Laboratory Evaluations

53. Several important aspects of field application were not addressed in this laboratory study. Topics beyond the scope of this investigation include scale-up factors, long-term stability of the solidified/stabilized sediment, and engineering economy. In the field, strengths may be lower than those obtained in the laboratory due to lower mixing efficiency and/or dosage control. The implementation strategy will affect mixing efficiency and dosage control. For this reason, these factors are best evaluated in a field demonstration. Temperature is another processing variable that was not investigated that can be important in the field.

1. 0.1:0.2:1.0	Portland:Firmix:Sediment
2. 0.15:0.15:1.0	Portland:Firmix:Sediment
3. 0.2:0.10:1.0	Portland:Firmix:Sediment
4. 0.05:1.0	Portland Cement:Sediment
5. 0.10:1.0	Portland Cement:Sediment
6. 0.20:1.0	Portland Cement:Sediment
7. 0.1:0.4:1.0	Lime:Flyash:Sediment
8. 0.40:1.0	Firmix:Sediment
9. 0.1:0.3:1.0	Lime:Flyash:Sediment
10. 0.50:1.0	Firmix:Sediment
11. 0.60:1.0	Firmix:Sediment
12. 0.1:0.6:1.0	Lime:Flyash:Sediment

Figure H13. Normalized organic carbon isotherms

54. Caution must also be exercised in extrapolating the desorption data to the field. The surface area for leaching in the field may be different from that in the serial, graded batch leach tests. Since the solidified/ stabilized sediment samples were ground, the surface area to mass ratio in the laboratory tests is probably higher than that in the field. However, the laboratory leach data are not necessarily conservative since the impact of grinding on contaminant mobility is poorly understood.

55. Chemical leach data from serial, graded batch leach tests and the methods of data analysis presented in this report were designed to provide a basis for evaluating the source term in permeant-porous media equations.

Permeant-porous media equations are mass transport equations that describe the generation of leachate as water percolates through a porous medium, such as solidified/stabilized sediment. Mass transport models with other assumptions and equations, such as the solid-phase diffusion approach (Cote and Isabel 1984), might also be applied to solidified/stabilized sediment and give reasonable results. The permeant-porous media model is probably a worst-case model, and the solid-phase diffusion model is probably a best-case model (Myers and Hill 1986). The lack of detailed field records, however, makes a definitive statement concerning the relative merits of the two approaches impossible.

Potential Implementation Scenarios

56. Solidification/stabilization technology can potentially be implemented in a variety of ways, depending on the design of the disposal facility and the manner in which the setting agents are added to and mixed with the dredged material (Francingues 1984). Two design concepts for disposal of the contaminated dredged material in an upland site are illustrated in Figures H14 and H15. Other designs and mixing concepts or modifications of those presented below may also be feasible.

Disposal site design

57. The layered concept shown in Figure H14 involves alternating layers of clean dredged material and contaminated dredged material that has been solidified/stabilized. The initial lift of clean dredged material would be dewatered to promote densification and consolidation to provide a low-permeability foundation. Once this layer has achieved the desired degree of consolidation, the solidified/stabilized dredged material would be placed on top. Conventional earthmoving equipment would be used for shaping as necessary before the solidified/stabilized material hardened.

58. One alternative to the layered design for a confined disposal facility is the liner concept. The liner concept incorporates solidification/stabilization as a treatment to produce a low-permeability foundation. A layer of solidified/stabilized dredged material is initially placed in the site; then, contaminated dredged material is disposed and dewatered. A clean layer of dredged material is used as final cover.

## CONCEPTUAL SKETCH FOR STRATIFIED DISPOSAL
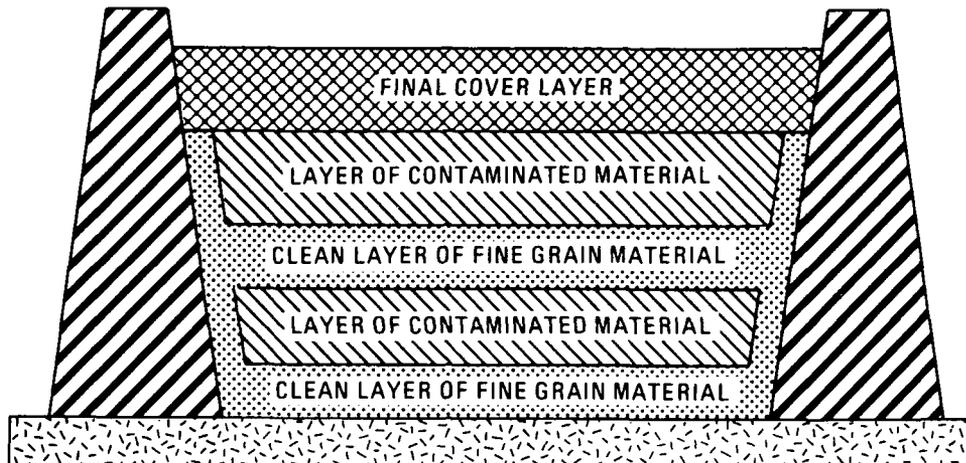
Figure H14. Disposal concept for alternating layers of
solidified/stabilized dredged material

59. The secure disposal concept shown in Figure H15 provides the
highest degree of environmental protection. A soil or flexible membrane
liner (or both) is used to line the bottom and sides of the disposal site. A
coarse-grain layer is used for leachate collection. Contaminated dredged
material that has been solidified/stabilized is then placed into the prepared
site so that a monolithic block develops as the material cures.

60. As an alternative to the secure facility, the liner and coarse-
grain layer could be deleted from the disposal site design if the permeability
and leachability of the solidified/stabilized dredged material are suffi-
ciently low. Laboratory permeabilities in the range of 10^{-11} to 10^{-5} cm/sec
have been achieved with solidification/stabilization of industrial waste
(Bartos and Palermo 1977). Soils with laboratory permeabilities of
10^{-7} cm/sec or less are considered for liner construction.

Addition and mixing methods

61. Three basic methods of agent addition and mixing are considered
feasible (Francingues 1984). These are in situ mixing, plant mixing, and area
mixing.

62. In situ mixing is suitable for dredged material that has been ini-
tially dewatered. In situ mixing is most applicable for the addition of large
volumes of low-reactivity setting agents. This method employs conventional

H46

DISPOSAL CONCEPT FOR STABILIZATION IN SECURE FACILITY
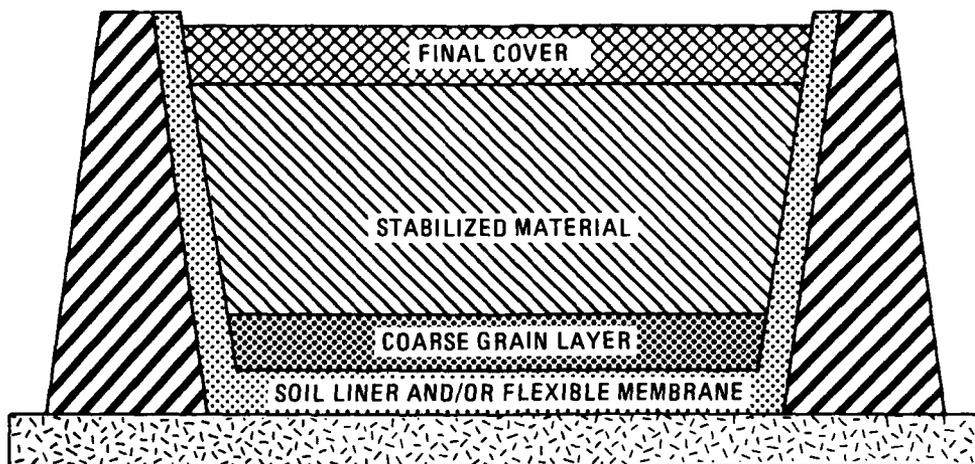
Figure H15. Disposal concept for solidification/
stabilization in a secure facility

construction machinery, such as a backhoe, to accomplish the mixing process. Where large containment areas are being treated, a clamshell dredge and/or draglines may be used. An alternative to conventional construction equipment involves agent addition and mixing by injection. Specially designed equipment that is commercially available can be used to inject and mix setting agents with the materials to be solidified/stabilized. The system moves laterally along the perimeter of a facility, solidifying the material within the reach of the injection boom. As soon as one pass is completed and the material has set long enough to support the injection carrier, the process is repeated. The equipment advances in this manner until the job is complete.

63. Plant mixing is most suitable for application at sites with relatively large quantities of contaminated material to be treated. In the plant mixing process, the dredged material is mechanically mixed with the setting agent(s) in a processing facility prior to disposal. If the volume of material to be processed does not justify the expense of a mixing plant, one alternative is to mix the setting agent(s) with the dredged material in a scow before it is unloaded. Mixing may be accomplished in route to a docking site, as shown in Figure H16, using a specially designed system mounted on the scow for this purpose, or by using a shore-based injection system, as shown in Figure H17. In the latter, track-mounted injection equipment would move along
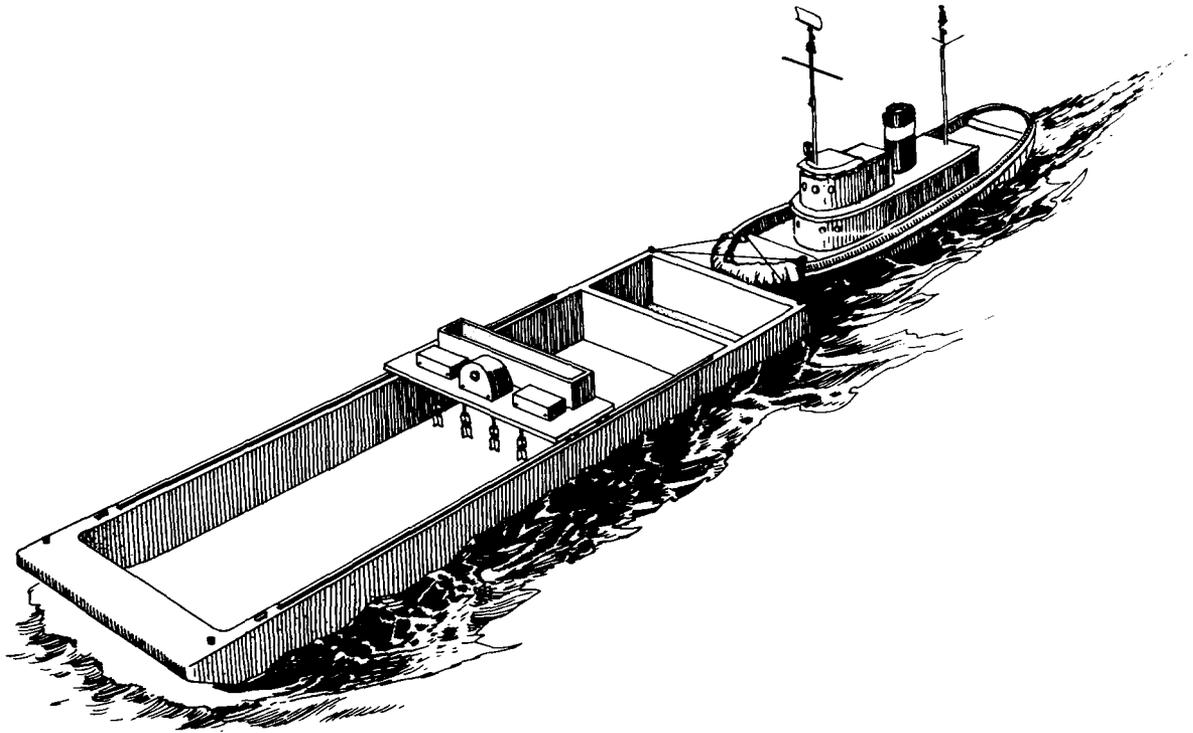
Figure H16. Conceptual sketch of scow fitted with mechanism for
mixing setting agents with dredged material
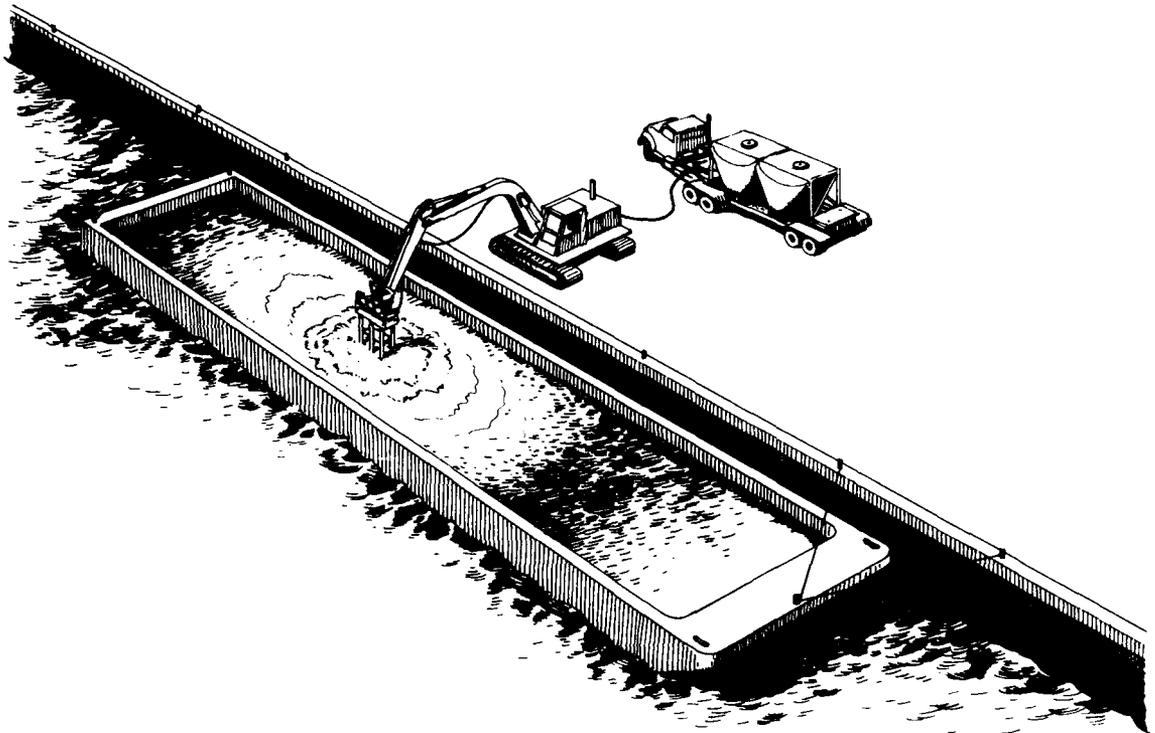
Figure H17. Conceptual sketch of shore-based mixing alternative

the dock and reach all parts of the scow. Solidifying agent in a dry state is piped directly from a tank truck to the injector. Since the setting process takes several days before freshly prepared, solidified/stabilized dredged material is hardened and cannot be rehandled, the risk of having the material set up before it can be removed from the scow is minimal.

64. Areawide mixing is applicable to those confined disposal sites where high-solids content slurries must be treated. Areawide mixing involves the use of agricultural-type spreaders and tillers to add and mix setting agent(s) with dredged material. Areawide mixing is land intensive and presents the greatest possibility for fugitive dust, organic vapor, and odor generation. Implementation of the areawide mixing concept will require that the dredged material be sufficiently dewatered to support construction equipment.

Cost

65. Actual project cost data are not available for solidification/stabilization of dredged material. Application of the technology to hazardous waste is estimated to cost $30 to $50 per ton (Cullinane 1985). The actual cost will vary with the amount of setting agent(s) required. The amount of setting agent(s) required depends on the implementation strategy and the performance criteria that are specified. Cost estimates must also take into consideration the volume increase due to the addition of setting agents(s) and future expenditures needed for end uses anticipated at the site. The cost-effectiveness of solidification/stabilization technology as an alternative to liners and leachate collection, treatment systems, or other ground-water pollution control strategies for upland disposal sites depends on the site-specific environmental constraints that are placed on disposal.

Conclusions

66. The range in 28-day UCS was 35 to 605 psi, depending on the agent(s) used for solidification and the dosage applied. The maximum strength recorded was 1,176 psi at 90 days. This range in product strength is indicative of the versatility of solidification as a physical stabilization process for Everett Bay sediment. The technology has the flexibility to meet

specifications for physical stability ranging from primarily immobilizing sediment solids in a low-strength product to producing a material suitable for end uses typical of soft concrete.

67. Solidification/stabilization reduced the leachability of selected metals. Arsenic and zinc were completely immobilized by the processes included in this study. Depending on the process and process formulation, 93 percent or greater of the cadmium, chromium, and lead in the solidified/stabilized sediment was resistant to leaching. Analysis of the leachate data indicates that solidified/stabilized Everett Bay sediment does not have a significant leaching potential for metals.

68. Solidification/stabilization did not significantly alter the sorption capacity of the sediment for organic carbon. Data were not available to evaluate the potential of solidification/stabilization technology to reduce the leachability of specific organic compounds.

69. Solidification/stabilization technology can be implemented in a variety of ways. The implementation strategy and the performance criteria selected impact cost. The cost-effectiveness of solidification/stabilization technology as an alternative to other leachate control strategies depends on the site-specific constraints for upland disposal.

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APPENDIX I: MONITORING PLANS

1. This appendix contains draft monitoring plans for dredging and dis-
posal operations for the Everett Homeport project. Separate plans are
included for dredging operations, contained aquatic disposal (CAD) placement,
contained aquatic disposal mound and cap behavior, and intertidal disposal.
The level of detail in the plans is intended to provide general guidance on
monitoring and the level of effort involved in the monitoring. Since some of
the alternatives for dredging and disposal are still under development, these
plans cannot be considered final and must be refined once final scheduling and
design for the project have been completed. A panel to include experts famil-
iar with local conditions should be formed to assist in refining the plans.
These monitoring plans have been revised from those presented in the Disposal
Alternatives report to reflect more recent information on the proposed
alternatives.

2. The objectives of the monitoring plans given here are the following:

a. To determine the degree of sediment resuspension at the point of
dredging during representative dredging operations.

b. To verify modeling predictions of dredged material behavior to
include mass release during open-water disposal for the CAD
alternative.

c. To determine the area of deposition of dredged material on the
bottom following each phase of disposal for CAD.

d. To determine the cap thickness immediately following disposal
and after initial consolidation for CAD.

e. To determine the effectiveness of the cap in chemically isolat-
ing the contaminated sediments for CAD.

f. To determine contaminant releases from effluent, surface runoff,
and leachate for confined upland or intertidal alternatives.

Since CAD is identified as the preferred alternative and designs for CAD have
been proposed, the monitoring plans are more detailed for CAD.

Biological Monitoring

3. The monitoring plans described here are restricted to physical and
chemical parameters. It is recognized that biological monitoring should be
considered as a part of the overall monitoring effort. Biological monitoring
should reflect the concerns of resource agencies and should be developed in

cooperation with biologists familiar with local species and conditions. Plans for biological monitoring can be finalized once a disposal alternative and final site design have been selected.

Monitoring Plan for Dredging Operations

Purpose and scope

4. The purpose of this monitoring plan is to define the sediment resuspension and contaminant release of a dredge plant operating in contaminated sediments. The plan is oriented toward clamshell dredging, which is the preferred method for the CAD alternative. The monitoring effort will identify the resuspension of sediments generated by the dredging operation and any possible release of contaminants from the sediment to the water column. A sample grid near the dredging operation will be defined where samples and measurements of the resuspended sediment plume will be collected. Discrete water samples, current measurements, and other parameters will be obtained at the sample grid points. The intent of this plan is to intensely monitor representative dredging operations over a 2-day period. The procedures described in this section are not intended for routine use throughout the entire dredging project.

Sampling procedure

5. <u>Sampling locations.</u> There will be 1 day of background sampling followed by 2 days of sampling during the dredging operation. The sample grid will be completed three times during each sampling day. Each sample set will be sampled in the same order as the previous set, such that the first station sampled on the first set will be the first station sampled on the second set. Background sampling will be done prior to the start of dredging and will include water samples for total suspended solids (TSS) determination and current measurements to describe the hydraulic regime of the area to be dredged.

6. The sample grid will consist of 10 sampling stations arranged in two perpendicular transects. The first transect will be parallel to the direction of flow in the area to be dredged with seven sampling stations located at geometrically increasing distances from the point of dredging. Stations will be located 100, 200, 400, 800 and 1,600 ft downcurrent from the point of dredging. One station 100 ft upcurrent from the point of dredging and a station on the dredge nearest the point of dredging will complete the first transect.

The second transect will be perpendicular to the first and located 200 ft downcurrent from the point of dredging. It will consist of three stations. A sketch showing the grid is presented as Figure I1.

7. Water column samples for suspended solids. At each sampling station, discrete water samples will be collected at the near-bottom (1 to 5 ft above bottom), middepth, and near-surface (1 to 5 ft below the surface). These water samples will be analyzed for TSS only, and should be of sufficient volume (approximately 200 ml) to perform the analysis.

8. Current measurements. After background data have established the general flow pattern, current measurements will be collected throughout the sample collection effort at the 100-ft upcurrent station, the 400- and 1,600-ft downcurrent stations, and the three stations that comprise the second transect. The current measurements will be obtained at similar depths (surface, middepth and near bottom) as the water column samples.

9. Water column samples for chemical analysis. On the first day of sampling, during the dredging operation, water samples will be collected for water quality analyses. The samples will be collected at four of the stations along the first transect: 100 ft upcurrent of the point of dredging, at the station nearest the downcurrent side of the point of dredging (either on the barge or 100 ft downstream), and at the 200- and 400-ft downcurrent stations. This sample set will be collected once at each station except for the first station downstream from the dredge, which will be sampled three times during the day. The water quality samples will be collected at the near-surface, middepth, and bottom at each station. Three replicates from each sampling depth will be obtained by sequential sampling at each depth. Each sample replicate will be of sufficient volume for the chemical analyses to be performed.

10. Labeling and field log. For the plume sampling, there are 10 sampling stations. A sample number consisting of four components will be assigned to each sample. The four components are: date, station, depth, and time. The date will be represented by a two-digit number depicting the day of the month. The station portion of the sample number will be assigned sequentially, such that the 100-ft upcurrent station will be 01; the station on the dredge, 02; the station 100-ft downstream, 03; and the rest as shown in Figure I1. The depths will be similarly numbered, 1 for surface, 2 for middepth, and 3 for bottom. The sampling time will be incorporated such that, for a sample collected on the first day of the month and at 0800 hr at the 200-ft
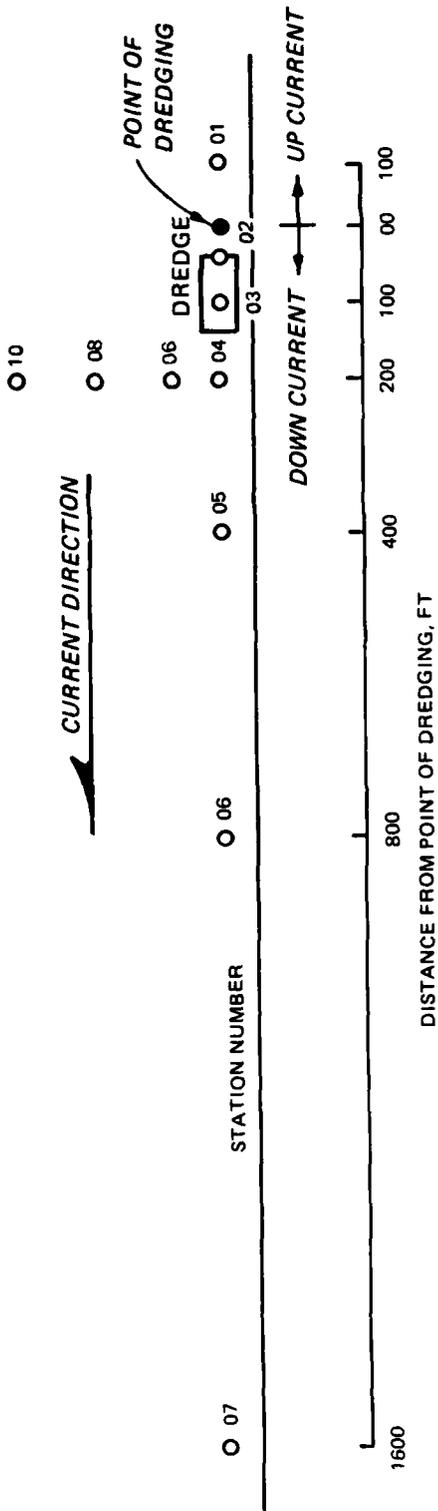
Figure I1. Sample grid for monitoring of dredging operations, Everett Homeport

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downstream station, the sample number would be 01-04-02-0800 if it were obtained at middepth.

11. A field log will be kept to outline sampling procedures and identify each sample. The field log will be arranged into sampling days. Each sampling day will begin by recording the names of the persons collecting the samples, a description of the weather condition (approximate wind speeds and direction, etc.), and a description and/or sketch of the dredging operation for that day. Each time the dredge makes a significant movement, such as changes in position in the channel, it will be recorded in the field log. Each sample will be identified by sample number, depth, time, and distance from the point of dredging. Other events recorded each day will include: cycle time of the dredge bucket, current measurements, any interruptions of the dredging operation, water temperature, any ship movement in the vicinity of the field study, and any other event the data recorder feels to be pertinent to the field study. Similar procedures for labeling and field logging should be used in other portions of the monitoring.

Laboratory testing

12. Total suspended solids. All the discrete water column samples will be analyzed for TTS in accordance with the AWWA-WPCF-PHS Standard Methods (total of 250 samples).

13. Chemical analysis of water column samples. All water quality samples collected at the station immediately downstream from the dredging operation (total of 27) will be analyzed for TSS, dissolved chemical concentrations (filtered or centrifuged subsamples), and total chemical concentrations. A dissolved sample will be defined as that passing 0.45-μ filters. This will yield a total of 54 water samples for chemical analysis. Both the total and dissolved subsamples will be analyzed for metals, nutrients, PCBs, and PAHs. A list of specific parameters for analysis will be provided by the Seattle District. The remaining water quality samples (27) will be split; subsamples will be filtered or centrifuged, preserved, and retained for possible later chemical analysis.

Report

14. The contractor will summarize the data collected in a report to include tables of all test results, descriptions of the test procedures used, copies of sample logs and field notes, and any other information pertinent to the sampling and testing.

Monitoring Plan for Dredged Material Placement
for the CAD Alternative

Purpose and scope

15. The purpose of this monitoring program is to determine actual dis-
position of dredged material during disposal for the CAD alternative and to
verify mathematical models used to predict such behavior. Verification of
modeling assumptions regarding the behavior of material during descent to the
bottom, surge along the bottom, and initial transport through diffusion will
be accomplished by intensely monitoring several barge dumps using arrays of
instrumentation in the water column and on the bottom. The area of deposition
following each phase of disposal will be determined by comparisons of bathy-
metric surveys taken before and after each phase of disposal, supplemented by
data from instrumentation on the bottom. The monitoring program outlined
could be applied with modifications to most coastal dredged material disposal
sites possessing similar water depths and maximum currents.

16. The data to be collected are needed to characterize the disposal
site and the properties of the material in the disposal vessel, as well as to
describe the descent of the material as it falls through the water column,
spreads over the bottom as a density current, and finally is transported by
the ambient current while undergoing turbulent diffusion. The instrumentation
required to accomplish the monitoring program, as well as the placement of
instruments around the disposal point, is described below. It is assumed that
disposal will be from bottom-dump scows. If a different dredging method is
selected, appropriate modifications to this plan must be made.

Field data collection program

17. To provide insight into the fate of dredged material disposed at
the designated disposal site as well as to furnish data for verifying math-
ematical models, field data must be collected throughout the placement pro-
cesses that occur during several disposal operations* and for a short period
of time after each operation. A major problem that must be overcome stems
from the fact that dredged material placement occurs through a series of rapid

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* For purposes of this monitoring program, a "disposal operation" is defined
   as the filling, transport, and subsequent release of a single load of
   dredged material.

three-dimensional processes that may be quite difficult to observe. The requirement for rapid and continuous observations of dredged material placement can best be met by optical transmittance and acoustic and water flow measurements.* Both continuous observations at one location and observation profiles through the water column must be made. Comparison with suspended solids concentration measured in simultaneously taken water samples will ensure reliability of transmissometer calibration. A survey echo sounder can be used to track dredged material through the water. If the boundary between the ambient water and water containing dredged material is a sharp one, the sounder permits flow velocities and layer thicknesses to be measured. Flow velocities of dredged material can also be measured directly with standard current meters. These methods of measurement will be used simultaneously during each disposal operation monitored.

Instrument requirements

18. <u>Transmissometers.</u> The requirements of the transmissometer design are mechanical rigidity and sufficient strength to withstand forces encountered during the release of dredged material. It is also necessary that the instruments operate at much higher sediment concentrations than are usual for optical methods. A total of six transmissometers must be used simultaneously during the monitoring program.

19. <u>Acoustic transducers.</u> Acoustic pulses of 200-kHz frequency return good echoes from small concentrations of fine-grain sediments. Based upon work by Proni et al.,** standard echo sounder equipment should suffice to detect the presence of dredged material. For example, Raytheon survey fathometers operating at 200 kHz with an 8-deg cone angle might be used. A total of nine transducers must be used simultaneously during the monitoring program.

20. <u>Current meters.</u> Fluid flow measurements are needed to determine the background current at the disposal sites and to record the velocity of the bottom surge and the speed of descent of the dredged material. Measurements of speed and direction of the background current can be made with an Endeco

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* H. J. Bokuniewicz et al. 1978. "Field Study of the Mechanics of the Placement of Dredged Material at Open-Water Disposal Sites," Technical Report D-78-7, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

** J. K. Proni et al. 1976. "Acoustic Tracking of Ocean-Dumped Sewage Sludge," <u>Science,</u> Vol 193, pp 1005-1007.

current meter, or equivalent, mounted on taut moorings at the desired distances above the bottom. Several types of flowmeters could be used to measure the speed of flow in the bottom surge, e.g., a standard Price meter of the type designed to measure flow in rivers. At least one current meter and seven flowmeters must be used simultaneously during the monitoring program.

21. Survey equipment. The monitoring program includes detailed bathymetric surveys. A Ratheon survey echo sounder, or equivalent, could be used.

22. Water pumps. Submersible electric pumps with a capacity of at least 0.01 m^3/min must be used to collect water samples during each disposal operation. At least six pumps must be used simultaneously during the monitoring program.

23. Range and bearings. The positions of observing points around the scow should be determined by electronic positioning equipment similar to Loran C positioning system or better. This equipment should be calibrated using fixed range markers and coordinates from navigational charts. Ranges can be taken with an optical range finder, and bearing compasses can be used as a field check on the electronic positioning.

24. Deposition samplers. Alternatives are available to measure the extent of depositions occurring from disposal activities. For example, one type sampler may consist of sediment collection vessels mounted at multiple levels on a tripod that will rest on the bottom. The lower vessels will reflect accumulation of material reaching the samplers due to the bottom surge. The uppermost vessel will reflect only the deposition of material due to transport-diffusion. A diagram of the sampler is shown in Figure I2. (This sampler is identical to that used by Mr. Glenn Earhardt, Baltimore District, in similar studies.) As a supplement or alternate, a sediment profiling camera such as REMOTES (Remote Ecological Monitoring of the Seafloor), or comparable, can be used to measure the thickness of the deposited sediments. Use of deposition samples is critical in measuring the extent of thinner layers of deposited material that would not be observable by surveys.

25. Sediment sampler. The properties of the dredged material in the barge are required for each disposal operation monitored. To determine properties of the material at various vertical locations in the barge, a syringe mounted on a long pole with the piston pointing up can be used. With this configuration, no material will enter until the syringe is at the desired
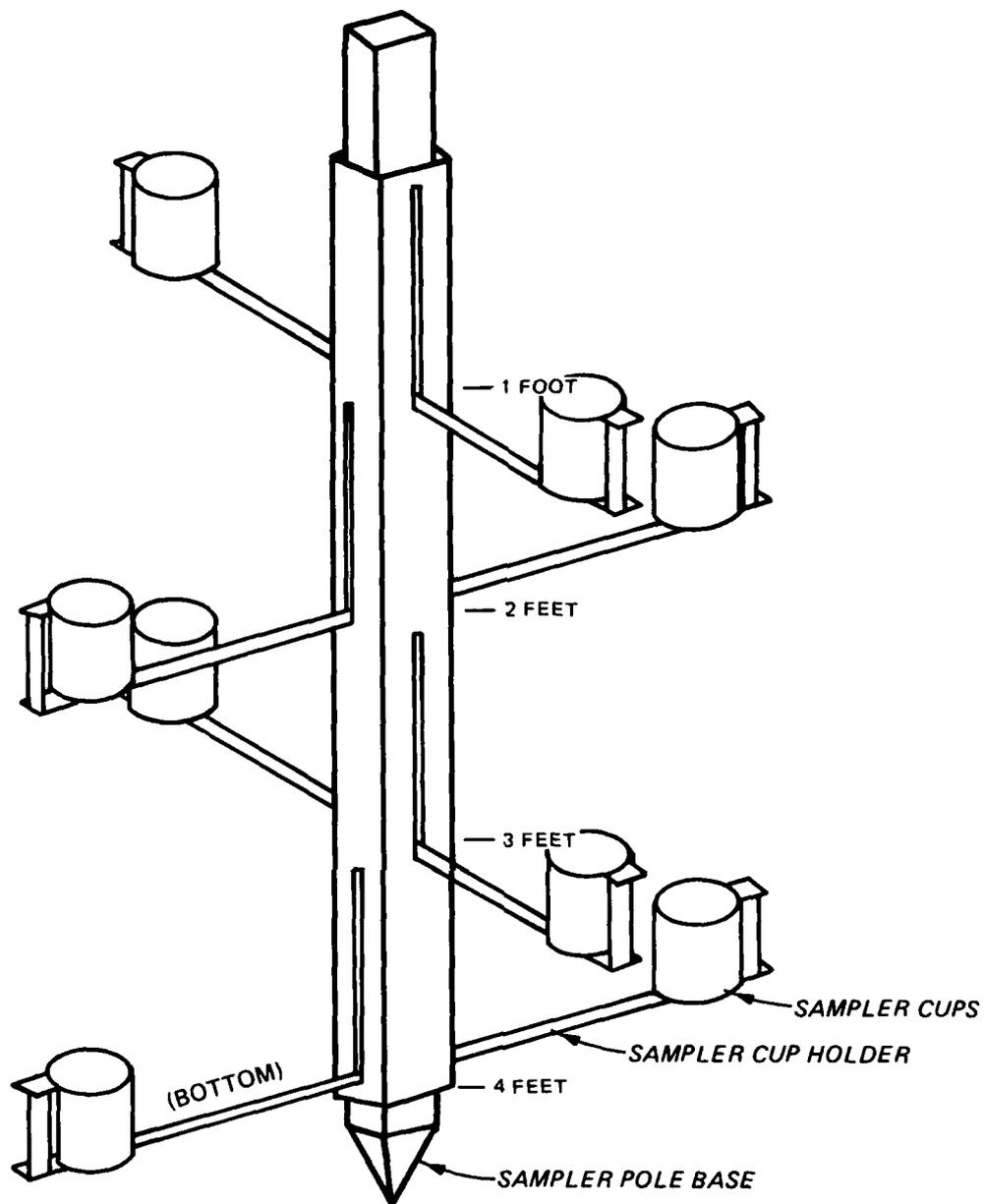
Figure I2.   Suggested deposition sampler, Everett Homeport

depth and the piston is pulled.   Samples of the dredged material from the
surface can be taken with a scoop.

26.   _Timed camera._   A stationary camera with time-lapse capability will
be used to record the filling of the barge and the subsequent release of
dredged material from the barge during each disposal operation monitored.   A
scale will be attached to the inside wall of the barge so that estimates of

volumes and rates of filling and release can be determined from the
photographs.

27. <u>Observation boats.</u> At least seven observation boats will be used
simultaneously during the disposal operation sampling period. The boats
should be large enough to accommodate three crew members, who will handle
equipment and record data, plus all necessary equipment. The observation
boats will serve as a working platform for the crew and should be stable under
expected working conditions. The boats should also be able to anchor in the
water depths anticipated at the site and be equipped with electronic
positioning equipment.

Description of disposal
operations to be monitored

28. The disposal barge will be stationary during the monitoring opera-
tion. A range of disposal operations consisting of varying volume and dredged
material possessing different sediment and water content should be monitored
(if applicable). In addition, disposals should be conducted at different
times in the tidal cycle, reflecting the maximum and slack current velocities
during the flood and ebb tides, and in different water depths (if applicable).

Data collection phases

29. Major factors affecting the short-term fate of dredged material
disposed in open water are the disposal site characteristics, the properties
of the disposed material, and the type of disposal operation. Data concerning
each factor must be collected. The behavior of the material can be separated
into three phases: convective descent, during which the dump cloud or dis-
charge jet falls under the influence of gravity; bottom collapse, occurring
when the descending cloud or jet impacts the bottom; and passive transport-
diffusion, commencing when the material transport and spreading are determined
more by ambient currents and turbulence than by the dynamics of the disposal
operation. Data describing the movement of the dredged material through each
of these phases will be collected.

30. <u>Bathymetry.</u> Bathymetric surveys will be obtained prior to disposal
and after the entire volume of dredged material has been placed in each phase.
Phases to be surveyed include the berm (if used), first contaminated mound,

first cap, second contaminated mound, and second cap. Other supplemental surveys would be desirable to determine progress during each phase.

31. The predisposal survey is to establish existing depth gradients and to serve as "prehistory" of the site prior to initial disposal. The post-disposal surveys will be used to help determine mound configuration and sediment volumes.

32. Disposal site characteristics. Current velocity and direction data from at least one station will be collected during the sampling period. Such data can then be converted to a local velocity field through a ratio of water depths. A sufficiently large density gradient in sufficiently deep water can result in arrest of the descent phase. Therefore, the vertical density profile at the time of maximum flood, ebb, and slack-water current velocities will be obtained at the deepest point in the disposal site. This will require the collection of salinity and temperature data.

33. Properties of dredged material. Data must be collected concerning the properties of the dredged material in the barge prior to all disposal operations that are monitored. Timed photographs should be taken as the barges are filled during dredging. Samples of dredged material, for subsequent laboratory analysis, must be taken from the barges with the syringe sampler previously discussed. In most cases the material will not be uniformly distributed over the depth; therefore, samples should be taken at the surface, at middepth, and near the bottom. These samples will be analyzed for the following parameters: moisture content, Atterberg limits, bulk density, specific gravity of solids, void ratio, and the particle size distribution. Chemical composition should also be determined.

34. Point of discharge. Control of the point of discharge will be important throughout the disposal operation. Appropriate control for the point of discharge will be specified in the plans and specifications and will be used to establish the points of discharge during the monitoring. Control for the point of discharge could be established by prelocated taut-line buoy, electronic positioning with onboard computer printout, or other appropriate means. The disposal barge during placement of contaminated sediments should be stationary during the release phase for each dump. This will assist in keeping the dredged material mass in a clumped condition for descent.

35. Disposal operation data. The quantity of material and the mode of operation of the bottom-dump doors must be provided for each disposal

operation monitored. Information concerning the time required to complete the discharge of material from individual barges as well as the time required for complete discharge is essential. In addition, the location of the doors below the water surface, the distance from the doors to the center of gravity of the dredged material, and the dimensions of the doors must be furnished. The rate of emptying of the barges can be determined by taking a series of timed photographs of the barges during discharge. Water level measured against a scale photographed in place in the barges can then be converted to volume of material with the aid of calibration curves available from builder's drawings. Timing of events during the monitoring efforts should be based on the time at which the scow doors are first opened. Observers should be placed on the scow to call or signal the time of discharge.

36. **Descent data.** Processes that occur during the descent of dredged material through the water column determine the impact velocity at the bottom, the location of the impact point, and the amount of material that reaches the bottom. Field observations using transducers and a flowmeter are intended to yield information on the descent velocity, size, and entrainment of the descending cloud or jet. The instruments to provide these data may be deployed as shown in Figure I3.

37. Release of much of the dredged material in the form of cohesive blocks or clods will occur if the material in the barges is cohesive and the water content is low. Evidence on the formation of clods during the release of the material must be provided. This can be obtained by either taking bottom photographs under the disposal vessel immediately after the disposal operation, through acoustic data, or both. A transducer looking downward alongside the disposal vessel will be used to detect the presence of clods during free-fall.

38. Detailed information on the descent of the dredged material will be obtained with transducers and flowmeters. The transducers should be used to produce beams directed downward, upward, and sideways. From the transducer data, the speed of the descending cloud or jet can be determined. The speed of the descending jet of dredged material will also be measured with a flowmeter. A low threshold propeller should be used to enable the measurement of flow velocities from almost zero to perhaps 3 to 4 fps. The flowmeter could be attached alongside the transducer as shown in Figure I3.
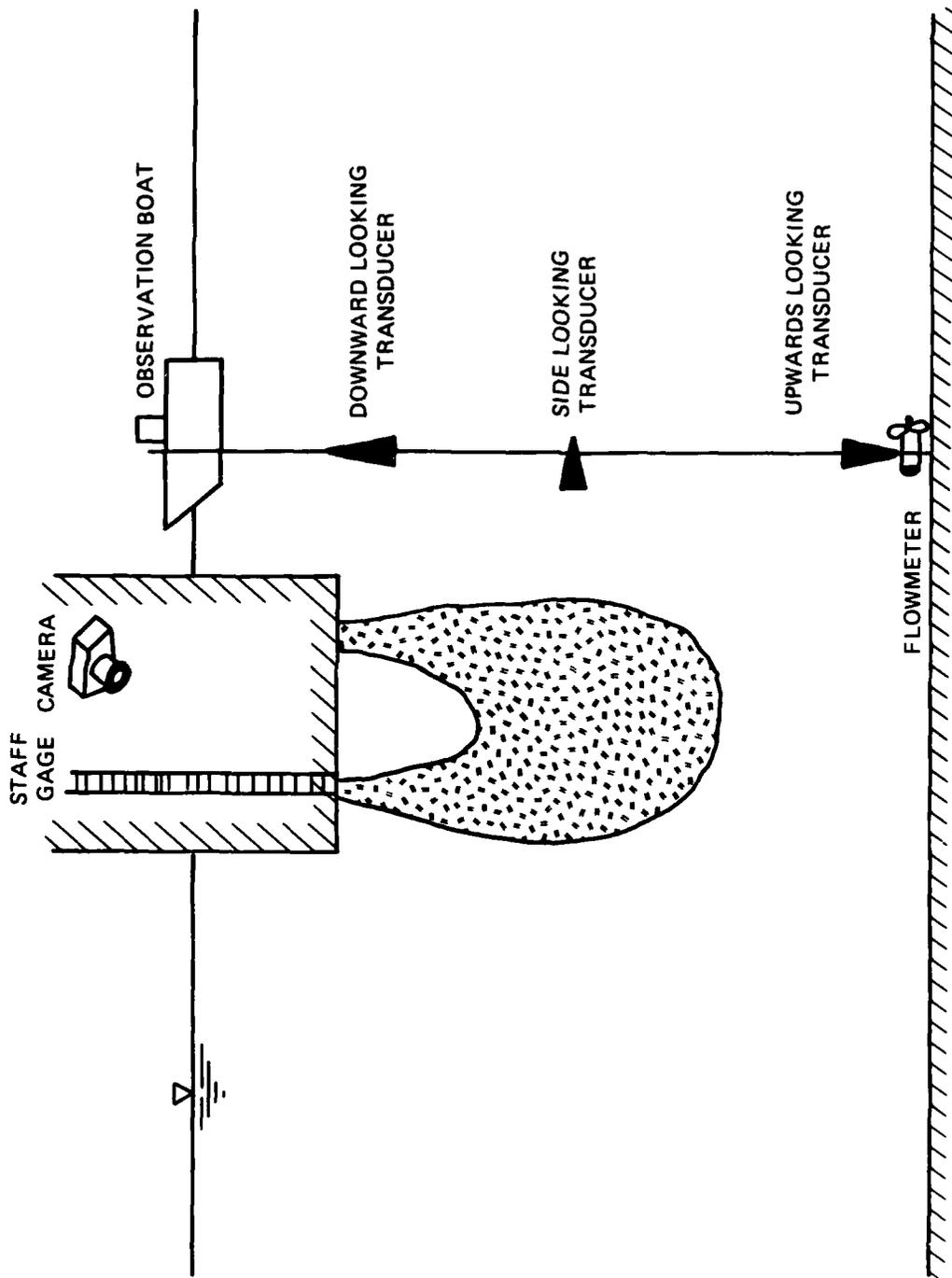
Figure 13. Typical configuration of instruments to provide data on the descent phase, Everett Homeport

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39. _Bottom surge and spread data._ Impact of the descending jet or cloud with the bottom deflects the flow of dredged material and entrained water to form a surge or density current that spreads away from the impact point. The surge spreads radially outward with both its thickness and speed decreasing as its radius increases. The entrainment of ambient water into the surge and friction eventually cause the velocity of the surge to decrease to the point where much of its contained sediment is deposited. The initial energy of the surge and the rate of energy dissipation determine the range of the surge, as well as the area of the bottom that will be covered by dredged material, the form, and the thickness of the deposit. To adequately describe the bottom surge it is necessary to know its velocity as a function of distance from the impact point, its thickness, and the concentration of solids contained. The rate at which the leading edge of the surge spreads outward from the impact area can be determined by noting the time at which the spreading surge of dredged material arrives at a number of stations various distances from the disposal vessel. Since the bottom surge resulting from the disposal of dredged material can be expected to spread over several hundred feet, the distribution of stations shown in Figure I4 will be used. Since the disposal is made over an essentially flat area of the disposal site, the surge should be symmetrical about the impact point. The station located 200 ft upcurrent of the descent impact point will be used to confirm this.

40. At each station, the arrival time of the surge will be detected with a transmissometer, a 200-kHz acoustic transducer, and a flowmeter or a bottom-mounted recording current meter. A typical configuration of instruments required to characterize the bottom surge is shown in Figure I5. The instruments must be secured in such a way as not to be displaced or damaged by the bottom surge.

41. The thickness of the surge and the change in thickness in time will also be measured by the acoustic transducers. Because of the suspended solids, the fluid in the bottom surge should return a good echo of the 200-kHz acoustic pulses.

42. To monitor the concentration of suspended sediment in the bottom surge as well as the suspended sediment concentrations in the transport-diffusion phase, both transmissometers and water samples collected with submerged pumps will be employed. The transmissometers and pumps should
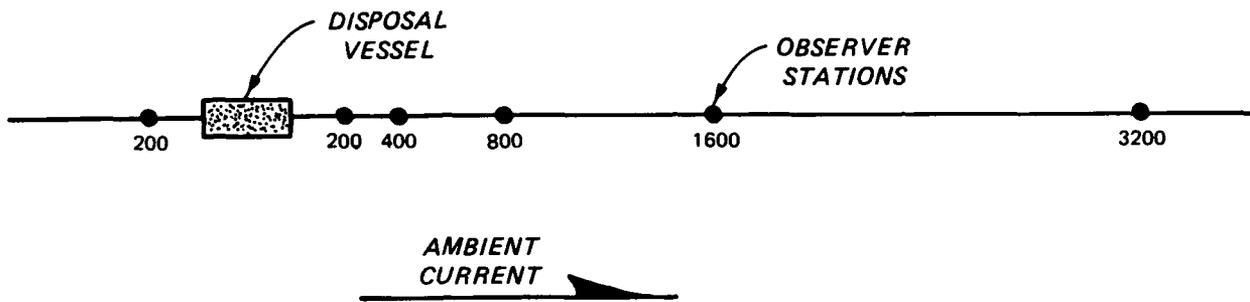
Figure I4. Distribution of deposition samplers, Everett Homeport
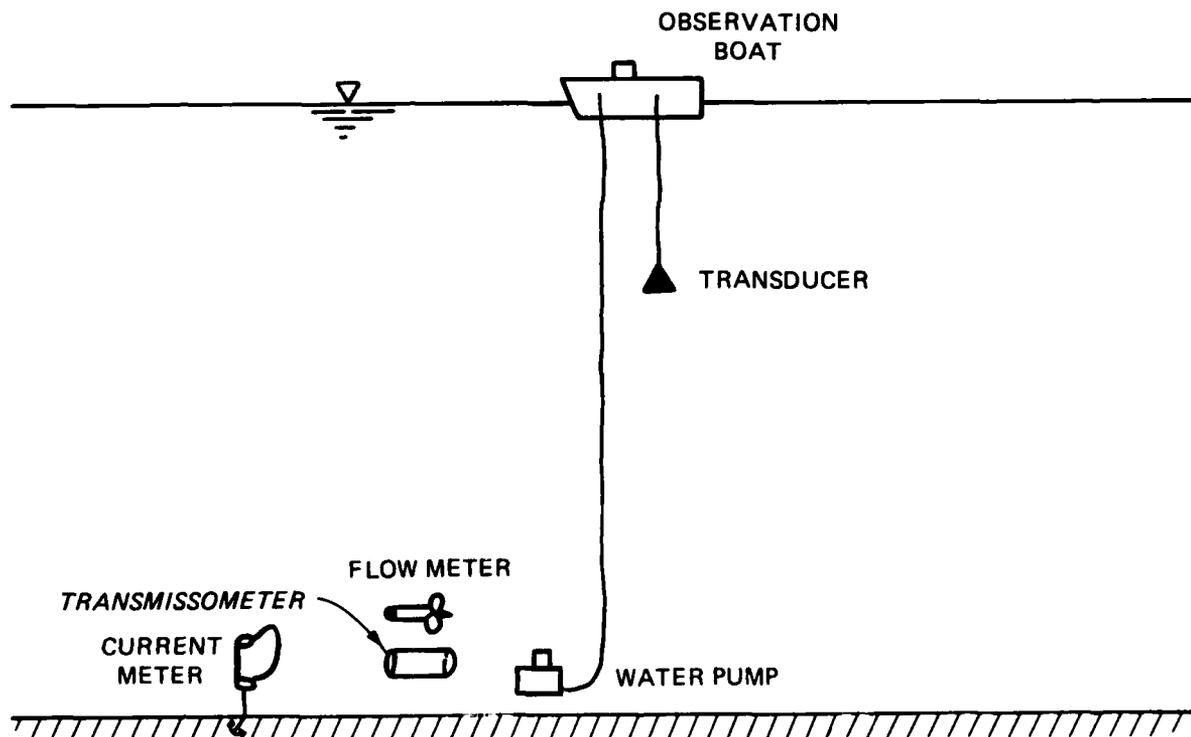
Figure I5. Data collection stations for observing the bottom
surge and transport-diffusion of the suspended sediment cloud,
Everett Homeport

initially be stationed about 2 ft above the bottom and continuously pump water to the observer boat above for purposes of monitoring the surge. Discrete water samples should be collected at the 200- and 400-ft stations at 30-sec intervals for the first 3 to 5 min, and every minute thereafter until the surge has passed. Water samples obtained simultaneously with transmittance readings should provide a check on the transmissometer calibration, and will be be particularly useful if the sediment concentration is too large to be measured by optical methods. The solids content of the water samples can be determined by filtration through millipore filters followed by weighing of the dried sediment. The bottom surge phase of the disposal operation should be over approximately 15 min after its initiation. Additional sample volumes for water quality should be taken at the 200-ft station during this period.

43. Transport-diffusion data. To provide information on the longer term of transport and diffusion of the suspended sediment cloud remaining after the energy of the bottom surge has been dissipated, sediment concentration and cloud thickness data should continue to be collected at all stations until the next disposal event. During this period, alternating transmissometer readings and water samples should be collected. The data should be obtained throughout the water column at near-surface, middepth, and near-bottom. A sampling interval of 3 to 5 min would probably be sufficient.

44. Deposition data. Deposition samplers should be installed or sediment profile samples collected at the same locations shown on the grid in Figure 14 to determine the quantity and distribution of settling from the disposal operation. A bathymetric survey of the dredged material mound should also be obtained at the time of the deposition data collection.

Water quality samples

45. Samples for water quality analysis will be collected at the station nearest the downcurrent side of the point of disposal. The water quality samples will be collected at the near-surface, middepth, and bottom at each station. Three replicates from each sampling depth will be obtained by sequential sampling at each depth. Each sample replicate will be of sufficient volume for the chemical analyses to be performed, including TSS, dissolved chemical concentrations (filtered or centrifuged subsamples), and total chemical concentrations. Dissolved samples will be defined as that passing 0.45-μ filters. This will yield a total of nine water samples for chemical

analysis for each disposal operation monitored. Both the total and dissolved subsamples will be analyzed for metals, nutrients, PCBs, and PAHs. A list of specific parameters for analysis will be provided by the Seattle District.

Data analysis and report

46. All data collected by the contractor will be furnished; however, the contractor will also analyze the data to provide the following information in either graphic or tabular form for each disposal operation monitored:

<u>a</u>. Water depths over the disposal site and a description of the relative roughness of the bottom.

<u>b</u>. Magnitude and direction of ambient current as a function of time and position in the water column at the background current station. The water depth at the current station must be provided.

<u>c</u>. Vertical profile of ambient density at maximum flood and ebb current velocities and slack-water periods of the tidal cycle.

<u>d</u>. Amount of dredged material disposed in each disposal operation, bulk density, vertical variation of density in the hopper, grain-size distribution, void ratio, and Atterberg limits of the material in the hoppers or scow. Drawings of the disposal barge showing the bottom doors and a detailed narrative describing the actual disposal operations, e.g., time required for disposal to be completed, etc. In addition, visual observations of the wind and sea conditions should be provided.

<u>e</u>. Time required for the disposed cloud or jet of material to strike the bottom, its growth while falling through the water column, its velocity at bottom encounter, an estimate of the amount of solids that falls as clods, and the average fall velocity of these clods must be provided.

<u>f</u>. Time history of the radial spreading of the bottom surge and a time history of the flow velocity, surge thickness, and suspended sediment concentrations at each of the stations.

<u>g</u>. Thickness of deposited material obtained from the deposition samplers. In addition, from the bottom photographs and the resurvey information, the volume of material deposited.

47. A written report describing the monitoring will be prepared, to include narrative descriptions of the conditions during monitoring, equipment utilized, monitoring techniques employed, results, and any other data pertinent to the monitoring effort.

Summary

48. The fate of dredged material released at an open-water disposal site is determined by disposal site characteristics, properties of the material, and the nature of the disposal operation. The objective of this

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monitoring program is to follow the path of the dredged material, to determine how much material reaches the bottom, in what form, and how long it takes for the placement processes controlled by the factors above to go to completion. Results from the field data collection will provide quantitative information on how much material will be retained in the site from individual disposal operations and the distribution of that material on the bottom. In addition, the detailed data collected during the descent, bottom collapse, and transport-diffusion phases will aid greatly in the calibration of mathematical models for predicting the short-term physical fate of dredged material during open-water disposal operations.

Monitoring Plan for Mound and Cap Behavior

General

49. This plan is intended to provide data for determining the final cap thickness immediately following disposal and after initial consolidation, and the effectiveness of the cap in chemically isolating the contaminated sediments. This will be accomplished by physical and chemical analysis of core samples taken through the cap at various time intervals. Information on material type, density, and void ratios must be obtained at various times before, during, and after the dredging and subsequent disposal and capping operations to quantify the amount and condition of materials involved. The monitoring effort would be similar to that carried out for the recent capping demonstration project on the Duwamish Waterway. Determination of the materials' in situ engineering properties over time is necessary. Also, chemical analysis of the sediments and the pore water will yield information on possible contaminants and any discernible migration of these contaminants through the cap into the water column. Several types of activities are necessary to obtain the required information.

50. In situ samples of the sediments must be obtained before dredging, during storage/transport in the barge, and at several times after placement at the disposal site. Core borings of the sediment/dredged material will provide information concerning types of materials involved in this disposal operation; this information will be useful in predicting anticipated behavior of the material and in interpreting and understanding observed field behavior, i.e., rate of consolidation and possible erodibility of the sediments. Sampling

will also provide data on void ratios/densities of the material at various times during the dredging/disposal operation; this will allow determination of the (average) effect of various dredging/disposal activities on sediment characteristics. Void ratio data will provide needed information about the conditions existing when consolidation begins.

Sampling and materials

51. Portions of the sampling requirements may be covered in other monitoring plans or sufficient data may be available from previous samples. However, all required sampling is discussed in this monitoring plan. Samples will be taken at selected locations within the contaminated shoal to be dredged within representative transport barges and at the disposal site. All core samples will be taken with a Vibracore, or equivalent, core sampler. A 20-ft vibracore sample, or a shorter sample if refusal is reached before 20 ft, will be taken at each sampling location. Within the barge, grab samples will be taken during barge loading. Portions of all samples taken prior to disposal operations will be available for chemical analysis, as deemed necessary by sediment chemists. Samples taken subsequent to disposal will be collected for the dual purposes of geotechnical and chemical analysis.

52. Vibracore samples of the foundation soils will be obtained from the disposal site before the disposal operation begins. Vibracore samples will be obtained at stations corresponding to these shown in Figure 13. The borings should be centered in the disposal site in the upslope to downslope direction. These samples are necessary for delineation of foundation materials from dredged material in future borings collected at the disposal site. Prior knowledge of the foundation material to be expected at the disposal site will be invaluable in identification of the foundation/dredged material interface, particularly if any intermixing of materials occurs during disposal or sampling operations.

53. After placement of both the contaminated material and the capping material, core borings will be taken at specified time intervals to provide profiles of engineering properties. This will provide a means of monitoring any changes in the capped site in both the spatial and time dimensions.

54. Initial samples at the capped site will be taken utilizing the Vibracore sampler. Whether or not this sampler is used for future core borings on this project is dependent upon (a) quality of the samples obtained initially from the capped site and (b) continued availability of the

equipment. Twenty-foot samples will be taken at locations selected to correspond with settlement plates that will have been placed in the disposal site before sampling occurs. Vibracore samples will be taken of locations. The schedule for sampling should be: immediately after cap placement and then at 6, 12, and 18 months after cap placement.

Laboratory testing (geotechnical)

55. The vibracore borings will be visually inspected and photographed soon after completion of the sampling operation. Portions of each boring will be selected for laboratory testing. Soil classification will be determined for each sample; testing will include water content, Atterberg limits, specific gravity, and grain-size distribution (hydrometer and/or sieve analysis). Consolidation tests will also be performed on selected samples. The number of samples selected for testing will be dependent upon results of the visual examination of the cores.

Settlement plates

56. Deployment and monitoring of settlement plates in the mound is desirable to differentiate between mound consolidation and mound erosion. Designs for settlement plates, monitoring requirements, diving plans, etc., were necessary for similar mound monitoring conducted at the Duwamish demonstration recently conducted in the Seattle District.

57. It is recognized that the water depth at the proposed CAD site would present significant problems for such a monitoring effort. Final decisions on deployment and monitoring of settlement plates should be made only after final CAD site design is complete and a more through evaluation of the potential problems for monitoring can be made.

Chemical migration through cap

58. Movement of contaminants through the cap and their rate of movement should be determined using a combination of water column and sediment core sampling. As contaminants move into the clean cap material from the contaminated sediment, they will be adsorbed by the clean material. As the adsorptive capacity of the lower cap layer is reached, the contaminants continue to move upward into cap sediment with remaining adsorptive capacity. Over time, the cap should become progressively more contaminated if contaminants are moving from the underlying material, and a discernible contaminant wave could be observed. If the contaminants exceed the adsorptive capacity of the cap, they will diffuse into the overlying water. To track and quantify these

contaminant movements, cores and water samples should be taken as soon after capping as possible (within 1 month), then at 12 and 24 months after capping.

59. Water samples must be obtained from as near the bottom as possible (within 1 m) and should include four samples taken in a transect across the site and an equal number of samples taken at an appropriate reference site. These samples must be filtered or centrifuged to remove particulate matter.

60. Sediment samples for chemical analysis will be obtained from vibra-cores. Four to six cores in a transect will be needed. Sampling will be concentrated in the cap material and the upper 30 cm of capped sediment. Beginning at the surface of the core, twenty-three 4-cm sections will be taken in each core. This will ensure that all cap material to the clean/contaminated interface will be sampled despite localized variations in the cap depth. In addition, one sample of capped material will be taken at a depth of 6 ft.

Monitoring Plan for Intertidal Disposal

61. Monitoring efforts for intertidal disposal sites should include effluent monitoring during filling operation, surface water monitoring during a representative storm event, and leachate monitoring using observation wells. Since design for intertidal sites is still under way, only descriptive plans are given here.

Effluent monitoring

62. Since the effluent discharged during filling operations potentially accounts for the majority of contaminant release from an intertidal site, routine monitoring should take place throughout the filling operations. The routine monitoring could be limited to suspended solids and perhaps representative chemical parameters to determine the overall efficiency of the site in retaining contaminants. The routine samples should be taken and analyzed on a daily basis for suspended solids and parameters such as dissolved oxygen. Routine samples should be taken on a weekly basis for chemical analysis. Each routine sample should be composited from several grab samples of the effluent taken from the discharge weir overflow. In addition to the routine sampling, a more intensive sampling effort should be carried out during one representative filling day early in the disposal operation. This sampling effort will be used to verify the accuracy of the modified elutriate test as a predictive

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technique for the project. On the intensive sampling day, a total of
12 influent and 12 effluent samples should be taken on an approximately hourly
basis. This will provide a basis for establishing the contaminant retention
efficiency of the site, as well as a basis for verifying the total contaminant
mass release from the site.

63. All samples taken for chemical analysis should be analyzed for
total and dissolved concentrations of the parameters of concern in addition to
suspended solids. Early routine monitoring can verify which parameters are
likely to be present in the effluent, and costs of monitoring could be sub-
sequently reduced by eliminating other parameters from the analysis.

Surface runoff monitoring

64. Monitoring of surface runoff quality should be conducted for a rep-
resentative storm event. It is assumed that runoff water from storms would be
ponded in the site by control of the weir boarding, and water would only be
released once suspended solids had settled from the ponded water to the
greatest possible degree. Therefore, the monitoring should be conducted by
sampling directly from the pond during or shortly after the storm event.
Three replicate samples would be taken from the pond at the weir structure.
The samples would be analyzed in the same manner as effluent samples taken
during filling as described above.

Ground-water monitoring

65. Escape of contaminants from nearshore disposal sites can occur due
to the close proximity to and movement of water adjacent to the site. Moni-
toring of contaminant escaping into adjacent waters and ground waters is com-
plex and costly. Tidal fluctuations at nearshore sites may affect the
direction and flow of ground water through the disposal sites. Since the con-
taminated dredged material will be placed at or below the ground-water level,
the contaminants will be in direct contact with the ground water, and the
potential for contaminant migration will exist. The results of testing have
indicated that the contaminants are sediment bound as long as the material
remains saturated; however, ground-water monitoring to confirm this would be
required. If the installation of liners to prevent contaminant migration is
required, monitoring to evaluate the effectiveness of the liner system both
below and outside the site would be necessary.

66. Ground-water monitoring wells should be established around the
entire site at both the East Waterway and Snohomish sites. From preliminary

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sketches, the total diked perimeters of the 100-acre Snohomish Channel site
and the East Waterway site are approximately 7,600 ft and 4,000 ft, respec-
tively. If wells are spaced at 500-ft intervals, this would require the
installation of 15 wells for the Snohomish Channel and 8 wells for the East
Waterway. These wells should be screened in the water-carrying stratum
around the site. Additionally, wells may also be installed in the dikes to
monitor seepage through the dikes. Monitoring wells installed inside the dis-
posal areas will evaluate leachate percolating through the base of the dis-
posal site. Monitoring wells installed outside the dikes when compared to
wells through the dikes could be used to evaluate the dilution factor at the
dikes.

67. The contaminants of concern have been identified by the Seattle
District as: chromium (Cr), nickel (N), copper (Cu), zinc (Zn), arsenic (As),
lead (Pb), cadmium (Cd), mercury (Hg), polychlorinated biphenyls (PBCs), poly-
nuclear aromatic hydrocarbons (PAHs), and 1- and 2-methylnaphthalene.
Sampling should begin before dredged material placement to evaluate background
conditions. Background conditions should be evaluated for tidal and seasonal
fluctuation. The sampling frequency should be more frequent during the begin-
ning of the dredging project to evaluate the initial impact of the contami-
nated sediments in the disposal sites. After disposal operations are
completed and the clean caps are in place, sampling may be performed less fre-
quently unless evidence of contaminant migration is seen.

68. Action threshold levels for contaminants of concern may be estab-
lished to indicate the probability of exceeding chronic saltwater criteria at
the dike face. This would indicate a failure of the disposal site and
controls to adequately contain the contaminants, and may justify initiating a
remedial action. A monitoring program frequency and threshold level similar
to the program used at the Port of Seattle for the Terminal 91 confined dis-
posal of contaminated sediments may be used.

69. A detailed monitoring program cannot be developed without detailed
data as to dike layout and construction, control measures to be constructed,
and dredged material placement schedules. When these data become available or
are developed along with more detailed information as to the hydrogeology of
the site, a more detailed monitoring program outlining well placement and sam-
pling strategy can be developed.