DREDGED MATERIAL RESEARCH PROGRAM

TECHNICAL REPORT D-78-45

EVALUATION OF THE ELUTRIATE TEST AS A METHOD OF PREDICTING CONTAMINANT RELEASE DURING OPEN-WATER DISPOSAL OF DREDGED SEDIMENTS AND ENVIRONMENTAL IMPACT OF OPEN-WATER DREDGED MATERIAL DISPOSAL

VOL. I: DISCUSSION

by

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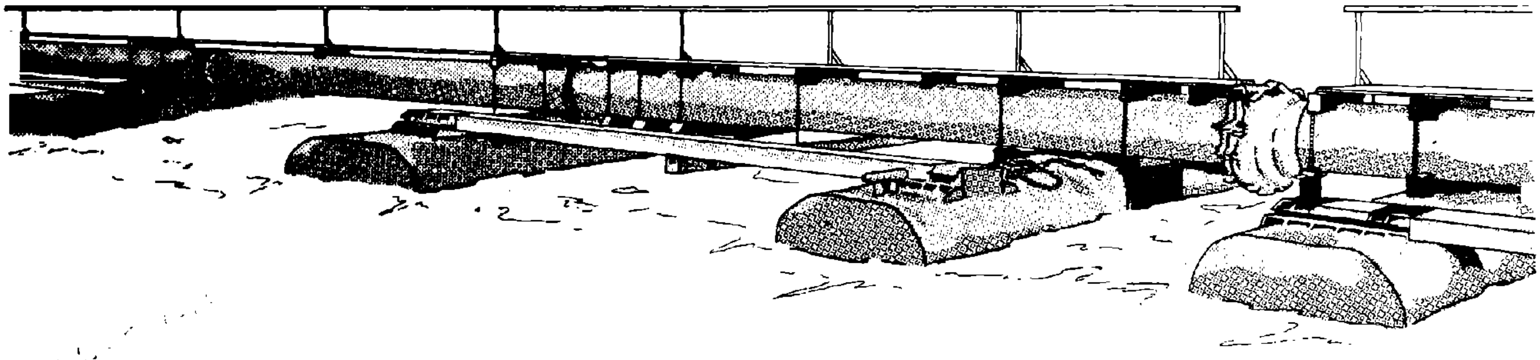

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15 November 1978

SUBJECT: Transmittal of Technical Report D-78-45 (Volume I)

TO: All Report Recipients

1. The Dredged Material Research Program (DMRP) was a broad, multifaceted investigation of the environmental impacts of dredged material disposal that included the development of new or improved disposal alternatives. In the early stages of the DMRP, it became apparent that an understanding of the actual pollution potential of dredging and discharging sediments required substantial state-of-the-art improvement in a number of fundamental biochemical areas. The procedures specified in Public Laws 92-500 and 92-532 for use in predicting the pollutional impacts of the aquatic disposal of dredged material include the elutriate test, bulk sediment analyses, bioassays, and bioaccumulation tests. Particularly critical were assessments of possible biological responses to the readily mobile and bioavailable fraction of dredged material, as well as that fraction of dredged material that may have a long-term impact on aquatic organisms as the material is carried away from the disposal site by currents. A knowledge of these effects would further support the use of these procedures as meaningful regulatory tools.

2. While developing and initiating the several-year-long program of relevant research, it was found that existing and proposed regulatory guidelines and criteria for dredged material discharges did not include techniques that adequately reflected an effective and implementable procedure for assessing potential environmental impact. Provided an opportunity to help direct the criteria development for the recently promulgated regulatory programs, the DMRP initiated research to develop biological as well as chemical evaluative procedures to assess the bioavailability and mobility of constituents from contaminated dredged material and to project their effects on the ecosystem. Moreover, these newly developed procedures required active field verification of their prediction potential.

3. This report (Volumes I and II) represents the results of the further refinement and field verification of regulatory criteria and guidelines required in the ecological evaluation of dredged and fill material discharges. Volume I (transmitted herewith) contains the main text, while Volume II is a data report. This study was one of several work units included under Task 1E (Pollution Status of Dredged Material) of the DMRP; in the DMRP's management structure, it was included as part of the Environmental Impacts and Criteria Development Project.

WESEV 15 November 1978
SUBJECT: Transmittal of Technical Report D-78-45 (Volume I)

4. The two-volume report discusses the factors influencing the results of the elutriate test and the reliability of this test in predicting the release of contaminants during actual dredged material disposal operations. Sediment samples were taken from 26 waterway locations representing marine, estuarine, and freshwater locations. Field investigations were conducted at eight active dredging and disposal operations in marine, estuarine, and freshwater areas so that a comparison could be made between the results of the standard and modified elutriate tests for water column concentrations during disposal operations.

5. Results of these assessments have shown that the standard elutriate test, involving 30-minute mixing and one-hour settling, is a reliable test for predicting the potential release of contaminants associated with hydraulically dredged sediments that are discharged into open water. These investigations have also shown that the open-water discharge of dredged sediments, including those that are highly contaminated with various types of chemical toxicants, would rarely cause an adverse effect on water column quality and aquatic organisms in the disposal site water column.

6. The information and data published in this report are contributions to the further understanding of the complex nature of sediment, water, and chemical/biological interactions and the report establishes a baseline from which to develop meaningful regulatory criteria. It is expected that the methodology employed in this study and the resultant interpretation of the biochemical interactions will be of significant value to those persons concerned with dredged material permit programs.

John Cannon
JOHN L. CANNON
Colonel, Corps of Engineers
Commander and Director

REPORT DOCUMENTATION PAGE

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The U. S. Army Corps of Engineers and the U. S. Environmental Protection Agency developed the elutriate test for the purpose of predicting the release of chemical contaminants from dredged sediments upon open-water disposal. This study was conducted to evaluate the factors influencing the results of the elutriate test and the reliability of this test in predicting the release of contaminants during actual open-water dredged material disposal operations. Sediment samples were taken from waterways located at or near Duwamish River-Elliott Bay-Puget Sound, Washington; San Francisco Bay, Mare Island, Rodeo Flats, Oakland Harbor, and Los Angeles Harbor, California; Galveston Bay Entrance Channel, Galveston Channel, Texas City Channel, Houston Ship Channel, and Port Lavaca, Texas; Mobile Bay, Alabama; Apalachicola Bay, Florida; Wilmington, North Carolina; James River, Virginia; Perth Amboy, New Jersey; Bay Ridge

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20. ABSTRACT (Continued).

and Foundry Cove, New York; Newport, Rhode Island; Norwalk and Stamford Harbors, Connecticut; Menominee River, Michigan; Upper Mississippi River near St. Paul, Minnesota; and the U. S. Army Engineer Waterways Experiment Station Lake, Vicksburg, Mississippi.

These samples were subjected to the standard and modified elutriate tests in order to examine the influence of various operating conditions on the results of the test. In addition, field studies were conducted at Elliott Bay-Puget Sound, Washington; Galveston Bay Entrance Channel Disposal Area, Texas; Mobile Bay, Alabama; Apalachicola Bay, Florida; James River, Virginia; New York Bight; and the Upper Mississippi River near St. Paul, Minnesota, in which comparisons were made between the results of chemical analyses of the standard and modified elutriates and contaminant concentrations in the water column during disposal operations. These studies showed that the standard elutriate test, involving 30-minute compressed air mixing and one-hour settling, is a reliable test for predicting the potential release to the water column of contaminants associated with hydraulically dredged sediments that are disposed of in open water. The air bubbles, in addition to mixing the solution, also keep the system oxic, thus simulating the conditions of importance for water column organisms at the dredged material disposal site.

It is recommended that the EPA and the Corps of Engineers continue to use this modified elutriate test to evaluate the potential release of contaminants to the open-water dredged material disposal site water column.

This study has also shown that the open-water discharge of dredged sediments, including those which are highly contaminated with various types of chemical toxicants and nutrients, would rarely cause an adverse effect on water quality and/or aquatic organisms in the disposal site water column. Generally, the rapid dilution and dispersion and the intermittent nature of open-water dredged material disposal operations result in rapid reduction of the concentrations of any released contaminants to levels below those which are known to have adverse effects on aquatic life and/or water quality.

V

SUMMARY

A study has been conducted to evaluate the reliability of the elutriate test as a means of predicting the release of contaminants during open water disposal of dredged sediment. In addition, studies have been conducted to evaluate the influence of various test conditions on the test results. Sediments and water samples have been collected from the Duwamish River and Elliott Bay-Puget Sound, Washington; San Francisco Bay, Mare Island, Rodeo Flats, Oakland Harbor, and Los Angeles Harbor, California; Galveston Bay Entrance Channel, Galveston Channel, Texas City Channel, Houston Ship Channel and Port Lavaca, Texas; Mobile Bay, Alabama; Apalachicola Bay, Florida; Wilmington, North Carolina; James River, Virginia; Perth Amboy, New Jersey; Bay Ridge and Foundry Cove, New York; Newport, Rhode Island; Norwalk and Stamford Harbors, Connecticut; Menominee River, Michigan; Upper Mississippi River near St. Paul, Minnesota; and the Corps of Engineers Waterways Experiment Station Lake, Vicksburg, Mississippi. These sediments have been subjected to the standard elutriate test and modifications thereof. Bulk sediment analysis and analyses of the elutriates have been made for approximately 30 contaminants, including selected heavy metals, selected chlorinated hydrocarbon pesticides and PCBs, various forms of nitrogen and phosphorus compounds and other selected parameters which could be of significance in open water disposal of dredged sediments.

Factors such as length of aeration period, duration of sediment storage prior to elutriation, sediment-to-water ratio, type of water used in the elutriate test (i.e., disposal site or dredging site), etc. have been examined to determine their influence on the release of contaminants during the test. The results of these studies have been compared to similar previously reported studies on sediments from Ashtabula Harbor-

Lake Erie; Bridgeport, Connecticut; Corpus Christi, Houston Ship Channel, Port Aransas, and Trinity River, Texas; Duluth Harbor, Minnesota; Lake Superior; and Mobile Bay, Alabama.

Comprehensive field studies have been conducted in conjunction with elutriate test studies at Elliott Bay-Puget Sound, Washington; Galveston Bay Entrance Channel Disposal Site, Texas; Mobile Bay, Alabama; Apalachicola Bay, Florida; James River, Virginia; New York Bight; and the Upper Mississippi River near St. Paul, Minnesota in order to compare the release of contaminants during open water disposal of mechanically and hydraulically dredged sediments to the release found in the elutriate tests of the same sediments. The results of these studies show that the elutriate test must be conducted under oxic conditions to properly assess the significance to aquatic organisms of toxic contaminant release during open water disposal of hydraulically dredged sediment. It is recommended that for the 30-minute mixing period specified in the standard elutriate test procedure, compressed air be vigorously bubbled through the sediment-water slurry.

The elutriate test generally overestimates the amount of contaminants released during barge dumping of mechanically dredged sediments. A recommended modified elutriate test procedure (plop test) has been developed for evaluation of the release of contaminants during open water disposal of mechanically dredged sediments.

Of the various operating conditions (for example, mixing and settling times, test water salinity, temperature, liquid:solid ratio, and the type of water used) that would likely affect the results of the elutriate test, the amount of solid used in the test proved, for a number of contaminants, to be the most important. However, a consistent pattern was not found for the amount of release as a function of liquid:solid ratio for the sediments studied. The 20 percent sediment of total elutriate volume currently specified for the elutriate test is probably too high for the typical conditions found in association with the open water disposal of hydraulically dredged sediments. It is recommended that a more dilute (5 to 10 percent) sediment volume of the total elutriate volume be used. This would greatly facilitate conducting the test and more properly simulate the conditions that prevail at the dredged material disposal site, when the concern is the release of contaminants that can have an adverse effect on aquatic organisms in the disposal area water column. The duration of the mixing and settling period appears to be appropriate. For some sediments, however, the finely divided particulate matter does not settle to a sufficient degree to allow rapid filtration. In these situations, the duration of the filtration step can greatly increase the contact time after mixing. With this type of sediment, decreasing the sediment:water ratio may reduce amounts of suspended material in the unfiltered elutriate and hence decrease filtering time. For many contaminants, based on the results of this study, increasing the length of the settling period by one to two hours did not significantly affect the results of the elutriate test, provided that the system remains oxic during the time of settling.

Dredging site water should be used in the elutriate tests for hydraulically dredged sediments. Disposal site water should be used for the modified elutriate tests (plop tests) for mechanically dredged sediments. In those areas where there are marked salinity differences between dredged channel surface waters and bottom waters, a mixture of the two waters, with a salinity intermediate between the two, i.e., approximate average content, should be used.

The magnitude of uptake or release of the various contaminants studied appeared to be highly site specific, with little or no correlation between the physical and chemical characteristics of the sediment such as its bulk chemical content, and the release of a particular contaminant in the elutriate test. It is impossible to predict with any degree of reliability the release of contaminants during open water disposal of mechanically and hydraulically dredged sediments based on the bulk chemical content of the sediment. A possible exception is for ammonia in sediments from a restricted geographical area. Even with ammonia, the reliability of such predictions is generally very poor and should not be used unless extensive studies have been done for a particular dredging site which could serve as a basis for establishing a correlation between the bulk sediment ammonia content and its release in the elutriate test.

Among the heavy metals studied, manganese was the only metal that was consistently released from the sediments in the elutriate test. Occasionally, relatively large amounts of iron were released; however, it is thought that this iron is in a colloidal form and not in true solution. Zinc frequently showed removal from the test water. For the other heavy metals (copper, cadmium, arsenic, nickel, lead, and chromium) small amounts of uptake or release occurred; however, no consistent patterns were found.

Ammonia was consistently released from the sediments in the elutriate test. Frequently the concentrations of nitrate present in the site waters were reduced slightly as a result of elutriation.

The release of phosphorus from the dredged sediments was highly site specific. It appears that phosphorus release, like that of many of the other contaminants, is tied to the ability of hydrous ferric oxide to remove phosphate by coprecipitation reactions.

The behavior of chlorinated hydrocarbon pesticides and PCBs was also site specific; however, in general, sediments having the greatest oil and grease content tended to release less of these compounds (pesticides and PCBs) in the elutriate test. Some of the greatest PCB releases were obtained from what would normally be considered from a classical pollutional standpoint the "cleanest" of the sediments. At several sites, monitoring programs had been conducted to evaluate the accumulation of chlorinated hydrocarbon pesticides and PCBs within aquatic organisms residing in the disposal area. It has been found that even though the sediments may contain very high concentrations of these compounds, none of the sites where this monitoring has taken place has shown high concentrations in marine organisms. It is therefore concluded that open water dredged material disposal operations are not, in general, having a significant effect on the accumulation of chlorinated hydrocarbon pesticides and PCBs within aquatic organisms.

Bioassay studies on unfiltered elutriates have shown that, in general, very limited toxicity is expected from the dredged sediments upon open water disposal. At many of the sites studied, most of the test organisms (grass shrimp or <u>Daphnia</u>) living in what is equivalent to a settled discharge from a dredging pump, survived a four-day exposure period. In nature, the normal dilution associated with open water disposal of dredged sediments would likely render even those sediments with the highest toxicity non-toxic to aquatic life residing in the water column at the disposal site. A bioassay screening procedure has been developed which is believed to be much simpler and less expensive than the US EPA/Corps of Engineers bioassay procedures released in July 1977 in accord with the January 11, 1977 <u>Federal Register</u>.

Studies on the water quality characteristics associated with open water disposal of dredged sediments have shown that, in general, few if any significant environmental quality problems due to the chemical contaminants present in sediments, would be associated with most open water dredged material disposal operations. The intermittent nature of the dumping operations and the relatively rapid dispersion of any released contaminants at the disposal site creates a situation where the likelihood of significant toxicity or bioaccumulation of sediment-associated contaminants is small.

The January 11, 1977, Federal Register specifies that the US EPA July 1976 water quality criteria shall be used to judge the significance of chemical contaminants released from ocean-dumped dredged sediments after consideration of mixing. The use of these chronic toxicity-based criteria results in an overly conservative estimate of water quality impact. The September 5, 1975 Federal Register specifies that the concentrations of various chemical contaminants at the edge of a mixing zone associated with the disposal of dredged material in estuarine and inland waters, must be compared to "appropriate water quality criteria." It is important to note that the July 1976 US EPA Quality Criteria for Water should not be used to judge the significance of contaminant release associated with disposal of dredged sediments in fresh water or marine water. The July 1976 US EPA water quality criteria are based on chronic-exposure situations, where the aquatic organisms are exposed to the available forms of contaminants for a significant period of their lifetimes. Dredged material disposal operations, in general, do not create chronic-exposure situations. This is especially true for dumping operations from barges or hopper dredges. In the case of pipeline disposal operations, it is conceivable that chronic-exposure conditions could be achieved. A site-by-site evaluation must be made in order to determine whether a particular disposal operation should come under chronic-exposure criteria or under criteria which are based on shorter times of exposure. In the case of the latter, in general, a much higher concentration of contaminant can be tolerated.

This study has shown marked differences in the potential for environmental impact of various methods of dredged material disposal. The least impact to water column organisms would likely be associated with mechanically dredged sediments dumped from a barge. This is followed, in the series of increasing potential environmental impact by hopper-dredge disposal of hydraulically dredged sediment. This is followed by pipeline disposal of hydraulically dredged sediments, where the discharge point is relocated within a few hours to a few days. The next most potentially damaging would be pipeline disposal of hydraulically dredged sediment where the discharge is fixed at essentially one point for a period of a few weeks to a few months or longer. The most potentially damaging situation would likely be where hydraulically dredged sediments are pumped to a confined disposal area with a supernatant detention time of a few hours or less to a day or so and where overflow waters are discharged to the nearshore areas of a watercourse. While this is the general order of increasing potentially adverse environmental impact associated with dredged material disposal, site specific conditions can change this order. Therefore, a complete evaluation must be made for each dredged material disposal operation and disposal method used.

The results of this study have shown that, in general, the elutriate test provides a reasonably accurate estimate of the concentrations of contaminants which will be released at an open water dredged material disposal site. The bulk sediment concentrations of various contaminants have been shown to have no relationship to the release of contaminants in the elutriate test of that sediment or to toxicity of the elutriate and sediments for a variety of waterway sediments collected throughout the US. The elutriate test combined with screening-type bioassays appear to be the best overall dredged material disposal procedure by which to judge the significance of chemical contaminants associated with hydraulically or mechanically dredged sediments which are disposed of in open water.

In general, it appears that confined disposal of dredged sediments in which there is overflow of supernatant water to the nearby watercourse may be equally, and in many instances, more adverse to environmental quality than open water disposal of these sediments. It is therefore recommended that before any change is made from open water disposal of dredged sediments to confined disposal in which there is no treatment of the overflow-supernatant water, a careful evaluation be made to ensure that the confined disposal method does not have equal, or greater, detrimental impact on water quality than open water disposal of the sediments. It is recommended that the US Army Corps of Engineers and the US EPA continue to use the elutriate test and a screening-type bioassay to evaluate the environmental impact of sediment-associated chemical contaminants upon open water dredged sediment disposal. Further studies need to be done to evaluate the actual environmental impact associated with confined or on-land disposal operations to ensure that the alternate, and in many cases, the more expensive, methods of disposal do not result in equal or greater environmental contamination than the previously used open water dredged material disposal methods.

PREFACE

The work described in this report was performed under Contract Number DACW 39-75-C-0102, titled "Sample Collection for Development of Dredged Material Disposal Criteria," dated February 1, 1975 through August 30, 1976, and Contract Number DACW 39-76-C-0117, titled "Field Testing and Verification of Dredged Material Disposal Criteria," dated May 1, 1976 through July 31, 1977, between the US Army Engineer Waterways Experiment Station (WES), Vicksburg, Mississippi, and the University of Texas at Dallas, Richardson, Texas. The research was sponsored by the Office of the Chief of Engineers, under the Civil Works Program "Dredged Material Research Program." This report presents a discussion of the results of a laboratory and field study of the factors influencing the response of the elutriate test for evaluation of the release of chemical contaminants from dredged sediments. Sediments were obtained from thirty waterways and harbors across the US and were subjected to standard and modified elutriate tests. The second phase of this study was devoted to evaluating the reliability of the elutriate test to predict the release of contaminants on open-water disposal of hydraulically dredged sediments. Field studies were conducted at seven sites in which a comparison was made between the concentrations of contaminants released in the elutriate tests and that actually found in the water column during the dredged material disposal operation. The study was conducted during the period February 1, 1975 through July 1977. The laboratory and field work which served as a basis for this report were conducted at the University of Texas at Dallas. The final phase of preparation of this volume of the overall report was completed at Colorado State University, Fort Collins, Colorado.

The principal investigator for this project was Dr. G. Fred Lee, who during most of this study held the position of Professor in the area of Environmental Chemistry at the University of Texas at Dallas and at the time of completion of this report, held the position of Professor of Civil Engineering at Colorado State University. The principle authors for this volume of the report were Drs. R. Anne Jones and G. Fred Lee. At the time of completion of this report, Dr. Jones held the position of Assistant Research Professor in the Department of Civil Engineering at Colorado State University, Fort Collins, Colorado. Drs. Lee and Jones were assisted by Pinaki Bandyopadhyay, Jeannie S. Butler, David H. Homer, and George M. Mariani, in developing sections of the report as indicated in the text. Many other individuals from the University of Texas at Dallas contributed to the success of this study. Their contributions were primaril made in connection with the development of Volume II, the Experimental Procedures and Results volume of this report. These individuals are acknowledged in the Preface to that volume of the overall report.

The editorial assistance of J. Hale and the secretarial assistance of M. Jaye are greatly appreciated. The secretarial assistance of S. Elder and P. Wernsing is also appreciated. We wish to acknowledge the assistance given this study by G. Max of the UTD Center for Advanced Studies, and other UTD staff. R. Ricamore of UTD provided drafting services for this report.

Numerous individuals on the Corps of Engineers District staffs have contributed significantly to this project. Of particular note are T. Wakeman--San Francisco District, A. Hall--Seattle District, P. Warren--Mobile District, R. Van--Norfolk District, Major M. Howell, Jr. and D. Dunn--Galveston District, D. Suszkowski and J. Zammit--New York District, V. Andreloures--New England District, and R. Whiting--St. Paul District. We also wish to acknowledge the assistance of the captains and crews of the dredges in the various Districts where field studies were conducted. Without their outstanding cooperation, the success of the field studies would not have been possible.

Several members of the US Army Engineer Waterways Experiment Station staff greatly aided in this study. Especially significant contributions were made by Drs. R. Engler and R. Peddicord, and Mr. B. Holliday. Also, in association with the Galveston studies, the assistance of D. Mathis and S. Cobb is acknowledged.

A special thank you is given to Dr. J. Harrison, Chief of the Environmental Laboratory of the Waterways Experiment Station, Vicksburg, Mississippi, for his support throughout the course of this study. Dr. J. Keeley of the Waterways Experiment Station deserves special recognition for his assistance in connection with the initiation of the work on the environmental impact of chemical contaminants in dredged sediments.

This study was conducted in part with support of EnviroQual Consultants and Laboratories, Inc. of Fort Collins, Colorado and the Department of Civil Engineering at Colorado State University, Fort Collins, Colorado.

This contract was monitored by Dr. R. Peddicord under the supervision of Dr. R. Engler.

Commander and Director of the Waterways Experiment Station during this study was COL J. L. Cannon, CE. Technical Director was Mr. F. R. Brown.

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PART I: INTRODUCTION

The US Army Corps of Engineers (CE) is responsible for maintenance of the US waterways and harbors. Since many of these waterways are in urbanized and industrialized areas, the sediments are often highly contaminated with municipal and industrial wastes. In addition, these sediments are contaminated with chemicals from urban and agricultural drainage. Prior to about 1970, sediments dredged from US waterways were generally disposed of in nearby open waters. Consideration in disposal site selection was given to the presence of fish spawning or commercially important shellfish areas. In the early 1970's, attention began to be devoted to the significance of the chemical contaminants in the dredged sediments and their potential impact on water quality at the dredged material disposal site. Federal water pollution control regulatory agencies developed dredged material disposal criteria which were based on the total chemical content for selected parameters in the dredged sediment. This resulted in many sediments being classified as "polluted" and thereby requiring alternate methods of dredged material disposal which almost universally increased the cost of waterway maintenance.

There are many who opposed the use of bulk criteria as a basis for judging the potential significance of chemical contaminants in dredged sediments to disposal site water quality upon open water disposal based on the fact that many contaminants in sediments do not become available to affect water quality. As a result of this situation, the US EPA and CE developed the elutriate test which was designed to estimate the release of chemical contaminants from dredged sediments that would occur in the water column during or shortly after the disposal operations at an open water disposal site. This test involves the mixing of one volume of sediment with four volumes of dredging site water for 30 minutes, settling for one hour, and then filtering. The filtrate is analyzed for the various chemical contaminants of interest. The original basis for the conditions used in the elutriate test were largely empirical. Little information was available on the various factors such as mixing time, method of agitation, settling time, etc. which could affect the results of the test. Further, at the time of its adoption, there was no information available on how well this test actually predicted the release of contaminants upon open water disposal of dredged sediments. In an effort to try to provide this type of information, the US Army Corps of Engineers' Dredged Material Research Program funded a study devoted to evaluating the factors influencing the release of contaminants associated with dredged sediments. A discussion of the results of this study is presented in this report. Sediments have been collected from approximately 30 different waterways across the US (See Figure 1) and subjected to standard and modified elutriate tests in order to determine the factors that influence the release of contaminants during the elutriate test.

During the course of this study it became clear that the elutriate test, if modified so that it was conducted under oxic conditions, was relatively insensitive to minor changes in test conditions and therefore showed considerable potential for use in estimating the release of contaminants associated with open water disposal of hydraulically dredged sediments. The next step in the development of the elutriate test was that of evaluating how well it actually predicted the release of contaminants at open water disposal sites. A series of seven field study sites were also investigated. Sediments from a waterway were collected prior to dredging and then normal dredging operations were conducted in the area from which the sediments were collected. An intensive water column monitoring

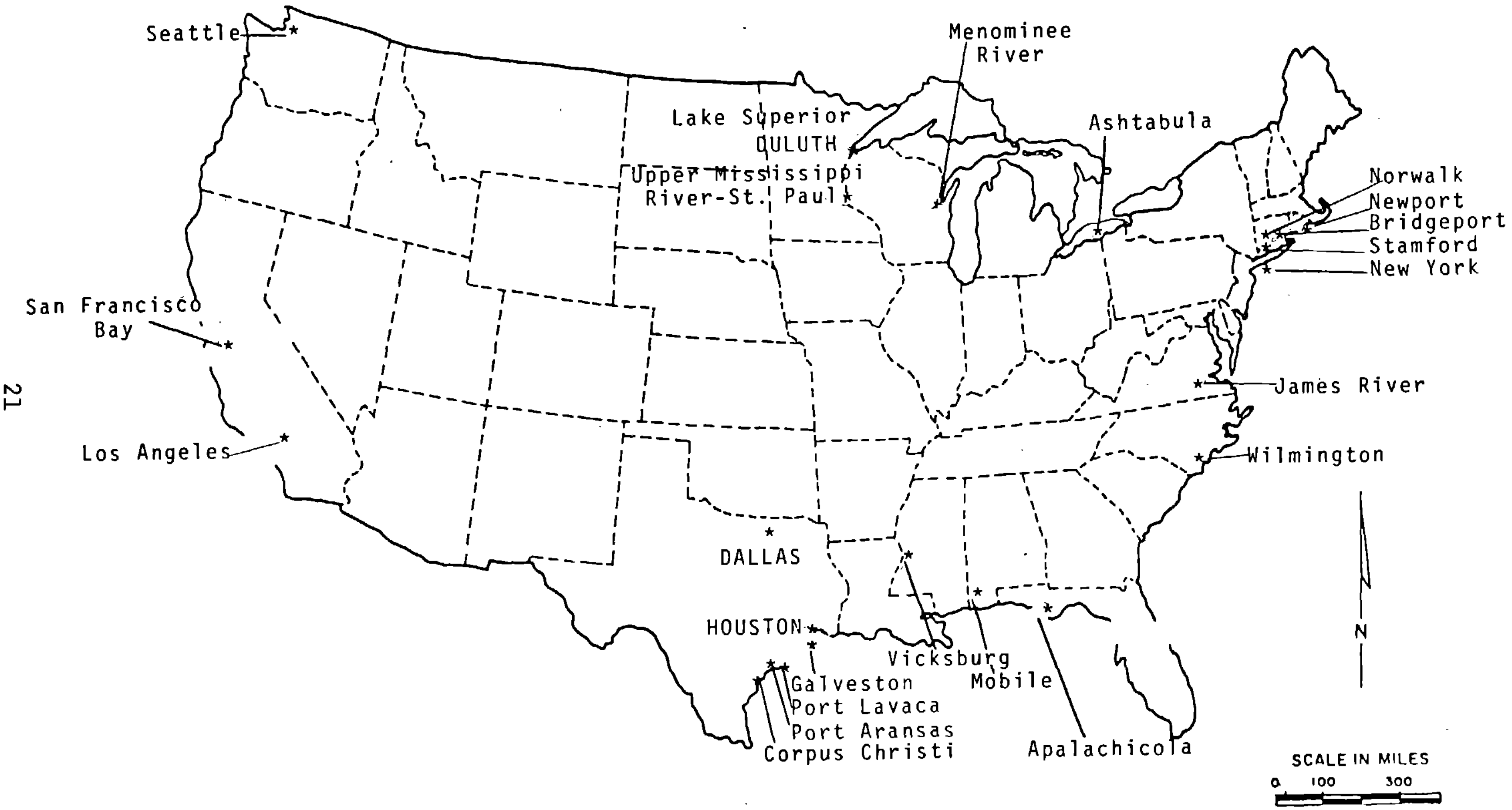

Figure 1
Sampling Locations

program was conducted during the dredging and disposal of these sediments in order to ascertain the magnitude of uptake or release of contaminants from the dredged sediments during open water disposal. Contaminants investigated included selected nitrogen and phosphorus compounds, heavy metals, chlorinated hydrocarbon pesticides, and PCBs, as well as other physical-chemical parameters.

It should be noted that a perfect match between elutriate test results and the results of these field studies would not be expected. As discussed in Volume II, the field studies were, in general, conducted so as to position the sampling vessel at some distance downcurrent from the discharge. It would be expected that some dilution of any chemicals released in the dredged sediment would occur between the point of discharge and the location of the sampling vessel. To completely define the concentration-time profiles for various contaminants associated with the various disposal operations monitored, it would be necessary to have a large number of ships from which samples could be collected simultaneously. As a result, instead of collecting several hundred samples per disposal, as was done in this study, several thousand samples would likely have to be collected. This would greatly increase the cost of the study program but would likely provide a better match between elutriate test results and field data.

The overall report covering results and discussion of this study is divided into two volumes. Volume I presents a discussion of the results of these studies and an interpretation of the data with conclusions and recommendations. Volume II is a companion volume devoted to a presentation of the experimental procedures and the results of these studies. Volume II should be consulted for background information to the discussions presented in this report. Throughout this discussion, reference is made to dredged material disposal and dredged material dumping. The term "disposal" includes both pipeline and hopper dredge or barge dumping, whereas the term "dumping" refers to the intermittent release of dredged sediment from a hopper or barge-scow.

PART II: OVERALL CHARACTERISTICS OF DREDGED MATERIAL DISPOSAL OPERATIONS IN RELATION TO ELUTRIATE TEST DEVELOPMENT

Introduction

The development of appropriate tests to estimate the potential environmental impact of the open water disposal of dredged material requires an understanding of the physical and chemical conditions that exist during dredging and disposal. The elutriate test was designed to simulate the various processes that occur during hydraulic dredging-open water disposal operations and thereby predict contaminant release upon open water disposal of dredged sediment. The development of this test required the specificiation of certain test operating conditions which could affect the results of the test.

Since the optimum hydraulic dredge pumping ratio is generally believed to be 1:4 (sediment:dredging site water), the elutriate test is run using 20 percent sediment by volume in dredging site water. The 30-minute mixing period specified in the elutriate test procedure is supposed to be representative of the mixing time of the dredged sediment and associated waters in the hoppers or pipeline. This is the time between the initial entry of the dredged sediment into the suction head of the dredging pump until its release to the dump site water. A one-hour settling period provides additional contact time in order to better predict the release of contaminants that may occur in the turbid plume. The settled sample is filtered since in the water column at the dump site, it is the soluble forms of contaminants that are of primary interest.

The following discussion of the appropriateness of the designated test conditions for various types of dredging-disposal methods is based on observations made by the authors during two hopper dredging operations involving open water dumping of the sediment (New York and Galveston), two mechanical dredging operations with open water barge-scow disposal (New York and Seattle-Puget Sound), and four different open water pipeline disposal operations of hydraulically dredged sediment (James River, Virginia; Upper Mississippi River near St. Paul, Minnesota; Mobile Bay, Alabama; and Apalachicola Bay, Florida).

Sediment to Water Ratio

Examination of the characteristics of hydraulic dredging operations shows that a 20 percent sediment pumping ratio is rarely maintained, due to the nature of the hydraulic dredging process. For many hydraulic dredging operations, those responsible for the operation of the suction head adjust the head depth to achieve the maximum percent sediment without plugging the system. In addition, in the case of hopper dredging, the suction head overlaps adjacent, previously dredged areas. Another factor which causes a variable percent sediment is the walking action associated with pipeline dredging.

During the course of this study, a series of samples was collected from the discharge of a dredged material disposal pipeline. Samples of the dredging pump discharge during hopper dredging were also collected. The percent settleable solids data for these samples (see Volume II) show that the percent settleable solids are highly variable, ranging from zero to 100 percent. Based on these data, it is concluded that there is no more technical validity in selecting 20 percent than any other value for running the elutriate test. For ease of operation, Lee _et al_.[1] recommended the use of five percent sediment by volume in elutriate tests. This recommendation was based primarily on ease of filtration since the use of five percent sediment typically results in lower suspended solids in the settled elutriate. The additional work that has been done during this study tends to support this recommendation.

Additional justification for changing the sediment-water ratio for hopper dredged sediments stems from the fact that few, if any, hopper dredges maintain a 20 percent sediment-water ratio in the hoppers during the 15 to 60 minute period of transport from the dredging to disposal sites. Even though overflow of supernatant water is against the Corps practice, it was found that some hopper dredges continue this practice with the result that the surface of the sediment within the hoppers is almost dry during the period of transport. While overflow may cause potentially significant aesthetic problems near the dredging site because of increased turbidity, the results of this study show that except for this aesthetic problem, no other significant water quality problems are likely to arise from overflow and/or pumping off of excess water from the hoppers.

Some sediments which tend to be highly cohesive are kept in suspension in the hoppers by high velocity jets of water or air in order to facilitate sediment removal at the dump site. It is therefore likely that the sediment:water ratio is different in the hoppers during transport than in the dredging pump discharge to the hoppers. The net effect is that the liquid:solid ratio in the hopper at the time of the dump is typically markedly different than the pumping ratio that was used to fill the hoppers. While to the knowledge of the investigators, no studies have been done to determine the actual liquid:solid ratios of the material in the hoppers just before the dump, it is likely to be much different and probably greater than the 20 percent used in the elutriate test. The effect of this reduced ratio on the release of contaminants is unknown at this time, but it is likely to result in less mixing of the dumped sediment with dump site water. This may justify using a lower sediment:water ratio for elutriate tests.

Since Lee et al.[1] and subsequent investigators have found that the liquid:solid ratios can be an important parameter in the release of some contaminants (e.g., ammonia), it is likely that the actual release that occurs from a hydraulically dredged sediment during open water disposal is dependent, to a much greater extent than thought, on the manner in which the dredge is operated. It is conceivable that the manner of dredging-disposal could be used as a means of minimizing adverse environmental impact of open water disposal of dredged sediment. It is probable that further studies of the relationships between method of dredge operation and release of contaminants could be highly beneficial in providing information on how best to dredge and dispose of sediments to minimize adverse environmental impact.

Obviously, mechanically dredged sediments have a markedly different character than hydraulically dredged sediment due to the fact that they have not been slurried with dredging site water as part of the dredging operation. In many instances, fine-grained mechanically dredged sediments have a paste-like cohesive character. It is certainly inappropriate to use an elutriate test which involves a slurrying of the sediment and water to estimate the release of contaminants from mechanically dredged sediments. A test which more properly simulates the barge dumping of mechanically dredged sediments such as a plop test, should be used. The plop test, developed during this study (See Volume II for details), was found to give a better estimate of the release of contaminants from mechanically dredged sediments than the standard elutriate test. For the purpose of uniformity with the standard elutriate test, the 20 percent sediment in the total elutriate volume was maintained for the plop test. This could be readily adjusted to five percent should the standard elutriate test be modified according to the recommendations of the Lee *et al.*[1] study. It was interesting to note that for many of the parameters measured, the five percent test sometimes yielded results which were similar to the 20 percent plop test.

Contact Time

The standard elutriate test provides 1.5 hours of contact between the sediments and the water. Actually, it is often difficult to hold to this contact time because of the difficulty in filtering the elutriate. Even with the the pressure filtration technique used in this study, contact times of up to several hours are frequently necessitated to properly process the sample. Lee *et al.*[1] found that for the elutriate tests on several different sediments, varying the contact times from 30 minutes to several hours had limited influence on the release of a number of contaminants. Examination of the contact time that actually occurs during hydraulic dredging and open water disposal shows that it can be as short as a few minutes for some pipeline disposal operations, to as long as several hours where long runs are made by hopper dredges from the dredging to disposal sites. If one critically examines open water disposal operations, it would be found that the contact time of interest is the time from when the sediments are first picked up by the dredge until the turbid cloud (associated with open water dumping) or the turbidity current (associated with pipeline disposal) is sufficiently dispersed so that any further release of contaminants from the suspended sediment would not have a significant adverse effect on water quality.

The typical dredged sediment dumping operation consists of the release of several hundred to several thousand cubic meters of sediment at a predetermined location. Usually, this is done while the hopper dredge or barge is moving. The dump itself usually takes place over a several minute period. Depending on the design of the hopper dredge, i.e., number of hoppers and the sequence and rapidity of their opening for disposal, a single mass or a series of masses of somewhat consolidated dredged sediment rapidly descends from the hopper dredge through the water column. A similar situation exists for the barges or scows. For both types of dumps, the majority of the dredged sediment does not, to any significant extent, interact with the disposal site water column. Even with hopper dredged sediment, there is not a complete slurrying of the dumped sediment with the water at the disposal site. The large, somewhat cohesive mass of sediment hits the bottom and may create a current of its own as it spreads across the bottom.

Associated with each dumping operation is a turbidity plume which arises from the peeling off of suspended sediment during the descent of the dumped sediment. This turbidity plume or cloud moves downcurrent at the rate of the ambient currents (i.e., it does not create its own current). It is rapidly (i.e., within a few hours) diluted to essentially ambient turbidity levels. As was demonstrated with both the Galveston and New York dumps, there is appreciable entrainment of dump site water associated with the descent of the mass of dredged sediment. This entrainment and the peeling off of suspended solids from the mass of the dumped sediment results in substantial mixing of dump site waters with the sediments at the outer edge of the mass of descending sediment.

While this study focuses on the significance of the short-term fate of contaminants in the water column during open water disposal of dredged sediment, mention should be made that most open water dump sites are, geologically speaking, relatively high energy areas in which there is rapid mixing and dispersion in the water column and within the sediments. In general, a low energy site would not be suitable unless it were very deep, due to the fact that the site would soon become unusable as a result of the accumulation of dredged sediments. The significance of this situation to the long-term impact of chemical contaminants present in dredged sediments is discussed in a separate section of this report.

An additional source of suspended sediment at the disposal site is the washing of the hoppers after disposal has taken place. Usually small amounts of sediments hang up within the hopper after disposal; many hopper dredges wash this sediment out with high speed jets of water. This usually creates a turbid cloud which floats on the surface. While the turbidity may be readily visible, it usually contains very small amounts of suspended solids, and except for impact on aesthetic qualities, it has little impact on overall water quality.

The results of drogue studies conducted during this investigation in addition to the results of studies conducted by others have shown that the turbid plume associated with open water dredged material disposal typically persists for approximately one to three hours. It is likely to be rare that the total contact time between dredging and the dissipation of dump-associated turbidity would be greater than about three hours, except for the area immediately adjacent to the bottom at the disposal site. The water quality next to the bottom may be adversely affected for periods of time longer than the passage of the turbid plume. As was found at the Galveston study site, the bottom currents are often of sufficient magnitude to erode freshly deposited sediments from the mound created as a result of dumping. This could cause the waters next to the sediments immediately downcurrent from the dump site to have elevated concentrations of chemicals for much longer periods (i.e., several hours).

Another factor which would tend to increase the time that the waters in the bottom part of the turbid plume take to pass a specific location is that in general the currents at the surface are considerably greater than those at the sediment-water interface. This results in an oval-shaped plume extending from near the dumped sediment mound outward down surface current from the mound. This was the shape of the turbid cloud at the GBEC disposal site. However, at the mud dump site in the New York Bight, the surface and bottom currents moved in opposite directions. There was little or no surface turbid plume; essentially all of it was restricted to below the thermocline-chemocline. It is obvious from these studies that the shape of the turbid cloud is highly site specific and dependent to a major extent on the physical characteristics of the dump site water column.

Based on the results of this study, it is concluded that the typical open water pipeline disposal operation results in the development of a turbidity current or mud flow which spreads in a fan shape downcurrent from the point of discharge. Most open water pipeline disposal operations are conducted in shallow water, frequently of a meters depth. Near the point of discharge and for several hundred meters from that point, the turbidity current is on the order of 0.5 to 1 meter thick. There is substantial mixing of the dredged sediment slurry with the disposal site water at the point of pipeline discharge. Further, there is mixing of the waters and sediment in the turbidity current with overlying waters along its path. This mixing, coupled with sedimentation of suspended solids and the spreading, leads to a fairly rapid dilution and dissipation of the turbidity current.

From an overall point of view, it is felt that the 1.5 hour contact time used in the elutriate test is appropriate for predicting the contaminant release in the upper part of the water column. If the concern at particular sites is for the sediment-water interface, then an extended contact time (a day or so) should be used in the elutriate test. It should be emphasized that the above discussion focuses on the potential impact of the chemicals released during open water disposal operations and does not address the long-term release of chemical contaminants associated with redeposited sediments.

Type of Water

There is considerable confusion about the type (i.e., source) of water that should be used for the elutriate test. Basically, a dredging site water should be used for all elutriate tests on hydraulically dredged sediments, and disposal site water should be used in the plop test for mechanically dredged sediment. This approach, while appropriate, could lead to erroneous conclusions about the potential for uptake or release associated with dredged material disposal. Dredging site water tends to have higher concentrations of many contaminants than the disposal site water. For those chemicals which may be sorbed by the dredged sediment, the elevated concentration of the contaminants in the dredging site water could lead to elutriate test results which show decreased concentrations in the elutriate compared with those in dredging site water. On the other hand, if the test had been conducted with disposal site water, the results may have shown a slight release. In a strict sense, the elutriate test should be modified to provide for a dilution of the mixed but not settled elutriate with disposal site water. The amount of dilution should be determined on a case-by-case basis considering the characteristics of the disposal operation and disposal site.

Redox Conditions

One of the most controversial aspects of the elutriate test procedure is the manner in which the sediment-water slurry should be mixed. The problem is that the redox conditions that are to prevail during the elutriate test are not prescribed. This situation could lead to highly variable results where minor variations in mixing technique could markedly change the results of the elutriate test. This is a result of the fact that if the test vessel is open to the atmosphere, the manner and degree of agitation controls the amount of oxygen introduced into the sediment-water slurry, Since many dredged sediments are from anoxic environments, the introduction of oxygen into the sediment-water slurry during the 30-minute mixing period could result in appreciable oxidation of iron and sulfide species. The oxidation of the ferrous iron typically present in sediments results in the formation of ferric hydroxide which in turn is an effective scavenger of many trace contaminants such as phosphate, heavy metals, and organics such as some chlorinated hydrocarbon pesticides and PCBs.

The scavenging ability of ferric hydroxide is dependent on many factors: its rate of formation; period of time since formation; the presence of various types of chemicals in the water, especially organic matter; pH; and a variety of other factors which are poorly defined at this time. While, in general, the trends of the impact of the dominant factors affecting the scavenging ability of ferric hydroxide are known, little information exists on the interrelationships among these factors. For further discussion of this topic, consult Lee.[2]

Lee <u>et</u> <u>al</u>.[1] recommend that the elutriate test be conducted with compressed air agitation during the mixing period. The compressed air provides adequate mixing and most importantly keeps the system oxic during this period. While with few exceptions the sediment-water slurry is anoxic during dredging and transport, shortly after disposal, the turbid plume becomes oxic. At that time, the ferrous iron is oxidized by the dissolved oxygen to the ferric form which results in the formation of ferric hydroxide. As noted above, the ferric hydroxide acts as a scavenger for many contaminants. While the rate of reaction between ferrous iron and oxygen is rapid in many waters, it is not instantaneous. There is some evidence (see phosphorus section) that the reaction is sufficiently slow at some dump sites so that released soluble orthophosphate remains in the water column several minutes after the dump.

If the elutriate test were designed to exactly mimic a typical disposal operation, it should, in general, be conducted with an anoxic mixing period followed by a period of aeration. At the end of the anoxic mixing period, which should last from about five minutes to one hour depending on the type of dredging operation that will be conducted (i.e., hydraulic, pipeline or hopper dredge), the slurry should be mixed with disposal site water. This mixture should be aerated for about five minutes and then allowed to settle for one hour. It may be necessary to use pure oxygen at that point to satisfy the oxygen demand of the sediments or use a longer mixing period with compressed air. As discussed by Lee et al.[1], it is important to keep the system oxic prior to filtration in order to properly simulate the conditions that will occur in the water column after disposal. The only exception to this would be if disposal were to take place in anoxic waters. The maintenance of oxic conditions prior to filtration is appropriate because with few exceptions the potential significance of the chemicals of concern are to aerobic organisms such as fish. Since these organisms will not be present under anoxic conditions, before a contaminant released from the dredged sediment could be of significance to them, it would have to be exposed to oxic conditions.

While the elutriate test recommended by Lee et al.[1] does not exactly mimic the redox conditions that will occur during typical open water disposal of dredged sediment, it does provide a much more realistic test than the currently prescribed elutriate test generally being used by the Corps of Engineers Districts. The failure to properly define the redox conditions during the elutriate tests is resulting in the production of a large amount of essentially meaningless elutriate test data across the US. It is unlikely that the differences between the results of the Lee et al.[1] test and one that would more closely mimic actual site conditions would be insignificant as far as predicting the potential for contaminant release during disposal. As indicated above, the primary controlling factor for the release of chemical contaminants during open water disposal of dredged sediments is the presence or absence of freshly formed iron hydroxide floc. In general, the characteristics of this floc would largely be independent of any anoxic mixing that occurs prior to the introduction of oxygen. Further, use of both dredging and disposal site waters should not be necessary if appropriate interpretation of contaminant concentrations in the elutriate is made with consideration of redox and mixing conditions at the disposal site. Therefore, the use of an elutriate test using dredging site water with compressed air mixing such as that recommended by Lee et al.[1] rather than a test that would more closely mimic actual disposal site conditions is justified based on the fact that the results should be essentially the same. Therefore, the more simple test should be the one that is used.

PART III: EVALUATION OF THE APPLICABILITY OF THE ELUTRIATE TEST

Introduction

Field studies were conducted at various waterways throughout the US in which sediment samples were collected from a location in a navigation channel that was to be dredged on the following day. These sediments were subjected to the elutriate test and various modifications thereof and also characterized by their bulk chemical content. A comprehensive disposal site water column monitoring study was conducted immediately prior to, during, and immediately following disposal of sediments dredged from the previously sampled waterway location.

Using data collected during this study, it is possible to interrelate several environmentally significant aspects of open water dredged material disposal: prediction of contaminant release or uptake using the elutriate test; actual contaminant release associated with dredged material disposal; and concentrations of contaminant in the dredged sediment, dredging pump discharge, hopper supernatant, and/or pipeline effluent. In several of the study areas examined, information for each of these components was obtained. From the other study areas, information on only selected components was gathered. Presented below is a discussion of the potential for release of selected chemical contaminants during open water dredged material disposal as predicted by elutriate tests; as related to concentrations in the sediment to be dredged; and as actually found in the water column at or near the disposal site during the disposal operations.

Predicting Release of Phosphate during Open Water Dredged Material Disposal*

Phosphorus is important to consider in the evaluation of a dredged sediment for open water disposal because of its ability to stimulate excessive growths of aquatic plants in many natural waters. This is especially true for freshwater systems such as the Great Lakes. This section discusses the capabilities of the elutriate test to predict phosphate release during open water dredged material disposal.

Hopper and mechanically dredged sediment - open water dumping

During the Perth Amboy Channel sediment dump, surface and mid-depth soluble ortho P concentrations about 100 m downcurrent from the dump appeared to have remained fairly constant. This would be expected since the surface current was moving in the opposite direction. The soluble ortho P concentrations in the near-bottom water increased by a factor of about 10 above ambient to about 0.24 mg P/l and remained elevated for about 10 minutes in association with the passage of the turbid plume arising from the dump event. A hundred or so meters farther down bottom current, the soluble ortho P increase was only to about 0.16 mg P/l and lasted for less than five minutes. This pattern between the two sampling vessels was expected as a result of dilution, dispersion, and other processes (such as coprecipitation with iron hydroxide floc) which would decrease the concentration within the plume over time.

The Perth Amboy Channel oxic elutriate test predicted soluble ortho P uptake, whereas the anoxic ones predicted release. The concentrations at the disposal site were greater than ambient after dumping under oxic conditions

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*For an expanded discussion of the phosphorus results, consult the Ph.D. dissertation of A. Jones.[3]

whereas the oxic elutriate predicted soluble ortho P uptake. However, it is actually uptake that is observed in the field after the initial release of dredged sediment, as indicated by the decrease in concentrations between the two sampling vessels.

Essentially the same pattern was seen with the Bay Ridge Channel sediment dumps. The approximate magnitude of the maximum concentration found about 100 m down bottom current from the dump (0.7 mg P/l) was approximated by the anoxic elutriate soluble ortho P concentrations (0.92 to 1.2 mg P/l). The oxic tests predicted uptake; a reduction in maximum concentration (from 0.7 to 0.47 mg P/l) was found between the two sampling vessels positioned about 100 and 200 m downcurrent from the dump.

The other location where the open water dumping of hydraulically dredged sediment was monitored was at the Galveston Bay Entrance Channel (GBEC) disposal site in the Gulf of Mexico. Two to four months before the GBEC sediment dumps were monitored, sediment and water samples were collected in the GBEC by Piwoni[4] for elutriate test analysis. The elutriate tests run on GBEC Buoys 1 and 9 samples (20 percent oxic) showed release of soluble orthophosphate to 0.064 and 0.024 mg P/l in the dredging site water.[4] In general, the maximum soluble ortho P concentrations found in the water column immediately following dumping of sediments from the GBEC Buoy 9 vicinity (GBEC Dumps 1, 2, and 3) and by those from the GBEC Buoy 1 area (GBEC Dumps 4 and 5) were not approached by the concentration found by Piwoni[4] in oxic elutriates of these sediments. For GBEC Dumps 1 - 4, the maxima were on the order of 0.5 mg/l but were, in general, based on a single peak value and may not be representative of the conditions that generally prevailed. Although too few samples were taken to clearly define a peaking concentration, the fact that a similar peak concentration was found during four dumps indicates that those values may not be spurious. GBEC Dump No. 5 showed an increase in soluble ortho P to about 0.13 mg P/l.

Anoxic elutriate tests had not been run on these sediment samples. These tests would have likely provided a reasonable estimate of the maximum concentration found immediately after the dump as was found with the New York-New Jersey sediments. Since the oxic tests run showed release of soluble ortho P, it is likely that in addition to the dilution of released soluble ortho P associated with dispersion of the turbid plume, there may be a small amount of additional soluble ortho P released within the turbid plume. Compared with the amount of dilution typically found at open water disposal sites, the amount of additional release shown by the GBEC elutriate tests (after 1.5 hours contact time with no dilution) would not likely be significant; the turbid plume would be expected to be completely dispersed in that amount of time. It should also be recalled that the elutriate tests were run on sediments collected several months prior to the monitored disposal operations. This may affect the comparability of the field and laboratory data for this site study.

In association with Texas City Channel Dump No. 2 (dredged material from the TCC Turning Basin), the surface water soluble ortho P concentrations decreased whereas those in the mid-depth and bottom waters increased to a maximum of 0.56 mg P/l. The 20 percent oxic elutriate tests run on TCC Sites 1 and 4 turning basin sediments showed an approximate doubling of soluble ortho P concentration to about 0.25 mg P/l. Tests on TCC-5 sediment showed essentially no change with resulting concentration of 0.15 mg P/l in the elutriate. The TCC-1 anoxic/oxic test overestimated disposal site release, showing a concentration of 2.1 mg P/l. The oxic elutriates showed the potential for continued release in the turbid plume.

In conjunction with the Perth Amboy Anchorage barge dump in the New York Bight, there was an increase in soluble ortho P at all three depths monitored at about 100 m down surface current from the dump, to about 0.13 to 0.18 mg P/l. This increase was not associated with an increase in turbidity and, therefore, may not be directly related to the disposal event and passage of the surface turbid plume. At about 100 meters down bottom current from the dump, soluble ortho P concentrations increased to 0.22 mg P/l in association with the passage of the turbid plume.

The Perth Amboy Anchorage elutriate tests showed varying results. The two and three-fold increases in concentration in the 20 percent oxic test and plop tests suggest that factors other than the iron system, such as desorption, may be controlling phosphate behavior with these sediments. The fact that there was soluble ortho P uptake in the five percent oxic test indicates that the pattern of oxic release found in the 20 percent elutriates may not occur in the turbid plume associated with the disposal. The plop tests indicated soluble ortho P release to 0.4 mg P/l. The results of the plop tests may have been influenced by the fact that dredging, rather than disposal site water was used for the tests.

The general trend during the two Duwamish River barge disposal operations was a decrease in soluble orthophosphate concentrations in the bottom waters in the turbid plume associated with the dump, with no detectable change at surface and mid-depths. Concentrations generally decreased to 0.03 to 0.04 mg P/l then returned to ambient levels (0.08 to 0.09 mg P/l).

The Duwamish River elutriates showed a general lack of reproducibility between replicates. The five percent oxic tests, in general, showed a tendency for soluble ortho P uptake, as did the plop test, to 0.03 to 0.05 mg P/l. This was the order of uptake found at the dump site during the passage of the turbid plumes.

Hydraulically dredged sediment - pipeline disposal

There appears to have been a measurable increase in soluble orthophosphate concentrations in the bottom waters during the James River disposal operation. Concentrations in the area of discharge (within 25-50 m) were in the 0.03 to 0.04 mg P/l range; although, in the bottom waters at the point of discharge, they were 0.13 to 0.35 mg P/l. This increase in soluble ortho P above the ambient <0.01 mg P/l appears to have been highly localized near the discharge; concentrations at sampling locations a few hundred meters from the discharge were at background levels. In many cases, samples containing elevated concentrations (including those at the discharge), contained large amounts of finely divided particulate matter which were not removed by filtering through a 0.45 μ pore size filter. Even after refiltration through a 0.2 μ pore size filter, some of what was measured as released soluble ortho P may have actually been associated with particulates.

All of the elutriate tests on the James River sediment generally showed a slight decrease or essentially no change in soluble orthophosphate concentration from that in the site water used. Concentrations in the elutriates were about the same as the maximum levels found in the vicinity of the discharge. It is likely that sorption on the finely-divided particulates associated with the James River sediment influenced the soluble ortho P behavior in the elutriates. If there was soluble ortho P release during dredged sediment transport through the pipeline, this was not indicated by the elutriate tests. It should be noted that compared with actual sediment-water contact time during dredging-disposal, the contact time provided in the elutriate test (over 1.5 hours) was quite long. This could have allowed the removal of phosphate by the particulates in the elutriate test so that any release occurring during mixing would not have been reflected in elutriate concentrations.

During the Mobile Bay pipeline disposal operation, the discharged dredged material flowed from the discharge pipeline in a relatively thin layer between 3 and 4 meters depth. This density current, evidenced by both high turbidity values and elevated soluble orthophosphate concentrations, was detected as far as 400 meters downcurrent from the discharge. It was not, however, found on either side of the direct downcurrent path. Soluble ortho P concentrations typically ranged from 0.07 to 0.44 mg P/l in the mud flow.

The soluble orthophosphate concentrations 400 meters upcurrent from the discharge appeared to have been within the general ranges of soluble orthophosphate concentrations found 400 to 800 meters downcurrent from the discharge. They appeared, however, to have been somewhat higher than those in the near-surface water immediately downcurrent from the discharge. This may imply that there may be some decreases in surface water concentrations related to the disposal operations.

Since it appeared that the D.O. concentration in the mud flow was generally near or below 0.5 mg/l at stations up to 140 meters directly downcurrent from the discharge, the anoxic test would likely be the best predictor of expected soluble orthophosphate behavior in the immediate area of the dump. The anoxic elutriate tests predicted release to 0.086 to 0.25 mg P/l. This was approximately the level of soluble ortho P found in the mud flow. The oxic tests showed slight release or no change in soluble ortho P concentrations, resulting in oxic elutriate levels of 0.03 to 0.1 mg P/l which may indicate that upon oxidation of the mud flow, a decrease in concentration would be dependent on dilution.

From the data collected, there did not appear to be a consistent pattern to the behavior of soluble ortho P during the five sampling sequences at Apalachicola Bay. The increased concentrations found in the disposal site water column showed no particular trend with increasing distance from the discharge. Because of the variable nature of pipeline disposal operations, it would have been necessary to use a greater number of sampling vessels and collect considerably greater numbers of samples than allowable within the framework of this study to define the release and fate of chemicals which had been associated with the dredged sediment. The field studies did indicate that there was not a massive release of soluble ortho P and that by a few hundred meters downcurrent from the discharge, concentrations were at ambient levels.

The samples collected in the surface waters about 50 meters from the discharge over a 2.5 hour period showed soluble ortho P concentrations to range from <0.01 to 0.13 mg P/l, indicating variable degrees of release. This fluctuation was likely related to the nature of pipeline disposal operations in which the percent sediment in the discharge is variable over time and discharge is often not continuous. It is also likely related to the highly stratified nature of the dredging site water column in which the surface waters were fresh and the bottom waters were saline. This would lead to a variable mixture during dredging and also discharge. This is demonstrated by the elutriate test results in which the surface water elutriates showed soluble ortho P release whereas those run with the bottom water showed uptake or essentially no change in concentration.

From soluble orthophosphate and turbidity data collected in conjunction with the monitoring of the Upper Mississippi River pipeline disposal operation, it appeared that the dredged material flowed from the onshore discharge area, downcurrent along the river bottom. With the increase in turbidity with depth at the sampling station nearest the discharge, there appeared to have been a decrease in soluble ortho P concentrations from about 0.25 (surface) to 0.09 mg P/l (near bottom). This was likely related to the fact that the disposal area remained oxic (D.O. = 3-4 mg/l near disposal area) during sampling, and under that condition, the iron present could act to remove soluble phosphate from the water. In general, however, the water columns at the sampling stations were well mixed with respect to soluble orthophosphate concentrations and generally were in the 0.20-0.25 mg P/l range.

Overall, there appeared to have been little effect on the soluble orthophosphate concentrations as a result of either the dredging or disposal activities in the Upper Mississippi River. Concentrations upcurrent from the dredge, downcurrent from the dredge, and within a few hundred meters of the disposal site were about the same. The elutriate tests run on the dredging site sediment proved to be good predictors of phosphorus behavior in the water column during disposal operations. Both the 5 percent and 20 percent oxic elutriates showed uptake of soluble orthophosphate to approximately the degree found in the field in the area of disposal (to about 0.1 mg P/l). The 5 percent tests were slightly better predictors than the 20 percent. This might be expected since some of the dredged sediment was retained on the disposal mound at the edge of the discharge island and did not enter the river.

From an overall point of view, the standard oxic and anoxic elutriate tests appear to be applicable for use in connection with pipeline disposal of hydraulically dredged sediments. They give a reasonable estimate of soluble ortho P behavior after disposal.

Comparison of field study results to expected environmental behavior of phosphate

One of the major factors controlling the behavior of phosphate as well as a number of other contaminants in natural waters is the iron system, with particular emphasis on the presence or absence of D.O. When oxygen is added to anoxic waters, ferrous iron becomes oxidized and can form an amorphous ferric hydroxide colloid-precipitate which tends to agglomerate, forming a floc which is an effective scavenger of dissolved phosphate heavy metals and many organic compounds. As discussed by Stumm and Lee[5], the rate at which ferrous iron is oxidized to the ferric form is directly dependent on the D.O. and Fe II concentrations and on the square of the $[OH^-]$ concentration. Their laboratory work has shown that at pH 7.2, essentially all of the ferrous iron in an approximately 30×10^{-6} M Fe II solution ($\sim$1.7 mg/l Fe II) is oxidized within about eight to ten minutes. This rate would be expected to be somewhat faster at a higher pH but lower with lower Fe (II) or oxygen concentrations. The rate and extent of uptake of phosphorus and other contaminants by iron hydroxide floc do not follow a stoichiometric ratio but are, according to Lee[2], dependent on the age of the floc. Considerably more phosphate can be incorporated if floc formation takes place in the presence of the phosphate than if phosphate is mixed with aged or preformed floc. Destruction of the floc will result in the release of the associated phosphate and other chemicals. This can only occur with very low pH or in the absence of oxygen.

The sorption and release of contaminants from clay minerals can also be an important factor controlling phosphate behavior in natural waters. The degree and duration of sediment-water contact typically control these reactions. The sediments of most of the waterways examined during this study were anoxic. It is expected that this is also true of most waterway sediments especially those near urban centers. The amount of oxygen added during the dredging process is generally believed to be insufficient to satisfy this oxygen demand and cause oxic conditions in the slurry in the hoppers or in the disposal pipelines.

There is limited contact of dredged sediment with oxygen during mechanical dredging and barge transport. Therefore, during dredging and transport of dredged sediment to the disposal site, anoxic conditions typically prevail. The mixing which occurs during hydraulic dredging and transport allows the release of interstitial water soluble orthophosphate as well as phosphorus which is loosely bound to sediment fractions. Since mixing during mechanical dredging-barge transport is limited, release during this type of dredged material disposal would be expected to be minimal. The waters of open water disposal sites are typically oxic. If there is a thermocline or chemocline present, hypolimnetic waters may be anoxic.

When the anoxic sediment-water slurry is dumped via hopper or barge, the majority of the mass of materials falls rapidly to the sediments and is not mixed to any significant degree with the oxic dump site waters. There is, however, considerable mixing of the oxic waters with the solids which form the turbid plume. This action promotes the oxygenation of the ferrous iron associated with the dredged sediment and results in the removal from solution of phosphate and other contaminants released during dredging and transport. While the rate of reaction between ferrous iron and oxygen is rapid in many waters, it is not instantaneous. At a dredged material dump site, therefore, it would be expected that relatively high soluble ortho P concentrations would be present in the hopper or barge-contained material. High concentrations would also be expected when the turbid plume is first created beneath the hopper or barge. As this turbidity cloud is moved with the currents, rapid removal of released soluble ortho P would be expected due to its association with freshly forming ferric hydroxide floc, dilution, and sorption on finely-divided particulate matter. The net result would be rapidly decreasing concentrations of soluble ortho P and other contaminants in the turbid plume with distance from the disposal vessel, i.e., increasing time for oxidation of iron, precipitation and sorption reactions to occur, in addition to dilution. Whereas soluble orthophosphate concentrations would be expected to decrease rapidly in the turbid plume after the dump, the total phosphorus concentration would be expected to decrease in proportion to the turbid plume dispersion since removal of the dissolved fraction would be onto fine materials within the plume. There is the possibility that additional soluble ortho P release could occur in the turbid plume, but it has been found[6,7] that release under oxic conditions in the presence of iron is slow; it would not likely occur before complete dissipation of the turbid plume. Additional release may occur if there is very limited iron present to remove released phosphate and if the behavior of phosphate is controlled by desorption-sorption mechanisms.

The same principles discussed above would also be applicable to open water pipeline disposal operations. One of the differences would be that there is generally less time between dredging and pipeline disposal for the release of sediment-associated phosphate. Also, a larger portion of the dredged material slurry can be mixed with the disposal site water. As oxygen from the surrounding waters is entrained or mixed into the turbidity current-mud flow, the ferrous iron would be expected to be oxidized to ferric with the concomitant coprecipitation of soluble orthophosphate and a number of other contaminants.

Predicting Release of Selected Heavy Metals during Open Water Dredged Material Disposal

Of all the chemical contaminants present in dredged sediments, the heavy metals have received the greatest attention. This is a result of the fact that certain forms of some metals are highly toxic to aquatic life. Also, some heavy metals tend to bioconcentrate in aquatic organisms to the point that they can make organisms unsuitable for use as food. Of greatest concern for both of these types of problems are Hg, Pb, Zn, Cd, Cu, and to a lesser extent, Ni. Other heavy metals such as Fe and Mn show little or no toxicity to aquatic life or man. In large amounts, however, they can cause water quality problems. These problems are primarily associated with the use of natural waters as domestic water supplies as high concentrations of Mn and Fe can cause problems such as staining of clothing, fixtures, etc. Also, there is a potential problem associated with large amounts of Mn in marine waters due to its bioaccumulation within marine molluscs which could render them unsuitable for use as food.[8]

For the purpose of this report, another element, arsenic, is included with the metals even though it is not a metal. This study has included measurement of all of the above heavy metals, including arsenic. A discussion of the results obtained in these studies is presented below.

Based on the review by Lee and Plumb[9] and the studies of Lee *et al.*[1], it was concluded that the only heavy metal of the group investigated that would likely be released in sufficient quantities to potentially affect water quality upon open water disposal of dredged sediment is Mn. Lee and Plumb[9] concluded that based on the aqueous environmental chemistry of heavy metals, there should be no relationship between the bulk content of a heavy metal in dredged sediment and the release of this heavy metal during open water disposal. The studies of Lee *et al.*[1] which investigated the factors influencing the results of the elutriate test, as well as this study generally confirm this conclusion. It is therefore of interest to examine the relationship between the elutriate test results and heavy metal concentrations in the disposal site water column during open water disposal of dredged sediments.

While the majority of the elutriate tests run during this study showed Mn release, there were a few, such as Duwamish River-Puget Sound, and Bay Ridge Channel which showed uptake. In general, concentrations of Mn in the turbid plume or density current near the points of discharge were somewhat above ambient. For most of the sites studied, the elutriate test results provided a general indication of the level of Mn release that was found in the field.

Many of the elutriate tests and disposal operations showed an apparent release of iron. This result is more likely due to an artifact of the analytical procedures used than to an actual increase in concentration of soluble iron. As discussed in the phosphorus section of this report, at the temperature and pH conditions that generally prevailed in the elutriate and disposal site waters, the ferrous iron present in the sediments should be rapidly oxidized to ferric by dissolved oxygen and precipitate as iron hydroxide. However, as discussed by Lee[2], appreciable amounts of colloidal ferric hydroxide are formed which can pass through a 0.45 μ pore size filter that was used in this investigation to separate "dissolved-soluble" from particulate species. It is therefore highly probable that the apparent release of iron is due to the presence of a colloidal ferric hydroxide.

The studies of Delfino and Lee[10] have shown that under natural water conditions, the oxidation of Mn is typically much slower than the oxygenation of Fe by dissolved oxygen. This leads to a situation in which soluble manganese present in the sediments could persist in the presence of D.O. in the elutriates and in waters near the disposal site. However, in time, it would be expected that manganese would be oxidized to MnO_2 which is highly insoluble. Both the ferric hydroxide and manganese dioxide would tend to have a strong scavenging effect on other contaminants which would remove soluble species from the water column and transport them to the sediments. For further discussion of this phenomenon, consult the work of Lee[2], Chen *et al.*[11], Khalid *et al.*[12], and Gambrell *et al.*[13]

In general, it was found that there were no or only slight changes in the concentrations of Cd, Cr, Ni, Pb, Cu, Hg and As as a result of 20 percent oxic elutriation. While the pattern for Zn was occasionally erratic, there tended to be a general trend for Zn removal during 20 percent oxic elutriation. Similar patterns to those found in elutriate tests were found for these heavy metals in the turbid plumes at essentially all of the various dredged material disposal sites that were monitored during this study. While there are a few aberrant heavy metal concentrations associated with the turbid plumes for the various disposal operations, the general trend for the data obtained for each of the disposal sites showed no release of Cd, Cr, Ni, Pb, Cu, Hg, As, and Zn.

From an overall point of view, it is concluded from these studies that the elutriate test is a valid tool to use for predicting the general trend of uptake and/or release of the various sediment-associated heavy metals and arsenic during open water disposal of dredged sediment. Further, these general trends are in accord with what would be expected based on the aqueous environmental chemistry of heavy metals under the conditions typically found in fresh and marine waters at dredged material disposal sites.

Predicting Release of Nitrogen Compounds during Open Water Dredged Material Disposal*

Nitrogen compounds are important to consider in an evaluation of the potential tendency of dredged sediments

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*This section of the report was based in part on draft materials submitted by P. Bandyopadhyay.

from two points of view. First, ammonia is highly toxic to aquatic life, and various forms of sediment-associated nitrogen can be slowly converted to ammonia. This point is discussed further in this section. The second reason for the importance of nitrogen compounds stems from the fact that both nitrate and ammonia can stimulate excessive growths of aquatic plants, especially in marine waters. Since most dredged sediments are anoxic, there is little concern about nitrate in these sediments; nitrate is not thermodynamically stable under anoxic conditions. It is apparent from a review of the significance of dredged sediment-associated nitrogen compounds in natural waters that primary concern should be directed toward the release of ammonia from dredged sediments during open water dredged sediment disposal. Lee _et_ _al_.[1] found that ammonia was readily released in elutriate tests run on various US waterway sediments. It is therefore of interest to compare the magnitude of ammonia release in elutriate tests to that actually found during disposal of dredged sediments. These comparisons for the sites investigated during the current study are presented below. In the discussion of these results, the term "ammonium", unless otherwise specified, represents the sum of ammonia and ammonium in the sample.

Hopper and mechanically dredged sediment - open water dumping

Dredged material disposal operations from a hopper dredge were monitored at the Galveston Bay Entrance Channel study site. Sediments dredged in the mid to upper reaches of the Galveston Bay Entrance Channel generally released little or no ammonium to the disposal site water column. Texas City Channel (TCC) Dump No. 2 showed ammonium release which was significantly higher than that associated with other disposal operations monitored involving sediments from the Galveston Bay Entrance Channel. The highest concentration found in the water column, 1.9 mg N/l, was in the bottom waters and persisted for less than two minutes while the turbid plume passed the anchored sampling ship. Measurable amounts of ammonium were detected in turbid plumes from other disposal operations, but they were lower than those found during Texas City Channel Dump No. 2. Elevated ammonium concentrations persisted in the water column at the monitoring site for only very short periods of time, usually on the order of a few minutes.

Increased concentrations of organic N were found in the turbid plumes. This was expected since organic nitrogen in natural water systems occurs primarily in a particulate form. In several disposal operations, nitrate content appeared to increase during passage of the turbid plume. However, the magnitude and duration of increased levels would not likely have any adverse effect on water quality in disposal site waters. The apparent increase in nitrate is probably due to analytical problems.

The ammonium release at the Galveston Bay Entrance Channel offshore disposal site followed the expected pattern predicted by the elutriate tests. The highest ammonium concentrations in the elutriates of sediment from this region was 15.7 mg N/1 in a Texas City Channel sample which corresponds well with the highest ammonium release (1.9 mg N/1) observed during the monitoring of the Texas City Channel dump. The difference between the elutriate test results and concentrations found in the field reflects in part the dilution that occurs at the point of discharge (during discharge) and during transport from the point of discharge to the monitoring site 40 m away.

At the New York Bight mud dump site, dredged material disposal into open water from a hopper dredge resulted in an ammonium release similar to that found for TCC Dump No. 2. At both of these sites, somewhat elevated (above ambient) ammonium concentrations persisted in the near-bottom waters for about one to two hours. Highest ammonium concentrations in the bottom water following the two hopper dumps ranged from 1.4 to 2.8 mg N/l, compared to elutriate concentrations ranging from 5.2 to 33.0 mg N/l. Ammonium concentrations in the samples collected directly from the hopper dredge showed considerable variation and ranged from 5 to 41 mg N/l for Perth Amboy Channel samples and from <0.05 to 70 mg N/l for Bay Ridge Channel samples. The fact that lower values were found in the disposal site water column is related to the dilution that occurred between the discharge point and the sampling point.

The barge disposal in Elliott Bay-Puget Sound of mechanically dredged sediment was monitored during this investigation. Except for a single data point of 2.6 mg N/l in the surface water following Dump 2B, ammonium release at the disposal site was less than that found at other dump sites investigated. Values for ammonium concentrations, as predicted by the elutriate test, ranged from 1.9 to 5.8 mg N/l, and were comparable to the highest ammonium concentration (2.6 mg N/l) in the disposal site water column. Ammonium concentration, predicted by the plop test (0.86 mg N/l), gave a closer estimation of the actual highest field concentration in the bottom water (0.2 mg N/l) than the elutriate test. This would be expected since the plop test was designed to more closely simulate the conditions that prevail during barge disposal of mechanically dredged sediment.

Mechanically dredged sediment from the Perth Amboy Anchorage was transported by barge and disposed of at the New York Bight mud dump site. The highest ammonium concentration observed in the disposal site water column during passage of the turbid plume was 2.4 mg N/l compared to 8.9 to 14.4 mg N/l estimated by the 20 percent elutriate tests. It appears that the actual ammonium release following barge disposal can be better estimated by conducting plop tests which predicted values of about 3 mg N/l.

From an overall point of view, it can be concluded that the elutriate test for hydraulically dredged sediments and the plop test for mechanically dredged sediment are useful tools to estimate the amount of ammonium released to the water column during open water disposal. These results are what would be expected based on the aqueous environmental chemistry of nitrogen compounds in natural water systems.

Hydraulically dredged sediment-pipeline disposal

The disposal operations in Apalachicola Bay did not significantly change the nitrogen compound concentrations in the water column even near the discharge point. The high ammonium and organic N concentrations observed in the samples of pipeline discharge were not seen in samples taken only 20 to 50 meters downcurrent from the point of discharge. Organic N and ammonium in samples of the pipeline discharge varied from 163 to 422 mg N/l and from 38 to 184 mg N/l, respectively. Organic N and ammonium in samples collected 20 to 50 meters downcurrent varied from <0.05 to 3.4 mg N/l and from <0.05 to 8.6 mg N/l, respectively. It appears that these variations were due to the variability in composition and volume of material discharged together with the variability associated with the operational procedures of the dredging and discharge. Ammonium and organic N decreased with increasing distance from the discharge point, indicating dilution. The ammonium and organic N concentrations were generally higher in the near-bottom water samples. This is due to the formation of a density current at the point of the pipeline discharge. However, none of the nitrogen compounds showed any correlation with turbidity or salinity values in the region of the discharge. Nitrate showed no pattern with respect to either depth or distance from the discharge point.

Ammonium concentrations in the elutriates ranged from 2.4 to 11.1 mg N/l. By comparing these concentrations to those in the water column near the discharge point can be concluded that the elutriate test provided a reasonably accurate estimation of ammonium released during dredged material disposal at the Apalachicola Bay Study site.

A density current was also formed by the dredged sediment discharge at Mobile Bay disposal operation. The elevated concentrations in the density current were found to decrease with distance downcurrent from the discharge. Ammonium in the bottom water at 50 m from the discharge point was approximately 6 mg N/l, whereas at 800 m downcurrent from the discharge point, the concentration was 1.6 mg N/l.

Nitrate and organic N showed no general pattern with respect to either distance from the discharge point or depth at any station. The concentrations of ammonium and organic N in the elutriate were slightly lower than those found near the discharge point. For example, 8.3 mg N/l ammonium was found near the disposal point compared to 6.1 mg N/l predicted by the elutriate test. Similarly, an organic N concentration of 5.4 mg N/l was found near the discharge point whereas a concentration of 1.4 mg N/l was estimated by the elutriate test.

At the James River study site, ammonium showed rapid dilution by short distances downcurrent from the discharge point. For example, at the point of discharge, ammonium in the bottom water (turbidity current) was 24.1 mg N/l; that at 25 m downcurrent ranged from <0.05 to 1.7 mg N/l and at 250 m was <0.05 mg N/l. Nitrate and organic N did not vary with respect to either distance from the discharge point or depth at any location. Turbidity correlated well with the ammonium concentrations.

The elutriate test did not provide as good an estimate of the release under actual field conditions in the James River as was found at the other disposal sites studied. The ammonium concentrations near the discharge point (as high as

24.1 mg N/l) were somewhat higher than the range predicted by the 20 percent elutriate test (3.9 to 5.8 mg N/l).

There was rapid dilution of elevated ammonium concentrations associated with the pipeline discharge at the Upper Mississippi River disposal site. Ambient concentrations were found within 200 m downcurrent from the discharge point. Ammonium concentrations in the elutriate (about 2 mg N/l for 20 percent tests) were about the same as the values observed in the discharge runoff from the island disposal (2.9 and 8.9 mg N/l).

From an overall point of view, it was found that for the pipeline disposal operations monitored, the general order of magnitude of ammonium concentration found near the point of discharge was predicted by the elutriate test. The prediction of the ammonium concentration near the discharge was, in general, good considering all the factors that could influence the results of the field studies such as variability in sediment composition, volume, and dredge operation.

Of all the reactions which affect concentrations of various forms of nitrogen in natural waters, those which are of potential concern in relation to open water dredged material disposal are those which affect the concentrations of ammonium. Of particular concern are the reactions that govern the conversion of organic nitrogen to ammonium and the conversion of ammonium to nitrate. As discussed by Brezonik and Lee[14], Austin and Lee[15] and Cowen _et al_.[16], both of these reactions are bacterially mediated and take place at relatively slow rates in natural water systems. They often require a week to a month or more for completion. This means that as far as a dredged material disposal operation is concerned, the changes in concentrations of ammonium that occur in the density current or turbid plume would be influenced little by these reactions. The primary factors governing the concentrations of ammonium in the discharge waters are the desorption reactions from the sediment particles and the dilution of the interstitial waters.

Predicting Release of Chlorinated Hydrocarbon Pesticides and PCBs during Open Water Dredged Material Disposal*

Chlorinated hydrocarbon pesticides (CHPs) and polychlorinated biphenyls (PCBs) are among the most hazardous chemicals present in natural waters. They are known to bioconcentrate in aquatic organisms to the point where these organisms become unsuitable for use as food. Widespread contamination of aquatic environments by these compounds has been found.[17,18,19] Due to their general hydrophobic nature, they tend to become associated with the sediments. The persistence of these compounds in sediments has caused considerable concern about the environmental impact of dredging and deposition in open waters of dredged material containing CHPs and PCBs. In this study, CHPs measured were lindane, heptachlor, aldrin, dieldrin, endrin, mirex, methoxychlor, both isomers of endosulfan, as well as op'DDT and pp'DDT and their isomers. Samples from the James River and Bailey Creek, Virginia, were also examined for the presence of kepone. This section of the report presents a discussion of the use of the elutriate test to predict the behavior of dredged sediment-associated PCBs and CHPs during open water disposal.

In general, the data showed that there was an inverse relationship between the oil and grease content of sediments and PCB and CHP release in the elutriate test. Further, for some sediments, there appeared to have been a general direct relationship between the percent sediment used in the elutriate test and the amount of CHP and PCB release. It should be noted that the apparent effect of percent sediment on release may be an artifact of the method used to determine release. This method was based on centrifugation. With

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*This section of the report was based in part on draft materials submitted by J. Butler.

higher percent sediment in the elutriate test, there would be a higher amount of finely divided particulate matter which would not be removed during centrifugation, thereby giving the appearance of greater PCB and CHP release with increasing percent sediment.

Hopper and mechanically dredged sediment - open water dumping

Samples of dredging pump discharge and hopper supernatant were collected during dredging of the GBEC Buoys 1 to 3 area. The pump discharge samples were separated into "soluble" and particulate fractions. The soluble PCB concentration in the dredging pump discharge (44 ng/l) was reduced to 34 ng/l by the time the hopper supernatant was sampled, i.e., after about 30 minutes of the materials standing in the hoppers.

In the elutriate test run on GBEC Buoy 1 samples, the PCB concentration was reduced from 210 ng/l in the site water to 170 ng/l in the elutriate. The elutriate test indicated no change or a slight reduction (16.0 ng/l to 14.7 ng/l) of CHPs. The field studies showed more sorption of CHPs than was indicated by the elutriate test. The elutriate test did indicate that the Buoy 1 sediment would tend to sorb CHPs and PCBs; the field studies verified this.

A comparison of the Bay Ridge Channel elutriate test and field study results is presented in Table 1. The 20 percent elutriate test for the Bay Ridge Channel sediments indicated release of PCBs and little or no release of CHPs. The concentration of PCBs in the elutriate was over ten times the amount found in the site water, whereas only slight release of PCBs and CHPs was found during dredged sediment disposal. The disposal site water samples were collected about 200 meters downcurrent from the disposal operations. The mixing and dilution that occurred between the point of dumping and the

Table 1

Comparison of CHPs and PCBs in the Water Column Before and After Bay Ridge Barge Dump, with 20 Percent Oxic Elutriates

Compound	Elutriate Test			Field Samples (ng/l)			
				Depth (m)			
	Sediment** (μg/kg)	Water (ng/l)	Elutriate (ng/l)	12 Pre Dump	24 Pre Dump	24 Post Dump	12 Post Dump
Aldrin	22.7	9.2	<0.4	1.0	1.2	26.0	2.5
pp'DDT	66.0	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0
pp'DDD	47.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
op'DDE	4.4	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
pp'DDE	13.0	<2.0	5.2	<2.0	<2.0	<2.0	<2.0
Heptachlor	14.6	<0.4	5.3	<0.4	<0.4	<0.4	<0.4
Lindane	16.7	5.0	4.5	1.2	1.5	<0.3*	2.8
PCBs	1820	9	86	14	10	17	12
		(mg/l)	(mg/l)	(mg/l)	(mg/l)	(mg/l)	(mg/l)
Oil and Grease	826(mg/kg)	<0.5	3.6	2.3	0.5	1.0	0.5
TOC	3.2%	15.0	79.0	5.0	2.6	17.6	2.5
Soluble TOC	NT	9.2	62.0	3.0	2.6	4.1	2.2

*Indicates compound detected but not measurable.

**Results are in terms of weight of sediment, air dried.

NT Not tested.

sampling ship account for the lower concentrations found in the field samples than predicted by the elutriate test.

The Duwamish River Site 2 sediment showed slight CHP and moderate PCB contamination compared to other sediments analyzed during this study. The 20 percent elutriate test indicated that this sediment released PCBs and CHPs, although the release was minimal considering the high sediment-to-water ratio used in the test.

The highest PCB concentration (36.0 ng/l) found in the water column during the Elliott Bay disposal operations was observed in the turbid plume from Dump No. 1. The PCB value in the elutriate, 33.4 ng/l, was about the same as that found in the turbid plume. The elutriate value is lower than that expected in the field because it represents the "soluble" (centrifuged) portion of the elutriate. The field water samples were not centrifuged, and the values reported for them include PCBs associated with particulate matter.

Hydraulically dredged sediments - pipeline disposal

Data presented in Table 2 shows that Mississippi River sediment near St. Paul, Minnesota had very low PCB and CHP concentrations (7 μg/kg and 4.7 μg/kg, respectively) compared to the other study sites, yet showed an apparent release during the 20 percent elutriate test. This release could be due to the high sediment to water ratio, the low oil and grease concentration, and/or the high percentage (85 percent) of sand-sized particles in the sediments. Two water samples were collected in the discharge runoff at two different times during sampling at this study site; their "soluble" PCB and CHP concentrations were compared to the total in concentrations of water samples collected one mile upstream from the dredging operation and 2.6 miles downstream from the discharge point, as well as to the "soluble" fractions in the elutriate. The PCB concentrations in the discharge runoff were four to

Table 2

Comparison of CHPs and PCBs in Pipeline Discharge, Dredging and Disposal Areas with 20 Percent Oxic Elutriates - Upper Mississippi River

Compound	Elutriate Test			Field Samples			
	Sediment	Water	Elutriate	1 Mile Up River	Discharge Runoff 10:10	Discharge Runoff 14:17	2.6 miles Down River
	(µg/kg)	(ng/l)	(ng/l)	(ng/l)	(ng/l)	(ng/l)	(ng/l)
Aldrin	0.7	3.4	1.3	<0.6*	4.0	2.9	1.4
op'DDE	1.6	<2.0	3.9	<2.0	3.3	8.8	<2.0
pp'DDE	<1.4*	<2.0	2.1	<2.0	2.8	8.3	<2.0
Lindane	1.0	2.3	3.1	<0.3*	<0.3	0.8	0.8
PCBs	7	<6	11	<6*	40	50	<6*
		(mg/l)	(mg/l)	(mg/l)	(mg/l)	(mg/l)	(mg/l)
Oil & Grease	60 mg/kg	0.5	2.0	0.6	0.9	2.5	<0.5
TOC	0.3%	13.0	15.0	15.0	338.0	768.0	17.0
Soluble TOC	NT	10.0	10.0	8.0	10.0	11.0	9.0

*Indicates compound detected, but not measurable.

**Results in terms of weight of sediment, air dried.

NT = Not tested.

five times greater than those in the elutriates. "Total" water CHP concentrations were also greater in the disposal area and discharge runoff than the "soluble" fraction of the elutriate. Although turbidity measurements were not made on these samples, it is likely the higher concentrations are associated with particulate matter since field water samples were not centrifuged prior to analysis. The elutriate test did, however, indicate that these sediments had a tendency to release PCBs and CHPs. Criteria based on US EPA bulk sediment criteria would have classified this sediment as "clean". Even though the concentrations in the sediments were low, release did occur and disposal of this sediment should demand as much concern and consideration as a highly contaminated sediment.

Results of the 20 percent elutriate test on the James River sediment indicated slight release of kepone and CHPs but no apparent change in PCB concentration. Two water samples collected within the turbid plume below the pipeline discharge point showed elevated kepone concentrations, 572 and 997 ng/l versus the 33 ng/l level predicted by the elutriate test (Table 3). The concentrations of the other pesticides in those water samples were in the same order of magnitude as predicted by the elutriate test. Disposal area water samples also showed elevated PCB levels. These samples had very high turbidity values due to the fine clay-like characteristics of the sediment, and it is plausible to conclude that much of the contaminant which was apparently released was associated with the particulate matter which did not settle during the analysis.

The 20 percent elutriate test run on Galveston Channel sediment indicated that CHPs and PCBs would be released upon open water disposal. The sediment used for the elutriate test may have had somewhat different characteristics from those being dredged at the time the pipeline effluent was

Table 3

Comparison of CHPs and PCBs in Water Samples Downstream from Pipeline Discharge with 20 Percent Oxic Elutriates - James River, Virginia

Compounds	Sediment*	Site Water	Elutriate	Site 7	Site 15
				(In the Turbid Plume)	
	(μg/kg)	(ng/l)	(ng/l)	(ng/l)	(ng/l)
Kepone	450	<20	33	572	997
Lindane	3.6	17.6	20	1.6	10.2
op'DDE	4.5	<2.0	2.9	5.5	8.3
pp'DDE	8.6	<2.0	5.7	4.8	4.1
pp'DDT	6.9	2.0	8.7	6.7	5.0
pp'DDD	<2.0	<3.0	<3.0	6.2	7.8
PCBs	45	<6	<6	9	28
		(mg/l)	(mg/l)	(mg/l)	(mg/l)
Oil & Grease	471 mg/kg	<0.5	0.6	0.9	0.8
TOC	2.7%	4.0	6.0	264	112
Soluble TOC	NT	NT	NT	12	3

*Results are in terms of weight of sediment, air dried.

NT = Not tested.

sampled because the sediment sample was collected some distance from the area being dredged. In a harbor situation such as the Galveston Channel, it is expected that sediment composition would be highly variable within relatively short distances. The CHP concentrations in the solid phase of the sediment sample and pipeline effluent were comparable, but the discharge solid phase had higher PCB and lower oil and grease levels than the sediment used for the elutriate test. The PCB concentration in the elutriate (261 ng/l) was 1.5 times greater than the concentration in the site water (171 ng/l) used for the elutriate test. The "soluble" phase of the pipeline discharge showed a PCB concentration of approximately one-half of the total concentration in the site water. The CHP values predicted by the elutriate test were comparable to the values found in the "soluble" phase of the discharge. The difference between the PCB values in the elutriate and "soluble" phase of the discharge is probably due to the difference in the amount of leachable PCBs in the sediments dredged and sampled.

Water samples were collected at a point about 250 meters downcurrent from the discharge of the Mobile Bay pipeline disposal operation. Total CHPs and PCBs were detected in both samples. The more turbid 3.5 meter sample showed higher concentrations of almost all the measured parameters. Considering the presence of 22.5 percent settleable solids in this sample as compared to <1 percent in the 2 meter sample, it is likely that a large fraction of these compounds was associated with particulate matter. An insufficient amount of sample was collected to run the elutriate test for chlorinated hydrocarbon analyses for this site.

In general, the total PCB and CHP concentrations in water samples collected near dredged sediment discharge areas were lower than the concentrations in the "soluble" fraction of the corresponding sediment elutriates. This is likely due to the dilution which occurred between the disposal site and the sampling location. Some pipeline discharge samples had higher concentrations of CHPs and PCBs than were found in centrifuged elutriates. It is impossible to directly compare the PCB and CHP concentrations in the elutriate test with either pipeline discharge or disposal site water samples since the former was centrifuged prior to analysis and the latter two were not.

These field studies have shown that generally the elutriate test is a viable tool for predicting potential release of this group of chlorinated hydrocarbons during disposal. In each case, the elutriate gave an indication of whether sorption or release may occur and, with the exception of kepone, had concentrations in soluble fractions which were within an order of magnitude of the total concentration of the contaminant concentration in the water column a short distance downcurrent from the disposal site.

Prediction of Dissolved Oxygen Depletion in the Disposal Site Water Column*

One of the major areas of concern associated with open water disposal of dredged sediments is the depletion of dissolved oxygen in the disposal site water column. Dredged sediments generally are anoxic and would typically be expected to have a high oxygen demand. As discussed in other sections of this report, the dissolved oxygen concentration is of importance to aquatic organisms in the disposal site water column directly through respiratory needs and indirectly through influencing the behavior of iron which controls the availability of many contaminants to aquatic organisms. It is therefore

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*This discussion is based in part on draft materials provided by D. Homer.

important to assess the magnitude of dissolved oxygen depletion that is likely to occur in association with various types of dredged material disposal operations. Further, it is important to develop a laboratory procedure that can be used to predict D.O. depletion during open water disposal of dredged sediment. This section os the report presents a discussion of the studies that were conducted in these areas during this study.

Lee et al.[1] developed a laboratory oxygen demand test to predict, prior to dredging and disposal operations, the extent of oxygen depletion expected in the disposal site water column during and shortly after disposal. Their recommended procedure, which was used and evaluated in this study, involved mixing a few cubic centimeters of a wet sediment with about 300 millimeters of water in a closed BOD bottle. The ratio of oxygen utilization was measured for one hour during vigorous mixing, using a polargraphic membrane electrode. This test provides an indication of the relative oxygen demands of the sediments tested. The oxygen demand measured by the test, when coupled with the hydrodynamic characteristics of the disposal site, should provide an estimate of the oxygen depletion in the turbid plume associated with dredged sediment disposal. The results of the studies evaluating the applicability of this test to open water disposal site conditions during dredged sediment disposal are discussed in this section. Also discussed is the relationship between selected bulk characteristics of the sediments and the results of the oxygen demand tests on them.

Summary of oxygen demand test results

There were certain characteristics seen in most of the oxygen demand tests performed in this study. The rates of reaction in each oxygen demand test could be generally approximated by two consecutive first-order reactions. The initial reaction exhibited a faster rate and usually had a duration of five to ten minutes. The rate of the second reaction was approximately five to ten times slower than the initial rate.

Table 4 presents the oxygen demand data collected during this study. Also presented are values for selected bulk chemical parameters which could have an effect on the sediment oxygen demand. The highest demand per gram dry weight after one hour, 1.95 mg O_2, was found in the sample from Apalachicola, Florida Site No. 4, while the sample from Perth Amboy Channel exhibited the highest demand per cubic meter of sediment for the first hour, 9.3×10^2 g O_2. The lowest oxygen demand per gram dry weight for the firt hour was 0.03 mg O_2 in the Duwamish River Site 1 sediment sample. The lowest demand per cubic meter for the first hour was observed in the Duwamish River Site No. 1 sediment, 42 g O_2. Many of the sediments evaluated had oxygen demand values on the order of 500 g O_2/m^3 sediment.

The reproducibility of the oxygen demand test after the one-hour test period was good. The highest standard deviation between the dissolved oxygen readings of triplicate runs of the same sample was seen with the sediment from the James River Site No. 1, 0.6 mg/l. The lowest was reported in the test of the Upper Mississippi River Site No. 2 sediment, 0.0 mg/l. The mean standard deviation was 0.2 with a 95 percent confidence interval of ± 0.2 mg/l.

The reaction rates were determined for all but one of the tests; these results are presented in Table 5. Since varying sediment to water ratios were used, ranging from 2 to 10 cc in 300 mls, the depletion ratios have been normalized to one volume. In normalizing, it was assumed that the relationship between reaction rates and sediment volume used was

Table 4

Oxygen Demand and Bulk Characteristics of Sediments

Sample Site	Eh (mv)	Sulfide* (mg/kg)		Fe * (mg/kg)		Oxygen Demand First Hour		Percent Dry Wt	Dry Wt/cc (g)
		$\bar{X}$	SD	$\bar{X}$	SD	g O_2/m^3**	mg O_2/g*		
Mare Island	-	48		47,900	4,300	$4.8x10^2$	0.84	43.5	0.583
Rodeo Flats	-	258		42,800	20,000	$5.7x10^2$	0.96	43.8	0.589
Oakland	-198	1568	8	37,300	1,000	$6.0x10^2$	0.95	46.8	0.634
Los Angeles A-7	-234	2565	259	17,478	946	$6.7x10^2$	1.15	43.6	0.583
C-2	-117	291	4	-	-	$3.2x10^2$	0.24	74.2	1.309
Newport A	+57	38	8	8,847	293	$6.2x10^2$	0.55	67.4	1.127
B	-92	93	12	-	-	$2.2x10^2$	0.22	64.0	0.979
Stamford	-173	2320	93	9,531	64	$5.6x10^2$	1.33	34.5	0.421
Norwalk 1	-151	2890	281	9,414	178	$5.3x10^2$	0.94	42.6	0.562
2	-160	2810	225	9,387	344	$6.4x10^2$	1.39	36.4	0.462
Apalachicola 2	-106	745	32	37,600	8,600	-	-	28.5	0.328
3	-129	674	79	9,648	25	$5.6x10^2$	1.53	30.8	0.365
4	-90	684	18	9,914	70	$7.5x10^2$	1.95	34.5	0.433
5	-117	883	190	9,990	102	$4.8x10^2$	1.15	33.6	0.416
Duwamish 1	-64	13	8	15,581	44	42	0.03	79.6	1.401
2	-91	254	10	16,079	88	$5.2x10^2$	0.64	56.9	0.812
3	-111	247	13	15,892	220	$5.6x10^2$	0.64	59.7	0.891

(Continued)

Table 4 (Continued)

Sample Site	Eh (mv)	Sulfide* (mg/kg)		Fe* (mg/kg)		Oxygen Demand First Hour		Percent Dry Wt	Dry Wt/cc (g)
		$\overline{X}$	SD	$\overline{X}$	SD	g O_2/m^3**	mg O_2/g*		
Texas City Channel 1	-38	1205	108	23,200	300	$5.0x10^2$	0.66	52.6	0.761
2	-20	1251	95	17,400	4,700	$4.8x10^2$	0.64	51.4	0.745
3	-	-	-	33,300	2,300	-	-	-	-
4	-100	2519	10	26,260	238	$6.0x10^2$	1.21	38.5	0.495
5	-136	237	10	26,000	274	$4.5x10^2$	0.76	44.1	0.592
6	-124	1557	140	25,792	477	$6.1x10^2$	1.17	40.0	0.520
Galveston Bay Entrance Channel									
Buoy 1	+100	446	57	40,000	1,300	$3.6x10^2$	0.67	40.7	0.561
Buoy 9	-	-	-	13,300	2,100	$1.5x10^2$	0.16	41.2	0.552
Buoy 11	-	-	-	94,100	6,900	80	0.14	60.0	0.962
Galveston Channel	-87	1078	35	26,848	262	$5.9x10^2$	1.28	36.3	0.460
Port Lavaca	+40	156	7	21,000	1,100	$2.5x10^2$	0.42	44.6	0.603
Houston Ship Channel 1	-96	2962	78	20,055	0	$5.8x10^2$	1.42	34.3	0.416
2	-137	3483	291	19,778	161	$6.0x10^2$	1.80	28.2	0.333
3	-104	633	35	29,790	488	$5.2x10^2$	0.88	43.6	0.594
Vicksburg WES Lake	-75	278	17	-	-	$8.1x10^2$	1.01	54.8	0.802
Wilmington	-86	72	4	8,145	124	$1.5x10^2$	0.11	73.3	1.320

(Continued)

Table 4 (Continued)

Sample Site	Eh (mv)	Sulfide* (mg/kg)		Fe* (mg/kg)		Oxygen Demand First Hour		Percent Dry Wt	Dry Wt/cc (g)
		$\overline{X}$	SD	$\overline{X}$	SD	g O_2/m^3**	mg O_2/g*		
Mobile 1	-51	327	25	49,400	2,300	5.1×10^2	1.18	34.9	0.434
2	- 9	577	20	64,400	0	6.3×10^2	1.62	31.4	0.388
Upper Mississippi River 1	-172	5	4	11,820	700	1.1×10^2	0.07	83.4	1.488
2	-209	15	5			1.7×10^2	0.11	85.9	1.608
James River 1	-215	25	0	59,750	440	6.2×10^2	0.97	46.6	0.639
2	-244	44	6			7.3×10^2	1.15	46.6	0.637
Bailey Creek	-2	166	5	37,100	4,500	4.0×10^2	0.27	80.0	1.493
Port of New York									
Bay Ridge Channel	-135	1190	27	15,900	10,300	6.6×10^2	0.90	47.6	0.735
Perth Amboy Channel	-109	1502	25	13,900	4,900	9.3×10^2	1.43	34.4	0.650
Perth Amboy Anchorage	-137	2373	121	14,800	7,300	8.4×10^2	1.29	39.7	0.652
Menominee 1	-77	212	80	12,400	100	5.0×10^2	1.25	30.6	0.399
2	-116	1133	26	9,600	100	5.4×10^2	1.14	38.8	0.476
Foundry Cove	-96	56	9	27,890	100	5.2×10^2	0.81	46.7	0.642

(Continued)

Table 4 (Concluded)

ample Site	Eh (mv)	Sulfide* (mg/kg)		Fe* (mg/kg)		Oxygen Demand First Hour		Percent Dry Wt	Dry Wt/cc (g)
		X	SD	$\overline{X}$	SD	g O_2/m^3**	mg O_2/g*		
Ashtabula	-	-	-	30,540	1,840	$3.6x10^2$	0.4	55	-
Corpus Christi									
1	-	-	-	8,300	10	$1.6x10^2$	0.2	70	0.80
2	-	-	-	27,400	8,780	$2.2x10^2$	0.6	33	0.37
3	-	-	-	20,830	5,890	$2.5x10^2$	0.3	57	0.83
4	-	-	-	29,270	860	$2.1x10^2$	0.6	30	0.35

Dash (-) indicates no data collected.

*Based on dry weight of sediment.

**Based on wet sediment.

A and B are replicate sediment samples collected at same location.

Table 5

Reaction Rates for Oxygen Demand Test

Sample Site	Initial (mg/l O_2 min^{-1})	Final (mg/l O_2 min^{-1})	Initial:Final Ratio
Oakland Harbor	-0.023	-0.011	2.09
Los Angeles A-7	-0.029	-0.014	2.07
C-2	-0.026	-0.003	8.67
Newport A*	-0.014		-
B	-0.006	-0.002	3.00
Stamford	-0.009		-
Norwalk 1	-0.009		-
2	-0.017	-0.009	1.89
Apalachicola 3	-0.026	-0.006	4.33
4	-0.034	-0.009	3.78
5	-0.016	-0.004	4.00
Duwamish 1	-0.001		-
2	-0.030	-0.006	5.00
3	-0.033	-0.007	4.71
Texas City Channel			
1	-0.020		-
2	-0.018		-
4	-0.023	-0.011	2.09
5	-0.024	-0.004	6.00
6	-0.064	-0.010	6.40
Galveston Entrance Channel Buoy 1	-0.014	-0.004	3.50
Galveston Bay Channel	-0.025	-0.009	2.78
Port Lavaca	-0.016	-0.002	8.00
Houston Ship Channel 1	-0.018	-0.008	2.25
2	-0.030	-0.008	3.75
3	-0.031	-0.007	4.43

(Continued)

Table 5 (Concluded)

Sample Site	Initial (mg/l O_2 min^{-1})	Final (mg/l O_2 min^{-1})	Initial:Final Ratio
Vicksburg, WES Lake	-0.064	-0.003	21.33
Wilmington	-0.005	-0.001	5.00
Mobile Bay 1	-0.031	-0.006	5.17
2	-0.031	-0.011	2.82
Upper Mississippi River 1	-0.001		-
2	-0.002		-
James River 1	-0.036	-0.009	4.00
2	-0.037	-0.010	3.70
Bailey Creek	-0.010	-0.004	2.50
Port of New York			
Bay Ridge Channel	-0.022	-0.009	2.44
Perth Amboy Channel	-0.024	-0.013	1.85
Perth Amboy Anchorage	-0.012		-
Menominee 1	-0.025	-0.003	8.33
2	-0.019	-0.005	3.80
Foundry Cove	-0.027	-0.003	9.00
Ashtabula Harbor	-0.012	-0.004	2.86
Corpus Christi 1	-0.006	-0.002	3.00
2	-0.012	-0.001	12.00
3	-0.006	-0.002	3.00
4	-0.006	-0.002	3.00

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*A and B are replicate sediment samples collected at same location.

Dash (-) indicates not applicable.

linear, i.e., a 2 cc volume of sediment in 300 ml would react at rates two-thirds that of a 3 cc sediment sample in 300 ml. The rates of oxygen depletion fit the following equation:

$$\text{Log } C_t = \log C_o + kt$$

where C_t is the dissolved oxygen concentration at time, t; Co is the initial concentration, k is the oxygen depletion rate constant; and t is time.

A wide range of initial rates was noted from a low of 0.005 mg/l O_2 min^{-1} in the Wilmington, North Carolina sample, to a high of 0.064 mg/l O_2 min^{-1} in the Texas City Channel Site No. 6, and WES Lake, Vicksburg, Mississippi samples. The final rate also had a large range of values, from a low of 0.001 mg/l O_2 min^{-1} for the Wilmington, North Carolina and Corpus Christi Site No. 2 samples, to a high of 0.014 mg/l O_2 min^{-1} seen in the sample from Los Angeles Harbor, Buoy A-7. The rates of reactions were controlled by various chemical species. Each of these species exerted a different amount of influence in each sediment, making for an extremely complex system in which the overall reaction is taking place.

The ratios of the initial and final first-hour rates of oxygen depletion for the sediments investigated were compared and were found to vary considerably among the sites. The ratios of initial to final rates are found in Table 5. They ranged from 1.85 for the sediment from Perth Amboy Channel to 21.33 for the WES Lake, Vicksburg, Mississippi sample. These ratios are important in determining the initial oxygen demand of the dredged sediment at the disposal site during the first few minutes after disposal. Those sediments with the highest ratios would be expected to show the highest depletion near the point of disposal. It should be noted, however, as discussed in a subsequent section, that the actual amount of depletion is determined to a major extent by the hydrodynamics of the dumping operation and receiving waters rather than the oxygen demand of the sediments.

The two consecutive first order reactions may be controlled by two different systems. The initial stage may be primarily the oxidation of reduced material such as Fe II, $S^=$ and polysulfide in the interstitial water. These constituents would be expected to be oxidized rapidly, as seen in the faster initial rates. The slower first-hour rate may be primarily the result of the oxidation of reduced particulate material. These surface-controlled reactions take place at a slower rate. The oxidation of pyrite, FeS, or similar reduced materials in particular form, would be an example of such a reaction.

For those reactions that fit a single slope on a semi-log plot, the rates also varied greatly. The slowest rate was seen with the sediment sample from Duwamish River Site No. 1 and the Upper Mississippi River Site No. 1, 0.001 mg/l O_2 min^{-1}. The fastest rate was found in the test with the sediment from Texas City Channel Site 1, 0.02 mg/l O_2 min^{-1}. The single stage reaction may primarily result from a surface reaction, or from reactions with constituents in the interstitial water. However, the reaction most likely involves both processes with approximately equal influence.

Relationship between bulk characteristics and the oxygen demand test

The following bulk characteristics of the sediment samples were determined: percent dry weight, oxidation-reduction potential, total sulfide and total iron content. The results of these analyses are presented in Table 4.

The percent dry weight values showed a large range, from 28.2 to 85.9 percent. The mean vlaue for all the sediments was 47.8 percent with a standard deviation of 15.6. It was noted by Lee et al.[1] that there was no correlation between the percent dry weight of the sediment and its oxygen demand after one hour. This observation was substantiated in this current study. The relationship between the oxygen demand and percent dry weight of the sediment was found to have an r^2 value of 0.37. This indicated that only 37 percent of the variation in the oxygen demand could be correlated to the dry weight.

The oxidation-reduction potential was determined using a platinum electrode referenced versus the H_2 electrode, but the readings were not corrected for pH. The range was from +100 millivolts (mv) to -244 mv. The mean for all sediments tested was determined to be -103 mv. There were only three sediments found to be in an oxidizing or positive Eh state: Newport, RI; GBEC Buoy 1; and Port Lavaca. It can be concluded that most sediments are in a reduced oxidation-reduction state and, therefore, would likely contain significant concentrations of chemicals which would exert an oxygen demand.

The reducing conditions of the sediments was also seen by the presence of the reduced sulfur species, sulfide. The range in the total sulfide concentrations was extremely large, from 5 mg/kg to 3483 mg/kg dry weight; these samples were from the Upper Mississippi River Site No. 1 and Houston Ship Channel Site No. 2, respectively. The mean sulfide concentration was 928 mg/kg dry weight, with a standard deviation of 999 mg/kg,indicating the wide variation.

It has been reported that the highest oxidation-reduction potential where sulfide species will be formed is +110 mv and the lowest is -500 mv.* The Eh values for all sediments investigated were within this range, and sulfide was found in each sample analyzed.

The relationship between Eh and total sulfide content was investigated. When compared statistically, there

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*Baas-Becking, L.G.M., Kaplan, I.R. and Moore, D., "Limits of the Natural Environment in Terms of pH and Oxidation-Reduction Potentials," J. Geol. 68:243-284. (1960).

was no significant relationship found between them. The evidence for this was found in a 0.08 r^2 value. The Eh is an indication of whether or not the sulfide can form, not a measure of the extent of formation. Formation is more related to the suitability for the microorganisms' response to sulfate reduction.

The other major reduced species in the sediment which would have a major impact on the immediate oxygen demand was iron, in the form of Fe II. The iron reported in Table 4 was total iron in the sediments as dry weight; no attempt was made to differentiate between Fe II and Fe III. Iron ranged from 8145 mg/kg in sediments from Wilmington, North Carolina to 94,100 mg/kg in the sediment sample from GBEC Buoy 11. The mean was found to be 25,348 mg/kg for all sediments tested. The standard deviation, 18,143 mg/kg, indicates the wide variation in their content.

Calculations were made to determine the theoretical oxygen demand of each sediment based on the sulfide and iron content. It was assumed that all of the iron in the sediment was in a reduced state. Also, it was assumed that the oxidation of the two reduced species followed these reactions:

$$HS^- + 2O_2 \longrightarrow SO_4^{-2} + H^+$$

$$Fe^{2+} + 1/4\ O_2 + H^+ \longrightarrow Fe^{3+} + 1/2\ H_2O$$

Presented in Table 6 are the theoretical oxygen demands based on the above equations, the demand realized after one hour, and the realized percentage of the theoretical demand. The percentage of the demand realized ranged with the sites from a low of 1.33 in the Duwamish River Site No. 1 sediment to

Table 6

Theoretical Oxygen Demand per Gram Dry Weight

Sample Site	Oxygen Demand (mg O_2)		
	Theoretical	Realized	Percent
Mare Island	6.96	0.84	12.07
Rodeo Flats	6.65	0.96	14.44
Oakland	8.48	0.95	11.20
Los Angeles A-7	7.63	1.15	15.06
Newport A *	1.34	0.55	40.95
B	1.45	0.22	15.14
Stamford	6.01	1.33	22.15
Norwalk 1	7.13	0.94	13.19
2	6.96	1.39	19.96
Apalachicola 3	2.73	1.53	56.04
4	2.79	1.95	69.94
5	3.20	1.15	35.97
Duwamish 1	2.26	0.03	1.33
2	2.81	0.64	22.77
3	2.77	0.64	23.10
Texas City Channel 1	5.73	0.66	11.51
2	4.99	0.64	12.82
4	8.80	1.21	13.75
5	4.20	0.76	18.10
6	6.81	1.17	17.18
Galveston Bay Entrance Channel Buoy 1	6.62	0.67	10.12
Galveston Channel	6.00	1.28	21.33
Port Lavaca	3.34	0.42	12.57
Houston Ship Channel 1	8.80	1.42	16.14
2	9.80	1.80	18.37
3	5.53	0.88	15.90

(Continued)

Table 6 (Concluded)

Sample Site	Oxygen Demand (mg O_2)		
	Theoretical	Realized	Percent
Wilmington	1.31	0.11	8.39
Mobile Bay 1	7.73	1.18	15.26
2	10.37	1.62	15.62
Upper Mississippi River 1	1.70	0.07	4.11
2	1.72	0.11	6.34
James River 1	8.61	0.97	11.27
2	8.65	1.15	13.30
Bailey Creek	5.65	0.27	4.78
Port of New York			
Bay Ridge Channel	4.66	0.99	21.25
Perth Amboy Channel	4.99	1.43	28.63
Perth Amboy Anchorage	6.87	1.29	18.79
Menominee 1	2.20	1.25	56.84
2	3.64	1.14	31.22
Foundry Cove	4.11	0.81	19.72

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*A and B are replicate sediment samples collected at same location.

69.94 in Apalachicola Site No. 4 sediment. The mean percent of the demand exerted by the sediments tested was 19.92 percent, with a standard deviation of 14.34 percent. The standard deviation indicates the large variation among the different sediments. From these data one can conclude that prediction of the oxygen demand of the sediments cannot be based on the total iron and total sulfide content alone.

Tests were also performed on the data to determine if a correlation exists between the bulk properties of total sulfide or total iron and oxygen uptake. The data for total sulfide content and oxygen demand of the sediments were compared. The r^2 value was found to be 0.32, indicating that only 32 percent of the oxygen demand can be correlated to the total sulfide concentration. When the total iron content was compared to the oxygen demand, an r^2 value of 0.01 was found. The total iron content could account for one percent of the variability in the oxygen demand. These r^2 values, total sulfide and total iron, indicated that as expected neither one alone can be used for estimating the oxygen demand of a sediment.

From these findings one can conclude that bulk characteristics such as total sulfide, total iron, oxidation-reduction potential, and dry weight cannot be used individually or in combination to estimate the one-hour oxygen demand of a sediment. Problems arise in trying to determine the actual concentration of reduced iron and iron sulfide species in the sediments since no reliable method exists. The system is also extremely complex in terms of the oxidation of iron sulfide. Oxygen demand can only be determined by empirical methods.

Relationship between elutriate test and oxygen demand test results

The elutriate test was intended to estimate the extent of the release of contaminants such as PCBs, chlorinated hydrocarbon pesticides, heavy metals and nutrients, and was not developed for estimation of the sediment oxygen demand. This section is devoted to a comparison between the oxygen demand seen during the elutriate test and that found in the one-hour oxygen demand test.

The data in Table 7 compare the oxygen demand test results with the demand measured in the elutriate test from the end of the mixing period to the end of the settling period. These data sets were tested to determine if any correlation existed between them. The r^2 value was calculated to be 0.05, showing that no statistically significant correlation existed between them.

The uptake during the elutriate test is based on the D.O. levels after the mixing portion of the test. The sediment may have a sufficiently high demand to bring about a low D.O. ($\leq$ 2.5 mg/l) in the flask after settling. At these low D.O. levels, the rate of oxygen uptake would be expected to be lower than at levels near saturation. If the entire oxygen supply were depleted, the uptake of 2.5 mg/l would be measured even though an unsatisfied demand remained. It would be extremely difficult to measure the total oxygen demand of sediments using the established elutriate test procedures.

Relationship between the oxygen demand tests and observed oxygen depletion during open water dredged sediment disposal

Dredged sediment dumping operations. The hopper-dredged sediment disposal operations monitored in the Gulf of Mexico near Galveston, Texas showed changes in water column D.O. ranging from a slight increase to a 1 to 2 mg/l decrease during the passage of the turbid plume. Where depletion occurred, the duration of reduced D.O. concentration ranged from 10 to 20 minutes before the level increased to ambient (5 to 7 mg/l).

Table 7

Comparison of Results from Oxygen Demand Test and Oxygen Uptake During the Elutriate Test

Sample Site		O_2 Demand mg O_2/g*	O_2 Uptake - 20% Oxic Elutriate Test (mg/l)
Mare Island		0.84	1.5
Rodeo Flats		0.96	1.3
Oakland		0.95	2.4
Los Angeles	A-7	1.15	1.5
	C-2	0.24	-
Newport	A **	0.55	0.8
	B	0.22	0.8
Stamford		1.33	2.4
Norwalk	1	0.94	2.6
	2	1.39	2.0
Apalachicola	3	1.53	2.7
	4	1.95	1.3
	5	1.15	2.2
Duwamish	1	0.03	2.1
	2	0.64	3.2
	3	0.64	1.8
Texas City Channel	1	0.66	2.0
	2	0.64	3.2
	4	1.21	1.5
	5	0.76	0.9
	6	1.17	3.3
Galveston Bay Entrance Channel			
Buoy 1		0.67	2.1
Buoy 9		0.16	1.7
Buoy 11		0.14	1.3
Galveston Channel		1.28	3.6

(Continued)

Table 7 (Concluded)

Sample Site	O_2 Demand mg O_2/g*	O_2 Uptake - 20% Oxic Elutriate Test (mg/l)
Houston Ship Channel 1	1.42	1.6
2	1.80	1.4
3	0.88	4.0
Port Lavaca	0.42	0
Vicksburg WES Lake	1.01	6.2
Wilmington	0.11	0.6
Mobile 1	1.18	2.5
2	1.62	3.1
Upper Mississippi River 1	0.07	1.8
2	0.11	
James River 1	0.97	2.2
2	1.15	
Bailey Creek	0.27	3.8
Port of New York		
Bay Ridge Channel	0.90	2.9
Perth Amboy Channel	1.43	2.6
Perth Amboy Anchorage	1.29	1.5
Menominee 1	1.25	3.0
2	1.14	4.3
Foundry Cove	0.81	1.4
Ashtabula	0.4	1.9
Corpus Christi 1	0.2	3.2
3	0.3	3.9

*Based on dry weight of sediment.

**A and B are replicate sediment samples collected at the same location.

Unlike the conditions found at the Galveston Bay Entrance Channel Disposal Site in the Gulf of Mexico, a thermocline-chemocline was found at the time of sampling at the New York Bight mud dump site. The thermocline-chemocline provided a barrier to the transfer of oxygenated surface waters to the bottom; the waters below this barrier had an ambient concentration of 1 to 2.5 mg/l. During passage of the turbid plume associated with the New York Bight hopper disposal operations, the D.O. concentrations in the waters below the thermocline-chemocline increased. This increase was caused by the entrainment of oxygen-rich surface waters during the descent of the dumped sediment. It is evident that the amount of entrained oxygen exceeded the short-term oxygen demand of the dumped sediments and water that became part of the turbid plume. A similar pattern of D.O. behavior was found in association with the barge dumping of mechanically dredged sediments at the mud dump site. At the Elliott Bay-Puget Sound dump site, the dissolved oxygen content in the near-bottom waters decreased by less than 1 mg/l during the passage of the turbid plume. It returned to ambient-predisposal levels shortly after passage of the turbid plume.

It is evident that the open water dumping of dredged sediment derived from either hopper or mechanical dredging operations would not be expected to cause a significant dissolved oxygen depletion at or near the dredged material disposal site. Further, because of the entrainment phenomenon, a dumping operation may actually increase the short term D.O. concentration in the hypolimnetic waters. Further, it is evident that because of entrainment and other hydrodynamic factors, there is no simple relationship between the results of the oxygen demand test developed by Lee _et_ _al_.[1] and the oxygen depletion that will occur in the water column at or near an open water dredged material dump site.

Pipeline disposal operations. At three of the four pipeline disposal operations monitored (James River, Apalachicola Bay, and Mobile Bay), a consistent pattern of D.O. behavior was found. There was generally an inverse relationship between the turbidity and the D.O. concentrations. At each of these sites, a strong density current was found as a result of the discharge of the dredged sediments; the dredged sediment slurry moved in a relatively thin layer (0.5 m or so thick) along the bottom in the direction of the prevailing current. The discharge from the pipeline was anoxic and had a high oxygen demand. The density currents were also anoxic near the point of discharge, but within 100 to several hundred meters from this point, the waters were oxic.

At the fourth pipeline disposal operation monitored (Upper Mississippi River), the discharge from the pipeline was oxic, containing several mg/l D.O. While the D.O. was depressed near the point of discharge within a few tens of meters, levels within the density current were ambient. While an insufficient number of disposal sites were monitored to establish a well-defined relationship between the results of the oxygen demand test and the area and amount of depletion of dissolved oxygen in the density current associated with the disposal operation, it is reasonable to suspect that a relationship of this type could be formulated if a much better understanding were available of the hydrodynamics of the density current associated with the pipeline disposal operations.

It has become clear from these studies that in order to predict dissolved oxygen depletion associated with open water dredged material disposal operations, a much better understanding of the hydrodynamics of density current and the disposal area must be available. It is also evident that while it would likely be rare that significant oxygen depletion problems would occur in association with open water dumping of dredged sediment, water quality problems related to D.O. depletion could develop near the point of discharge for a pipeline disposal operation. Of particular concern would be the effects of depleted dissolved oxygen in the density current on benthic and epibenthic organisms.

PART IV: FACTORS INFLUENCING THE RESULTS OF THE ELUTRIATE TEST*

Review of Elutriate Test Development

With the general environmental awakening of the late 1960's came the realization that the sediments of many US waterways contained elevated concentrations of various chemical contaminants which had, in general, been associated with environmental degradation. Of particular concern were the chlorinated hydrocarbon pesticides, heavy metals, and aquatic plant nutrients such as nitrogen and phosphorus compounds. This situation led to concern by water pollution regulatory agencies about the potential environmental impact of dredging of US waterways and in particular, the open water disposal of dredged sediments.

Since, in general, higher concentrations of contaminants lead to greater environmental degradation, bulk chemical criteria were developed by water pollution regulatory agencies and, in particular, the Federal Water Pollution Control Administration (FWPCA), as a means of minimizing the adverse environmental impact of chemical contaminants associated with dredged sediments. This led to what are commonly referred to as the Jensen criteria.[20] With the release of these criteria by the FWPCA, considerable controversy developed within some segments of the water pollution control field on the appropriateness of the use of bulk criteria as a tool to minimize the adverse environmental impact of chemical contaminants associated with dredged sediment disposal. Within a very short period of time after release of the Jensen criteria in the early 1970's, various EPA regions adopted these criteria or modifications thereof as a basis for classifying their

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*This section of the report was developed based on materials provided by a number of individuals: heavy metals, Lopez[21]; nitrogen draft materials submitted by P. Bandyopadhyay; chlorinated hydrocarbon pesticides and PCBs draft materials submitted by J. Butler.

waterway sediments as "polluted" or "non-polluted". In some parts of the US, the use of the Jensen criteria by the US EPA regions resulted in a significant increase in the cost of maintenance of navigable depths in the waterways of the region because alternate methods of disposal had to be adopted. An example of this type of situation occurred in the US-Canadian Great Lakes region where the US EPA Region V's adoption of bulk chemical criteria as a basis for classifying the degree of pollution of waterway sediments, resulted in the expenditure of several hundred million dollars for alternate methods of dredged material disposal. The cost of dredging in some parts of the US portion of the Great Lakes increased from about 50¢/cu. yd. to over $10/cu. yd. In a number of US Great Lakes locations, maintenance dredging of waterways was halted because of the high cost. The socio-economic impact of this situation was temporarily minimized due to the high water levels in the Great Lakes during that time.

In an effort to try to develop a more technically valid approach for assessing the environmental significance of chemical contaminants in dredged sediment upon open water disposal, the US EPA and CE developed the elutriate test. This test was designed to simulate the conditions that would prevail during hydraulic dredging and subsequent open water disposal of the dredged sediments. Lee and Plumb[9] and Lee *et al.*[22] have reviewed the development of the elutriate test and discussed its application and use. Further, Lee *et al.*[1] Piwoni[4], and Lopez[21], conducted a comprehensive study of the factors influencing the results of the elutriate test.

While the primary purpose of the study presented in this report was to evaluate the use of the elutriate test as a tool for predicting contaminant release to the disposal site water column during open water dredged material disposal, some data were obtained on the factors influencing the results of the elutriate test. This part of the report reviews the conclusions of Lee et al.[1] and includes an updated discussion of the factors influencing elutriate test results based on the additional work done during this investigation. One of the major factors found by Lee et al.[1] to affect the results of the elutriate test described by Keeley and Engler[46], was the amount of D.O. present in the slurry during the test. Because this parameter was uncontrollable in the original test procedure, elutriate test results were erratic for some sediments and were highly dependent on the technique used in the mixing step. Lee et al.[1] controlled this condition by using compressed air mixing, maintaining oxic conditions during the mixing step. Using the controlled redox tests, they evaluated several other parameters that might affect elutriate test results (percent sediment, type of water, agitation time, redox conditions, and settling time.

Percent sediment in elutriate mixture

The use of 1:4 sediment-to-water ratio was chosen for the elutriate test because it is generally believed that this is near the optimum pumping ratio for hydraulic dredging. Since it was found by Lee et al.[1] that this pumping ratio was actually highly variable, they investigated the effect of percent sediment in the dredged material slurry on elutriate test results. This parameter was investigated further in this study; a discussion of the general trends found is presented below.

Phosphorus. Piwoni[4] reported that for soluble orthophosphate, there appeared to be a general pattern of decreasing elutriate soluble ortho P concentration with increasing percent sediment. He reported that using five percent sediment, there was release and using 10, 15, and 20 percent sediment, there was increasing uptake. He suggested that this response may have been due to an increasing amount of sediment iron to scavenge phosphate with increasing percent sediment. Careful examination of his data, however, does not support this conclusion. With most sediments evaluated in this manner, release was about the same independent of percent sediment used (see Appendix Table A1).

Approximately 20 additional pairs of 5 percent and 20 percent oxic elutriate tests were performed during this study. It appeared from these additional tests that there was no consistent relationship between the percent sediment in the test and the soluble ortho P concentration in the elutriate. In general, concentrations in both 5 and 20 percent elutriates were about the same or the 20 percent elutriate had a greater soluble ortho P concentration than the 5 percent.

Heavy metals. From the data presented in Appendix Table A2, it is found that for many of the sediments studied, the amount of Mn released during the elutriate test increased with increased percent sediment used. In many cases where Zn removal occurred, greater removal was observed with increased sediment content in the test. Heavy metal release was not, in general, proportional to the amount of sediment used, and it is not possible to predict without study of a particular sediment, the change in magnitude of release or uptake that would result from a change in sediment-water ratio. Lopez[21] found an average 20 percent sediment:5 percent sediment Mn release ratio of two in the oxic elutriate test. Using the much larger data base available in the current study, the average ratio is slightly over three, while the range of ratio values was from 0.05 to 57. It is apparent that only rarely is there a consistent relationship between the amount of sediment used in the test and the amount of Mn released during the test. (See also Figures 7 and 8)

The response of sediment-associated Cu, Cd, Ni, Cr, Pb, Hg, and As to oxic elutriation was not, in general, affected by the sediment-water ratio used in the test. The elutriate concentrations of these metals were usually not significantly different from those found in the test water.

Nitrogen. Lee *et al.*[1] reported that the release of ammonium is nearly directly proportional to the percent sediment of the total elutriate volume. They indicated that the release of ammonium in the 20 percent sediment mixture can be estimated from the release observed with the 5 percent sediment. Throughout this study (see Appendix Table A3), release found using 20 percent sediment was 1.4 to 4 times higher than that found using 5 percent sediment in the total elutriate volume. This verifies that percent sediment is a major factor affecting ammonium release. However, it does not appear that there is a direct proportionality between releases obtained in 5 and 20 percent sediment elutriate tests. Increasing the amount of sediment used in the elutriate test generally increased organic nitrogen release. (Also see Figure 6)

Chlorinated hydrocarbon pesticides and PCBs. No direct comparison was made during this investigation between the PCB and CHP release from 5 percent and 20 percent elutriation of the same sediment sample. However, some inference about the possible effect of sediment-to-water ratio on CHP and PCB behavior during elutriation can be made from studies conducted on Texas City Channel sediments. Sediments from adjacent sampling locations in the Texas City Channel were used in 5 and 20 percent elutriate tests. A 5 percent elutriate test was run on the TCC-1 sediment which contained 304 mg/kg oil and grease, 7426 µg/kg PCBs, and 15.5 µg/kg CHPs. The TCC-4 sediment which contained comparable oil and grease (533 mg/kg) and CHPs (12.5 µg/kg) concentrations, and less than half the PCB concentrations (3010 µg/kg) than that of the TCC-1 sediment, was used in a 20 percent elutriate test. A comparison of the results of the two tests (Appendix Table A-4) shows that the release was much higher in the 20 percent test, even though the PCB concentration was less. Since the oil and grease content was essentially the same for these two sediments, the degree of release could possibly have been dependent on the sediment-to-water ratio. However, it could also have been dependent on the composition of the PCBs and the different modes of binding of PCB in the two sediments.

Bulk chemical content

A number of investigators have recently reported that there is no relationship, or reason to believe there may be a relationship, between the bulk chemical content of a sediment and the release and/or resulting concentrations of these chemicals during the elutriate test or immediately following open water disposal of the sediments. As will be discussed in a subsequent section, however, over half of the US EPA regions are currently using the bulk sediment content as a basis for judging the acceptability of dredged sediments for open water disposal. The relationships between bulk sediment content and the release of selected contaminants during elutriate tests was therefore investigated during this study.

Phosphorus. Figure 2 presents the relationship between sediment total P content and the degree of uptake or release relative to site water during elutriation. This relationship was based on approximately 80 20 percent oxic elutriate tests run on 46 waterway sediments. It can be seen from the examination of this plot that whether a sediment releases or takes up soluble orthophosphate is not dependent on its total P content.

Heavy metals. Since there was little or no release of As, Cd, Cr, Cu, Hg, Ni, Pb or Zn during elutriate tests, there is obviously no relationship between the bulk heavy metal content for any of these metals in the sediments and their release during open water disposal of the sediments. An

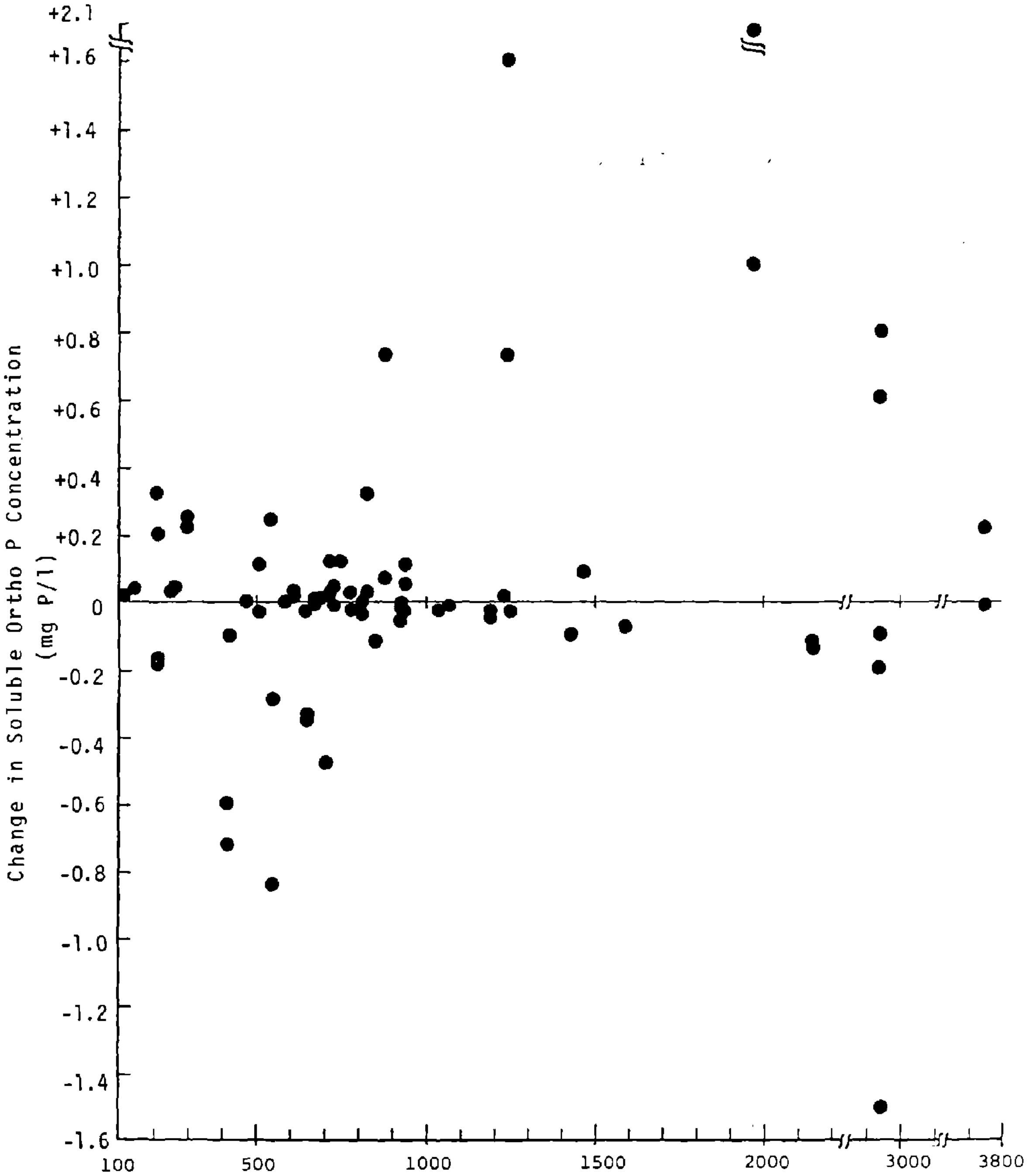

Figure 2

Change in Soluble Ortho P Concentration Resulting from 20% Oxic Elutriation in Relation to Total P Content in Sediment examination of the data for Mn, the only heavy metal that was frequently released during oxic elutriate tests, shows that there is no relationship between the total Mn content of the sediments and the tendency for release during open water disposal operations.

Nitrogen. Figure 3 shows the relationship between ammonium release in each 20 percent oxic elutriate test and total kjeldahl nitrogen (TKN) content of the sediment used. Examination of this figure and Table A3 shows a general trend of increasing release with increasing sediment TKN, but with high scatter in the data (correlation coefficient, r^2=0.20). This indicates that the magnitude of ammonium release cannot be reliably predicted based on the bulk sediment content of total kjeldahl nitrogen. A correlation, however, was not expected between ammonium release and total kjeldahl nitrogen, which includes both ammonium and organic N, in the sediments. The conversion of organic N to ammonium is a biochemical process that takes place at a relatively slow rate; the ratio between ammonium and total kjeldahl nitrogen would be variable and specific for each sediment. Lee et al.[1] suggested that there might be a relationship between the extractable ammonium content of a sediment and ammonium release in the elutriate test. In Figures 4 and 5, the ammonium concentrations in the elutriates for 20 percent oxic and anoxic tests are plotted against the sediment ammonium content. Figures 4 and 5 and Table A3 show a general trend of increasing elutriate concentration with increasing ammonium content in the sediment. There is considerable scatter, however, indicating that one cannot reliably predict ammonium release based on bulk sediment content of ammonium. However, for a specific site, such as one of the channels in New York Harbor, there appears to be a reasonable correlation between the ammonium content of the sediments and ammonium release.

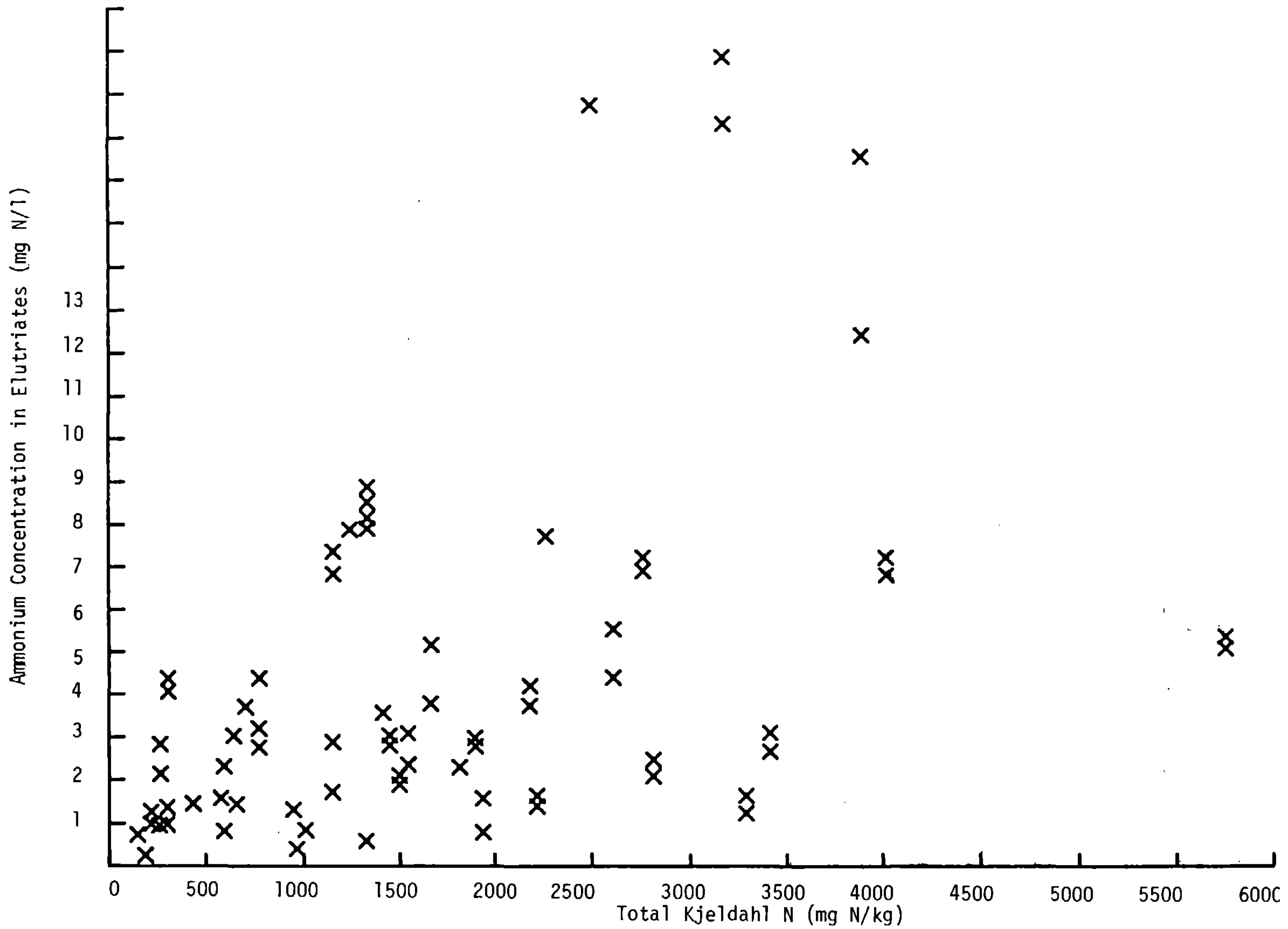

Figure 3

Relationship Between Ammonium Concentration in 20 Percent Oxic Elutriate Tests (mg N/1) and Total Kjeldahl N in Sediment (mg N/kg)

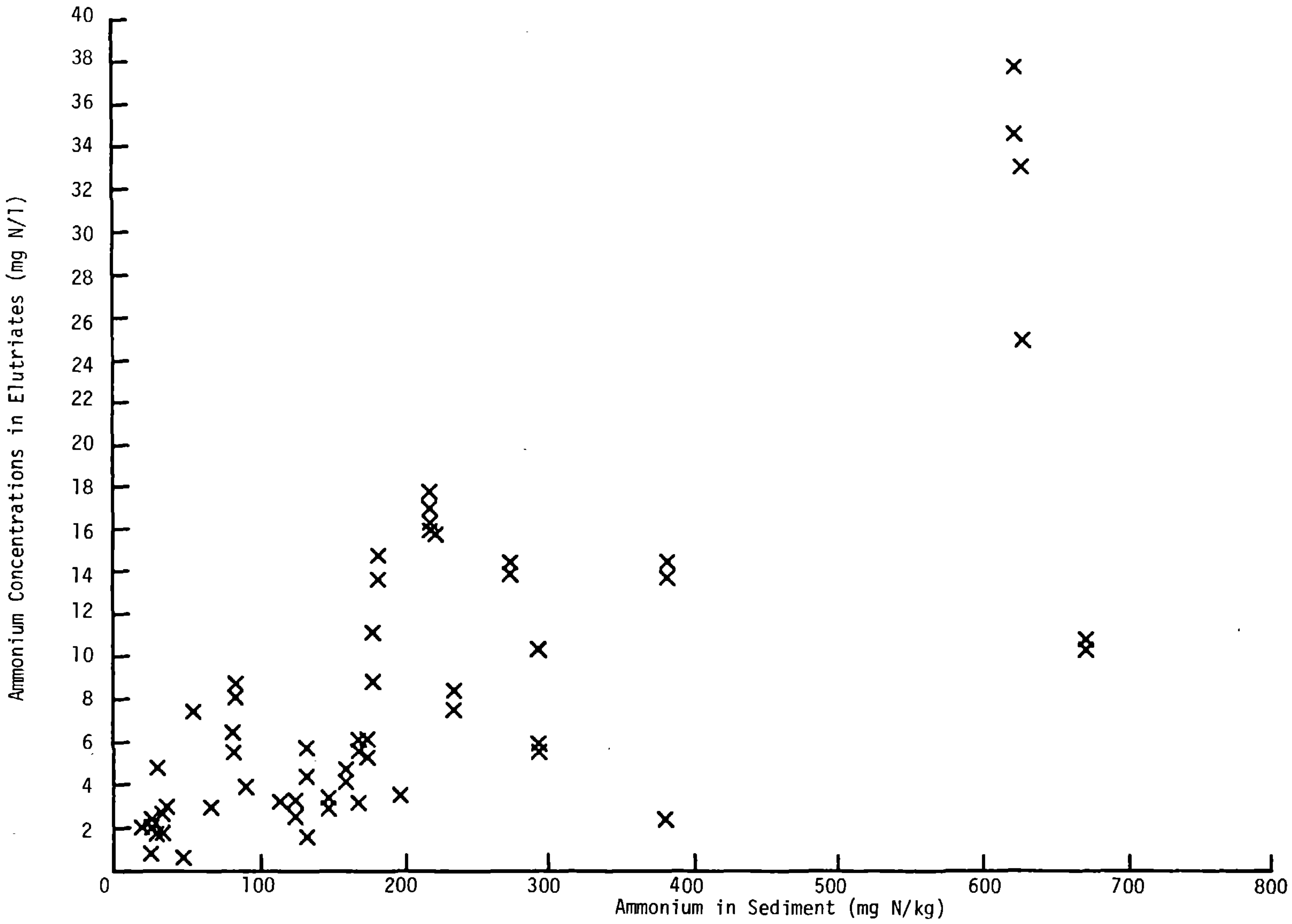

Figure 4
Ammonium Concentration in 20 Percent Oxic Tests (mg N/l) Compared to Ammonium in Sediment (mg N/kg)

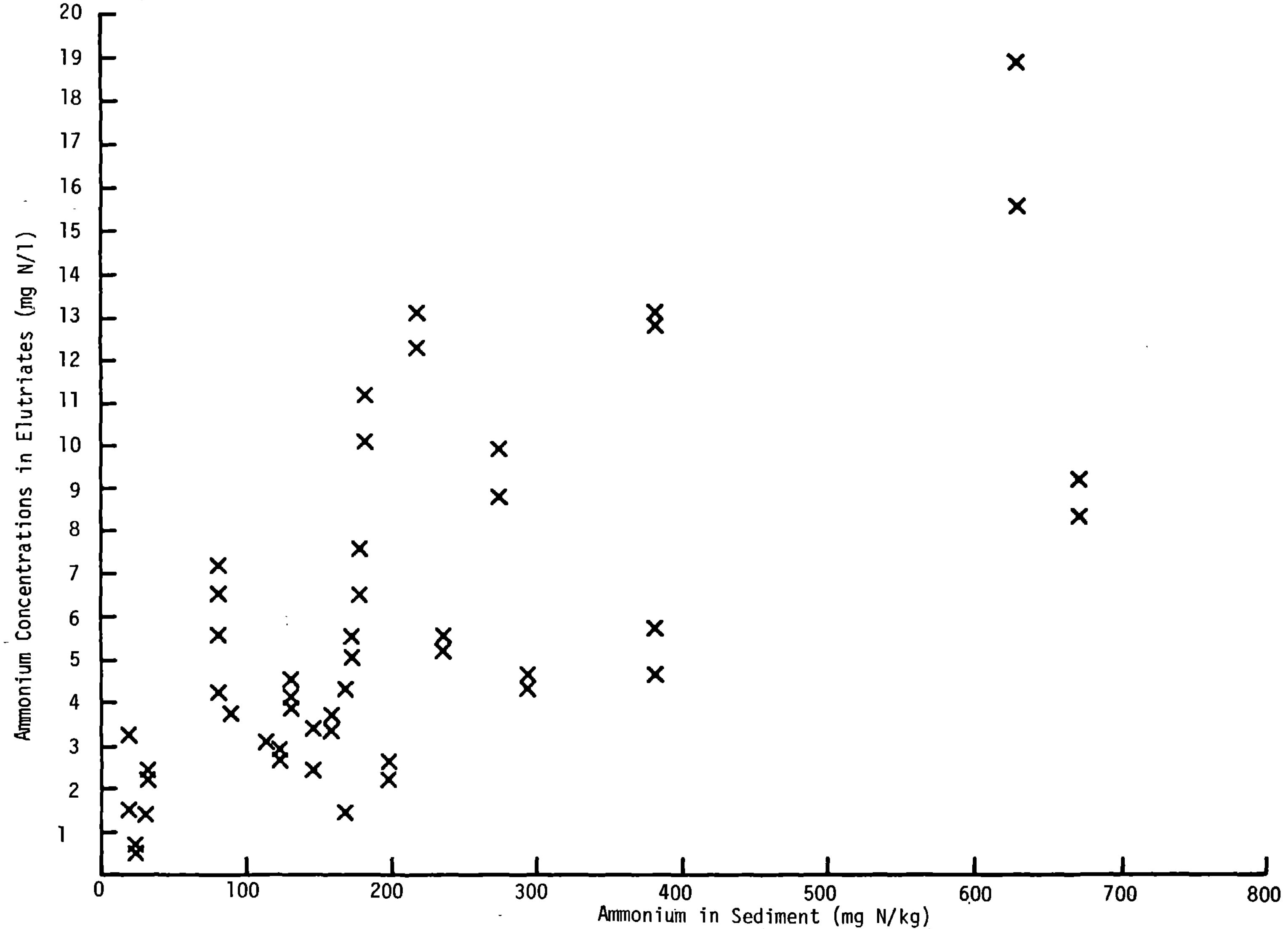

Figure 5
Ammonium Concentration in 20 Percent Anoxic Tests (mg N/l) Compared to Ammonium in Sediment (mg N/kg)

From an overall point of view, it is evident from this study that the bulk sediment content of total kjeldahl nitrogen, organic N, or ammonium is not a reliable predictor of the release of nitrogen compounds in the elutriate test or during open water disposal of dredged sediments. It is concluded that the bulk nitrogen compound content of sediments should not be used as a criterion upon which to judge the potential environmental impact of the nitrogen compounds upon open water disposal of dredged sediments.

Chlorinated hydrocarbon pesticides and PCBs. Appendix Table A4 and Table 8 show that for a variety of sediments there is no relationship between bulk sediment content of PCB, total CHP, or selected CHP compounds and release of these compounds in elutriate tests. Further, there did not appear to have been a direct relationship between the bulk PCBs concentration in a particular sediment and the release during elutriation. As discussed previously, for the Texas City Channel system, there appeared in fact to have been an inverse relationship between the two parameters for sediments that were collected in adjacent areas of the channel.

Table 9 compares the release and sorption of PCBs and CHPs during elutriation. In general, the behavior of PCBs and CHPs was about the same for individual sediments tested. That is, when there was PCB release from a sediment, there was also CHP release, and when there was PCB sorption, the CHPs were sorbed as well. Aldrin-dieldrin and heptachlor generally, and the DDT group always, behaved as the PCBs. Lindane behavior did not appear to be related to that of the CHPs or PCBs.

It can be seen from Table 8 that for the 5 percent tests there is an inverse relationship between PCB release and the amount of oil and grease contained in the sediment. For the 20 percent tests, however, the same correlation does

Table 8

Release and Sorption of PCBs during the Elutriate Test

Sampling Site	Oil and Grease in Sediment (mg/kg)	Total PCB Concentration Sediment (μg/kg)	Total PCB Concentration Water (ng/l)	Total PCB Concentration Elutriate (ng/l)	Percent Sorption/ Release
PCB Sorption in 5 Percent Elutriate Test					
GBEC Buoy 1	24	192	210	170	19
Bridgeport-Eaton's Neck	1450	1500	48	17	65
Rodeo Flats	2227	90	85	25	71
HSC Turning Basin	1580	3280	3368	400	88
Mare Island	3120	95	225	21	91
PCB Release in 5 Percent Elutriate Test					
Texas City Channel 1	304	7426	130	150	15
Corpus Christi 3-4	166	45	73	165	126
Corpus Christi 3-3	166	45	94	259	176
Ashtabula	81	1660	16	50	218
Port Aransas	19	41	27	213	689
PCB Release in 20 Percent Elutriate Test					
Galveston Channel	437	33	171	261	53
Upper Mississippi River	60	7	<6	11	83
Duwamish River 2	702	406	12	33	190
Oakland Harbor 2	1436	360	<6	20	233
Stamford Harbor W	1877	2064	7	26	271
Perth Amboy Anchorage	2091	1760	17	81	376
Bailey Creek	1059	235	<6	34	467
Los Angeles A7	2438	4430	<6	36	500
Bay Ridge Channel	826	1820	8	86	975
Perth Amboy Channel	4044	1078	<6	69	1050
HSC 2	3634	7918	64	872	1262
Norwalk Harbor	2506	2820	<6	104	1633
Texas City Channel 4	533	3010	52	9619	18,398
HSC 1	5868	5325	<6	11,755	195,817
PCB Sorption in 20 Percent Elutriate Test					
Apalachicola 1	345	13	24	14	42
Apalachicola 3	423	454	38	<6	84

Table 9

Comparison of Release and Sorption for PCBs and CHPs

Study Site	Total PCBs	Total CHPs	Total DDT	Aldrin-Dieldrin	Heptachlor	Lindane
Rodeo Flats	-					
Mare Island	-					
Oakland 1	ND	+	+		+	-
Oakland 2	+	-			-	-
TCC 1	+	+		+		+
TCC 4	+	+	+	+		+
GBEC Buoy 1	-	N	N	-		N
Galveston Channel	+	+		+		+
Los Angeles A7	+	-	+			-
Newport, R.I.						
Stamford,Washington	+	+	+			
Norwalk South	+	+	+	+	+	+
Apalachicola 1	-	-				-
Apalachicola 3	-	+			+	+
Duwamish River	+	+	+		+	
Perth Amboy Channel	+					
Perth Amboy Anchorage	+	+	+		+	+
Bay Ridge Channel	+	N	+	-	+	N
HSC 1	+	+	+	+		-
HSC 2	+	+	+	+	+	-
Mississippi River	+	+	+	-		+
James River		+	+			+
Bailey Creek	+	+	+			-

ND = No Data

N = No Change

+ = Release

- = Sorption

Blank = Below Detection Limit

not exist. It appears that when the sediment-to-water ratio is high, other factors, in addition to oil and grease, influence the behavior of PCBs during the elutriate test. Fulk et al.[23] found in a laboratory leaching study that there was a direct correlation between PCBs remaining in the water column and the amount of oil and grease remaining in suspension. In general, as can be seen from Tables 8 and A4, it appears that the oil and grease in the sediment is an important factor affecting release of CHPs and PCBs.

Type of water

Based on the aqueous environmental chemistry of contaminants in natural waters, it is likely that the type (i.e., source) of water used in the elutriate test may affect the amount of release of some chemical contaminants during elutriation. Lee et al.[1], therefore, recommended that dredging site water be used for elutriation of those sediments which would be hydraulically dredged and disposed of in open water, and that disposal site water be used for sediments dredged mechanically and dumped via barge. In situations where the dredging site waters are highly stratified, it is likely that use of surface and bottom water for elutriate tests would show different patterns of contaminant release. In those cases, a mixture of surface and bottom waters should be used for elutriation since the hydraulic dredging process generally mixes, to some extent, both the surface and bottom waters with the sediment.

Phosphorus. The work done by Piwoni[4] and Lee et al.[1] on the effect of source of site water on elutriate test results showed that, in general, for phosphorus there was little difference in elutriate test P release when different site waters were used. During the current study, three sediments, Apalachicola Bay Sites 3, 4, and 5, were evaluated using two types of water. At that site, the dredging site surface and bottom waters had markedly different salinities. The elutriate tests run on Sites 3, 4, and 5 sediment using surface water all showed soluble ortho P release, whereas those run on the same sediments using bottom site water showed soluble ortho P uptake. It is likely that in the case of Apalachicola Bay sediment elutriate tests, the difference in salinity was a major factor affecting the phosphate behavior. This would be expected due to the tendency of more saline waters to flocculate and remove phosphate from solution.

Heavy metals. Lee *et al.*[1] and Lopez[21] reported that experiments conducted with Corpus Christi Site 2 sediment indicated that the characteristics of the elutriate test water can affect the release of heavy metals. Similar results are shown in Appendix A2, especially for sediments exposed to Reconstituted Sea Water versus dredging site water. The greatest influence would be expected in dredging situations where marked salinity differences exist between the dredging and disposal sites, especially with estuarine dredging and disposal at sea. In this type of situation, different elutriate test results may be obtained by using dredging site or disposal site water as the test water.

Nitrogen. From examination of the results of GBEC Buoys 9 and 11, elutriate tests for nitrogen compound release (Table A3), it appears that although use of disposal site versus dredging site water generally produced somewhat different elutriate concentrations of organic N, ammonium and nitrate, the direction of concentration change was not affected by the site water used. Comparison of elutriate test results for Apalachicola Sites 3, 4 and 5 surface and bottom water tests shows that overall, the general behavior and final elutriate concentration of ammonia was not affected by the different site waters even though considerably different concentrations were found in the site waters. Concentrations of nitrate and organic N were, however, generally considerably lower in the bottom water elutriate than in that of the surface water.

Agitation time

The time for mixing (30 minutes) was prescribed based on an estimation of duration of mixing that typically takes place during hopper and pipeline dredging.[6] This time in actuality varies from site to site as discussed previously. Although no investigation of the effect of varying this parameter on elutriate test results was made during the current investigation, a number of investigators have made this evaluation.

Phosphorus. Tests conducted by Piwoni[4] indicated that increasing removal of soluble ortho P may have been occurring with increased agitation time. According to him, however, elutriate soluble ortho P concentrations were not statistically different among elutriates which had been mixed for 15, 30, 60, and 90 minutes (See Table A1, Mobile Bay tests). It should be noted, however, that those tests run by Piwoni were mixed using mechanical shaking rather than compressed air agitation. The results may not, therefore, be comparable with results of aerated tests.

Heavy metals. Lee *et al.*[1] and Lopez[21] subjected Mobile Bay, Alabama sediments to elutriation using 15, 30, 60, and 90 minute mixing (shaking) periods. Results showed substantial release of Mn irrespective of the length of mixing period. The selected heavy metal concentrations in all elutriates were essentially identical, showing that for these compounds, the elutriate test was not sensitive to the duration of agitation. Response of other metals also appeared to be independent of mixing time. Results using mixing periods shorter (15 minutes) or longer (60 or 90 minutes) than 30 minutes tended to show larger variances. O'Connor[24] observed different elutriate-seawater Zn concentration ratios when mixing the suspensions for 30 minutes as compared to 24 hours. From this and other supportive data, he concluded that elutriate analyses do not represent an equilibrium between sediment and water. He further suggested the importance of mixing time and scavenging effects of ferric hydroxide within the isolated elutriate in controlling final heavy metals concentration in the elutriate.

Nitrogen. Lee *et al.*[1] and Piwoni[4] reported that based on the Mobile Bay studies, increased settling time generally increased ammonium release and showed a decrease in nitrate concentration in the elutriate.

Settling time

The one-hour settling time specified in the elutriate test procedure allows for contaminant release from slow settling sediment. Some attention was given to the effect of varying this operating condition on contaminant release in the elutriate test by Lee *et al.*[1] They conducted a series of elutriate tests in which they used both one-hour and 24-hour settling periods for 5, 10, and 20 percent oxic tests on several sediments. For ammonium, nitrate, and the selected heavy metals investigated, they found that the duration of settling did not affect release. For soluble orthophosphate, they observed greater increases with the longer settling period. They related this increased release to the decreased oxygen at the sediment-water interface and pointed out that due to current movement typically found at disposal sites, this behavior would not be expected to be found in the field.

As part of the study of phosphate behavior in the elutriate test included in this investigation, it was observed that in a number of cases where duplicate tests were run, the second replicate elutriate had a higher soluble ortho P concentration than the first. Further, there was a tendency for increasing elutriate concentration with each of the five replicate elutriate tests run on Houston Ship Channel sediments (Table A1). This is likely related to the fact that often, due to the slow filtering of some elutriates, the second and following elutriates were allowed to stand unfiltered for periods longer than the prescribed 1.5 hour contact time while other samples were being processed.

Sample storage

Lee _et_ _al_.[1] showed that the storage of sediment and water samples prior to elutriation can alter oontaminant release patterns. To investigate this further for a number of different samples, following the elutriation of freshly collected samples (less than two weeks old), samples were returned to storage for periods of 1.5 to 2.5 months; elutriate tests were then re-run.

Phosphorus. Examination of storage study results in Table A1 shows that there was generally a change in the soluble ortho P content of the site water over the storage period. Further, there were generally changes in the pattern and amounts of soluble ortho P release resulting from elutriation. In many cases, results of tests after storage were directly opposite of those found in elutriate tests on fresh sediment. It is concluded that in order to obtain as accurate a prediction as possible of the behavior of soluble ortho P during dredged material disposal operations, elutriate tests should be run as soon as possible after collection. Sediments and water samples should be kept cold (but not frozen) and in the dark between sampling and testing to minimize chemical and biological activity.

Nitrogen and heavy metals. Lee _et_ _al_.[1] reported that extended storage of sediment and site water samples prior to elutriation resulted in higher ammonium release in the elutriates. Results of storage studies conducted as part of the current investigation (Table A2) showed that there was no consistent effect of sample storage prior to elutriation on the release of the nitrogen compounds. It appeared that elutriate concentrations were about the same in many cases but the concentrations in the site water had changed during storage. In the case of TCC Site 6 studies, there was less release than there had been in tests run before sediment storage.

There was no consistent pattern to the differences in heavy metal concentration in elutriates from tests run before and after storage.

Overview of Elutriate Test Applicability

Because of the lack of technical validity in the use of bulk sediment criteria for judging potential water quality problems resulting from dredged sediment disposal, the elutriate test procedure was developed. The operating conditions specified for the elutriate test such as percent sediment used, mixing and settling time, etc., were for the most part based on general impressions of the dredging and disposal process. In order to assess the widespread applicability of the elutriate test and to ensure that it is only the sediment characteristics which are affecting test results, an evaluation was made of the sensitivity of the test results to the various operating conditions. If the test results proved to be highly sensitive to these conditions, then it would lose general applicability because of normal differences among laboratories and because of the fact that many of the prescribed operating conditions are highly variable among dredging and disposal operations.

Lee *et al.*[1] evaluated the response of contaminant release to alterations in the type of water and percent sediment used, method and duration of mixing, and duration of settling. They concluded that unless the redox conditions were specified for the mixing period, elutriate concentrations could be highly erratic among replicate tests and be dependent on mixing technique rather than on characteristics of the sediment being tested. They therefore recommended that compressed air agitation rather than mechanical shaking be used for mixing so that oxic conditions would be maintained. They found that using this modified procedure, elutriate test results could be reproduced by replicate testing of sediments. Although for some contaminants other specified conditions affected release, the differences were generally insignificant, i.e. the difference in results would not be expected to change a decision regarding the potential pollutional tendency of the sediment should it be disposed of in open water. Because of this, and the fact that it was found that the test was often easier to perform using a lower percent sediment that was still justifiable based on dredging-disposal characteristics, Lee et al.[1] recommended that 5% rather than 20% sediment be used.

During the current study, a larger number and wider variety of sediments were evaluated than included in the Lee et al.[1] study. This evaluation supported the general conclusions of Lee et al.[1] and also demonstrated that for the most part, the specified mixing and settling periods were appropriate based on field evaluation of the test. In the case of phosphate, because of the relationship of its behavior to the iron system and the presence of dissolved oxygen, it is appropriate to conduct anoxic (N_2-mixed) as well as oxic tests to determine the range of water column concentrations possible in the turbid plume after disposal. If it is suspected that phosphate release would cause a water quality problem in the disposal area, a case-by-case approximation of redox conditions and contact time for the elutriate test would also be advisable since those parameters tended to have some effect on the amount of phosphate released in elutriate tests.

During this investigation, further attention was given to the effect of storage of sediment and water samples prior to elutriation on elutriate test results. Extended storage often resulted in unpredictable differences in the amount and often direction of release of various contaminants measured. In order for the elutriate test to provide an appropriate prediction of the behavior of a dredged sediment, elutriate tests should be run as soon as possible (at least within 2 weeks) after sample collection.

Because of the differences in sediment-water mixing between hydraulic dredging-pipeline or hopper dredge disposal and mechanical dredging-barge disposal, the elutriate test (designed for hydraulically dredged sediment) was modified so the same test principle could be applied to mechanically dredged sediments as well. It was found that this "plop test" was suitable for use to predict the behavior of chemical contaminants during barge disposal of mechanically dredged sediment.

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Subsequent to the completion of this report, a report authored by Wyeth and Sweeney* was received. One of their conclusions was,

> "currently employed elutriate test procedures were judged to be of limited value for evaluating the potential release of contaminants from the proposed dredged materials. Comparison of elutriate and on-site monitoring data yielded incompatible results. Those parameters which increased in the elutriate tests did appear in elevated concentrations during disposal monitoring. The reverse, however, was not true. During disposal monitoring all parameters that showed maxima or minima were not evident during the elutriate test results. "

However, examination of the approaches and data upon which this conclusion was based showed that the elutriate test procedure

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*Wyeth, R.K. and Sweeney, R.A., "Aquatic Disposal Field Investigations, Ashtabula River Disposal Site, Ohio - Appendix C: Investigation of Water-Quality and Sediment Parameters," Technical Report D-77-42, 1978, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

used by Wyeth and Sweeney was not appropriate. They used a mechanical shaking technique even though it has been repeatedly demonstrated that compressed air agitation must be used for elutriation in order to obtain reliable, interpretable results. Further, although many of the details of the field procedures used are unclear from their report, there appear to be a number of deficiencies in them. While these deficiencies would be insignificant with respect to the evaluation of the overall impact of dredged material disposal on disposal site water quality, they would likely be highly significant with respect to evaluating the reliability of the elutriate test.

From an overall point of view, it is concluded that Wyeth and Sweeney's statement regarding the suitability of the elutriate test is inappropriate and not technically valid based on the data presented in their report.

PART V: ENVIRONMENTAL IMPACT OF TURBIDITY AND SUSPENDED SOLIDS ASSOCIATED WITH DREDGED MATERIAL DISPOSAL OPERATIONS*

The US EPA Quality Criteria for Water[8] established a criterion for water column settleable solids and turbidity for the protection of fish and other aquatic life where the addition of suspended solids should not reduce the seasonal average depth of the compensation point for photosynethetic activity by more than 10 percent from the seasonally established norm. The compensation point is the depth at which photosynthesis equals respiration in a water column. Examination of the data on optical properties of the water column obtained in the Galveston Bay Entrance Channel (GBEC) study shows that based on Secchi depth data, the compensation point would be decreased by more than 10 percent in the turbid plume associated with disposal. The overall extent and duration of the turbid plume, determined by aerial photography and transmissometer data, was such that the GBEC disposal operations would probably not be in violation of the US EPA Quality Criteria for Water for suspended solids and turbidity because of the intermittent nature of the dumping operations and the short duration of the existence of the turbid water following dumping.

The duration and degree of increased turbidity at the other open water disposal sites where the impact of dredged sediment dumping was investigated, were similar to those obtained in the GBEC study. Therefore, it is appropriate to conclude that, in general, open water dredged material dumping would not have a long-term significant adverse impact on water clarity because of the intermittent nature of dredged sediment dumping operations. Also, at most open water disposal sites, dredging and disposal usually occur for only a short period of time each year or every few years while dredging is being conducted in the nearby channels. There are some locations,

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*This section of the report is based on draft materials submitted by G. Mariani.

such as the New York Bight mud dump site, where dredged sediments from many different channels are brought to one location for disposal throughout the year. Even at this site, dumping is still intermittent and the mixing in the region would be expected to rapidly dilute the turbid plume created from each dump so that there would be no significant long-term impact of the dumping operation on the optical properties of the water column which would be adverse to aquatic organisms.

It should be noted that the AFS panel which reviewed the suspended solids section of the US EPA Quality Criteria for Water released July 1976, has recommended that the US EPA criterion of a ten percent change in the compensation depth be changed to something that would be more workable.[25] It is likely, based on this recommendation, that future revisions of these criteria will attempt to establish critical concentrations of suspended solids for aquatic life.

Although it is possible that organisms inhabiting the water column at an open water dredged material disposal site may be affected by a brief exposure to increased concentrations of suspended material, it is more likely that the most adverse effects resulting from dredged material disposal would occur in the near-bottom waters. In both the Galveston Bay and New York Bight dumps, elevated turbidity in the near-bottom waters, resulting from the disposal of dredged material, lasted for more than one hour at the sampling site but was not detected the following day.

Lee *et al*.[26] concluded after consideration of the impact of suspended solids released to the water column during the GBEC disposal operations, that the water column organisms would not be adversely affected by dumping. This conclusion was based on the work of Peddicord *et al*.[27] who have found that, in general, aquatic organisms are relatively insensitive to moderate concentrations of suspended solids.

Many aquatic organisms are adversely affected only by concentrations in the grams-per-liter range or greater and then only when the elevated levels persist for several days to several weeks. The results of the Peddicord et al.[27] studies are similar to the conclusions of Lee and Plumb[9] who reviewed this topic as part of their literature evaluation of the environmental impact of dredged material disposal on water quality. Further, Stern and Stickle[28] recently published a literature review on the effects of turbidity and suspended solids in aquatic environments on water quality and aquatic organisms. Their literature review was specifically directed to the potential environmental impact of dredging and dredged material disposal. They concluded that except for the burial of organisms at a disposal site, most dredged material disposal operations likely have limited environmental impact on water quality and aquatic organisms. However, the turbidity and suspended solids released during dredged material disposal near coral reefs could have adverse impact on coral reef communities. Additional discussion of the effects of turbidity on water column organisms has been provided by Pequegnat.[29]

While dumping of dredged sediments does not appear to create situations where the concentrations of suspended solids in the water column would be sufficient to be adverse to aquatic organisms at or near the disposal site, the pipeline discharge of hydraulically dredged sediments could readily create adverse conditions for aquatic organisms. Depending on the frequency and distance of movement of the pipeline discharge point, a turbidity current-mud flow could be present in a region for several days. The concentrations of suspended solids in the mud flow are sufficient to be adverse to aquatic organisms. While there is potential for harm to some aquatic organisms due to open water pipeline discharge of dredged sediments, the ecological significance of mud flows would, in general, likely be small since the organisms that inhabit the areas are frequently exposed to large concentrations of suspended sediments associated with wind-induced mixing of the sediment into the shallow water column.

Lee et al.[26] pointed out that the increased turbidity in the turbid plume associated with disposal would have an insignificant effect on benthic and epibenthic organisms when compared to the amounts of suspended solids frequently stirred into the water column during storms. Sly[30] reported that for the Great Lakes, dredging-disposal and shipping-induced turbulence suspend large amounts of waterway sediments; however, he indicated that these increases are small when compared to sediment resuspension caused by wind-wave action.

From an overall point of view, turbidity and suspended solids associated with disposal operations involving dumping, are judged to have an insignificant effect on dredged material disposal area water quality. As discussed by Lee et al.[26], there would be some burial of organisms at the disposal site. Even this seems to be of relatively minor consequence in most situations because many benthic organisms can migrate considerable vertical distances upon burial. Further, studies on the repopulation of dredged material disposal sites show that within a relatively short time, many of the disposal sites show a rapid recolonization with a variety of benthic and epibenthic organisms. For further discussion of this topic, consult Oliver et al.[31], Serafy et al.[32], and Richardson et al.[33] Special consideration must be given to the open water disposal of dredged sediments in areas where there could be burial of coral reefs, oyster beds, and disruption of spawning areas. Disposal operations should not be permitted in these areas where significant harm to the ecosystem could occur.

PART VI: BIOASSAYS*

The development of the modified elutriate test bioassay as part of this study was prompted by difficulties in using chemical data generated through elutriate tests to predict possible adverse effects of dredged sediment-associated contaminants on aquatic organisms during open water disposal. These difficulties resulted primarily from the lack of information on the toxicity to and/or bioaccumulation within aquatic organisms of contaminants present in the elutriates or associated with the sediment fraction. The difficulty in interpreting results of the elutriate test to reliably predict potential adverse effects of various forms and concentrations of contaminants present in dredged sediments is further complicated by the fact that the US EPA's <u>Quality Criteria for Water</u>[8] are not valid for evaluation of all the different forms of a particular compound that can be present in the liquid, suspended particulate, and solid phases of dredged sediments. In addition, additive, synergistic and antagonistic toxicities and toxicity associated with unmeasured contaminants cannot be assessed from the information obtained from routine chemical analysis of the elutriates.

The bioassay for marine and estaurine organisms developed and evaluated in this study involved exposing <u>Palaemonetes pugio</u> to the settled elutriate with sediments present, typically for 96 hours. For freshwater, <u>Daphnia magna</u> were exposed to filtered elutriate also typically for 96 hours. A summary of the results of the modified elutriate test bioassays on 43 sediment samples from 19 US waterways conducted during this study are presented in Appendix Table A5.

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*This section of the report was based in part on draft materials submitted by G. Mariani.

Various aspects of the information contained in this table are discussed below.

Reproducibility of test results

Examination of the data on the toxicities between replicate bioassay tests (A and B), as presented in Table A5, indicates that the bioassay procedure used as part of this study is capable of yielding highly reproducible results under standard procedures and conditions. Mortalities associated with the exposure of *D. magna* and *P. pugio* to elutriates of sediment in duplicate tests for 96 hours did not differ by more than 20 percent in 88 of the elutriate test bioassays conducted. It should also be noted that only one bioassay had mortality greater than 10 percent in the controls.

Comparison of bioassay and heavy metal release results using dredged discharges and modified elutriates

Two studies were conducted to determine the differences between the results of bioassays conducted with elutriates of dredged sediments and bioassays using dredged sediment pipeline discharges collected at the associated discharge site. Sites evaluated in this manner were the Upper Mississippi River near St. Paul, Minnesota and the James River near Hopewell, Virginia. The results of these studies showed that the responses of the test organism, *D. magna*, to the dredged sediment discharges and the biaossay elutriate waters were essentially the same (Table A5). Further, examination of the heavy metals data for the dredged sediment discharge waters and modified elutriate bioassay waters (see Vol II) shows, with the exception of maganese, that the concentration of each heavy metal monitored was only slightly greater in the modified elutriate bioassay waters. These results indicate that the modified elutriate test bioassay procedure provides conditions which are similar as far as heavy metal release is concerned to those encountered during hydraulic dredging-pipeline discharge operations.

Variability in toxicity of dredged sediments from the same general area

Although the procedures used as part of this investigation proved to be reproducible when using subsamples of a sediment, the toxicity associated with the exposure of the test organisms to sediment samples taken at different sites in the same general sampling area proved to be somewhat variable. This is evident from the Perth Amboy Channel, New York bioassay results (Table A5), in which sediment samples taken at different specific locations (300 meters apart) within the same channel produced somewhat different acute lethal effects on the test organism, *P. pugio*. Examination of the Perth Amboy Channel data presented in Table A5 shows limited toxicity to the grass shrimp using sediments from Sites 1 and 3. However, the bioassays using sediment from Site 2 showed a higher mortality, with 40 percent of the test organisms dead in each of the triplicate tests after 96 hours.

Evaluation of the chemical and physical characteristics of the bioassay elutriates following the one-hour settling period shows that the Perth Amboy Site 2 elutriates contained six to eight times more ammonium (18 to 31 mg N/l) than the elutriates of sediments from Sites 1 or 3 (Vol.II). The heavy metals data for all three site elutriates were similar. Whether or not the increased levels of ammonium in the Site 2 bioassay elutriates were responsible for the higher mortalities cannot be properly evaluated at this time. However, these results indicate that variations in the chemical composition of sediments from the same general area may be great enough to display markedly different toxicities to organisms.

The only other bioassays conducted on a number of samples from different sites in the same general area were those run on the Texas City Channel Sites 1 through 6 sediments. Examination of the results for this region of the

Texas City Channel shows no large variations in the observed toxicities. In addition, the chemical and physical characteristics of the bioassay elutriates for the Texas City Channel bioassays show no large variations in the values of the parameters measured, among these sampling sites. From these results, it appears that the variability of the toxicity of sediments would, as expected, be highly site-specific. For any given region, study needs to be done to determine the variability of the toxicity of sediments from the same general area. Based on this variability, an appropriate sampling program should be established to yield representative bioassay results for the area.

Studies of the toxicity of the sediments from Bailey Creek near Hopewell, Virginia provided an opportunity to examine the potential changes in toxicity associated with sampling of the same site at two different times of the year. The Bailey Creek sediments were known to be contaminated by the pesticide kepone. They were sampled on April 3, 1976 and July 15, 1976. The results of the Bailey Creek sediment (April 3, 1976) bioassay revealed limited toxicity at the above normal test temperature of 30°C. Only one of the 20 _Daphnia_ subjected to the elutriate died during the 96-hour exposure period. Based upon the results of this bioassay, it is concluded that the higher toxicities previously reported by the Waterways Experiment Station staff of the US CE may have been a result of improper acclimatization of the test organisms or procedural differences.

The bioassays conducted on the James River, Virginia and Bailey Creek sediments collected in July showed higher toxicities than previously observed using the Bailey Creek sediment sample collected April 3. The results of the bioassays conducted on the July sediment samples show that the acute lethal response of _D. magna_ was greater in the 5 percent sediment elutriates than in the 20 percent elutriates. The Bailey Creek 5 percent sediment elutriate bioassay showed a 30 percent mortality for *D. magna* over the 96-hour test period, while in the 20 percent sediment elutriates only one of the ten test organisms died.

The reason for the differences in toxicity of Bailey Creek sediment between the two sampling dates is unknown. It could be due to the heterogeneity of the sediments at the sampling site and/or changes in the characteristics of the sediments during the time between sampling. These results again point to the importance of conducting an adequate sampling program to properly characterize the sediments in both time and space.

Relationship between toxicity and chemical characteristics of the sediments

Detailed examination of the data generated in these studies showed that there was no relationship between the bulk composition of the sediments for various contaminants or groups of contaminants and the toxicity of these sediments to aquatic organisms. This is to be expected since no relationship was found between the bulk content of the sediments for any chemical parameters and the release of that chemical to the water in the elutriate test. Without this type of relationship there is little likelihood that there would be a correlation between the bulk chemical content of a sediment and the toxicity of that sediment to aquatic organisms.

Chemical analyses of all the bioassay elutriate waters following the one-hour settling period, showed that ammonium, manganese, and iron were released from most of the sediments tested. Of these, ammonium was the only chemical contaminant that was consistently released from the sediments as a function of sediment percentage of the total elutriate volume. Figure 6 depicts the relationship between total ammonium release and percentage of sediment.

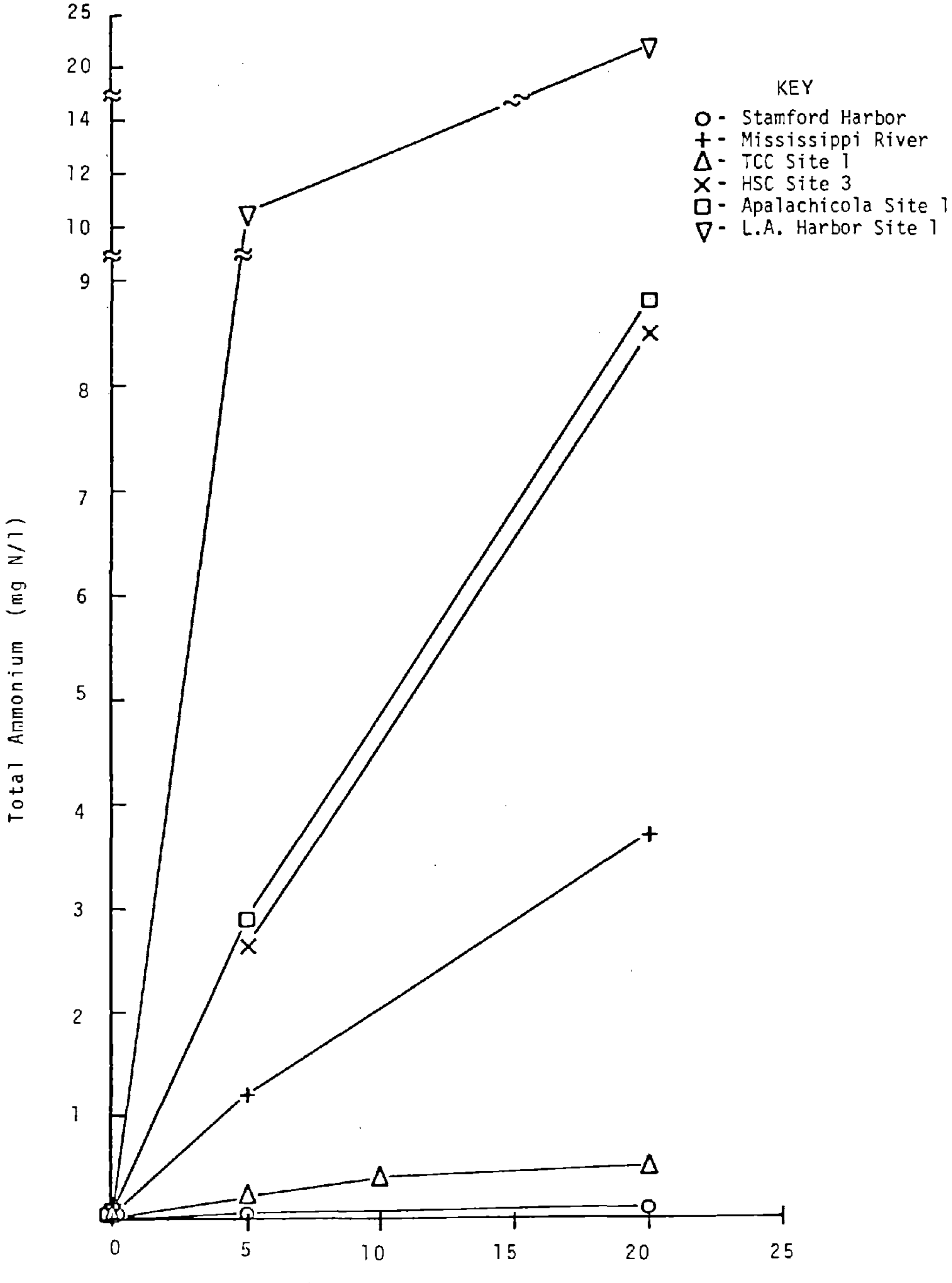


Figure 6
Release of Total Ammonium to Bioassay Elutriate Waters

used for selected sediments in the modified elutriate test bioassay, while Figures 7 and 8 show the variable release patterns for iron and manganese, using the same sediments. Although manganese and iron were released from most of the sediments tested, the results indicate that there is no direct relationship between the release of these metals and the percentage sediment of total elutriate volume used in the modified elutriate test bioassay.

In addition, examination of the total heavy metal content of these sediments shows that the amount of iron or manganese released to the bioassay elutriates is not directly related to the amount of these metals contained in the sediments. It is concluded that factors governing the fate of heavy metals, such as precipitation, redox, complexation, and sorption-desorption processes, appear to play a greater role than bulk content in determining whether or not a metal is released to the bioassay elutriate waters. These conclusions are supported by the studies presented in other parts of this report.

Figure 9 shows the response of the test organisms used in this investigation (P. pugio and D. magna) to dredged sediments collected from harbors and waterways across the United States. Each sediment is characterized based upon its bulk metal composition depending on how many of the metals in the sediment sample analyzed exceed proposed limits for heavy metals established by US EPA Regions V and VI (Table 15 in Part VIII) for determining the acceptability of dredged sediment for open water disposal. Responses of the test organisms are presented as the average of two replicate 96-hour static bioassay tests.

Examination of Figure 9 shows that 19 of the 25 sediments that exceeded proposed limits for two or more heavy metals showed toxicities of 20 percent or less to the test organisms used in this investigation. In addition, the results

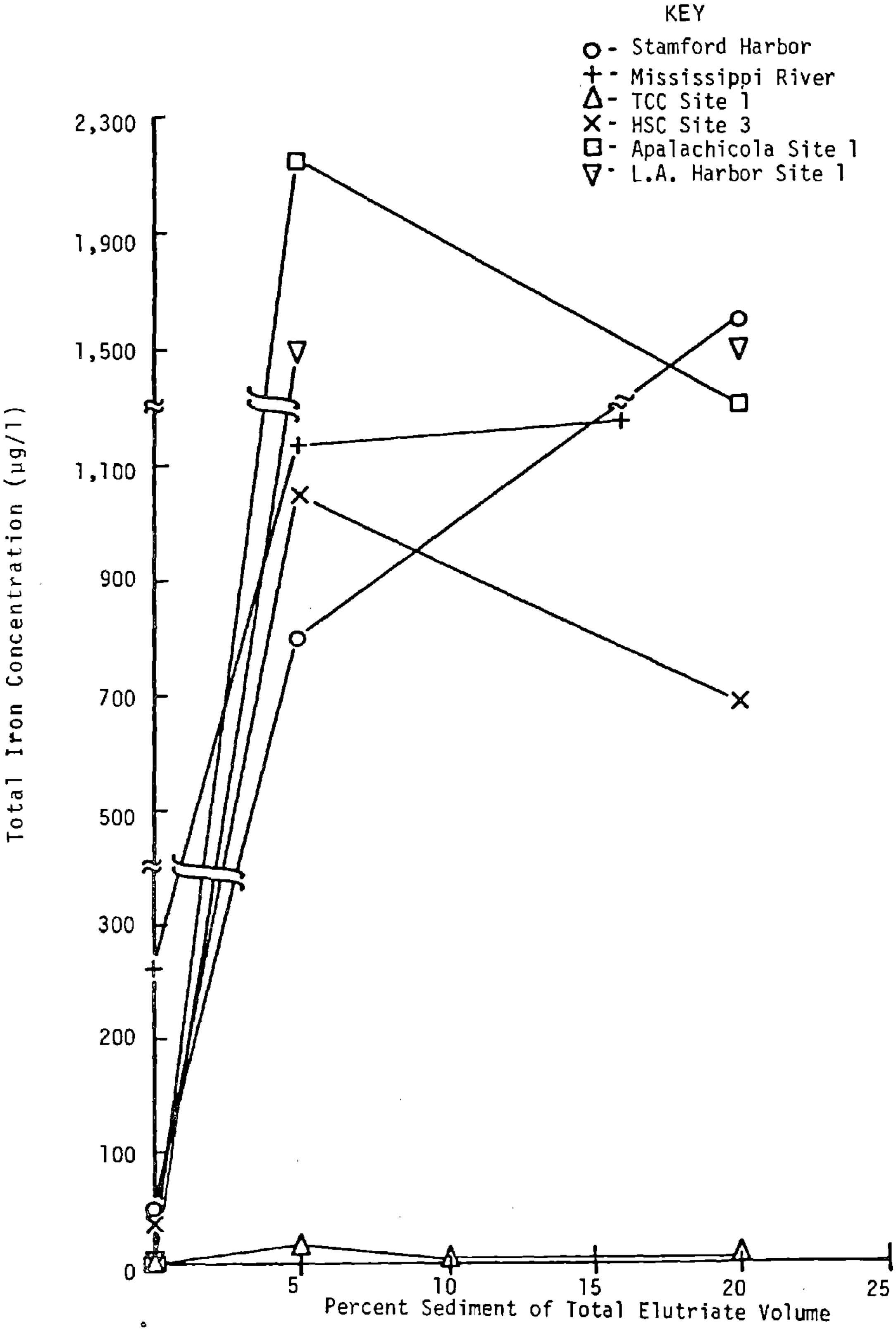


Figure 7
Release of Iron to Bioassay Elutriate Waters

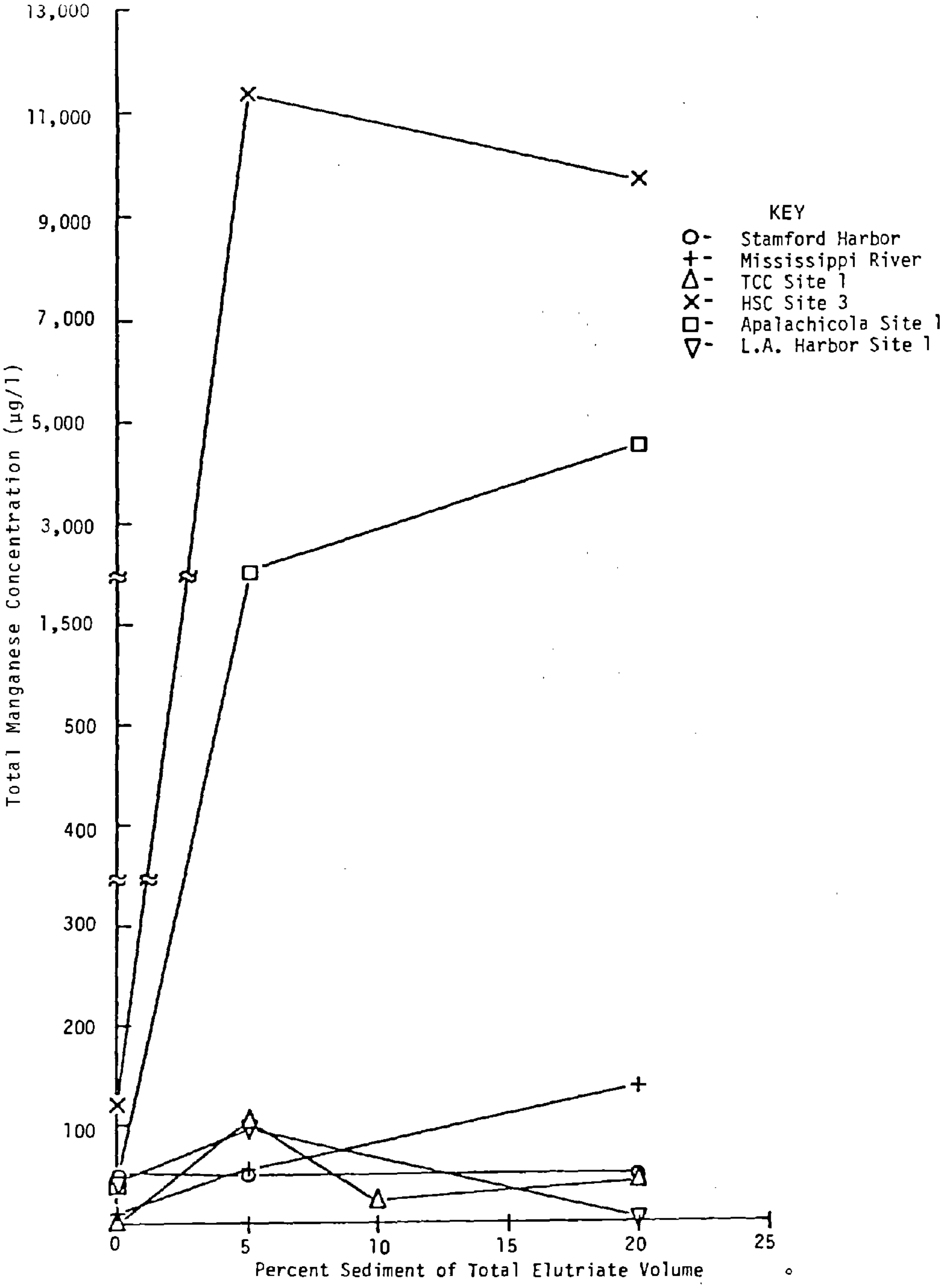


Figure 8
Release of Manganese to Bioassay Elutriate Waters

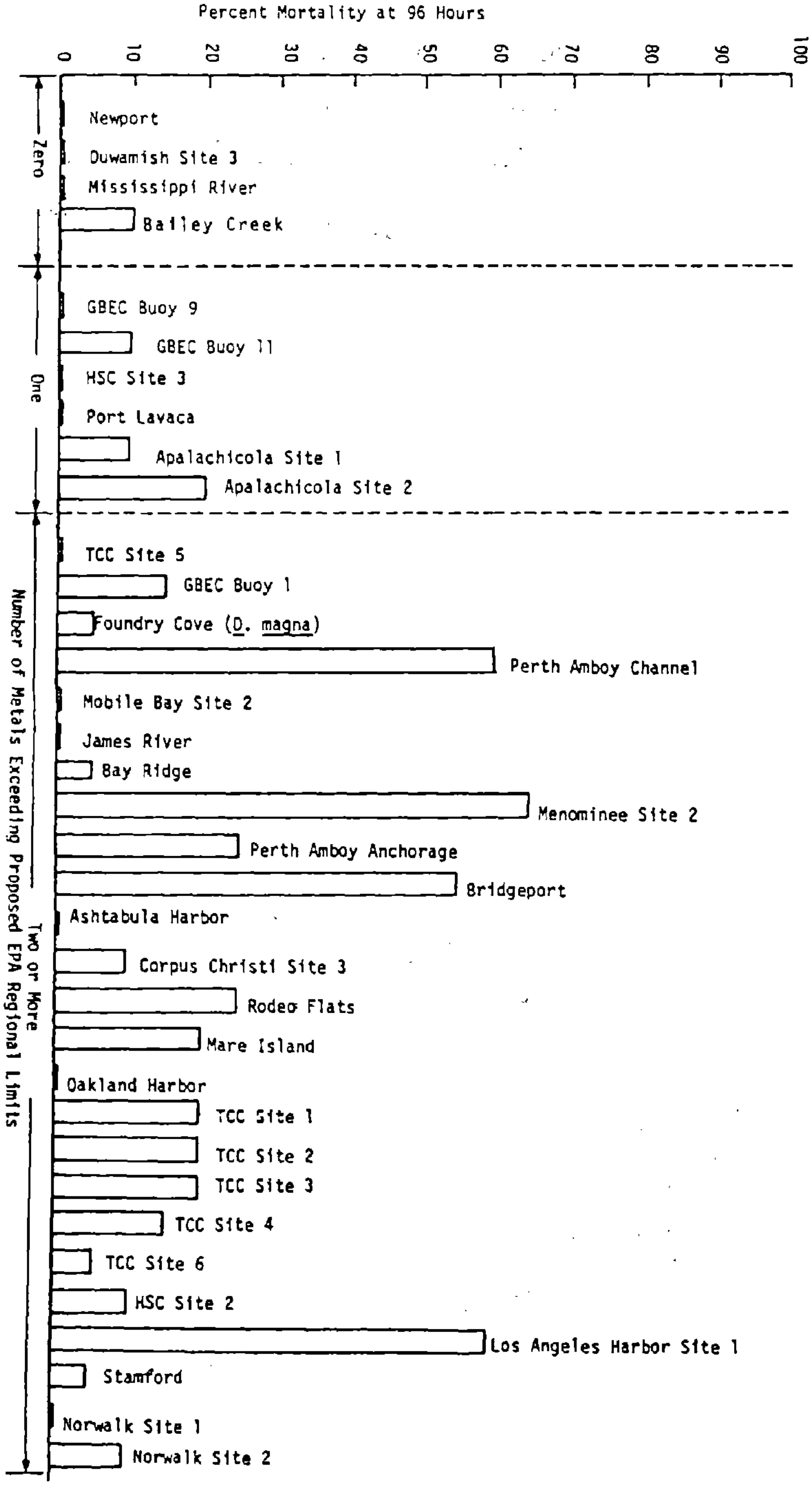


Figure 9

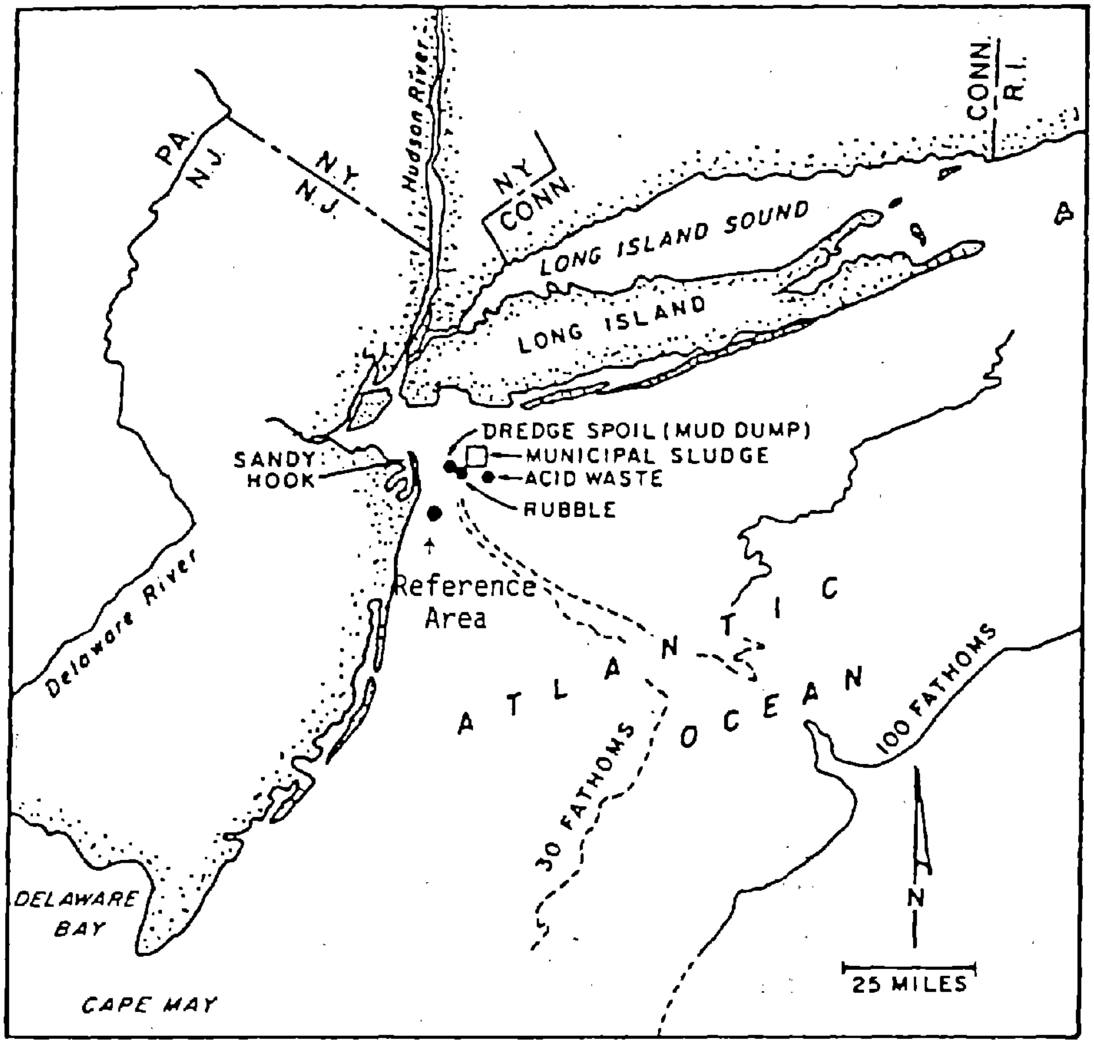
Response of D. magna and P. pugio to 20 Percent Sediment Elutriates of all the bioassays on elutriates of sediments which exceeded the limits for one metals showed limited or no acute toxic effects on the test organisms. Based upon these results, there does not appear to be a relationship between the guidelines as proposed by Regions V and VI and observed toxicity during the modified elutriate test bioassay, thus demonstrating the inappropriateness of bulk chemical characterizations in evaluating the potential biological significance of open water dredged material disposal.

PART VII: BIOACCUMULATION OF PERSISTENT CONTAMINANTS

Introduction

One of the areas of concern with respect to dredged material disposal is the accumulation of chemical contaminants in aquatic organisms which would be detrimental to the organism or to higher trophic level organisms which use the contaminated organism as a food source. While this was not a focal point of this study, some information on this topic was gathered by the authors in a companion study of the environmental impact of dredged material disposal in the New York Bight.[34] Benthic and epibenthic organisms from the mud dump site in the New York Bight (Figure 10) and adjacent areas were collected and analyzed for mercury, cadmium, selected chlorinated hydrocarbon pesticides and PCBs. The results of this study are presented below since they provide valuable additional information on the potential environmental impact of the open water disposal of highly contaminated sediments at one of the major study sites of the current investigation.

Few commercially important benthic organisms with limited mobility, such as clams, were found at the mud dump site at the time of collection in the spring of 1977. Evidently, the large-scale die-off of benthic organisms, associated with low dissolved oxygen conditions in large areas of the New York Bight that took place during the summer of 1976, significantly depleted the clam populations in the mud dump site and nearby areas such that few specimens were available for collection. A number of crabs and bottom-dwelling fish were collected at the New York Bight mud dump site for analysis. Even though these organisms are mobile, it was felt that since they were collected at the mud dump site and, therefore, had a potential opportunity to accumulate elevated concentrations of contaminants released from dredged sediments, the analysis of their



After Mueller et al.[35]

Figure 10
New York Bight Area

tissue would give an indication of the potential for significant bioconcentration problems in the area. If excessive concentrations of selected contaminants were found in a vareity of organisms collected from the mud dump site, then steps should be taken to ascertain the role of dredged sediments in contributing to these elevated concentrations. If, however, these organisms had low concentrations of contaminants compared to FDA guideline concentrations, then either of two situations exists. One is that the organisms which were captured at the mud dump site had not had sufficient time to accumulate an elevated concentration of contaminants within their tissues. The other alternative is that the contaminants present in the sediments, water and lower trophic level organisms were not being made available to the organisms at a sufficient rate to cause the concentrations within their tissue to exceed critical values.

It is sometimes suggested that certain non-commercially or recreationally important benthic species be used in this type of bioconcentration study.[36] This approach should not be followed since, as discussed in another part of this report, at this time it is impossible to interpret the significance of concentrations of contaminants in the tissue of these organisms. One of the types of organisms frequently suggested for use in bioconcentration studies is a benthic worm, such as the polychaete. In addition, to the above-mentioned problem of not knowing what the critical levels of contaminants in polychaete tissue are, there is the problem of properly purging the organism of the sediments within its intestinal tract prior to chemical analysis. It is, therefore, felt that little in the way of useful-interpretable information would be obtained by collecting and analyzing polychaetes and similar types of organisms from the mud dump site and adjacent areas. Therefore, this approach was not used in this study.

The details of the sampling and analytical procedures used have been presented by Lee and Jones.[34] In general, they are similar to procedures used in this study as reported in Volume II of this report.

Results

Of the organisms collected in the New York Bight, six rock crab (C. irroratus), five whiting (M. bilinearis), and three ling (U. chuss) from the mud dump site and three whiting, three flounder (P. americanus) and three ling from a reference area were analyzed (edible portions only) for the following constituents: mercury, cadmium, selected chlorinated hydrocarbon pesticides and PCBs.

The US FDA 1976 guideline concentrations[37] for mercury, chlorinated hydrocarbon pesticides and PCBs in aquati organisms are listed in Table 10. Table 11 presents the concentrations of these substances found in the edible portions of rock crab (Cancer irroratus) collected from the New York Bight mud dump site on April 26, 1977. Mercury concentrations in the edible portions of the five mud dump site crabs analyzed ranged from <0.02 to 0.19 mg/kg (wet weight of edible portion). These values are well below the US FDA guideline concentrations for mercury in fish (0.5 mg/kg based on edible portion wet weight). As reported by Saila and Segar[38], although the highest concentrations of mercury found in edible tissues of various fish and shellfish in the Bight was 2.3 mg/kg, average concentrations were <0.5 mg/kg. While the detection limit applicable to the analyses made by Saila and Segar were higher than those applicable to analyses made for this study, it appears that the two sets of values are at least of the same order of magnitude and do not indicate any significant environmental problem assiciated with mercury in the New York Bight.

Table 10

US FDA Guideline Concentrations for Mercury, Cadmium, Chlorinated Hydrocarbon Pesticides and PCBs in Aquatic Organisms

Concentrations in mg/kg (ppm)

Parameter	US FDA* (1976)
Heavy Metals	
Mercury	0.5
Cadmium	NL
Chlorinated Hydrocarbon Pesticides	
Lindane	0.3
Heptachlor	0.3
Heptachlor Epoxide	
Aldrin	0.3
Dieldrin	
Endrin	0.3
Endosulfan I & II	NL
Benzene Hexachloride	NL
Toxaphene	NL
Methoxychlor	NL
Mirex	0.1
Chlordane	NL
DDT and Metabolities	5.0
Total PCBs	5.0

*Based on wet weight of raw edible portion.

NL = No limits.

From Corneliussen[37]

Table 11

Heavy Metal, Chlorinated Hydrocarbon Pesticide and PCB Content of Rock Crabs (Cancer irroratus) From New York Bight Mud Dump Site*

Parameter \ Crab Number**	1	2	3	4	5
Heavy Metals mg/kg (ppm)					
Mercury	< 0.06	<0.06	0.10	0.19	< 0.02
Cadmium	0.12	0.13	0.08	0.13	0.01
Chlorinated Hydrocarbon Pesticides µg/kg (ppb)					
Lindane	4.1	4.9	5.6	4.5	2.0
Heptachlor	3.1	1.9	1.9	2.4	1.8
Aldrin	1.6	2.2	2.9	2.6	2.0
Dieldrin	< 2.5	< 2.5	< 2.5	< 2.5	< 2.5
Endrin	< 3.0	< 3.0	< 3.0	< 3.0	< 3.0
Endosulfan I	< 3.0	< 3.0	< 3.0	< 3.0	< 3.0
Endosulfan II	< 3.0	< 3.0	< 3.0	< 3.0	< 3.0
Methoxychlor	<20.0	<20.0	<20.0	<20.0	<20.0
Mirex	<10.0	<10.0	<10.0	<10.0	<10.0
Chlordane	<20.0	<20.0	<20.0	<20.0	<20.0
op'DDT	< 8.0	< 8.0	< 8.0	< 8.0	< 8.0
op'DDD	< 5.0	< 5.0	< 5.0	< 5.0	< 5.0
op'DDE	< 4.0	< 4.0	< 4.0	< 4.0	< 4.0
pp'DDT	18.8	14.7	11.8	11.0	16.2
pp'DDD	7.1	6.3	5.4	< 4.0	17.4
pp'DDE	20.1	20.4	17.1	10.1	7.6
Total DDT***	46.0	41.4	34.3	21.1	41.2
Total PCBs µg/kg (ppb)	203	229	385	287	548
Wet Weights (g)					
Total	145.1	95.5	61.4	57.3	74
Edible Portion	30.0	19.9	14.2	13.7	10.5

*Samples collected April 26, 1977.
All concentrations based on wet weights of edible portions of the organism.

**Values presented for pesticides and PCBs in crab Nos. 1-4 are results of single analyses of individual crabs. Values presented for pesticides and PCB's in crab No. 5 are the means of duplicate analyses of a composite sample of edible portions of two small crabs. See Table 14 for results of individual analyses of crab No. 5. Crab Nos. 1-3 analyzed for cadmium and mercury in duplicate; 4 and 5, singly.

***Exclusive of "less than" values.

<Indicates less than detection limit.

Concentrations of cadmium in the rock crab collected from the mud dump site ranged from 0.01 to 0.13 mg/kg (wet weight of edible portion). At this time, the FDA does not have a guideline concentration or action level concentration for cadmium in fish flesh. As reported by Saila and Segar[38], literature values for the concentrations of cadmium in edible portions of fish and shellfish in the New York Bight have been as high as 0.81 mg/kg. Average concentrations were reported by them to be generally less than 0.1 mg/kg. These are about the levels found in the crabs from the mud dump site analyzed during this study.

Concentrations of heptachlor, aldrin, dieldrin, and endrin in the mud dump site rock crabs analyzed were, in general, 0.005 to 0.02 of the FDA limits for these compounds in fish flesh. Concentrations of total DDT compounds were considerably below FDA limits. Similarly, concentrations of lindane and mirex were 0.01 and less than 0.1, respectively, of the FDA guideline concentrations established for these compounds. There are no FDA limits for the other chlorinated hydrocarbon pesticides determined. The levels of these compounds in the crab flesh were all below the detection limits of the analytical procedures used (See Table 11). The FDA limit for PCBs in fish flesh (5 mg/kg wet weight of edible portion) is from 10 to 25 times higher than the levels found in edible flesh of the rock crabs collected from the New York Bight mud dump site. The levels found in these crabs (203 to 548 μg/kg) are considerably less than those reported by Young _et_ _al_.[39] for edible flesh of striped bass (1 to 6.7 mg/kg PCBs) from the New York Bight.

The concentrations of mercury, selected chlorinated hydrocarbon pesticides, and PCBs in seven fish consisting of two bottom-dwelling species (whiting - _Mercluccius_ _bilinearis_ and ling - _Urophycis_ _chuss_) from the mud dump site are presented in Table 12. The concentrations of mercury in these fish were

Table 12

Heavy Metal, Chlorinated Hydrocarbon Pesticide and PCB Content in Fish from New York Bight Mud Dump Site*

Fish Type** / Parameter	Whiting (*Merluccius bilinearis*)						Ling (*Urophycis chuss*)				
	A		B	C	D	E	A		B		C
	$\bar{X}$	SD					$\bar{X}$	SD	$\bar{X}$	SD	
Heavy Metals mg/kg (ppm)											
Mercury	< 0.02	-	< 0.02	< 0.02	< 0.02	< 0.02	<0.02	-	<0.02	-	< 0.02
Cadmium	0.01	-	0.01	< 0.01	< 0.01	< 0.01	0.02	-	0.03	-	< 0.01
Chlorinated Hydrocarbon Pesticides µg/kg (ppb)											
Lindane	1.6	0.28	< 0.5	< 0.5	< 0.5	< 0.5	< 0.5	-	< 0.5	-	< 0.5
Heptachlor	5.3	2.68	< 1	1.6	< 1	< 1	< 1	-	< 1	-	< 1
Aldrin	4.3	1.2	17.3	< 1	< 1	< 1	2.9	0.78	2.3	0	< 1
Dieldrin	< 2.5	-	< 2.5	< 2.5	< 2.5	< 2.5	< 2.5	-	< 2.5	-	< 2.5
Endrin	< 3	-	< 3	< 3	< 3	< 3	< 3	-	< 3	-	< 3
Endosulfan I	< 3	-	< 3	< 3	< 3	< 3	< 3	-	< 3	-	< 3
Endosulfan II	< 3	-	< 3	< 3	< 3	< 3	< 3	-	< 3	-	< 3
Methoxychlor	<20	-	<20	<20	<20	<20	<20	-	<20	-	<20
Mirex	<10	-	<10	<10	<10	<10	<10	-	<10	-	<10
Chlordane	<20	-	<20	<20	<20	<20	<20	-	<20	-	<20
op'DDT	< 8	-	< 8	< 8	< 8	< 8	< 8	-	< 8	-	< 8
op'DDD	< 5	-	< 5	< 5	< 5	< 5	< 5	-	< 5	-	< 5
op'DDE	< 4	-	< 4	< 4	< 4	< 4	< 4	-	< 4	-	< 4
pp'DDT	6.5	0.28	42.2	13.1	13.2	15.3	14.6	2.5	< 8	-	< 8
pp'DDD	9.2	1.83	6.6	5.8	6.8	8.4	12.5	5.1	< 4	-	< 4
pp'DDE	6.4	1.91	36.3	6.9	21	26.7	8	0.78	7.1	0.78	< 4
Total DDT***	22.1	-	85.1	25.8	41	50.4	35.1	3.4	7.1	0.78	-
Total PCBs µg/kg (ppb)	69	-	362	333	161	162	194	70.7	66	4.95	26
Wet Weights (g)											
Total	262	-	627.9	418.2	470.5	191.6	175	-	411	-	257.5
Edible Portion	63	-	194.5	115.5	134.2	62.4	46	-	133	-	60.2

*Samples collected April 26 and May 4, 1977.

All concentrations based on wet weight of edible portion of organism.

Mean ($\bar{X}$) and standard deviation (SD) based on duplicate analyses of one sample. Mercury and cadmium concentrations based on single analysis as are values which appear without a standard deviation column.

**Letter designations (A-E) refer to individual organisms.

***Exclusive of "less than" values.

Dash (-) indicates not appropriate.

all less than 0.02 mg/kg wet weight of edible portion (detection limit). These levels are well below the FDA limits for mercury in aquatic organisms of 0.5 mg/kg (Table 10). Concentrations of heptachlor, aldrin, dieldrin, and endrin in the edible fish flesh were all less than 0.06 of the FDA guideline concentrations for these compounds in aquatic organisms (Table 10). Lindane and methoxychlor concentrations were less than 0.1 of the FDA guideline concentrations for aquatic organisms. Total DDT compounds and PCBs in the fish flesh analyzed were also considerably below the FDA guideline concentrations for aquatic organisms (Table 10). The other chlorinated hydrocarbon pesticides determined have no FDA limits; levels of these compounds were all below analytical detection limits on the order of several µg/kg wet weight of edible portions (Table 12).

Table 13 presents the concentrations of mercury, cadmium, selected chlorinated hydrocarbon pesticides and PCBs in the flesh of nine fish consisting of three species of bottom-dwelling fish collected from the reference area in the Bight. Mercury concentrations were generally equal to or below the 0.02 mg/kg wet weight of the edible portion detection limit, which is considerably below the FDA guideline concentration. Cadmium concentrations were about the same as or greater than concentrations in fish from the mud dump site.

In some cases, there appeared to have been fairly wide variability in concentrations of some of the chlorinated hydrocarbon pesticides among fish of the same species. Concentrations of the selected chlorinated hydrocarbon pesticides in the reference site fish were, in general, somewhat greater than those found in mud dump site fish. However, concentrations of all constituents measured were considerably below FDA guideline levels for the selected chlorinated hydrocarbon pesticides and PCBs in aquatic organisms. Several of the chlorinated hydrocarbon pesticides and PCB concentrations

Table 13

Heavy Metal and Chlorinated Hydrocarbon Pesticide Content of Fish* from Reference Site

Fish Type	Whiting (Merluccius bilinearis)					Flounder (Pseudopleuronectes americanus)			Ling (Urophycis chuss)			
	A		B		C	A	B	C	A		B	C
Parameter	X̄	SD	X̄	SD					X̄	SD		
Heavy Metals mg/kg (ppm)												
Mercury	< 0.02	-	< 0.02	-	< 0.02	< 0.02	< 0.02	< 0.02	0.02	-	0.03	< 0.02
Cadmium	0.10	-	< 0.01	-	0.01	< 0.01	0.04	0.04	0.02	-	<0.01	0.01
Chlorinated Hydrocarbon Pesticides µg/kg (ppb)												
Lindane	< 0.5	-	< 0.5	-	< 0.5	< 0.5	< 0.5	< 0.5	< 0.5	-	< 0.5	< 0.5
Heptachlor	9.2	1.1	25	17	18.2	19.9	94.7	9.6	< 1	-	2.9	3.9
Aldrin	10.6	0.5	15.8	10.5	10.2	19.6	36.5	6.2	4.6	0.21	3.8	6.2
Dieldrin	< 2.5	-	< 2.5	-	< 2.5	< 2.5	< 2.5	< 2.5	< 2.5	-	< 2.5	< 2.5
Endrin	< 3	-	< 3	-	< 3	< 3	< 3	< 3	< 3	-	< 3	< 3
Endosulfan I	< 3	-	< 3	-	< 3	< 3	< 3	< 3	< 3	-	< 3	< 3
Endosulfan II	< 3	-	< 3	-	< 3	< 3	< 3	< 3	< 3	-	< 3	< 3
Methoxychlor	<20	-	<20	-	<20	<20	<20	<20	<20	-	<20	<20
Mirex	<10	-	<10.7	-	<10	<10	<10	<10	<10	-	<10	<10
Chlordane	<20	-	<20	-	<20	<20	<20	<20	<20	-	<20	<20
op'DDT	< 8	-	< 8	-	< 8	< 8	< 8	< 8	< 8	-	< 8	< 8
op'DDD	< 5	-	< 5	-	< 5	< 5	< 5	< 5	< 5	-	< 5	< 5
op'DDE	< 4	-	< 4	-	< 4	< 4	< 4	< 4	< 4	-	< 4	< 4
pp'DDT	26.2	0.8	50	1.6	44.6	15.1	20.4	17.3	< 8	-	10.6	13
pp'DDD	21.2	0.9	44	7.4	53	8.3	7.7	5.9	< 4	-	4.6	8.9
pp'DDE	16	1.4	48	1.4	44.4	8.7	19.2	11.9	< 4.4	-	4.5	14.2
Total DDT ***	63.4	-	142	-	142	32.1	47.3	35.1	-	-	19.7	36.1
Total PCBs µg/kg (ppb)	158	-	388	205	578	130	237	92	191	0.7	52	98
Wet Weights (g)												
Total	501	-	258	-	295	218.2	216.8	192.8	413	-	345.9	387.8
Edible Portion	161.5	-	61.4	-	78.6	57.3	56.3	47.5	126	-	90.5	92.7

*A, B and C are individual organisms.

Concentrations based on wet weights of edible portion.

**Means and standard deviation for pesticides in whiting and in ling (A) are based on duplicate analyses of one sample. Dash in standard deviation columns indicates not appropriate. All other values resulted from single analysis.

***Exclusive of "less than" values.

<Indicates below detection limit.

appeared to have been higher in the whiting than in the ling from the same locations. Since a limited number of samples were analyzed, however, it is difficult to determine if this difference is real.

Mean and standard deviation data presented in Table 14, as well as in Tables 12 and 13, give an indication of the reproducibility of the analytical procedures used for the organism flesh. In general, the duplicate analyses of a sample of organism flesh showed good reproducibility.

The levels of chlorinated hydrocarbon pesticides, PCBs and mercury in the edible portions of crab and fish collected at the New York Bight mud dump site and reference area were well below the FDA limits for these compounds in aquatic organisms. It, therefore, appears that there has not been significant accumulation of these compounds in those organisms analyzed. This may be because the particular mud dump site organisms collected had not been present for an adequate period of time at the mud dump site to accumulate sufficient amounts of contaminants to cause elevated concentrations. Alternatively, this may have resulted from the contaminants not being available to accumulate within the organisms to a sufficient degree to cause elevated concentrations within edible portions of the flesh.

From an overall point of view, it appears that dredged material disposal at the mud dump site does not have an adverse impact on organisms present at or near the disposal area. However, additional monitoring should be conducted to ascertain whether the results obtained in this study are representative of the general conditions that prevail at the mud dump site throughout the annual cycle.

It should be noted that in addition to being exposed to elevated concentrations of PCBs in the sediments at or near the mud dump site, organisms were also exposed to elevated concentrations of PCBs in the water column. As reported in

Table 14

Replicate Analyses of Edible Portions of Rock Crab (*Cancer irroratus*) Flesh

Parameter	A*	B*	Mean**	Standard Deviation
Chlorinated Hydrocarbon Pesticides µg/kg (ppb)				
Lindane	2.0	2.1	2.1	0.07
Heptachlor	1.6	1.9	1.8	0.21
Aldrin	2.8	1.2	2.0	1.10
Dieldrin	< 2.5	< 2.5	< 2.5	-
Endrin	< 3.0	< 3.0	< 3.0	-
Endosulfan I	< 3.0	< 3.0	< 3.0	-
Endosulfan II	< 3.0	< 3.0	< 3.0	-
Methoxychlor	<20.0	<20.0	<20.0	-
Mirex	<10.0	<10.0	<10.0	-
Chlordane	<20.0	<20.0	<20.0	-
op'DDT	< 8.0	< 8.0	< 8.0	-
op'DDD	< 5.0	< 5.0	< 5.0	-
op'DDE	< 4.0	< 4.0	< 4.0	-
pp'DDT	15.6	16.9	16.3	0.92
pp'DDD	16.7	18.1	17.4	0.99
pp'DDE	7.3	7.9	7.6	0.42
Total DDT***	39.6	42.9	41.3	2.3
Total PCBs µg/kg (ppb)	539	557	548	12.7
Wet Weights (g)				
Total	41	107	74	-
Edible Portion	8.5	11.6	10.5	-

[*] A and B are replicate sub-samples of a composite of the edible portions of two crabs.

[**] Mean and standard deviation based on analysis of replicate sub-samples A and B. All concentrations based on wet weight of edible portions of organisms.

[***] Exclusive of "less than" values.

Dash (-) indicates not appropriate or not analyzed.

Volume II of this report, from 10 to 20 ng/l total PCBs were found in water samples obtained from this region. It is evident that both the fish and crabs analyzed in this study are exposed to total concentrations of PCBs which, if available, could readily bioconcentrate to excessive concentrations within the organism tissue. It, therefore, must be concluded that substantial parts of the PCBs in both the sediments and water column in the region of the New York Bight mud dump site are, in general, not readily available for bioconcentration. The results of these studies demonstrate the importance of conducting field studies to measure realized bioconcentrations rather than trying to estimate this from laboratory studies or extrapolate it from water concentrations.

PART VIII: DREDGING - DISPOSAL LEGISLATION AND CRITERIA

Presented below is a discussion of the current legislation, including some of the foundation legislation, pertaining to regulation of dredged sediment disposal in open waters. Also discussed are the criteria currently being used by various EPA Regions, as well as the approaches that should be used.

Legislative Background

The authority of the Department of the Army to regulate dredging and dredged material disposal is based on the Rivers and Harbors Act of 1899. Section 10 of this Act specifies that any structures built in navigable waters and any alterations and/or modifications of existing waterways must meet with the approval of the Chief of Engineers and be authorized by the Secretary of War. According to Hollis[40], until recently, the US Army Corps of Engineers has administered this Act based almost exclusively on navigational impairment considerations.

In the late 1960's, the US Army Corps of Engineers Buffalo District, at the request of the Federal Water Quality Administration (FWQA, predecessor to the US EPA) began a study of classical pollutional characteristics of Great Lakes harbor sediments and found many of the sediments to be grossly polluted.[41] This led to the Rivers and Harbors Act of 1970, creating special provisions for dredging/disposal activities in the Great Lakes. It also authorized the Corps of Engineers to initiate a comprehensive evaluation of the environmental impact of dredged material disposal. At about this time, the National Environmental Policy Act of 1969 (NEPA) (PL 91-190) was being formulated. As discussed by Hollis[40] and Bradley[42], this Act required that prior to undertaking any federal action which will significantly affect the quality of the human environment, the agency responsible must prepare a detailed statement to the President's Council on Environmental Quality in which consideration is given to:

- *a*. The environmental impact of the proposed action,
- *b*. Any adverse environmental effects which cannot be avoided should the proposal be implemented,
- *c*. Alternatives to the proposed action,
- *d*. The relationship between short-term use of man's environment and the maintenance and enhancement of long-term productivity, and
- *e*. Any irreversible and irretrievable commitments of resources which would be involved in the proposed action should it be implemented.

The FWQA proposed criteria, known as the "Jensen Criteria," for judging the acceptability of dredged sediment for disposal in US waterways. They were designed to provide interim guidance until final guidelines and criteria could be established. The criteria specified, among other considerations, maximum bulk sediment content of volatile solids, chemical oxygen demand (COD), total kjeldahl nitrogen, oil and grease, mercury, lead and zinc. They stated that if one of the above parameters exceeded the criterion, the sediment would be considered polluted and unacceptable for open water disposal.[20] It was also recommended in these criteria that a second group of parameters, including phosphorus, TOC, immediate oxygen demand, sulfides, Fe, Cd, Cu, Cr, As, Ni, pesticides, bioassay, and settleability be considered as well. Although this portion of the criteria was supposedly designed to predict long-term and short-term effects on receiving water, according to Lee[41], no attempt was made by the FWQA to determine what the potentially available fractions of those sediment-associated contaminants were. In January 1971, the FWQA had the Corps of Engineers issue Circular 1165-2-97 which stated that the "Jensen Criteria" should be applicable to sediments dredged from all US waters. As a result of this, many US waterway sediments were classified as polluted and other disposal methods had to be used.[41]

Section 404 of Public Law 92-500, the 1972 amendments to the Federal Water Pollution Control Act, indicated that the Secretary of the Army may issue permits for dredged material disposal through application of guidelines developed by the EPA Regional Administrator and Secretary of the Army through the District Engineers on the basis of criteria comparable to those for ocean discharge. As stated in Section 403 of PL 92-500, these shall give consideration to:

- *a*. the effect of disposal of pollutants on human health or welfare, including but not limited to plankton, fish, shellfish, wildlife, shorelines, and beaches;
- *b*. the effect of disposal of pollutants on marine life including the transfer, concentration, and dispersal of pollutants or their byproducts through biological, physical, and chemical processes; changes in marine ecosystem diversity, productivity, and stability; and species and community population changes;
- *c*. the effect of disposal of pollutants on aesthetic, recreation, and economic values;
- *d*. the persistence and permanence of the effects of disposal of pollutants;
- *e*. the effect of the disposal at varying rates, of particular volumes and concentrations of pollutants;
- *f*. other possible locations and methods of disposal or recycling of pollutants including land-based alternatives; and
- *g*. the effect on alternate uses of the oceans, such as mineral exploitation and scientific study.

Where these guidelines would prohibit the specification of a disposal site, permits may be issued, based on economic impact of the site in terms of navigation and anchorage. Further, the US EPA Administrator may, after consulting with the Secretary of the Army, deny, revoke, or restrict a disposal permit if he determines that there will be adverse environmental effect arising from it.

To comply with Section 404 of PL 92-500, the US EPA Administrator on May 6, 1975 proposed guidelines for evaluation of proposed discharge of dredged or fill material. The most recent version of these guidelines ("interim final" guidelines) appears in the September 5, 1975 Federal Register[43] and are for use for navigable fresh and estuarine waters. The regulations and criteria specified are discussed in a forthcoming section.

The Marine Protection, Research, and Sanctuaries Act (PL 92-532) was passed on October 23, 1972. Section 103 of this Act recommended that ocean dumping of polluted dredged sediments be phased out as soon as possible. Highly polluted sediments should still be dredged only when necessary to navigation, and even then, benefits should be weighed against damages. It required that the EPA Administrator and Secretary of the Army establish a permit program to eliminate dumping of dredged materials into the oceans that may result in "unreasonable" degradation or endangerment of the marine environment or human health. Permits for disposal would be issued, according to the statute, on a case-by-case basis where there has been evidence provided showing that there will be no degradation of water quality as outlined in the statute. The January 11, 1977 Federal Register[44] presents the final regulations and criteria for ocean dumping of dredged sediments. These criteria are discussed below.

Current Criteria - Guidance for Assessing Suitability of Dredged Sediment for Open Water Disposal

The "interim final" guidelines set forth by the US EPA[43] in the September 5, 1975 Federal Register are applicable to all dredged and fill material discharged in navigable waters of the US within the baseline from which the "territorial seas" begin. From this baseline seaward, the guidelines published by the US EPA[44] in the January 11, 1977 Federal Register are applicable to dredged material disposal. Both sets of guidelines indicate that the need for each proposed open water disposal must be carefully reviewed and alternate methods of disposal be assessed. Further, the District Engineer, with input from the US EPA Regional Administrator, may at any phase of testing require additional testing of the dredged sediment under investigation. Both regulations specify three conditions under which dredged sediment is considered environmentally safe without further testing. Briefly, these include: 1) if material is predominantly sand, gravel, or rock; 2) if material is for beach nourishment and is composed of sand, gravel, rock, or a particle size compatible with beach; and 3) if material is substantially the same as dump site sediment and is not from an area historically or recently polluted; and for fresh and estuarine waters, if it is reasonably certain that currents will not transport the deposited material that may cause environmental damage, from the disposal site. If the sediments proposed for dredging do not meet any one of these specifications, the following procedures must be used.

Fresh and estuarine waters

An elutriate test may be run by vigorously shaking a one-part dredged sediment plus four parts dredging site water (vol:vol) for 30 minutes, allowing the slurry to settle for one hour, and filtering the supernatant through a 0.45 μ pore size filter. Constituents deemed critical by the District Engineer will be determined in the elutriate (filtered supernatant from elutriate test). Elutriate concentrations are to be compared to disposal site data and information regarding rate and type of discharge and hydrodynamic regime at the disposal site. The District Engineer may require an "appropriate" benthic bioassay which "will be of value in assessing ecological effects . . ." A District Engineer may also require analysis of the bulk characteristics of the dredged and disposal site sediments for comparative purposes and investigations of the biological community structures at both dredging and disposal sites. A list of dumping conditions that should be avoided are also included. Among others, these include: activities that disrupt chemical and biological integrity of the aquatic ecosystem; activities that significantly disrupt the food chain; and activities in the area of shellfish production, fish spawning, and nursery areas. Further, release of nutrients should be minimized to prevent eutrophication and impairment of recreational uses. The US EPA guidelines[43] also discuss designation of a mixing zone which is defined as the smallest practicable zone within a specified disposal site, within which the desired concentrations of constituents must be achieved after disposal. Parameters that must be considered in formulating such a zone are also presented. In addition to chemical-biological evaluation, evaluation of physical effects of open water disposal must be considered.

In accordance with the September 5, 1975 <u>Federal Register</u>[43], the Corps of Engineers formulated a procedures manual[45] to provide guidance for fulfilling the requirements outlined by the US EPA[43] and hence PL 92-500, Section 404. Specific detailed procedures were presented for performing the elutriate test and collecting necessary samples for the test, formulating mixing zones, and performing bioassays and total sediment analyses.

The elutriate test procedure had previously been detailed by Keeley and Engler[46] with respect to the ocean dumping criteria. This test was recommended for use in fresh and estuarine waters in the draft implementation guidelines for Section 404 of PL 92-500.[47] Lee[48] and Lee *et al.*[1] reviewed and commented on these draft guidelines and noted that the major problem with the elutriate test as described by Keeley and Engler[46] was that the oxygen conditions during the test were not controlled. The elutriate test results could, therefore, for certain types of sediments, be unreproducible as the oxygen status would be dependent on the shaking technique and vigor used. Minor changes in the shaking procedure could, then, cause significant changes in the amount of the release of contaminants. Lee et al.[1], therefore, proposed that a 30-minute compressed air mixing period be used. Even with these comments and the results of the Lee et al.[1] studies, the US EPA failed to correct the procedural deficiency in the elutriate test in their September 5, 1975 "interim final" guidelines.[43] As pointed out in another section of this report, this situation has resulted in the elutriate test being conducted improperly with many hundreds of thousands of dollars being wasted. As a result, there are many within the pollution control field who have questions about the reliability of the elutriate test as a tool to predict contaminant release associated with open water dredged sediment disposal. As noted below, a similar situation developed with respect to the ocean dumping criteria.

The elutriate test outlined by the US CE[45] is essentially the same as that presented by US EPA[43] and Keeley and Engler[46], except that dredging site rather than disposal site water is prescribed for elutriate testing and an option is given to allow compressed air mixing rather than mechanical shaking. Compressed air mixing is allowed if the mechanical mixing causes anoxic conditions and the investigator feels that anoxic conditions will not be encountered at the disposal site. It is also allowed under conditions where reproducibility of elutriate test results is a problem. While these modifications tended to correct the procedural deficiency of the elutriate test, conducting the test under oxic conditions should have been made mandatory since only under oxic conditions can proper interpretation of the potential toxicity be made.

Keeley and Engler[46] discussed the use of a 1.5 factor as described by the US EPA[49] to evaluate whether or not an elutriate test indicated a "polluted" sediment. If the soluble constituents in the elutriate exceeded those of the disposal site water by 50 percent (1.5 factor), then the sediment was judged not fit for open water disposal. According to the US CE[45], the results of the elutriate analyses approximate concentrations of dissolved constituents at the moment of discharge, and the elutriate constituents' concentrations should be interpreted based on applicable water quality standards at the edge of a mixing zone. The "water quality standards," as presented by US CE[45] are discussed only in terms of toxicity effects; no recommendations or reference is given for biostimulatory, i.e., nutrient, criteria.

The use of a mixing zone concept provides a means to account for rapid dilution that often occurs at open water disposal sites.[45] The approach described by US CE[45] was to calculate the volume of disposal site water necessary to dilute a proposed dredged material discharge to acceptable levels by using elutriate and disposal site water constituent concentrations and applicable water quality standards. Various mixing zone shapes were presented for different types of disposal operations with the mathematical formulae for determining the volume of each. By substituting known information and the desired dilution volume into the volume equation, the dimensions of the mixing zone can be determined.

There are a number of deficiencies and major assumptions used in this concept. First, it appears to have been designed to model the transport of the cohesive mass of dumped or discharged material. As discussed previously, this mass is generally not the major concern with respect to the effects of disposal on water column water quality. Rather, it is chemical release associated with the turbid plume of finer suspended materials that arises from the disposal which is of concern. This portion is not considered in this model. Second, it is assumed that the chemical contaminant of interest behaves as a conservative element, that the only process occurring is dilution. There are a number of processes which cause the rapid removal of phosphate and other contaminants from solution, one of the most important of which is the oxygenation of ferrous iron. For example, upon disposal of anoxic sediments into oxic site water, phosphorus released during dredged sediment slurry transport could be rapidly removed during the oxidation of sediment-associated ferrous iron to ferric iron. Further, there is a possibility of continued release from or uptake by the finely divided particulates in the turbidity plume formed on disposal which is also not considered in the model. It is possible that these reactions could mask effects of or at least occur concurrently with dilution. Third, for some contaminants, such as phosphate, there are no numerical criteria with which to compare the elutriate concentration in determining the dilution volume.

Marine waters

Sediments proposed for disposal in the territorial seas, which do not meet the substrate requirements for disposal without further testing, must be subjected to elutriation.[44] The elutriate test supernatant is poured off and a portion reserved for testing as the suspended particulate phase; a portion of the supernatant is filtered through a 0.45 μ pore size filter and assayed as the liquid phase. The settled sediment from the elutriate test is the solid phase. The dredged material needing further testing will be considered environmentally safe for disposal only when 1) the major constituents of the liquid phase are in compliance with "applicable marine water quality criteria" (defined as US EPA Quality Criteria for Water - 1976) after allowance for initial mixing, or bioassays indicate that it will not exceed the "limiting permissible concentration," 2) bioassays on the suspended particulate and solid phase indicate that limiting permissible concentrations will not be exceeded, and 3) the sediments are in compliance with Section 227.6 of the January 11, 1977 Federal Register[44] dealing with contaminants allowed in only trace amounts.

In accordance with US EPA[44], the Corps of Engineers and US EPA developed an implementation manual for executing these evaluation procedures[36], providing specific procedures for conducting bioassays indicated in the January 11, 1977 Federal Register. A critique of these procedures is presented below.

US EPA/CE Bioassay Procedures Manual Critique. In response to a request from the New York District of the Corps of Engineers, G.F. Lee and R.A. Jones prepared a critique of the US EPA/CE July 1977 "Ecological Evaluation of Proposed Discharge of Dredged Material into Ocean Waters" implementaion manual, for the January 11, 1977 Federal Register. This critique was released to the public in December 1977.[50] The key points of this critique are presented below.

The Bioassay/Bioaccumulation Subcommittee of the EPA/CE Technical Committee on Criteria for Dredged and Fill Material developed the bioassay manual released in July 1977.[36] This manual provides a description of the bioassay procedures that the Corps of Engineers and the US EPA suggest should be used to evaluate the significance of dredged sediment-associated contaminants during ocean dumping of dredged sediments. However, a review of this situation shows that the January 11, 1977 Federal Register presents an overly complex approach for determining the significance to aquatic organisms of chemical contaminants associated with ocean dumping of dredged sediments. As discussed below, a much simpler, more readily interpretable bioassay procedure can provide essentially the same degree of environmental protection at a considerably lower cost.

a. Procedures. The January 1977 Federal Register contained several approaches to assessing the significance of chemical contaminants present in dredged sediments which are not technically valid. These inappropriate approaches have a profound impact on the development of the bioassay procedures designed to implement the provisions of the January 11, 1977 Federal Register. The provision which, as has been interpreted by the EPA and Corps of Engineers Subcommittee, requires separate bioassay tests for liquid phase, suspended sediment phase, and solid phase, using different organisms for each test, is unnecessary. A single screening test procedure similar to that developed in this study would serve essentially the same purpose and would promote, rather than discourage, the use of bioassay procedures.

The primary difficulty with the multiple test approach is that it greatly increases the cost of the bioassays without providing the District Engineer and Regional Administrator with any significant increase in information that could aid in determining whether or not the disposal of a particular sediment should take place at a given dump site. It is estimated that the cost of a complete EPA/CE bioassay series would be on the order of $1,500 to $2,000 per sample. At the present time, and for many years in the future, it is likely that the decision to dump or not to dump sediment at a particular site will be determined by many factors other than the results of a battery of bioassay tests conducted on a proposed dredged sediment. It is difficult to envision a situation where the District Engineer would be aided significantly by knowing that, for example, two crustaceans (such as grass shrimp), all the algae and clams, and no polychaetes died in a particular set of tests compared to knowing the results of a single test conducted on the grass shrimp. It is strongly recommended that the various Corps districts immediately initiate the steps necessary to have the provisions of the January 11, 1977 Federal Register modified so that a simple test with an appropriately sensitive organism (e.g., *P. pugio* and *D. magna*) could be used instead of the multiple tests currently required. In those few particular instances where, based on a known organism(s) at the disposal area, the single test approach does not appear to provide for adequate environmental protection and the bioassay results play a major role in determining whether or not disposal of a particular sediment may take place at a particular location, then bioassay tests specific to the characteristics of the sediment and the disposal area may be used.

The discarding of the results of a bioassay if more than 10 percent of the controls die, as is recommended in the bioassay manual, is excessively restrictive. While for certain test organisms there is no problem meeting this requirement, for others, especially some of those for which there is limited experience in laboratory handling, this requirement may result in having to have many tests rerun because one of the ten organisms died in the controls during the test period. It is recommended that 20 percent be used as the upper limit on control mortality.

b. Test organism. The manual states that bioassay test organisms should be collected from a reference area. While this is desirable, it greatly increases the cost and, in many instances, key test organisms may not be available in the general area. This suggested requirement should be changed so that it reflects the fact that it may be equally desirable to use a standard test organism with a known sensitivity, for bioassay testing. It should be noted that with respect to the water pollution control field and, for that matter, environmental toxicology in general, the approach of using a standard test organism is the one normally used to check out the toxicity of a particular chemical or situation rather than attempting to use the indigenous species of the region.

Little useful information can be gained from the use of algae as a bioassay organism in the evaluation of the significance of chemicals associated with dredged sediments. In general, algal bioassays are not appropriate for assessing the toxicity of dredged sediments. Further, algal bioassays as proposed in the EPA/Corps of Engineers manual are of limited value in assessing algal stimulation. They are of potential value in a modified procedure directed toward evaluating a long-term release of nutrients for phytoplankton growth in the general area where disposal has taken place. It is recommended that the various Corps of Engineers districts not use algal bioassay procedures for assessing the significance of chemical contaminants associated with dredged sediments to be disposed of in the ocean.

c. Elutriate test conditions. The July 1977 bioassay manual[36] fails to specify the redox conditions that must be maintained during the mixing period of the elutriate test. Markedly different results will be obtained from the bioassay procedures if the elutriate test system is oxic versus anoxic.[1] In order to properly simulate the situation associated with dredged material disposal, where release of chemical contaminants from the dredged sediments would be of concern to aquatic organisms, oxic conditions must be maintained during the elutriate test. Failure to adopt this approach could result in highly variable results which would add to the confusion associated with the use of elutriate tests which exists today in the water pollution control field.

Sediments which are to be hydraulically dredged should be mixed 1:4 with dredging site water. However, sediments that are mechanically dredged have a markedly different consistency than the 1:4 ratio normally said to be encountered in hydraulic dredging. A different ratio should be specified, a different mixing procedure should be used, and disposal site water should be used if the elutriate test procedure is to simulate mechanically dredged, barge disposal operations.

d. Applicable water quality criteria. Mention is made of the use of the US EPA July 1976 Quality Criteria for Water[8] as a basis upon which to judge the significance of chemical contaminants released to the water column during dredged material disposal operations after allowance is made for initial mixing (four hours) as specified in the January 11, 1977 Federal Register. This approach is not technically valid. The US EPA Quality Criteria for Water, released in July 1976, are based on chronic exposure or critical life stage exposure. Almost without exception, this involves months to years of exposure. Aquatic organisms can be exposed to much higher concentrations for short periods of time than they may tolerate for chronic exposure situations.

The US EPA Quality Criteria for Water have little or no applicability to determining the significance of chemical contaminants associated with ocean dumped dredged sediments and should not be used for this purpose. What should be used are quality criteria for water based on the time of exposure-concentration of available forms relationships that exist for each contaminant of interest, at the disposal area of interest, for each particular type of disposal operation. Much larger concentrations of chemicals than would be allowed based on chronic exposure concentrations, can be released to the water column without causing an adverse effect on water quality in the disposal area. A more realistic assessment can be made using time-available form concentration relationships. This situation has been discussed in detail by Lee *et al*.[1,26]

e. *Interpretation of bioassay results*. The approach that is advocated in the manual for interpreting solid phase bioassay results is one in which any statistically significant increase in the number of bioassay organism deaths in the test system over those in the control systems, is to be interpreted as a potentially adverse situation. While an attempt is made several places in the manual to qualify this statement, it will undoubtedly lead to governmental agencies' adopting an overly conservative approach for interpreting bioassay results.

The bioassay manual states that any adverse effect of sediments to be dredged on bioassay organisms should be considered to be due to "trace contaminants" because of legal constraints and that there is no method to distinguish among those contaminants unless bioaccumulation tests are conducted. It is not technically correct to assume that any toxicity found in bioassay tests is the result of the presence of "trace contaminants." Further, the concept that sediments which show toxicity in the bioassay tests should not be ocean dumped is inappropriate. The problem with the "trace contaminants" situation is that the US entered into an international treaty regarding ocean dumping of these certain materials. Because of this treaty, these materials require special attention. Second, it is not necessary to arbitrarily assume that any toxicity is due to "trace contaminants" as proposed in this manual. The significance of these materials can likely be evaluated by a standard additions technique. The method of standard additions is one in which those contaminants which are of concern under the ocean dumping treaty could be added individually in small amounts to see if there is a proportionate increase in the effect on bioassay organisms. If there is not a proportionate effect, then it is reasonably certain that the adverse effect is not due to the trace contaminant tested, but to some other material.

Simply finding toxicity, as implied in several locations in the manual, and assuming that disposal of those sediments tested would result in significant adverse effects on water quality is not a valid approach. This is due to the fact that the bioassay tests exaggerate the effects on organisms of chemicals released during a dredged material disposal operation. Toxicity under these bioassay conditions does not imply that there will be toxicity in the field associated with the actual disposal operations.

The procedures presented in the January 11, 1977 Federal Register for determining the size of the mixing zone do not appear to be very realistic. In this study, several attempts have been made to apply the Corps of Engineers' estimated mixing zone computation approach. Efforts were terminated, however, due to problems inherent in the models. It is recommended that the Corps of Engineers districts conduct studies to determine what the actual extent of the mixing zone is for various types of dredged material disposal operations that occur in ocean waters.

f. The approach advocated in the July 1977 bioassay manual of using any increase in the concentration of chemicals in any organism as indicating a potentially adverse situation is inappropriate. This approach is far too strict and should not be used. Bioaccumulation must be examined in terms of the concentrations in the higher trophic level organisms in the region of concern. It must be determined whether or not these concentrations impair the organisms or the beneficial use of them. One cannot utilize the fact that there were increased concentrations in lower trophic level organisms as a basis for judging the significance of bioaccumulation in higher trophic level organisms, as implied in the manual.

The bioassay manual states that mobile epifauna such as crab, lobsters, fish, etc. should not be used in bioaccumulation studies; instead, the recommendation is made to use polychaetes and other less mobile organisms. Information on bioaccumulation in polychaetes is of essentially no value since no one is in a position to interpret it in a meaningful way. What is of importance is the accumulation of hazardous chemicals, in general, within the so-called mobile epifauna such as fish, crabs, etc., i.e., the commercially important species. These data can be interpreted in terms of the FDA limits; data on contaminant concentrations in polychaetes are uninterpretable. The fact that an organism is mobile should not preclude its use in bioaccumulation studies.

What should be considered is whether or not the organisms of the region of concern have excessive concentrations of selected contaminants. If a properly conducted survey shows that organisms of concern do not have excessive concentrations, then all sources of that particular contaminant can be considered insignificant, including the dredged sediment. If an organism of the area has a concentration of contaminant in excess of that known to be harmful to man or exceeding an FDA limit, then further study needs to be done to determine the source of this excessive concentration with particular reference to the dredged sediments.

The statement is made in the manual that mollusks are very useful for bioaccumulation studies. While mollusks readily bioaccumulate, the interpretation of the bioaccumulation results is very difficult. Therefore, they are not, in general, good bioassay organisms for determining bioaccumulation potential or amounts of available forms of contaminants associated with dredged material disposal operations which could have an adverse effect on environmental quality. Further, because of the fact that mollusks can go into a phase of reduced or minimal activity for periods of days to weeks, in response to adverse conditions, their bioassay results may, under highly toxic conditions, yield a false sense of environmental protection. Small fish should be used if bioaccumulation tests are to be conducted in the laboratory. A far better approach is one of analyzing organisms from a disposal area where dredged material similar to that proposed for dumping has been previously dumped, as recommended in the July 1977 implementation manual.

In addition to the numerous problems in details of the analytical procedures prescribed, there are also significant problems in the statistical approaches prescribed in this manual. The primary problem with these statistical techniques is that according to Odell[51], they are inappropriate for the situations to which they are applied. A report to the New York District of the Corps of Engineers prepared by Lee *et al.*[52] discusses these problems in detail.

g. Overall conclusions. From an overall point of view, it has been found that the approach advocated in the US EPA/CE July 1977 bioassay manual is overly complex and will, therefore, result in a very significant increase in the cost of bioassays compared to what is needed. Further, the approach advocated with respect to interpretation of data in some instances is not technically valid. It is recommended that the Corps of Engineers adopt a screening-type bioassay procedure similar to the one developed in this study. and discussed in this part of this report.

Regional Dredged Material Disposal Criteria

It is of interest to compare federal legislation and guidelines with the various EPA region criteria presently in use for determining the suitability of a sediment for open water disposal. Each of the ten US EPA regions was contacted and asked specific questions concerning the type of criteria used and their numerical values for criteria where applicable.

Even with the federal guidelines and numerous studies showing the lack of validity of the bulk sediment criteria for assessing the effect of dredged material disposal on water quality, six of the ten EPA regions are still relying on bulk chemical analyses of dredged sediment to determine suitability of proposed dredged sediment for open water disposal. Two of those regions (IX - San Francisco and VII - Kansas City) did not specify how they used bulk criteria, or what the parameters measured and limiting values were. The other four provided numerical values for criteria used; these are presented in Table 15 along with the "Jensen Criteria" and those used by the Ontario Ministry of the Environment. Comparison of the various bulk criteria shows that the non-acceptable levels for several parameters are quite different. For example, for total phosphorus, there was no numerical level specified in the Jensen Critiera[20], nor is it considered in the Region I (Boston) criteria[53]. Total P sediment concentration of 650 mg P/kg dry weight is considered by Region V (Chicago) to indicate that a sediment is "heavily polluted" with phosphorus.[54] The Region V criteria were based in some undefined way on analysis of 260 sediment samples from 34 Great Lakes' harbors during 1974-75, but is is interesting to note that the average crustal abundance value for total P (655 mg P/kg) exceeds the Region V "heavily polluted" level. Further, as discussed by Jones[3] total P concentration in open water Great Lakes sediments are from about 1.5 to 5 times the level listed as "heavily polluted" in the Region V criteria. The overall sediment classification of a Region V sediment sample is based on the most prominent classification of the individual parameters. Region V also uses the elutriate test, with the "shaking" technique, in addition to, and to modify the results of, the bulk analyses. Further, the source of contamination, and characteristics such as odor, color, texture, detritus, and oiliness are used as indicators of suitability for open water disposal

Table 15

Bulk Sediment Criteria in Use

Parameter (mg/kg dry wt)	Jensen[a] Criteria	Region V[b]			Region I[c]			Region[d] III	Ontario Ministry of the Environment	Region VI[e] Screening level
		Non-Polluted	Moderately Polluted	Heavily Polluted	I	II	III			
Volatile Solids (%)	6	<5	5-8	>8	-	-	-	11	6	8
COD	50,000	<40,000	40,000-80,000	>80,000	-	-	-	100,000	50,000	50,000
TKN	1,000	<1,000	1,000-2,000	>2,000	-	-	-	2,500	2,000	1,000
Oil & Grease	1,500	<1,000	1,000-2,000	>2,000	<5,000	5,000-10,000	>10,000	5,000	1,500	-
Mercury	1	-	-	≥1	<0.5	0.5-1.5	>1.5	1	0.3	1
Lead	50	<40	40-60	>60	<100	100-200	>200	100	-	50
Zinc	50	<90	90-200	>200	<200	200-400	>400	300	-	75
Ammonia	-	<75	75-200	>200	-	-	-	-	-	-
Arsenic	-	<3	3-8	>8	<10	10-20	>20	-	-	5
Barium	-	<20	20-60	>60	-	-	-	-	-	-
Cadmium	-	-	-	>6	<5	5-10	>10	2	-	2
Chromium	-	<25	25-75	>75	<100	100-300	>300	80	-	100
Copper	-	<25	25-50	>50	<200	200-400	>400	80	-	50
Cyanide		<0.1	0.1-0.25	>0.25	-	-	-	-	-	-
Iron	-	<17,000	17,000-25,000	>25,000	-	-	-	-	-	-
Manganese	-	<300	300-500	>500	-	-	-	-	-	-
Nickel	-	<20	20-50	>50	<50	50-100	>100	-	-	50
Vanadium	-	<75	75-125	>125	-	-	-	-	-	-
PCBs	-	-	-	>10	<0.5	0.5-1	>1	-	-	-
Phosphorus	-	<420	420-650	>650	-	-	-	2,500	1,000	-
TOC	-	-	-	-	-	-	-	20,000	-	-
Reference:	Boyd *et al.*[20]		Tebo[54]			Adams[53]		DeBenedictis[59]	Int'l. Working Group[60]	Elliott[61]

Dash (-) indicates no standard or criterion given.

[a] If one or more of these levels are exceeded, the sediment is considered polluted in all cases and unfit for open water disposal. If these levels are not exceeded, it may be subject to further investigation. Other factors and characteristics are investigated.

[b] From "Guidelines for the Pollutional Classification of Great Lakes Harbor Sediments," US EPA Region V, Chicago, IL, April 1977. Overall classification of a sample based on most predominant classification of the individual parameters. Additional factors such as elutriate test results, source, color, and odor are considered. Results are interpreted in a somewhat subjective manner.

[c] From "Proposed Interim Plan for the Disposal of Dredged Materials in Long Island Sound," CT DEP and NY DEC, April 1977.

Class I:	Considered clean material acceptable for beach nourishment or open water disposal at a site of similar lithologic background.
Class II:	Considered to be of a questionable pollution level and additional information (not specified) must be obtained to determine if the material is to be treated as Class I or Class III material.
Class III:	Considered highly polluted and is generally prohibited from open water disposal. May be dumped if there is a compelling necessity to accomplish the dredging; no upland sites are available even at considerable cost; and special mitigating measures will be taken at the dump site to prevent adverse environmental effects.

[d] From "Guidelines for Determination of the Bulk Chemical Characteristics of Sediments Considered for Open Water Disposal," Recommendations of the Chemical Subcommittee, US EPA Region III, June 1975.

[e] Each sediment is review on a case-by-case basis. Other factors considered in evaluation, in addition to bulk criteria, include: volume of spoil, method and time of year of disposal, disposal site conditions, physical & biological character of material. Bulk sediment values for TOC, TON, chlorinated hydrocarbons and oil & grease need to be formulated.

even though these quantities, according to Lee[55] provide essentially no information that would indicate potential water quality problems resulting from their open water disposal. The benthos are also used in a nebulously-defined way to modify bulk content with the worst case being the lack of benthic organisms in the sediment proposed for disposal. After compilation of this type of information, a somewhat subjective decision is made as to the suitability of the sediment for open water disposal. It should be noted that Region V's continued use of bulk criteria has been a major part of controversy in the Great Lakes area. Recently, the Research Advisory Board of the International Joint Commission for the Great Lakes Water Quality [56] has concluded, based on recent research results, that the use of bulk criteria in the Great Lakes Basin may have caused the expenditure of considerably more funds for dredged material disposal than necessary.

Although Region V uses bulk sediment characteristics to make an immediate decision regarding disposal of dredged material, the addition of a sediment bioassay to the evaluation procedures is planned.[54] The Region V bioassay does not follow the US EPA/CE July 1977 bioassay procedure[36]; it follows the approach developed by Prater and Anderson.[57] This bioassay approach is an extension of the bulk sediment characterization and does not attempt to determine the availability of chemical contaminants being introduced to the disposal site water column and sediments as a result of dredged material disposal.

Region I (Boston) also primarily uses bulk chemical analysis to evaluate a proposed dredged material, with the New York-Connecticut proposed classification system as a guide to demarcation levels[53] (Table 15). Sediments classified in Class I are considered clean and suitable for beach nourishment and open water disposal on similar lithological background. Class II contains questionable pollutant levels and additional information must be obtained. Class III is highly polluted and generally prohibited from open water disposal. No indication was given as to allowable distribution of parameters in classes. Elutriate tests are used in conjunction with confined disposal supernatant overflow in Region I, but not for open water disposal because most of what is disposed of in open water, according to Adams[53], has been mechanically dredged. It should be noted in this regard that the elutriate test in its current form should not be used to assess contaminant release in overflow waters from confined disposal operations.[41, 58]

The June 1975 Region III (Philadelphia) guidelines[59] indicated that although they recognized the many shortcomings of bulk sediment criteria, they use bulk sediment analysis since they claim they cannot deal with these materials in any other manner at this time. Their criteria (Table 15) were based on 1) examining other regions' criteria, 2) conducting a detailed survey of Chesapeake Bay sediments looking at physical/chemical regimes in polluted and non-polluted areas of the Bay, and 3) reviewing historical data on Chesapeake Bay and other estuarine areas. They reportedly set their guidelines at least 50 percent above background levels. DeBenedictis also provided a procedure for an elutriate test which specified mechanical mixing with a five-hour settling period but gave no indication as to how or when it is used. Region III criteria are applied on a case-by-case basis, relying on professional judgment as well as test results.

Region IX (San Francisco) has been using bulk analysis for inland disposal but is planning on switching to bioassay procedures now in use for ocean disposal criteria.[62] No indication was given as to what the numerical values of the bulk criteria were. Region VII (Kansas City, Mo) uses the Jensen Criteria in conjunction with Region V guidelines, state standards for water supply, Quality Criteria for Water of 1976[8], STORET data on stream conditions, Soil Conservation Service input on soil samples near disposal sites, and Food and Drug Administration guidelines on parameters for constituents in food products.[63] No indication was given by Camin[63] as to how these sources of information were used to evaluate the suitability of a sediment for open water disposal.

Region II (New York) uses a grain size analysis prescreening test for proposed dredged sediments as outlined by US CE-New York[64] and Simon[65]. Dredged sediment is deemed acceptable for open water disposal if it falls into one of the following three categories:

a. Material composed predominantly of sand, gravel, rock or other naturally occurring bottom material with particle sizes larger than silt, and found in areas of high current or wave energy;

b. Material for beach nourishment or restoration and composed predominantly of sand, gravel or shell with particle sizes compatible with material on the receiving beaches; or

c. Material substantially the same as the substrate at the proposed disposal site; and source of dredged material is far removed from existing and historical sources of pollution so as to provide reasonable assurance that such material has not been contaminated by such pollution.

If the proposed sediment does not fit into any of these categories, a three-phase bioassay as outlined by US CE-New York[64] and Simon[65], as well as details about the disposal operation and vessel, are required.

Region IV (Atlanta) used to rely on bulk criteria but now uses the elutriate test in conjunction with US CE[45] interpretation guidelines[66]. Region VI (Dallas) reportedly uses US EPA [43,44], as well as US EPA/CE[36] as a basis for judging acceptability of sediments for open water disposal.[61] They have also prescribed bulk sediment screening levels for certain parameters as presented in Table 15. Region VIII (Denver) reported that disposal of chemically contaminated sediments is not a significant problem in the region and they, therefore, have no criteria for disposal.[67] Region X (Seattle) indicated that a case-by-case approach is used, and that there were no specific criteria or standards used.[68]

Great Lakes Legislation Regarding Open Water Dredged Material Disposal

Special attention has been given to dredged material disposal in the US-Canadian Great Lakes. Section 123 of the Rivers and Harbors Act Amendments of 1970 (PL 91-611) created the Great Lakes Confined Disposal Program. This is a ten-year US program in which the Corps has been directed to place dredged materials, designated by the US EPA to be contaminated, behind retaining dikes. The Jensen Criteria and more recently the Region V criteria have been the basis by which these dredged materials have been assessed.

In 1972, the "Agreement Between Canada and the United States of America on Great Lakes Water Quality" was signed by both the US and Canada. In compliance with this agreement, the International Working Group on the Abatement and Control of Pollution from Dredging Activities was established. This group was created to review existing dredging practices, laws, and regulations in order to develop criteria for the characterization and disposal of polluted dredged material.[60] It was concluded by the Working Group that the criteria (bulk sediment) being used by the US EPA and Ontario Ministry of the Environment are limited at best in their application to predicting potential hazard of open water disposal of dredged sediment and that there are no single parameter values that are applicable throughout the Great Lakes. They recommended a site-specific assessment of potential hazard associated with dredged sediment disposal. The basic criteria for dredging activities should be, according to them, the preservation of water uses and the optimization of net socio-economic benefits to society. It is likely that, as a result of recent IJC Research Advisory Board recommendations,[56] dredged material disposal practices in the Great Lakes area will be thoroughly reviewed. Revised dredged material disposal practices will likely result from this review.

It is evident from the above review that there is definitely a tendency to abandon the use of bulk sediment criteria for assessing pollution tendency of contaminants present in dredged sediments. A number of EPA regions continue to use bulk chemical criteria, primarily because of the ease of administration. While there is no doubt that this approach is easy to administer, there is also no doubt that these criteria are not technically valid and may, as in the case of the Great Lakes, result in the expenditure of additional hundreds of millions of dollars for dredged material disposal with little or no improvement in Great Lakes water quality. Further, in some instances, alternate, more expensive methods of dredged material disposal used because of results of bulk sediment analysis, may have resulted in greater water quality deterioration than would have resulted from the less expensive open water dredged material disposal used in the past.

It is also evident from this review of the national and regional criteria, that there are a number of problems with them in addition to bulk sediment analysis. This investigation provides considerable information which can be used as a basis for developing more technically valid dredged material disposal criteria than are in existence today. A discussion of the approaches that should be used are presented in the next section of this report.

Recommended Approach for Development of Dredged Material Disposal Criteria to Assess the Significance of Chemical Contaminants in Dredged Sediments

It is readily apparent from the above discussion of dredged material disposal criteria that there are a number of potentially significant problems with the existing criteria. It is appropriate to use the results of the current investigation, coupled with other information on the environmental behavior and significance of chemical contaminants present in dredged sediments, to recommend an approach that should be used to develop more ecologically valid, cost-effective criteria than are in existence today. This section of this report presents a discussion of the approaches that should be used to develop such dredged material disposal criteria.

Need for criteria

The first subject that should be addressed in a discussion of the approaches that should be used to develop dredged material disposal criteria is that of the need for these criteria. The basic justification for having criteria has been to provide for the protection of the environment. Since dredged sediments are known to contain a wide variety of chemical contaminants, often in seemingly large amounts, and since it has been generally believed by some water pollution regulatory agencies that one of the most potentially ecologically damaging methods of dredged sediment disposal is open water disposal, it has been assumed that there is a need to place some restriction on the dredged sediments disposed of in open water. Further, the effects of dredged sediment-associated contaminants during open water disposal had not been studied and demonstrated to have insignificant adverse environmental impact.

In many parts of the US, various methods of disposal are economically feasible. Since some dredged sediments contain significant amounts of chemical contaminants which could, if they were in available forms, have a detrimental effect on environmental quality at a dredged material disposal site. The US EPA Regional Administrator and the CE District Engineer as well as other pollution control regulatory agencies that have jurisdiction in a particular area, need assistance in evaluating the potential for environmental degradation resulting from various methods of disposal. It is important to emphasize that the disposal site selection process should involve an evaluation of the potential ecological damage and cost effectiveness resulting from various disposal methods.

There has not, however, in the long history of open water disposal been a single documented case where dredged sediment-associated contaminants caused a significant impairment of disposal site water quality. One might get the impression, based on the controversy and legislation that have developed around dredged material disposal practices in the past few years, that major problems have been identified with past practices. However, no real problems due to the presence of chemical contaminants in dredged sediments have yet been identified. There have been problems associated with dredged sediment disposal as related to burial of various aquatic organism habitats. These problems are not, however, related to the chemical contaminants present in dredged sediments. It appears that many regulations have been developed to alleviate an alleged problem which has, as yet, not been found to exist. While this approach would be appropriate as a precautionary measure for new situations, the use of this approach is questionable for a situation such as open water dredged material disposal where the practice has been carried out for many years without any evidence of environmental degradation due to chemical contaminants. It is obvious that if there was large scale, significant environmental damage due to chemical contaminants present in dredged sediments that these problems would have been identified many years ago. It must be concluded that even though there is a potential for significant environmental degradation associated with various methods of dredged material disposal due to the presence of large amounts of chemical contaminants present in dredged sediments, if realized, this potential would be manifested in subtle changes in environmental quality. Further, based on recent studies, in general, the majority of the contaminants present in dredged sediments do not become available to affect water quality at or near open water disposal sites. Since it has been generally accepted that there is a need for dredged material disposal criteria even though no real problem has been identified, more appropriate criteria than those in use today for open water disposal need to be developed.

Confined disposal criteria

One of the greatest difficulties with the existing criteria and the way they are being implemented is that they are primarily designed to evaluate the suitability of open water disposal; they do not give adequate consideration to the potential environmental degradation associated with the alternates to open water disposal. As discussed by Lee[69] and the Research Advisory Board of the International Joint Commission[56], many of the methods of disposal which have been chosen to be alternates to open water disposal may cause as much, if not more, environmental degradation as open water methods if disposal. The work of Hoeppel et al.[58] has provided some information on chemical characteristics of "confined" disposal effluent. These studies must be considered at best a meager start toward work on this topic. As pointed out by Lee[41,70], one of the greatest deficiencies in knowledge on the environmental impact of dredged material disposal is the effect on water quality of effluent from confined disposal operations. Any US EPA Regional Administrator or Corps of Engineers District Engineer or other who finds that open water disposal is not appropriate because of potential ecological damage, should be prepared to demonstrate that so-called "confined" disposal is actually less environmentally damaging than open water disposal.

From an overall point of view, it would be rare that open water disposal of dredged sediments will cause significant environmental degradation as a result of the presence of chemical contaminants in the sediments. In those instances where there is a great potential for environmental degradation associated with open water dredged material disposal, no shift to "confined" disposal should be made until it is certain that the environmental degradation for this type of disposal is significantly less than that associated with open water disposal.

An approach for developing open water disposal criteria

There are two major areas of concern with respect to the presence of chemical contaminants in dredged sediments. One of these is water column effects which are associated with the disposal operation; the other is the long-term effect associated with re-deposited sediments.

General types of water quality problems include: toxicity, ranging from organism death to impairment of reproduction and growth; stimulation of excessive growths of aquatic plants; and bioconcentration of contaminants. The significance of chemical contaminants present in redeposited sediments must be judged based on: direct toxicity to benthic and epibenthic organisms; long-term release of contaminants to the water column which could have an adverse effect on water column organisms; and the food web transfer of contaminants present in the sediments to higher trophic level organisms, thereby mobilizing contaminants which would not otherwise be mobilized.

The approaches that should be considered in establishing criteria to minimize these types of problems in association with open water disposal are discussed below. Since more is known today about developing meaningful criteria for phosphate than most other chemical contaminants, a more detailed discussion is presented for dredged sediment-associated phosphorus criteria than other contaminants.

Aquatic plant nutrients (phosphorus and nitrogen). The excessive fertilization (eutrophication) of water bodies is one of the major causes of water quality deterioration. The growth of planktonic and attached algae and aquatic macrophytes is generally controlled by nitrogen or phosphorus compounds. In fresh water, phosphorus is frequently the growth-limiting chemical element; in the marine environment; nitrogen frequently assumes this role. Since dredged sediments often contain large amounts of nitrogen and phosphorus compounds, it is important to develop an approach which can be used to assess the potential significance of nitrogen and phosphorus compounds present in dredged sediments in stimulating excessive growths of aquatic plants at or near disposal sites.

Both the September 5, 1975[43] and January 11, 1977 Federal Registers[44] mention eutrophication as an area that must be considered in evaluating the potential impact of open water dredged material disposal. Neither provides specific guidance on the approach that should be used. The implementation manuals[36,45] developed in accord with these Federal Registers provide little in the way of additional guidance. Attempts to use the guidance given are met with some difficulty as it is primarily directed toward assessing the effects of toxicants.

The US CE[45] recommends the use of algal bioassays for determination of growth stimulation resulting from release of nutrients in the elutriate test. The US EPA[71] algal assay procedure is recommended for use on dilutions of the filtered elutriate. Essentially no guidance was provided for the interpretation of such bioassay results other than to indicate that if growth stimulation is observed in the assays, dilution and mixing zone characteristics must be considered. As discussed by Lee et al.[50], standard algal assay techniques such as those proposed by US CE[45] are not applicable to predicting the environmental effect of short-term nutrient release since the maximum in situ exposure of algae to elevated concentrations is likely to be on the order of one to two hours; standard algal assays are typically run using exposures of from one to three weeks.

The Register and implementation manual[36] governing evaluation of sediments for ocean dumping are difficult to interpret in terms of phosphate concentrations and release. "Applicable marine water quality criteria," as defined by US EPA[44] are those included in the US EPA July 1976 Quality Criteria for Water.[8] There is no single numerical criterion for phosphate in this publication. Instead, a discussion is presented on various methods that have been used to estimate the impact of phosphorus on eutrophication-related water quality. As discussed in the American Fisheries Society review of the Quality Criteria for Water phosphate section (Lee et al.[72]) there are a number of deficiencies in the approaches discussed. With respect to the critical phosphorus concentration approach for lakes or tributary streams, as Lee and Jones[73] pointed out, one cannot today, with any degree of reliability, predict the water quality problems due to algae based on phosphorus concentrations at one time during the year. Listing of numbers such as 25 μg/l or 50 μg/l as critical concentrations for phosphorus, as indicated by US EPA [8], will tend to promote outdated approaches for establishing water quality standards.

The second approach discussed by US EPA [8] and from an overall point of view, the more appropriate for assessing the effect of phosphorus on water quality (Lee and Jones[73]), is the phosphorus loading approach originally developed by Vollenweider.[74] On behalf of the US EPA, as part of the Organization for Economic Cooperation and Development (OECD) Eutrophication Program, Rast and Lee[75] conducted a detailed review of phosphorus load-lake and impoundment response relationships for a variety of water bodies across the US. They found that the modified Vollenweider approach[76] involving the relationship between the areal phosphorus load to a water body and the mean depth and hydraulic residence time of the water body, is a valid approach to predict water quality characteristics of those water bodies in which algal growth is or can be made to be limited by phosphorus.

There are, however, a number of deficiencies and difficulties with applying this approach, as it is presented in US EPA[8], to dredged material disposal. Vollenweider has revised his phosphorus load-response relationship, arriving at different "permissible" and "critical" loadings than those presented by US EPA[8] for a water body with a given mean depth/hydraulic residence time ratio. Further, no guidance is given for the use of this approach other than presenting a table of outdated "permissible" and "critical" loadings.

As discussed by Rast and Lee[75], it is important to point out that the "permissible" and "excessive" loading curves do not represent sharp boundary lines. The fact that a lake has a load that is slightly above the critical loading value does not mean that it has significantly different water quality than a lake that is just below the critical loading level for the same morphological and hydrological relationships. As discussed by Rast and Lee[75], for a series of lakes in which algal growth is phosphorus limited and which have the same depth/hydraulic residence time ratios but different areal P loadings, there is a gradation of water quality among them which is proportional to the area P load. The best water quality would be found in lakes which have the lowest areal P load. Conversely, the worst water quality would be found in those water bodies with the highest areal P load. It should be further pointed out that the "permissible" and "critical" loading curves are, in general, based on impairment of the recreational use of water bodies due to planktonic algal growth.[75]

There is an additional consideration that needs to be made with respect to the use of the suggested US EPA Red Book phosphate criterion approach in connection with open water disposal of dredged sediment. The US EPA[8] stated that there were a number of exceptions to the threat of phosphorus in eutrophication. These exceptions included waters "heavily laden" with natural silts or color which would reduce light penetration; waters having certain morphological properties with no history of aquatic plant problems; waters where another nutrient limits aquatic plant growth and the limiting nutrient is not expected to increase; or where phosphorus cannot be made to be the limiting nutrient by controlling phosphorus load.

It has generally been found that in many coastal marine waters, the amount of available nitrogen controls planktonic algal growth. However, phosphorus release concomitant with nitrogen release from dredged sediment during disposal may allow a greater utilization of nitrogen and hence greater algal production even if the growth never becomes limited with respect to phosphorus. This situation could require that the Vollenweider approach be modified since, according to Rast and Lee[75], it should be applicable to water bodies in which algal growth is phosphorus limited.

Another factor that must be considered in using a phosphorus load approach for dredged material disposal is the fact that this approach is based on total phosphorus load. A large part of sediment-associated phosphorus is not solubilized into available forms during dredged material disposal. Further, unlike suspended material that may be introduced into a water body via a river or stream, dredged material introduced into the water column during hopper and barge disposal is in a fairly cohesive mass with a small percentage of the sediment remaining suspended in the surface water for any significant period of time. This would further decrease the amount of release to the disposal site water column expected, since many of the estimations of expected phosphorus release are based on completely mixed laboratory systems using a relatively high sediment-to-water ratio.

In order for the US EPA[8] Red Book phosphate criterion approach to be applicable to the legislative requirements regarding criteria for dredged material disposal, a critical available phosphorus concentration must be established for each water of concern. Whereas it is not possible to develop a universally applicable critical concentration, it should be possible based on the work of Rast and Lee[75] and Vollenweider[76] to develop such a level for a single water body or area of a water body by using its phosphorus load in conjunction with its morphologic and hydrologic characteristics. This critical concentration should be calculated for each period of concern rather than over an annual cycle. The work of Jones and Lee[77] has demonstrated that the Vollenweider[76] and Rast and Lee[75] approaches can be applied in such a manner.

According to US EPA[44], if there are no applicable marine water criteria, a liquid phase bioassay must be run to show whether or not the sediment can be dumped so as to not exceed the "limiting permissible level." Further, bioassays must be run, independent of the liquid phase bioassays, on the suspended particulate and solid phases also to show that the sediment can be discharged so as not to exceed the "limiting permissible concentration." The "limiting permissible concentration" is defined by the US EPA[44] in terms of toxicity or applicable marine criteria. Site and case-specific phosphate criteria must be determined in order to follow the procedures outlined in the case of phosphorus. Standard algal assays of the type recommended for use by the US EPA[44], i.e., those outlined by US EPA/CE[36], are not appropriate for evaluating the environmental impact of nutrients released during open water disposal of dredged sediments. This stems from the fact that hopper and barge disposal of dredged sediments are generally intermittent in nature and result in a short-lived pulse of increased nutrient concentration. Even in terms of predicting long-term impact, such bioassays may be of limited value since disposal areas have generally been chosen based partly on a short flushing time which would tend to minimize long-term impact of long-term nutrient release.

Another problem arises in evaluating dredged sediment in terms of phosphate in the manner outlined by US EPA[44] and US EPA/CE.[36] This is the use of an initial mixing period. According to the Federal Register[44], initial mixing is that dispersion of dredged material that occurs within four hours of disposal. The limiting permissible concentration cannot be exceeded beyond the boundaries of the disposal site during the mixing period and shall not be exceeded anywhere after initial mixing. The US EPA/CE discussed a number of methods for using elutriate test results to estimate the maximum concentration of the liquid and suspended particulate phases at a dredged material disposal site. The same type of problems arise when trying to apply this concept to phosphate release as discussed previously in applying a mixing zone to the fresh water criteria.

It is obvious upon examination of phosphate criteria presented in the Red Book [8] that they are not directly applicable to determining the potential for adverse environmental impact of dredged material disposal. Likewise, in the marine waters procedures manual[36], no guidance is given on how to properly use existing water quality criteria for phosphorus for making a meaningful interpretation of elutriate test results. It is most likely due to this lack of guidance that over half of the US EPA regions continue to use some type of bulk sediment criteria or no specifically defined criteria for judging the suitability of a dredged sediment for open water disposal.

Little is understood about the mechanisms of and the factors affecting algal uptake and assimilation of phosphorus. Likewise, the effects of a pulse nutrient input on the rate or extent of algal growth are not well understood. In order to determine the effect of a pulse of increased nutrient concentration on the rate and extent of algal growth, the increased concentration must be translated into a concentration-time of exposure relationship. It is not technically valid to make any judgment about potential eutrophication-related impact based on comparing a phosphorus concentration at some point in a water body (for example, at the edge of a mixing zone or so many hours after disposal) to some criterion or critical P concentration. Since this is the approach that has been advocated, in which the elutriate test results are used to predict concentration changes, a new, more technically valid approach for assessing the potential for eutrophication-related water quality problems to arise from the open water disposal of dredged sediments must be developed.

Out of work with the OECD Eutrophication Program, Vollenweider[74,76,78] and Rast and Lee[75] have found that the areal P load to a water body can, in general, be related to water quality as measured by average summer chlorophyll a, Secchi depth, and hypolimnetic oxygen depletion rate. Further, it has been found that changes in the load of phosphorus to a water body can be translated, using this approach, into resultant changes in mean chlorophyll a, Secchi depth, and hypolimnetic oxygen depletion rate. It appears that these relationships and approaches can be modified to be used for predicting eutrophication-related water quality response to dredged material disposal. The development of these approaches and modifications necessary for application to dredged material disposal are discussed below.

The Organization for Economic Cooperation and Development (OECD) initiated work on eutrophication effects and control in the mid 1960's. This initial work led to the classical report by Vollenweider[74] on the eutrophication of natural waters. Subsequently, the Water Management Sector Group of the OECD initiated a four-year, multination cooperative study directed toward the development of quantitative nutrient load-eutrophication response relationships for lakes and impoundments. The program has consisted of a several year, data-gathering effort, including determination of nutrient loads and resultant eutrophication responses as expressed by a variety of chemical and biological parameters.

The US portion of the OECD Eutrophication Project consisted of the examination of the nutrient load-eutrophication response relationships for 38 US water bodies. The water bodies range in surface area from that of Lake Michigan to water bodies of a few acres. They range in depth from two to over 300 meters and span the range of trophic conditions from ultra-oligotrophic to ultra-eutrophic.

R. Vollenweider (Chairman of the OECD Eutrophication Project Technical Bureau) has developed a phosphorus loading approach to eutrophication management which has been used as one of the primary eutrophication assessment methodologies in the OECD studies. It is an early version of this work which appears in the phosphorus criteria section of the Red Book.[8]

Vollenweider[76] found that a water body's mean depth and hydraulic residence time play dominant roles in determining the impact of a given phosphorus load on the fertility of a water body. He demonstrated that when the areal P load to a water body is plotted as a function of its mean depth:hydraulic residence time ratio, points for lakes of a similar trophic status tend to cluster together. Rast and Lee[75] verified this for the US OECD water bodies as shown in Figure 11.

In order to produce a relationship more useful for water quality management purposes, Vollenweider[76] extended the phosphorus loading concept to include its effects on chlorophyll a concentrations in water bodies. Vollenweider determined that the average summer epilimnetic chlorophyll a levels in water bodies could be correlated with the lakes' areal phosphorus loads as normalized by mean depth and hydraulic residence time.

Rast and Lee[75] have expanded this P load-chlorophyll relationship and derived models for determining the Secchi depth (water clarity) and hypolimnetic oxygen depletion rate in a water body based on its phosphorus loading characteristics. Figure 12 illustrates these three relationships for the US OECD water bodies upon which their correlations were based. These relationships can readily be applied to water bodies in which primary production is mainly due to planktonic algae, inorganic turbidity and color are low, and for which water quality is judged in terms of recreational use of the water body. Further, this approach is only applicable to water bodies in which algal growth during the growing season is P limited. If it is N limited, then release of additional P would not be expected to have a significant impact on eutrophication-related water quality. Recently, Jones, Rast, and Lee[79] have further extended these relationships to include predictions of the maximum summer chlorophyll concentration based on the phosphorus load, mean depth, and hydraulic residence time of the water body.

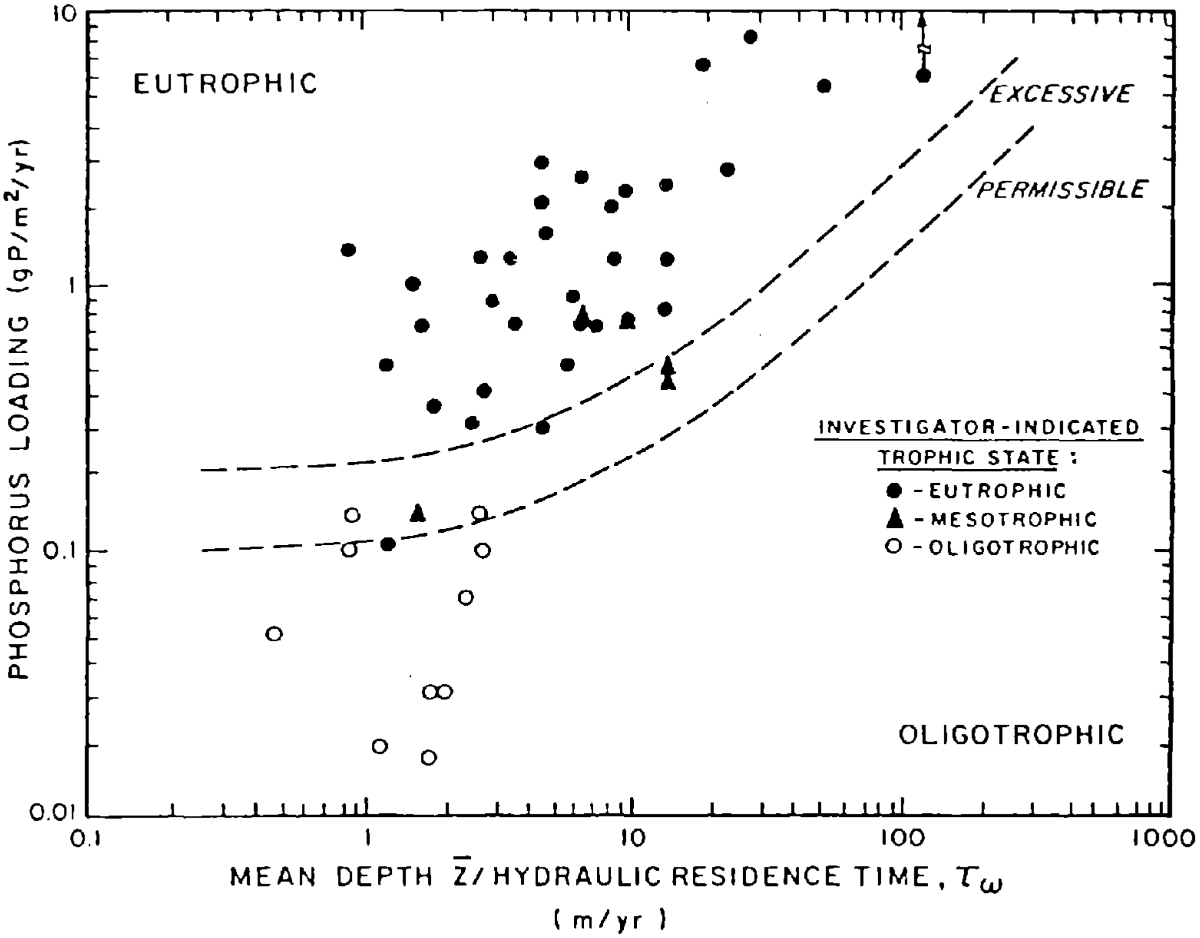


Figure 11
US OECD Data Applied to Vollenweider Phosphorus Loading-Mean Depth/Hydraulic Residence Time Relationship

After Rast and Lee[75]

Definition of Terms

$L(P)$ = Surface area total phosphorus loading ($mg/P/m^2/yr$)

q_s = Hydraulic loading (m/yr) = $\bar{z}/\tau_\omega$

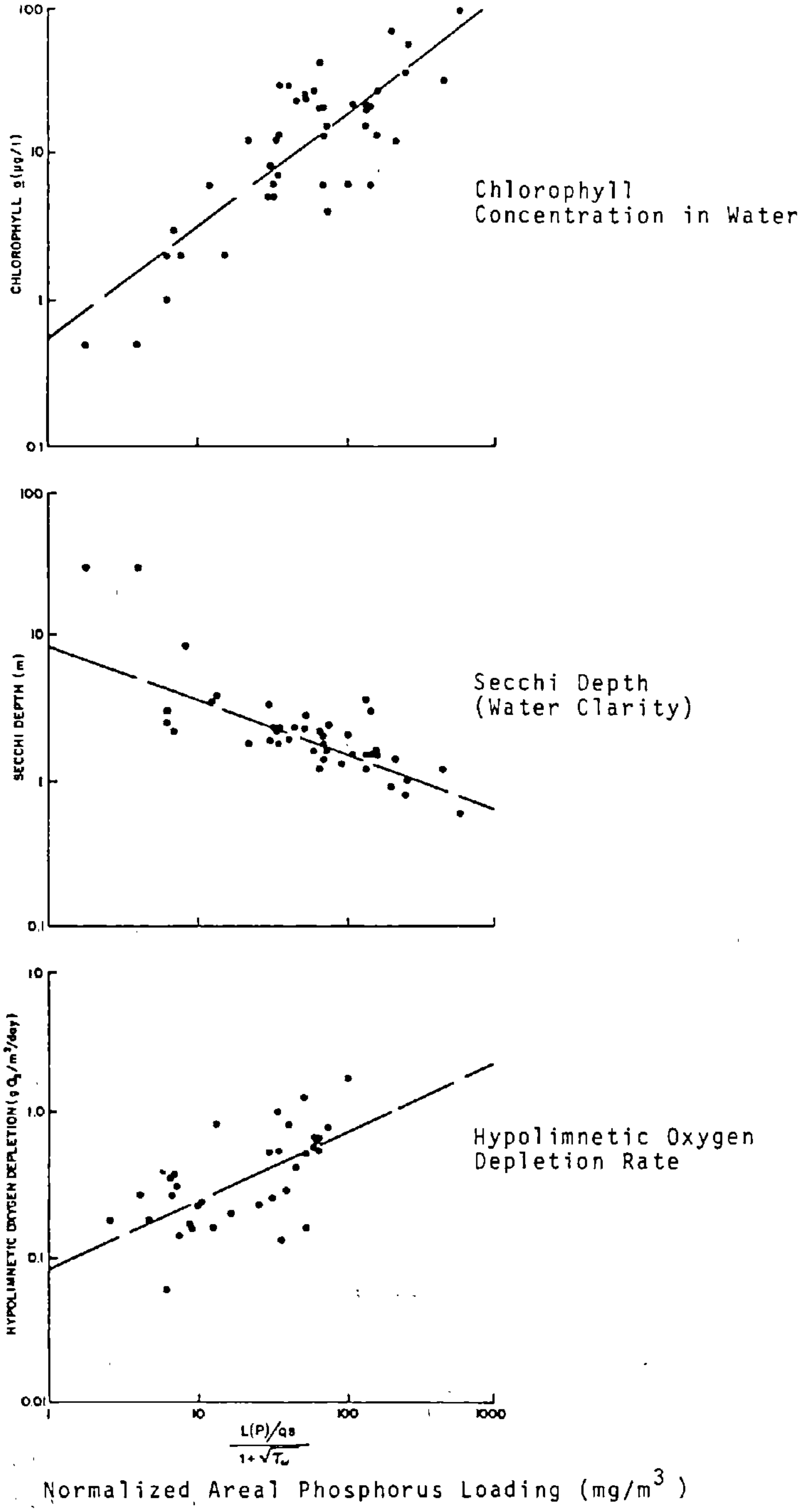
$\bar{z}$ = Mean depth (m) = water body volume (m^3)/surface area (m^2)

τ_ω = Hydraulic residence time (yr) = water body volume (m^3)/annual inflow volume (m^3/yr)

Figure 12

US OECD Data Applied to Phosphorus Loading--Mean Chlorophyll a--Secchi Depth, and--Hypolimnetic Oxygen Depletion Rate Relationships

After Rast and Lee[75]



It should be noted that since publication of the Rast and Lee report, these relationships have been tested for approximately 100 additional water bodies across the US, in Japan and in Europe. Except in cases where there have been high concentrations of color in the water, these additional water bodies have also been consistently found to obey these relationships.

The utility of these relationships can best be seen in terms of predicting the changes in water quality that would be expected to result from a change in the phosphorus load to a water body. As discussed by Rast and Lee[75], an example of the predictive capabilities of this approach can be seen for Lake Washington near Seattle, Washington. The phosphorus load to that water body has been reduced as a result of sewage treatment plant effluent diversion. The resultant steady-state value for mean chlorophyll a was accurately predicted by these methods. These methods have also been used by Lee[80], and Lee *et al.*[81] to predict the changes in Great Lakes' water quality that could result from a detergent P ban in the Great Lakes' states and from a removal of 90 percent of the P from sewage treatment plant effluent entering the Great Lakes or their tributaries. Although additional work needs to be done to verify the predictive capabilities of the Vollenweider and Rast and Lee nutrient load-response relationships, lakes across the country and in other areas of the world have been found to obey these relationships. This indicates that, while the aqueous environmental chemistry of phosphorus in natural waters is very complex, as far as the impact of phosphorus on primary production (as manifested in the growth of planktonic algae) is concerned, overall, lakes and impoundments respond to phosphorus loads in a relatively simple manner. The primary factors determining the effects of the utilization of the phosphorus load in a given water body are the mean depth (which for most lakes is related to their volumes) and the hydraulic residence time. These factors are the primary factors in determining the effective phosphorus concentration in a water body as a result of a given phosphorus load.

Another interesting aspect of the US OECD eutrophication project results is that, in general, the water quality in lakes and impoundments is remarkably insensitive to changes in phosphorus loads. All of the various relationships are double logarithmic plots which indicates that large changes in phosphorus loads will produce relatively small changes in average chlorophyll levels, water clarity, or hypolimnetic oxygen depletion rates. With respect to water clarity, a plot of chlorophyll a versus Secchi depth illustrates that at chlorophyll levels of about 5 μg/l or greater, little change in Secchi depth is noted for relatively large changes in chlorophyll.[75] Since the effects of excessive fertilization are usually most visible, from a management point of view, when chlorophyll levels are greater than about 10 μg/l, the control of phorphorus loads to most eutrophic lakes, while it may have a significant impact on the average chlorophyll levels, will likely have limited impact on water clarity until chlorophyll levels are reduced to below 5 μg/l.

In order to use the nutrient load-eutrophication response relationships outlined above, several basic pieces of information are needed. These include the water body's mean depth, hydraulic residence time (i.e., filling time) and areal phosphorus load. Information on mean summer chlorophyll a concentrations, Secchi depth, and hypolimnetic oxygen depletion rate are necessary to determine how well the particular water body in question obeys the established relationship (i.e., how close to the US OECD line of best fit the point for the water body plots), and to provide the initial conditions from which changes in the various parameters can be estimated. If such information is not available, and there are no detectable unusual features of the water body (such as high color or massive aquatic macrophyte gorwths), a rough estimate of the position of the water body can be obtained by plotting the water body at the intersection of its normalized P load and the US OECD line of best fit.

When the magnitude of the P load change is determined and normalized according to the P load term $(L(P)/q_s)/(1 + \sqrt{\tau_\omega})$, the new position for the water body will be at the intersection of a line drawn through the original point, parallel to the US OECD line of best fit, and a vertical line drawn through the new normalized P load. The resulting change in water quality can be determined quantitatively in terms of mean (or maximum) chlorophyll a, Secchi depth, and/or hypolimnetic oxygen depletion rate. There are, however, a number of aspects of this modeling approach which must be reconsidered in order to make it applicable to predicting changes in water quality that may result from disposal of dredged sediments in open water.

Probably the most important modification of this phosphorus load-response relationship approach that must be made is in the P load term. The nutrient load-eutrophication response relationship outlined above uses total P load. Since it has been found in the course of this study that very little of the total P associated with dredged sediment is mixed with disposal site water, it is inappropriate to use the total dredged sediment P load value when considering effects of a change in the P load to a water body. From elutriate test reresults, the mass of soluble ortho P released per unit mass of sediment as a result of elutriation, can be determined. It is suggested that for hopper dredge and barge disposal, this mass be multiplied by 20 to 25 percent of the total mass of dredged sediment dumped, giving an estimate of the maximum change in available P load to the water body that would result from the dredged sediment dump. The 20 to 25 percent figure was chosen based on the work of Gordon.[82] He found that approximately 18 percent of the dredged material discharged from a barge was transported outside a 30 m radius of the dump. It has been assumed for this discussion that material remaining within the 30 m radius is not sufficiently mixed with the area water to be transported and thus would not be included in the mass of sediment mixed at the dump site. The 25 percent estimate should be suitable for hopper dredge disposal operations since a greater amount of mixing of dumped sediment and disposal site water likely takes place during this type of operation than during barge dumping.

For pipeline disposal operations, pipeline output can be determined by knowing the size of the dredge pipe and the discharge velocity.[83] Multiplying this output by the percentage sediment composition will provide dredged sediment output. The total sediment mass added to a water body by pipeline disposal can be determined by multiplying output by dredging time. It is recommended that until more refined estimates are made, a 20 percent sediment composition be used (based on the optimum pumping ratio) and that a discharge of about 5000 cu yds/hr (3800 m^3/hr) be used as an average discharge rate. Twenty-five percent of this mass should be multiplied by the mass released during elutriation.

Another adjustment that may have to be made to apply the Vollenweider-Rast and Lee approaches is in the volume of water body to be considered and the associated hydraulic residence time. In the case of offshore open lake water disposal, such as for hopper dredge and barge disposal, use of the whole lake characteristics may be appropriate. For offshore ocean dumping or nearshore pipeline disposal, the volume, mean depth, and hydraulic residence time of some portion of the whole water body would be more appropriate to use. Decisions of that type would have to be made on a site by site basis and would depend on the major area of concern. It has been suggested[75] that water bodies with short annual hydraulic residence times, several weeks or less, may not follow the general nutrient load-response relationships developed by Vollenweider, and Rast and Lee since the nutrients added to the water are not in the area for a long enough period to be utilized by the algae. Jones and Lee[84] have found that in some cases, water bodies with short annual hydraulic residence times may experience periods of longer residence time, especially during the summer months. They suggested an approach for using these nutrient load-response relationships for modeling periods of less than one year. A discharge site should be examined for large changes in hydraulic residence times over the year. If it is found to experience large changes, the water body should be evaluated for effect of the added P load during each period or for the period when proposed dumping would take place.

If disposal takes place in a highly stratified area where there is very limited mixing between surface and bottom waters for extended periods, adjustments may have to be made in the mean depth-hydraulic residence time term and also in the percentage of the disposed dredged material that is mixed with disposal site water. If the material is dumped or discharged and remains trapped below a thermocline or chemocline, the mean depth term should reflect the epilimnetic mean depth and hydraulic residence time; the percentage of the disposed material mixed with disposal site water should be adjusted down to one to five percent. This figure is also based on the work of Gordon[82] in which he found that only aproximately one percent of the barge-dumped material was in the surface turbid plume.

The waters at many shallow nearshore open water disposal sites, such as are typically used for pipeline disposal operations, often contain high levels of inorganic turbidity. The presence of high levels of turbidity or color can reduce the rate at which algae reach maximum density based on available nutrients present because of reduced amounts of light available to the algae. This would increase the possibility of available phosphorus moving through the nearshore region to open waters or to the sediments without having been used by algae to the maximum extent possible. How these turbid waters should be handled in these relationships is still under investigation at UTD.

One other factor that must be considered in the interpretation of predictions made possible using this approach is what the public perceives as changes in water quality. In general, the public will perceive greater changes in water quality when the water quality is generally good than when it is poor. For example, a 2 μg/l change in chlorophyll content is perceived as a much greater improvement when the average chlorophyll is 5 - 10 μg/l than when it is 30 - 35 μg/l. Changes in nearshore and open water water quality as a result of a reduced P load must, therefore, be determined on a case-by-case basis.

Another factor that influences the public's perception of a change in water quality is the overall background clarity of the water. A change in water clarity resulting from decreased algal chlorophyll concentration is much more readily detectable in water without high background levels of inorganic turbidity than in more turbid waters often found in shallow nearshore areas.[85]

When evaluating the significance of the load of available P added to a water body during dredged sediment disposal, consideration must be given to the potential for an increase in internal P load from the redeposited sediment. As discussed by Jones[3], sediments are, in general over the annual cycle, sinks for phosphorus, i.e., phosphorus tends to accumulate in the sediments. However, there is some recycle from the sediments. Under oxic conditions (i.e., oxygen present in overlying waters) iron hydroxide tends to hold phosphorus in the sediments. However, as discussed by Lee *et al.*[6], there is some recycle of phosphorus to the overlying waters as a result of mineralization of organic P, principally from dead algae. When the overlying waters become anoxic, in addition to the mineralization reactions, ferric iron is reduced to the ferrous form with a concomitant release of phosphate. It is important to emphasize that under either condition, even though there is release of P from deposited sediments, normally over the annual cycle the net flux is strongly from the water column to the sediments. From an overall point of view, it is unlikely that the long-term, slow release of phosphorus from redeposited sediment would represent a significant source of phosphorus for overlying waters.

Nitrogen compounds can be of significance in surface waters from two points of view: as toxicants and as aquatic plant growth stimulants. The toxicant problems with nitrogen compounds are discussed in a subsequent section. As noted above, the growth of algae and other aquatic plants in many marine and some surface fresh waters is controlled by the availability of nitrogen compounds that can be used by aquatic plants.

Many of the same problems exist for developing criteria for nitrogen as an aquatic plant growth stimulant as were found with phosphorus, with one important addition: the US EPA *Quality Criteria for Water* of July 1976[8] provide no guidance on the approach that should be used to establish the critical concentration or load of nitrogen compounds that may cause excessive fertilization problems in natural waters.

While in principle, exactly the same approach as developed by Vollenweider[76] and expanded on by Rast and Lee[75] for phosphorus, can be used, at this time the load-response relationships for nitrogen have not been developed. Therefore, before meaningful dredged material disposal criteria can be developed for evaluating the significance of nitrogen compounds as aquatic plant stimulants, work must be done on developing load-eutrophication response relationships for nitrogen similar to those discussed for phosphorus.

Toxicants (heavy metals, ammonia, chlorinated hydrocarbon pesticides, PCBs and other compounds). Frequently, concern has been expressed about the open water disposal of dredged sediments due to the elevated concentrations of heavy metals, ammonia, chlorinated hydrocarbon pesticides and PCBs often associated with waterway sediments. This concern has largely been based on a supposition that the heavy metals and other toxic contaminants present in dredged sediments are in, or can readily be transformed into, forms which could have an adverse effect on water quality at an open water dredged material disposal site. The results of the current study have shown that this is clearly not the case for a variety of US waterway sediments and that very little of the heavy metals, chlorinated hydrocarbon pesticides and PCBs are released during open water disposal of dredged sediment. There was, however, considerable ammonia released during open water disposal operations monitored. Because of the similar nature of the potential adverse environmental impact of these toxic contaminants, the development of criteria for heavy metals, chlorinated hydrocarbon pesticides, PCBs and other toxicants should follow the same general approach.

In order to properly assess the potential for significant adverse impact to disposal site water quality associated with the release of toxic contaminants during open water dredged sediment disposal, an oxic elutriate test should be run on the sediment and appropriate site water. The elutriate should be analyzed for contaminants of potential significance. This would mean that during the first several times any waterway sediment is evaluated, a large array of contaminants should be assessed. In addition, the first several evaluations should include a screening-type bioassay in order to detect toxicity resulting from additive or synergistic effects,or from contaminants not measured. If the chemical and bioassay results are consistent among evaluations, then the extent of subsequent evaluations could be reduced to only those contaminants for which release may be marginal, unless a major change has taken place within the watershed between evaluations which would alter the input of contaminants to the waterway. If potentially significant amounts of a contaminant are released, then a screening bioassay of the type previously described should be run to determine the actual potential for toxicity of the elutriate. Bioassays should also be run where there is a question about the presence of some toxic contaminant which has not been determined. Since it has not been adequately addressed in the procedures manuals[36,45], there is a need for guidance on how to use the results of these and additional bioassays as may be necessitated based on screening test results to make a decision about the suitability of a sediment for open water disposal.

Results of chemical analyses on elutriates must be viewed in light of the characteristics of the disposal operation and disposal site. Using this information, an estimate of the rate of dilution of the turbid plume must be made in order to develop available concentration-duration of exposure relationships that exist in the turbid plume for any toxic contaminants released to the elutriate in potentially significant amounts. With respect to the available concentration-time of exposure relationships for aquatic organisms in the turbid plume, three situations must be considered. The worst case situation is one in which a planktonic or nektonic organism moves with or stays within the turbid plume until the turbidity has been dispersed. A second case involves nektonic organisms which swim through the turbid plume and thereby receive a short-term exposure to elevated contaminant concentrations. The third case is where the nektonic organisms avoid the turbid plume and thereby receive no exposure to elevated concentrations of contaminants. The relationships found to exist within the turbid plume in association with each of these conditions should then be compared to concentration-time of exposure relationships for the contaminant(s) that are known to have adverse effects on water quality. As previously discussed, information of this type is not readily available. In general, what information is available on the critical concentrations of chemicals for both fresh and marine water organisms is presented in the US EPA Quality Criteria for Water released in July 1976. This publication was largely based on an update of the information provided in Water Quality Criteria, 1972, prepared by the National Academy of Science-National Academy of Engineering.[86] While the 1976 criteria do not cover all possible contaminants and all possible types of aquatic organisms potentially affected, they do provide a sufficient body of knowledge to be useful in making a reasonable assessment of the potential ecological impact of chemical contaminants on water quality. Based on the aquatic toxicology studies that have been conducted thus far, it is improbable that a new chemical would show toxicities to aquatic life significantly different from the ranges of toxicities reported in the US EPA Quality Criteria for Water. Further, it is improbable that any untested species would show a significantly greater sensitivity to a contaminant than that reported in the US EPA Quality Criteria for Water.

In utilizing the numerical values presented in the US EPA Quality Criteria for Water, consideration must be given to the fact that, in general, these values were derived from chronic exposure situations where the organism was exposed to the available form of the contaminant throughout its lifetime or a significant part thereof. Because of the rapid dispersion of the turbid plume associated with the disposal operations and the intermittent nature of dredging and disposal, it would be virtually impossible for any water column organism to encounter a chronic exposure situation at most, if not all, open water disposal sites.

It should also be noted that while the US EPA Quality Criteria for Water numerical values are generally based on concentrations of soluble, available forms, regulatory agencies frequently apply them to the total content of a water sample. The elutriates are 0.45 μ pore size filtered samples. While these filters are typically used to separate "soluble" from "particulate" fractions, it is possible that there may be considerable differences in the available concentration of a contaminant in an elutriate and the "soluble" concentration. This would be likely encountered where the sediments contain finely divided particulate matter, such as was found with the James River sediments studied. A majority of the contaminants present in the turbid plume created by disposal has been found to be in an unavailable form which will not have an adverse effect on aquatic life and water quality [9, 87, 88]. Further discussion concerning the appropriate use of US EPA Quality Criteria for Water may be found in Lee [90], Lee *et al.*[89], and Lee and Jones [72].

While the above discussion is primarily directed toward dumping operations, the same general approaches are applicable to pipeline disposal operations as well.

While not within the framework of this study, it is important to consider, at least in a general way, the potential significance of redeposited dredged sediments on benthic, epibenthic, planktonic and nektonic organisms at the disposal site. First, for the water column organisms, the question must be asked whether the disposal of the dredged sediments in open waters results in a long-term release of contaminants to the overlying waters which may have a significant adverse effect on water column organisms. In general, it is unlikely that the rate of release of contaminants from redeposited sediments would be sufficient in most open water disposal systems to result in excessive concentrations of contaminants in the overlying waters. There is very little evidence for this kind of problem in the water pollution field and, in fact, except for mercury near a point where large amounts have been deposited from industrial waste discharges to sediments, few contaminants present in sediments have been found to cause excessive concentrations in overlying waters. Even where excessive concentrations do occur, they are usually close to the source of contamination. Hence, it is difficult to ascertain whether the contamination comes from the sediments or from the direct discharge of municipal and industrial wastes.

A second area of concern with respect to redeposited sediments is that of the direct uptake of the contaminants from sediments by benthic and epibenthic organisms and the possible transport to higher trophic level organisms. In order to determine whether this is a problem at a particular disposal site, samples of commercially important species should be collected from the disposal site. The concentrations of contaminants found in them should be compared to critical body burdens for the organisms and to critical concentrations or limits imposed for their use as sources of food for higher organisms. Additional information pertinent to developing criteria for dredged sediment-associated toxicants is provided in the bioassay section of this Part of the report.

Special considerations for the development of dredged material disposal criteria for selected contaminants.

The approaches discussed above are generally applicable to the development of dredged material disposal criteria for heavy metals, ammonia, chlorinated hydrocarbon pesticides, PCBs and other toxicants. Certain contaminants or groups of contaminants have characteristics which require special consideration for the development of dredged material disposal criteria. These considerations are discussed below for the potentially toxic contaminants investigated in this study.

Heavy metals. This study has shown that for a wide variety of US waterway sediments, there is little need for concern about sediment-associated heavy metals as they may affect the disposal site water column. Ordinarily, the only heavy metal which could be of potential concern is manganese; the dilution that normally occurs at an open water disposal site, however, will likely readily eliminate any adverse impact that might result from release of Mn during open water disposal of dredged sediment. While a wide variety of waterway sediments was tested in this investigation and showed no apparent problems, it is conceivable that an unusual environmental condition such as that associated with the discharge of a combination of industrial wastes containing heavy metals, complexing agents, etc., could create a situation where a heavy metal present in dredged sediments would be released in sufficient amounts to cause water quality problems at the dredged material disposal site. It is therefore appropriate, as discussed above, to use the elutriate test and bioassay tests to screen for this type of unusual conditions.

The one area that has not been investigated in this study is that of bioconcentration. Certain heavy metals such as mercury and cadmium are of potential concern because of bioconcentration within aquatic organisms to concentrations which, although not known to be adverse to the organism, could potentially be adverse to man when he uses these organisms for food. Studies have to be conducted at each of the dredged material disposal sites in order to determine whether sufficient concentrations of heavy metals could accumulate within aquatic organisms of the region to be considered adverse. This would have to be done on a site-by-site basis since it is not possible, based on the information available today, to extrapolate from laboratory studies to field conditions for bioconcentration of heavy metals associated with dredged sediment. It is important that a field evaluation be made of each existing disposal site rather than relying on the characteristics of the dredging site as proposed by the CE and US EPA in the bioassay manual[36].

Ammonia. Ammonia is likely to be the only one of the chemical contaminants studied in this investigation to represent a potential hazard to disposal site aquatic organisms. It would be generally expected that dilution would eliminate most potential for adverse environmental impact resulting from ammonia release during dumping operations. However, because of the relatively large ammonia release found in elutriate tests conducted during this study, there is a potential for water quality problems to arise from ammonia release during pipeline disposal operations. Although evaluative procedures outlined previously are applicable, it is important that the potential effects of ammonia release be carefully evaluated for all pipeline disposal operations to both open water and confinement areas. Studies conducted by several investigators (see Hoeppel, et al.[58]) have shown that considerably higher concentrations of ammonia are found in the effluent of confined disposal areas than in the influent. This is due to the conversion of organic nitrogen to ammonia within the confinement area. Since it is only the unionized form of ammonia which is toxic to aquatic organisms, the unionized fraction of ammonia must be determined in order to assess the potential toxicity of the disposal site water. To make this conversion, pH, ionic strength (salinity) and temperature of the receiving water must be known. It should be noted that for fresh water at 20°C and pH of 8 approximately 10 mg/l of ammonia would cause the death of half the fish in bioassay tests over a 96-hour period. Since an exposure period of 4 days would be rare at most pipeline disposal sites, a much higher concentration of total ammonia can be tolerated by organisms residing in that area. Therefore, even though large amounts of ammonia may be released from some sediments during open water disposal, the likelihood of this release causing significant water quality degradation is small.

The dredged material disposal criteria for nitrogen compounds should be directed toward ammonia rather than other forms of nitrogen since it is the only common chemical form of nitrogen which is likely to cause toxicity in natural waters.

Chlorinated hydrocarbon pesticides and PCBs. The development of dredged material disposal criteria for chlorinated hydrocarbon pesticides (CHPs) and PCBs should focus on potential problems of bioconcentration. Even though these compounds can be toxic to aquatic organisms at low levels, water quality criteria for many of these compounds are based on bioconcentrations rather than toxicity. At this time there is no adequate laboratory bioassay procedure that can be used to reliably predict the amount of bioconcentration that will occur at an open water disposal site. As discussed previously, the only reliable approach is that of measuring the concentrations of such contaminants in organisms of interest at existing disposal sites.

Caution should be exercised in using the US EPA *Quality Criteria for Water* for certain CHPs and PCBs since these criteria have been based on the total concentration of the individual compounds. For some waters part of the total PCBs and total individual CHPs is not available for uptake by organisms. It appears that the binding of these compounds to suspended sediment is stronger than through uptake and storage within aquatic organisms.

Dissolved Oxygen. Because of the high oxygen demand of some dredged sediments it is important to develop criteria which would predict the extent of deoxygenation that would be associated with dredged sediment disposal. Of particular concern is the deoxygenation that may occur in the turbidity current-mud flow associated with open water pipeline disposal. There is need to develop adequate information on the mixing characteristics at various types of disposal sites. Only with this information can the results of oxygen uptake tests be used to reliably predict the extent of deoxygenation at dredged material disposal sites. Further there is need for work on the significance of the exposure of organisms to short periods of low dissolved oxygen concentrations in order to properly evaluate water quality impact of the oxygen demand of dredged sediments upon open water disposal.

Bioassay.* The results of bioassays developed in this study can provide information which can be useful as a basis for screening dredged sediments for their potential adverse effects on aquatic life. The test developed in this study is not a definitive test designed to measure the response of aquatic organisms at the dredged material disposal site. Instead, it is a "worst case" test that is designed to easily and inexpensively screen potentially hazardous sediments from those that are innocuous.

In order to implement this test as a management tool, criteria must be established for interpretation of bioassay test results so they can be used as a "red flag" to indicate potentially excessive toxicity associated with disposal of dredged sediments. A case-by-case evaluation of the potential conditions that may exist at the dredged material disposal site should be used in implementing these criteria. A single numerical value criterion for organism mortality applied to all dredging and disposal operations should not be used. As discussed in other sections of this report, there are markedly different conditions associated with various types of dredging and disposal operations.

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*This section is based on draft materials submitted by G. Mariani.

Further, there are a number of site-specific characteristics which can influence the impact of a contaminant or group of contaminants at the disposal site.

The interpretation of the results of screening bioassay tests on dredged sediments requires considerable experience and skill. In the situations where all the test organisms live throughout the 96-hour test period, the likelihood of acute or chronic toxicity to aquatic life as a result of dredged material disposal is small. On the other hand, if all the organisms die within a few hours after exposure, the likelihood of significant toxicity at the disposal site is great. The results of 43 sediment bioassays conducted as part of this investigation show that generally only a few organisms die in the 96-hour test period in the test systems.

In order to evaluate the significance of extensive mortality in bioassays to disposal site water quality, it is important to consider the conditions that will prevail at the disposal site with respect to both the water column and the sediments. In evaluating the water column effects, it is important to consider the rate and amount of dilution that will occur at the disposal site. For most dredged material disposal operations involving dumping, the rate and extent of dilution would rarely permit toxicity to aquatic organisms in the disposal site water column. For most water column organisms, the likelihood of being exposed to elevated contaminant concentrations that may result from dredged material disposal, for sufficient periods of time to be harmed, is small. However, it is conceivable that disposal of sediments grossly contaminated with large amounts of readily available toxicants could result in acute toxicity under open water disposal site conditions. It should be noted, however, that even though many highly contaminated sediments have been tested in this study, gross acute toxicity to test organisms has not been encountered.

The area of concern with respect to effects on benthic and epibenthic organisms is the edge of the deposition area, not on the dredged material disposal mound. The disposal mound represents an area which has been set aside for disposal operations, and therefore toxicity should be allowed at the disposal site. Primary concern should focus on whether or not the materials that are transported from the disposal site have a significant adverse effect on water quality outside the designated disposal area. Such an approach requires some understanding of the rate and conditions of transport that prevail at the edge of the designated disposal area. It would be rare that the conditions that prevail there are not significantly different from those present in the disposal mound immediately after disposal. For example, the concentrations of sediment-associated contaminants should, in general, be considerably less than that present in the sediments that were originally dredged due to dilution with sediments of the region. Further, for many contaminants, the characteristics of the environment at the edge of the disposal site should generally be such that they would likely make the contaminant less available to affect water quality.

Another factor that must be considered in interpretation of bioassay results designed to evaluate the significance of disposal of contaminated sediments in natural water systems is that the sediments at the edge of the designated deposition zone are probably moving with the currents of the area. Therefore, although a slug of contaminated sediment may move through an area, long-term chronic exposure of benthic organisms that inhabit that area would be unlikely.

In those situations where moderate levels of toxicity are found in the screening bioassay test, additional more site-specific studies will be necessary to evaluate the significance of the toxicity. This will require specific evaluation of the significance of a certain level of toxicity to the test organisms compared to what actually occurs in the field. In general, the bioassay test developed in this study is conservative with respect to its predictions of potential environmental impact that will occur at the disposal site. It would be rare that toxicities greater than those encountered in the bioassay test, considering an appropriate application factor to relate acute lethal to chronic safe levels, would occur in environmental situations involving dredged material disposal in open waters. The conditions under which the modified elutriate test bioassays are run represent the "worst case" situation at a disposal site and the results of such a test should be used to indicate whether or not a particular sediment should be examined more closely before being dredged and disposed of at a particular site. However, in order to help characterize the source of toxicity, it may be beneficial to run bioassays on other fractions of the elutriate, such as on the filtered supernatant, or on the settled sediment-water slurry. The use of modifications of the modified elutriate test bioassays must be determined on a case-by-case basis depending on the characteristics of the particular situation. They should be designed to properly simulate the actual time (duration)-available concentration relationship at the edge of an appropriate mixing zone at the disposal site. The elutriate waters should be diluted with representative disposal site water in order to simulate as closely as possible the situation that will prevail in the disposal site water column. Bioassays should be run on various mixtures of the dredging site sediments with the sediments of the disposal region in order to simulate the dilution that will normally occur at the edge of the designated area. Care must be exercised to ensure that the physical and especially the chemical environment that exists at or near the disposal site water column and sediments is properly maintained in the bioassay tests.

Since many water pollution regulatory agencies use fixed numerical criteria rather than case-by-case evaluation, it is suggested that initially a 30 percent death in the 96-hour bioassay be used as a screening level; sediments which show toxicities of greater than 30 percent, using a modified elutriate test bioassay, should be classified as potentially hazardous and should be tested further as described above before disposal in open water is allowed.

The choice of 30 percent as a criterion for "red flagging" sediments was based on the results of the many bioassays conducted as part of this investigation as well as consideration of the many factors that play a role in determining the degree of toxicity that the disposal of a particular sediment may cause. Some of the variables that are important for determining the degree of toxicity include type of dredging-disposal; changes in the response of organisms to seasonal variations such as temperature; changes in the chemical form or availability of sediment associated toxic contaminants that may occur during dredging; and the hydrodynamic characteristics of the disposal area. If greater than 30 percent mortality is found for a sediment, further tests should be conducted to determine to what extent these other factors reduce the toxicity at the particular disposal site and to determine the likelihood of chronic toxicity resulting from the contaminants present in the dredged material.

An application factor which relates the acute lethal toxicity of a contaminant to aquatic life to the chronic safe level can be used to estimate the potential chronic toxicity to aquatic life of the contaminants present in the dredged sediments. It has been found [86] that for many chemical contaminants in natural waters, the 96-hour LC_{50} multiplied by 0.01 is equal to the chronic safe concentration for the contaminant. Examination of the data presented in Table A5 shows that for all the sediments tested, 50 percent or greater mortality occurred in the 96-hour period for only three sites, under conditions where organisms were placed in a settled slurry with characteristics similar to those of undiluted dredged material discharge from the pump of a hydraulic dredging operation. Since there would be substantial dilution and dispersion of any short-term or long-term contaminant release, it appears that there is a very small likelihood of chronic toxicity to aquatic organisms living within the water column above the dredged material disposal site or in the sediments near the dredged material disposal site.

PART IX: CONCLUSIONS AND RECOMMENDATIONS

Conclusions

Based on the results of this investigation, the following conclusions can be drawn:

a. Sediments dredged from US waterways typically contain sufficient amounts of chemical contaminants to cause the potential for significant water quality problems when the sediments are disposed of in open waters.

b. Studies on the elutriate test release of selected chemical contaminants from waterway sediments from across the US have shown that there is no relationship between the release of available forms of these constituents from a sediment and the sediment's total content of these constituents.

c. Studies on the relationship between the results of the elutriate test and the results of field monitoring studies have shown that in general the elutriate test is a useful tool for predicting the direction and approximate magnitude of contaminant release to the disposal site water column during disposal. There is no relationship between the bulk contaminant content of dredged sediments and the amount of contaminant that would be released upon open water disposal of dredged sediment.

d. Based on the results of the elutriate tests and field monitoring of dredged material disposal operations, it is concluded that the chemical contaminants present in dredged sediments would rarely, if ever, have a significant detrimental effect on water quality-aquatic biota in a disposal site in association with open water disposal of dredged sediment.

e. Based on the results of bioassay studies conducted as part of this investigation, it can be concluded that chemical contaminants present in dredged sediments are not generally available to affect aquatic life. It is concluded that little or no acute toxicity, and except under rare conditions chronic toxicity, would occur to aquatic organisms inhabiting the water column at or near open water dredged material disposal sites. Further, it would likely be rare that the chemical contaminants present in dredged sediments would have a significant adverse effect on benthic and epibenthic organisms residing outside of the designated open water disposal area.

f. The July 1977 bioassay procedure developed by the US EPA and CE, which was designed to implement the requirements of the January 11, 1977 Federal Register governing the ocean dumping of dredged sediments is overly complex and unnecessarily expensive for the determination of the potential significance of the chemical contaminants present in dredged sediment to aquatic organisms. The screening bioassay procedure developed in this study will likely prove to be a much more cost-effective, readily-interpretable procedure for assessing effects of dredged sediment-associated contaminants on aquatic life.

g. The bulk chemical criteria that are still being used by some EPA Regions are not technically valid for determining the pollutional tendencies of dredged sediment. They should be immediately abandoned; a combination of an elutriate test and screening bioassays should be substituted for these criteria. The elutriate test should be interpreted in terms of the concentration-time of exposure relationship that exists for each contaminant of concern outside of the designated disposal area. Consideration must be given to the sensitivity of the various water column organisms of interest based on their response to available forms of contaminants and the time of exposure of organisms to elevated concentrations.

h. The US EPA Quality Criteria for Water released in July 1976 which are often specified or used to judge the significance of the release of chemical contaminants are not a valid basis upon which to make decisions concerning the suitability of a sediment for open water disposal. These criteria are based on chronic exposure situations which rarely occur at open water dredged material disposal sites.

i. This study of the factors influencing the elutriate test has shown that the standard test required by the US EPA and CE has a number of potentially significant deficiencies which primarily relate to the oxygen content during the test. The failure of the CE and EPA to follow the recommendations of Lee *et al.*[1] to specify the redox conditions under which the test is conducted has resulted in the production of large amounts of essentially meaningless elutriate test data. This has contributed to a belief on the part of some individuals that the elutriate test is not a valid tool for predicting contaminant release upon open water disposal of dredged sediment. This study has shown that the use of a 30-minute compressed air mixing period in the elutriate test produces results which can generally be used to estimate the magnitude of release or uptake of contaminants during the disposal of hydraulically-dredged sediments.

j. The other recommended modification in the elutriate test is the use of 5 percent sediment of the total elutriate volume rather than 20 percent sediment. The 5 percent sediment content yields an elutriate which is often much easier to work with and also provides sufficient volume (within a reasonable period of time) for chlorinated hydrocarbon pesticides and other compounds to be determined with a sufficient degree of sensitivity for detection of potentially significant levels in the disposal site water column. In some instances, a 20 percent sediment mixture did not settle sufficiently during the one-hour period to allow rapid filtration of the elutriate. As a result, the expense associated with the test would have to be greatly increased due to the increased processing time and materials. Also, time of settling and filtration would vary from test to test depending on the type of sediment tested.

k. For a number of chemical contaminants, such as heavy metals, organic compounds and phosphorus, elutriate test results can often be explained in terms of the aqueous environmental chemistry of iron. As would be expected, the major factor affecting test results is the redox condition maintained during the test because of its influence on the forms of iron present. Other factors found to be of importance include sediment:water ratio and contact time, and length of sample storage.

l. The elutriate test should only be used to evaluate sediments to be hydraulically dredged where disposal is to be in open water. It should not be used for mechanically dredged sediments or to estimate the quality of the effluent from confined disposal areas. The plop test developed in this study should be used to estimate contaminant release during open water dumping of mechanically dredged sediments.

m. Dredged sediments typically have a high oxygen demand which could cause significant oxygen depletion in the water column associated with open water dredged material disposal. This study has shown that only a small part of the total oxygen demand is exerted during dumping operations and that in general it would be rare that open water dumping of either hydraulically or mechanically dredged sediment would cause sufficient oxygen depletion in the disposal site water column to be significantly detrimental to aquatic organisms inhabiting the area. There could, however, be significant oxygen depletion within several hundred meters of a pipeline dredged sediment discharge.

n. An oxygen demand test has been developed and evaluated as part of this overall study which, when used in conjunction with information on the hydrodynamics of the disposal operation and the receiving waters, can generally be used to predict the magnitude of oxygen depletion that will occur upon open water disposal of dredged sediment.

o. Of the heavy metals studied (Cu, Zn, Cd, Ni, Pb, Fe, Cr, As, Hg, and Mn), only manganese was consistently released in elutriate tests in potentially significant amounts. It is not anticipated that this manganese would, under typical open water disposal operations, be of sufficient magnitude to cause significant water quality deterioration at or near an open water dredged material disposal site. The elutriate test predicted the behavior of sediment-associated heavy metals found during open water disposal monitoring operations.

p. Of all the compounds studied in this investigation, ammonia was the only one released in elutriate tests in sufficient quantities to indicate the potential for adverse effects on disposal site water column organisms. The elutriate test was found to predict, with a reasonable degree of reliability, the general order of magnitude of release found in the field. The dilution normally associated with open water disposal would be expected to reduce elevated concentrations sufficiently rapidly so that no adverse effects on water column organisms would be expected. A case-by-case evaluation would have to be made to evaluate the potential for ammonia released from the disposal of dredged sediment to stimulate excessive growths of aquatic plants and thereby contribute to eutrophication-related water quality problems.

q. Results of the studies comparing dump site water column soluble ortho P concentrations during open water dredged material disposal and the release of soluble ortho P in various types of elutriate tests showed that the elutriate test can provide an indication of the behavior of sediment-associated phosphorus during open water disposal of dredged material. The soluble orthophosphate concentrations in the elutriate can also often give a general estimate of the soluble ortho P concentration found in the turbid plume created by the disposal, 100 to 200 m from a hopper or barge dump site. Proper consideration of disposal operation characteristics must be given in selecting the appropriate redox conditions, sediment to water ratios, and total contact time for the elutriate test.

From an overall point of view, it does not appear that the short-term release of available phosphorus from dredged material during open water disposal will result in significant water quality deterioration. On the other hand, use of confined disposal areas can, based on their design and operation, and on what is known about the behavior of sediment-associated phosphorus, readily result in eutrophication-related water quality problems in nearshore areas.

r. In general, with respect to phosphate, the criteria in use today for judging the suitability of a sediment for open water disposal are inappropriate and/or inadequate. The OECD-Vollenweider-Rast and Lee approach should be used to judge the significance to eutrophication-related water quality of an input of phosphorus from dredged sediments.

s. Some sediments showed release of chlorinated hydrocarbon pesticides and PCBs in elutriate tests. In general, the greatest release was found from those sediments with the lowest oil and grease content. It appears that sediments appear to retain PCBs within the hydrocarbon fraction of the sediment. The magnitude of the releases when they did occur, were sufficiently small so that no water quality deterioration would be expected to result.

This study has shown that the current US EPA water quality criteria for PCBs, which are based on the total PCB content of a water, are inappropriate. This is based on the fact that concentrations of PCBs far in excess of the criteria were observed in waters where indigenous organisms had bioaccumulated less PCBs than expected based on the water concentrations. It is evident that a substantial part of the PCBs present in suspended sediment is not readily available to aquatic organisms. These results raise questions about the general applicability of the EPA criteria for chlorinated hydrocarbon pesticides and PCBs where the concern is bioconcentration of these compounds within aquatic organisms.

t. The results of this study strongly indicate that it should be possible to alter dredging and dredged material disposal procedures to minimize or eliminate significant adverse environmental impact.

u. The adoption of bulk criteria by EPA regions and their continued use has, in many parts of the US, resulted in the use of alternate, frequently more expensive methods of disposal. These alternate methods are not necessarily more ecologically protective and in many cases may have resulted in greater environmental degradation than the less expensive, previously used open water disposal methods. The selection of a dredged material disposal site, i.e., confined on-land or open water disposal, should be based on a critical evaluation of the environmental impact associated with each potential site. It should not be assumed, as has frequently been done in the past, that "confined" disposal is more ecologically sound than open water disposal.

Recommendations

The following recommendations are made based on the results of this study:

a. Use of bulk chemical criteria as a basis for determining suitable locations for dredged material disposal should be terminated. In lieu of these criteria, the general approach outlined in the September 5, 1975 and January 11, 1977 Federal Registers should be adopted with appropriate modifications as delineated in this report. One of the key modifications that must be made is in the elutriate test. It should be conducted with a 30-minute compressed air mixing period rather than using mechanical shaking.

b. The elutriate test should be utilized to estimate the potential for release of chemical contaminants in dredged sediments to the disposal site water column. In addition, bioassay tests should be utilized to estimate the potential for significant bioaccumulation of chemical contaminants and significant adverse effects on benthic and epibenthic organisms. Appropriate criteria should be developed for interpreting the results of elutriate tests and bioassays.

c. It is recommended that a substantial research effort be initiated designed to develop information on the mixing and transport of water and sediments associated with open water dredged material disposal operations.

d. Long-term monitoring programs should be initiated at selected dredged material disposal sites in order to evaluate long-term subtle environmental effects of the contaminants present in dredged sediment. These studies should include investigations of the long-term environmental impact of both open water and "confined" disposal.

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APPENDIX A

Table A1

Summary: Phosphate Release From Dredged Sediment

Sample Designation		Modifications	Soluble Ortho P (mg P/l)		Total P (mg P/l)	
			$\overline{X}$	SD	$\overline{X}$	SD
SAN FRANCISCO AREA						
Rodeo Flats						
Site Water		-	0.058	0.013	0.19	0.02
5% Oxic		5% sediment	0.023	0.006	0.18	0.03
20% Oxic		-	0.030	0.006	0.28	0.06
Mare Island						
Site Water		-	0.053	0	0.44	0.02
5% Oxic		5% sediment, 20 hr settle	0.040	0	0.68	0.06
20% Oxic		20 hr. settle	0.11	0	0.63	0.04
Site Water		-	0.16	0	0.28	0.01
20% Anoxic	A**	Prepared, mixed, settled, and centrifuged under N_2, then filtered under air	0.54	0.02	1.2	0
	B	Prepared in air, then mixed, settled, centrifuged and filtered under N_2	0.50	0.01	0.98	0.02
	C	Processed completely under N_2	0.50	0.01	0.98	0.02
Mare Island (Storage Study)						
Site Water		44 day storage	0.063	0.003	0.12	0
5% Oxic		5% sediment, 44 day storage	0.070	0.008	0.14	0

(Continued)

Table A1 (Continued)

Sample Designation	Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
		$\overline{X}$	SD	$\overline{X}$	SD
Oakland Harbor					
Site Water	-	0.37	0.001	0.39	0.01
20% Oxic A	-	0.017	0	-	-
B	-	0.030	0.002	-	-
20% Anoxic/Oxic A	15 min. N_2 + 15 min. air mixing	0.052	0.003	-	-
B	15 min. N_2 + 15 min. air mixing	0.060	0.001	-	-
Oakland Harbor (Storage Study)					
Site Water	43 day storage	0.40	0.002	0.42	0.01
20% Oxic	43 day storage	0.11	0.001	-	-
20% Anoxic/Oxic	43 day storage, 15 min. N_2 + 15 min. air mixing	0.12	0.002	0.22	0.02
LOS ANGELES					
Los Angeles - Site 1					
EPA RSW	-	0.12	0.001	-	-
20% Oxic A	RSW used	0.34	0.005	-	-
B	RSW used	0.11	0.001	-	-
NEW ENGLAND					
Newport					
Site Water	-	0.056	0.001	-	-
20% Oxic A	-	0.31	0	-	-
B	-	0.28	0.002	-	-
20% Anoxic A	N_2 mixing	0.22	0.002	-	-
B	N_2 mixing	0.29	0.001	-	-

(Continued)

Table A1 (Continued)

Sample Designation			Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
				$\overline{X}$	SD	$\overline{X}$	SD
Bridgeport[a,b]							
Site Water				0.092	0.005	0.07	0
5% Oxic	(1 hr)		5% sediment	0.02	0.004	<0.02	-
	(24 hr)	A	5% sediment, 24 hr. settle	0.023	0.003	<0.02	-
		B	5% sediment, 24 hr. settle	0.018	0.001	0.03	0.01
10% Oxic	(1 hr)		10% sediment	0.027	0	0.06	0.01
	(24 hr)		10% sediment,24 hr. settle	0.059	0.003	0.07	0.02
20% Oxic	(1 hr)		-	0.063	0.001	0.15	0.06
	(24 hr)		24 hr. settle	0.34	0.003	0.37	0.01
Stamford							
Disposal Site Water			-	0.16	0.001	-	-
20% Oxic		A	-	0.043	0.001	-	-
		B	-	0.041	0.001	-	-
20% Anoxic		A	N_2 mixing	1.2	0	-	-
		B	N_2 mixing	1.4	0	-	-
Norwalk - Site 1							
Site Water			-	0.11	0.002	-	-
20% Oxic		A	-	0.013	0.001	-	-
		B	-	0.010	0.002	-	-
20% Anoxic		A	N_2 mixing	0.98	0	-	-
		B	N_2 mixing	0.92	0	-	-
Norwalk - Site 2							
Site Water			-	0.11	0.002	-	-
20% Oxic			-	0.030	0.002	-	-
20% Anoxic			N_2 mixing	0.83	-	-	-

(Continued)

Table A1 (Continued)

Sample Designation	Modifications	Soluble Ortho P (mg P/1) $\overline{X}$	Soluble Ortho P (mg P/1) SD	Total P (mg P/1) $\overline{X}$	Total P (mg P/1) SD
ASHTABULA [a,b]					
Ashtabula					
Site Water	-	0.011	0.002	<0.02	-
5% Oxic	(1 hr) 5% sediment	0.013	0.004	0.03	0.02
	(24 hr) 5% sediment, 24 hr.settle	<0.01	-	0.04	0.01
10% Oxic	(1 hr) 10% sediment	<0.01	-	0.06	0.01
	(24 hr) 10% sediment, 24 hr.settle	0.039	0.003	0.08	0.01
20% Oxic	(1 hr) -	<0.01	-	0.07	0.02
	(24 hr) 24 hr. settle	<0.01	-	0.11	0.02
MENOMINEE RIVER					
Menominee River - Site 1					
Site Water	-	0.039	0.001	-	-
20% Oxic A	-	1.6	0	-	-
B	-	0.77	0	-	-
20% Anoxic A	N_2 mixing	0.38	0	-	-
B	N_2 mixing	0.38	0.004	-	-
Plop A	Plop test procedure followed	0.33	0.001	-	-
B	Plop test procedure followed	0.19	0	-	-
Menominee River - Site 2					
Site Water	-	0.010	0	-	-
20% Oxic A	-	0.065	0	-	-
B	-	0.12	0.001	-	-
20% Anoxic A	N_2 mixing	0.081	0	-	-
B	N_2 mixing	0.039	0	-	-
Plop A	Plop test procedure followed	0.024	0	-	-
B	Plop test procedure followed	0.019	0	-	-

(Continued)

Table A1 (Continued)

Sample Designation		Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
			$\overline{X}$	SD	$\overline{X}$	SD
WES LAKE						
WES Lake						
Site Water		-	0.043	0.001	-	-
20% Oxic	A	-	0.067	0.001	-	-
	B	-	0.016	0.001	-	-
WILMINGTON						
Wilmington						
Site Water		-	0.19	0.001	-	-
20% Oxic	A	-	0.51	0	-	-
	B	-	0.40	0.002	-	-
MOBILE BAY						
Mobile Bay - Site 1						
Site Water		-	0.061	0.001	-	-
5% Oxic	A	5% sediment	0.058	0	-	-
	B	5% sediment	0.10	0.001	-	-
20% Oxic	A	-	0.053	0	-	-
	B	-	0.063	0	-	-
20% Anoxic	A	N_2 mixing	0.17	0.001	-	-
	B	N_2 mixing	0.25	0	-	-
Mobile Bay - Site 2						
Site Water		-	0.020	0.001	-	-
5% Oxic	A	5% sediment	0.031	0	-	-
	B	5% sediment	0.031	0.001	-	-
20% Oxic	A	-	0.040	0	-	-
	B	-	0.043	0	-	-
20% Anoxic	A	N_2 mixing	0.11	0.001	-	-
	B	N_2 mixing	0.086	0.001	-	-

(Continued)

Table A1 (Continued)

Sample Designation	Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
		$\overline{X}$	SD	$\overline{X}$	SD
Mobile Bay (August 1974)[a,b]					
Site Water	-	0.023	0	0.41	0.03
5% SH	Shaken	0.021	0.001	0.42	0.01
10% SH	Shaken	0.014	0.001	0.38	0.02
15% SH	Shaken	0.012	0	0.53	0.03
20% SH	Shaken	0.011	0.003	0.40	0.02
Site Water	-	0.031	0.002	0.15	0.11
20% SH	15 min. SH	0.017	0.004	<0.16	-
20% SH	30 min. SH	0.015	0.001	<0.12	-
20% SH	60 min. SH	0.010	0.002	0.08	0.02
20% SH	90 min. SH	0.012	0.003	0.09	0.02
Site Water	-	0.043	0.002	0.07	0.01
20% ST (1 ℓ)	Shaken	0.014	0.005	0.16	0.03
20% SH (1 ℓ)	Stirred	0.041	0.004	0.32	0.03
20% ST (3 ℓ)	Stirred	0.022	0.003	0.24	0.04

(Continued)

Table A1 (Continued)

Sample Designation	Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
		$\overline{X}$	SD	$\overline{X}$	SD
APALACHICOLA					
Apalachicola - Site 1					
EPA RSW	-	0.12	0.001	-	-
20% Oxic A	RSW used	0.19	0.001	-	-
B	RSW used	0.85	0.003	-	-
20% Anoxic/Oxic A	RSW used; 15 min. N_2 + 15 min. air mixing	0.044	0.001	-	-
B	RSW used; 15 min. N_2 + 15 min. air mixing	0.038	0.001	-	-
Apalachicola - Site 2					
EPA RSW	-	0.056	0.001	-	-
20% Oxic A	RSW used	0.040	0.001	-	-
B	RSW used	0.054	0.001	-	-
20% Anoxic/Oxic A	RSW used; 15 min. N_2 + 15 min. air mixing	0.094	0.001	-	-
B	RSW used; 15 min. N_2 + 15 min. air mixing	0.034	0.001	-	-
Apalachicola - Site 3					
Surface Site Water	-	0.039	0	-	-
20% Oxic	-	0.083	0.001	-	-
20% Anoxic	N_2 mixing	0.024	0.001	-	-
Bottom Site Water	-	0.055	0	-	-
20% Oxic	-	0.043	0	-	-
20% Anoxic	N_2 mixing	.054	0.001	-	-

(Continued)

Table A1 (Continued)

Sample Designation	Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
		$\overline{X}$	SD	$\overline{X}$	SD
Apalachicola - Site 4					
Surface Site Water	-	0.039	0	-	-
20% Oxic	-	0.042	0.001	-	-
20% Anoxic	N_2 mixing	0.086	0	-	-
Bottom Site Water	-	0.055	0	-	-
20% Oxic	-	0.017	0.001	-	-
20% Anoxic	N_2 mixing	0.032	0	-	-
Apalachicola - Site 5					
Surface Site Water	-	0.039	0	-	-
20% Oxic	-	0.068	0	-	-
20% Anoxic	N_2 mixing	0.063	0	-	-
Bottom Site Water	-	0.055	0	-	-
20% Oxic	-	0.029	0	-	-
20% Anoxic	N_2 mixing	0.040	0.001	-	-
UPPER MISSISSIPPI					
Upper Mississippi River					
Site Water	-	0.24	0.001	-	-
5% Oxic A	5% sediment	0.13	0	-	-
B	5% sediment	0.10	0	-	-
20% Oxic A	-	0.067	0	-	-
B	-	0.063	0	-	-

(Continued)

Table A1 (Continued)

Sample Designation		Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
			$\bar{X}$	SD	$\bar{X}$	SD
JAMES RIVER						
James River						
Site Water		-	0.040	0	-	-
5% Oxic	A	5% sediment	0.032	0	-	-
	B	5% sediment	0.028	0	-	-
20% Oxic	A	-	0.024	0	-	-
	B	-	0.020	0	-	-
20% Anoxic	A	N_2 mixing	0.028	0.001	-	-
	B	N_2 mixing	0.039	0	-	-
BAILEY CREEK						
Bailey Creek						
Site Water		-	0.040	0.001	-	-
5% Oxic	A	5% sediment	0.010	0	-	-
	B	5% sediment	0.010	0	-	-
20% Oxic	A	-	0.037	0	-	-
	B	-	0.038	0	-	-
20% Anoxic	A	N_2 mixing	0.12	0.001	-	-
	B	N_2 mixing	0.12	0	-	-
TEXAS GULF COAST						
GBEC - Buoy 1						
Buoy 1 Site Water		-	<0.02	-	<0.02	-
5% Oxic		5% sediment	<0.02	-	0.03	0.01
20% Oxic		-	0.064	0.004	0.07	0.01

(Continued)

Table A1 (Continued)

Sample Designation	Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
		$\bar{X}$	SD	$\bar{X}$	SD
GBEC Disposal Site Water		0.009	0.002	0.05	0.001
GBEC - Buoy 9					
Buoy 9 Site Water	-	0.005	0.001	0.05	0
5% Oxic (Buoy 9 Water)	5% sediment	0.024	0.001	0.10	0.01
5% Oxic (Disposal Site Water)	5% sediment	0.023	0.002	0.09	0.01
20% Oxic (Disposal Site Water)	-	0.023	0.002	0.09	0.01
GBEC - Buoy 11					
Buoy 11 Site Water	-	0.013	0.002	0.09	0
5% Oxic (Buoy 11 Water)	5% sediment	0.028	0.002	0.09	0.01
5% Oxic (Disposal Site Water)	5% sediment	0.029	0.001	0.10	0.01
20% Oxic (Disposal Site Water)	-	0.038	0.003	0.11	0.01
Texas City Channel - Site 1					
Site Water	-	0.095	0.021	0.10	0
5% Oxic (1 hr)	5% sediment	0.024	0.005	0.27	0.01
(24 hr)	5% sediment, 24 hr.settle	0.030	0.008	0.30	0.01
20% Oxic (1 hr)	-	0.22	0.008	0.24	0.01
(24 hr)	24 hr. settle	1.3	0.03	>1.0	-

(Continued)

Table A1 (Continued)

Sample Designation	Modifications	Soluble Ortho P (mg P/1) $\overline{X}$	Soluble Ortho P (mg P/1) SD	Total P (mg P/1) $\overline{X}$	Total P (mg P/1) SD
Texas City Channel - Site 1 (Storage Study)					
Site Water	70 day storage	0.082	0.003	-	-
5% Oxic	5% sediment, 70 day storage	0.15	0.004	-	-
20% Oxic	70 day storage	0.36	0.003	-	-
Site Water	72 day storage	0.11	0.002	0.11	0
20% Anoxic/Oxic	20 min.N_2+10 min.air mixing, 72 day storage	2.1	0	1.6	0.03
25% Anoxic/Oxic	25% sediment, 20 min.N_2+ 10 min.air mixing, 72 day storage	2.1	0	1.8	0.03
Texas City Channel - Site 2					
Site Water	-	0.11	0.006	0.12	0.01
5% Oxic	5% sediment	0.048	0.002	0.09	0.01
10% Oxic	-	0.11	0	0.24	0.01
Texas City Channel - Site 3					
Site Water	-	0.10	0.003	0.13	0.01
5% Oxic	5% sediment	0.054	0.005	0.13	0.01
20% Oxic	-	0.074	0.003	0.15	0.01
Texas City Channel - Site 4					
Site Water	-	0.17	0	-	-
5% Oxic	5% sediment	0.28	0.018	-	-
20% Oxic	-	0.26	0.001	-	-
Texas City Channel - Site 4 (Storage Study)					
Site Water	40 day storage	0.23	0.001	-	-
20% Oxic A	40 day storage	0.17	0.002	-	-
B	40 day storage	0.10	0.001	-	-

(Continued)

Table A1 (Continued)

Sample Designation	Modifications	Soluble Ortho P (mg P/1) $\overline{X}$	Soluble Ortho P (mg P/1) SD	Total P (mg P/1) $\overline{X}$	Total P (mg P/1) SD
Texas City Channel - Site 5					
Site Water	-	0.18	0.003	-	-
5% Oxic	5% sediment	0.13	0.003	-	-
20% Oxic	-	0.15	0.001	-	-
Texas City Channel - Site 5 (Storage Study)					
Site Water	40 day storage	0.21	0	-	-
20% Oxic	40 day storage	0.27	0	-	-
Texas City Channel - Site 6					
Site Water	-	0.20	0.002	-	-
5% Oxic	5% sediment	0.12	0.001	-	-
20% Oxic	-	0.22	0	-	-
Texas City Channel - Site 6 (Storage Study)					
Site Water	40 day storage	0.18	0	-	-
20% Oxic	40 day storage	0.042	0.002	-	-
Houston Ship Channel (August 1974)[a,b]					
Site Water	-	1.7	0	2.0	0.05
20% Oxic A	-	1.1	0.05	1.6	0.06
B	-	1.6	0.04	2.4	0.10
C	-	1.1	0.04	4.7	0.06
D	-	0.98	0.02	1.4	0
Site Water	-	1.8	0	1.9	0.1
20% Anoxic A	N_2 mixing	-	-	8.7	0
B	N_2 mixing, air contamination	3.5	0	5.8	0
20% Oxic	-	0.6	0	1.7	0.1

(Continued)

Table A1 (Continued)

Sample Designation		Modifications	Soluble Ortho P (mg P/1) $\bar{X}$	Soluble Ortho P (mg P/1) SD	Total P (mg P/1) $\bar{X}$	Total P (mg P/1) SD
Houston Ship Channel - Site 1						
Site Water		-	1.7	0.015	-	-
5% Oxic	A	5% sediment	0.97	0.006	-	-
	B	5% sediment	0.76	0.006	-	-
20% Oxic	A	-	2.7	0	-	-
	B	-	3.8	0.006	-	-
20% Anoxic	A	N_2 mixing	9.8	0.07	-	-
	B	N_2 mixing	9.0	0.035	-	-
Houston Ship Channel - Site 2						
Site Water			2.1	0.015	-	-
5% Oxic	A	5% sediment	0.32	0.002	-	-
	B	5% sediment	0.28	0.001	-	-
20% Anoxic	A	N_2 mixing	11.0	0	-	-
	B	N_2 mixing	9.0	0.058	-	-
20% Oxic	A	5 replicate tests made	0.63	0.007	-	-
	B	-	1.9	0.006	-	-
	C	-	2.0	0.006	-	-
	D	-	2.9	0.006	-	-
	E		2.7	0	-	-
Site Water			0.079	0.001	-	-
Plop Test	(2ℓ)	Plop test procedure followed, 2 ℓ tot.vol.; 1 hr. mixing; 30 min. settle	0.39	0.009	-	-
	(4ℓ)	Plop test procedure followed, 4 ℓ tot.vol.; 1 hr. mixing; 30 min. settle	0.32	0	-	-
	(8ℓ)	Plop test procedure followed, 8 ℓ tot.vol.; 1 hr. mixing; 30 min. settle	0.26	0.001	-	-

(Continued)

Table A1 (Continued)

Sample Designation		Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
			$\overline{X}$	SD	$\overline{X}$	SD
Houston Ship Channel - Site 3						
Site Water		-	1.2	0.006	-	-
5% Oxic	A	5% sediment	0.21	0.001	-	-
	B	5% sediment	0.32	0.001	-	-
20% Oxic	A	-	0.91	0.023	-	-
	B	-	0.36	0.001	-	-
20% Anoxic	A	N_2 mixing	1.2	0.006	-	-
	B	N_2 mixing	1.4	0	-	-
Morgan's Point						
Site Water		-	0.57	0.004	0.86	0.01
5% Oxic		5% sediment	0.13	0.01	0.20	0.05
20% Oxic		-	0.095	0.001	0.21*	0.01
Galveston Channel						
Site Water		-	0.17	0.002	-	-
20% Oxic		-	0.16	0.002	-	-
Trinity River[a,b]						
Site Water		-	5.0	0.05	8.7	1.0
20% SH	A	Shaken	0.13	0.02	7.7	1.1
	B	Shaken	0.12	-	8.7	2.1
Port Lavaca						
Site Water			0.023	0.001	0.06	0.01
5% Oxic		5% sediment	0.082	0.001	0.20	0.016
20% Oxic		-	0.058	0	0.38	0.028
5% Anoxic/Oxic	A	5% sediment; 20 min. N_2+ 10 min. air mixing	0.26	0.005	0.24	0.017
	B	5% sediment; 20 min. N_2+ 10 min. air mixing	0.10	0.002	0.15	0.03
20% Anoxic/Oxic	A	20 min. N_2+10 min. air mixing	0.086	0.003	0.20	0
	B	20 min. N_2+10 min. air mixing	0.12	0	0.27	0.007

(Continued)

Table A1 (Continued)

Sample Designation		Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
			$\bar{X}$	SD	$\bar{X}$	SD
Port Lavaca (Storage Study)						
Site Water		54 day storage	0.080	0	0.09	0.009
20% Oxic		54 day storage	0.095	0.002	-	-
Corpus Christi - Site 1^{a,b}						
Site Water		-	0.017	0.004	0.04	0.02
5% Oxic	A	5% sediment	0.044	0.001	0.09	0.01
	B	5% sediment	0.036	0.002	0.10	0.02
Corpus Christi - Site 2^{a,b}						
Deionized Water		-	< 0.01	-	<0.02	-
5% Oxic	A	Deionized water used, 5% sediment	0.54	0	0.13	0.01
	B	Deionized water used, 5% sediment	0.085	0.004	0.15	0
Site 2 Water		-	0.065	0.004	0.006	0.01
5% Oxic	A	5% sediment	0.065	0.002	0.08	0.01
	B	5% sediment	0.060	0	0.08	0.04
Site 4 Water		-	0.026	0	0.03	0
5% Oxic	A	5% sediment	0.059	0.006	0.08	0.02
	B	5% sediment	0.057	0.003	0.10	0.06

(Continued)

Table A1 (Continued)

Sample Designation		Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
			$\bar{X}$	SD	$\bar{X}$	SD
Corpus Christi - Site 3[a,b]						
Site 3 Water		-	0.10	0	0.12	0.02
5% Oxic		5% sediment	0.22	0.01	0.26	0.01
10% Oxic		10% sediment	0.23	0	0.31	0.02
20% Oxic		-	0.34	0	0.52	0.01
Site 4 Water		-	0.26	0	0.04	0
5% Oxic		5% sediment	0.14	0	0.20	0.03
10% Oxic		10% sediment	0.21	0	0.32	0.03
20% Oxic		-	0.30	0.01	0.41	0.02
Site 4 Water		-	0.28	0	0.06	0
5% Oxic	A	5% sediment	0.080	0.002	0.12	0.02
	B	5% sediment	0.10	0.002	0.13	0.02
Port Aransas[a,b]						
Site Water		-	<0.02	-	0.03	0.01
5% Oxic	(1 hr)	5% sediment	0.06	0	0.08	0.01
	(24 hr)	5% sediment, 24 hr.settle	0.05	0.01	0.06	0.02
10% Oxic	(1 hr)	10% sediment	0.06	0.01	0.07	0
	(24 hr)	10% sediment,24 hr.settle	0.06	0	0.07	0
20% Oxic	(1 hr)	-	0.08	0	-	-
	(24 hr)	24 hr. settle	0.07	0	0.11	0.01

(Continued)

Table A1 (Continued)

Sample Designation		Modifications	Soluble Ortho P (mg P/1)		Total P (mg P/1)	
			$\overline{X}$	SD	$\overline{X}$	SD
SEATTLE						
Duwamish River - Site 1						
Site Water		-	0.079	0.001	-	-
5% Oxic	A	5% sediment	0.054	0	-	-
	B	5% sediment	0.13	0	-	-
20% Oxic	A	-	0.19	0	-	-
	B	-	0.050	0.001	-	-
20% Anoxic	A	N_2 mixing	0.029	0.001	-	-
	B	N_2 mixing	0.042	0.001	-	-
Duwamish River - Site 2						
Site Water		-	0.079	0.001	-	-
5% Oxic	A	5% sediment	0.072	0.001	-	-
	B	5% sediment	0.039	0	-	-
20% Oxic	A	-	0.11	0.002	-	-
	B	-	0.40	0	-	-
20% Anoxic	A	N_2 mixing	0.23	0.001	-	-
	B	N_2 mixing	0.42	0.003	-	-
Duwamish River - Site 3						
Site Water		-	0.079	0.001	-	-
5% Oxic		5% sediment	0.032	0.002	-	-
20% Oxic		-	0.020	0.001	-	-
20% Anoxic		N_2 mixing	0.041	0.003	-	-
Plop Test		Plop test procedure followed	0.030	0.001	-	-

(Continued)

Table A1 (Continued)

Sample Designation		Modifications	Soluble Ortho P (mg P/1) $\overline{X}$	Soluble Ortho P (mg P/1) SD	Total P (mg P/1) $\overline{X}$	Total P (mg P/1) SD
NEW YORK						
Perth Amboy Channel						
Site Water		-	0.025	0.001	-	-
5% Oxic	A	5% sediment	0.04	0	-	-
	B	5% sediment	0.04	0	-	-
20% Oxic	A	-	0.11	0	-	-
	B	-	0.05	0	-	-
20% Anoxic	A	N_2 mixing	0.32	0.001	-	-
	B	N_2 mixing	0.73	0	-	-
Perth Amboy Anchorage						
Site Water		-	0.14	0.001	-	-
5% Oxic	A	5% sediment	0.053	0.001	-	-
	B	5% sediment	0.055	0	-	-
20% Oxic	A	-	0.29	0.001	-	-
	B	-	0.23	0.001	-	-
20% Anoxic	A	N_2 mixing	4.2	0.01	-	-
	B	N_2 mixing	3.6	0.02	-	-
Plop Test	A	Plop test procedure	0.46	0.001	-	-
		followed	0.31	0.006	-	-
	B	Plop test procedure followed	0.31	0.006	-	-
Bay Ridge Channel						
Site Water		-	0.094	0.001	-	-
5% Oxic	A	5% sediment	0.042	0	-	-
	B	5% sediment	0.53	0.004	-	-
20% Oxic	A	-	0.061	0.001	-	-
	B	-	0.046	0.001	-	-
20% Anoxic	A	N_2 mixing	1.2	0	-	-
	B	N_2 mixing	0.92	0	-	-

(Continued)

Table A1 (Concluded)

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*Centrifuged prior to analysis.

**Elutriate samples were treated as follows:

A - Sample was prepared, mixed, settled, and centrifuged under N_2, then filtered under air.

B - Sample was prepared in air, then mixed, settled, centrifuged and filtered under N_2.

C - Sample was processed completely under N_2.

***Twenty-four hour settling period. All other Mare Island elutriates settled one hour.

Dash (-) indicates not analyzed or not applicable.

Mean and standard deviation calculated from duplicate or triplicate analyses of one sample.

[a] Lee _et_ _al_.[3]

[b] Piwoni[4]

Table A2

Summary of the Elutriate Test Results for Heavy Metals*

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
SAN FRANCISCO AREA												
Rodeo Flats												
Site Water		-	30	0.3	-	32	-	2.3	2.2	3	-	-
5% Oxic		5% sediment	138	0.4	-	1.5	-	1.7	1.8	4	-	-
20% Oxic		-	160	0.5	-	1.1	-	<1	1.6	14	-	-
Rodeo Flats (Bioassay Elutriates)												
EPA RSW	A	-	25	13.5	-	6	-	<0.5	1.8	16	<0.01	-
	B	-	25	13.5	-	6	-	<0.5	1.8	16	<0.01	-
5% Oxic	A	5% sediment	50	13	-	10	-	<0.5	4	468	0.05	-
	B	5% sediment	70	13.3	-	11	-	<0.5	4	326	0.04	-
10% Oxic	A	10% sediment	175	12.7	-	6	-	<0.5	4	269	0.10	-
	B	10% sediment	50	12.7	-	6	-	<0.5	2.7	896	0.10	-
20% Oxic	A	-	160	12.7	-	14	-	<0.5	4	151	0.03	-
	B	-	190	11.5	-	8	-	<0.5	2.7	232	0.03	-
Mare Island												
Site Water		-	<30	0.3	-	20	-	<1.0	2.6	8	-	-
5% Oxic		5% sediment, 20 hr. settle	32	0.1	-	0.9	-	<1.0	4.4	81	-	-
20% Oxic		20 hr. settle	<30	0.3	-	1.4	-	<1.0	3.6	62	-	-
Site Water		-	13.7	1.2	-	117.8	117	5.6	43.9	74	-	-
20% Anoxic	A	Prepared, mixed, settled, and centrifuged under N_2, then filtered under air	6.9	0.6	-	18.7	117	8	25.7	500	-	-
	B	Prepared in air, then mixed, settled, centrifuged and filtered under N_2	9.1	1.4	-	17.4	118	7.3	19.5	463	-	-
	C	Processed completely under N_2	5.2	0.6	-	17.2	20	9.5	21.5	393	-	-

(Continued)

Table A2 (Continued)

Sample Designation	Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Mare Island (Storage Study)											
Site Water	44 day storage	23.3	-	-	51.9	26	4.5	68.1	10	0.067	-
5% Oxic	5% sediment, 44 day storage	15.5	-	-	2.1	34	6	32.2	18	0.033	-
Mare Island (Bioassay Elutriates)											
EPA RSW A	-	<30	0.2	-	20.2	-	<1	2.6	85	-	-
B	-	<30	0.2	-	20.2	-	<1	2.6	85	-	-
5% Oxic A	5% sediment	37	0.2	-	1	-	<1	4.5	69	-	-
B	5% sediment	30	0.1	-	1	-	<1	4.3	94	-	-
20% Oxic A	-	<30	0.2	-	1.3	-	<1	3.3	59	-	-
B	-	<30	0.3	-	1.5	-	<1	3.8	65	-	-
Oakland Harbor											
Site Water	-	153	1.3	<2	20.1	3.2	9.2	8.1	3	0.050	<2
20% Oxic A	-	621	1.5	<2	3.1	11.6	7.8	2.6	6	0.035	<2
B	-	588	1.4	<2	3.5	6.4	6	2.1	18	0.035	<2
20% Anoxic/Oxic A	15 min. N_2 + 15 min. air mixing	1450	0.7	<2	2.8	10	8.3	2	10	0.035	<2
B	15 min. N_2 + 15 min. air mixing	3300	0.7	<2	5.4	4.3	2.9	1.1	10	0.035	<2
Oakland Harbor (Storage Study)											
Site Water	43 day storage	24	1.3	<2	27.3	7.1	0.8	9.9	<5	<0.005	<2
20% Oxic	43 day storage	346	0.6	<2	9.8	9.9	1.5	3.6	6	<0.005	<2
20% Anoxic/Oxic	43 day storage; 15 min. N_2 + 15 min. air mixing	359	0.6	<2	12.5	4.4	1.6	3.9	11	<0.005	<2
Oakland Harbor (Bioassay Elutriates)											
EPA RSW A	-	202	1.8	<2	14.2	15.7	16.5	6.3	27	0.023	-
B	-	180	2	<2	29	16.1	19.2	8.3	17	0.008	-
5% Oxic A	5% sediment	270	1.9	<2	13.3	21.7	46.7	21	2600	0.030	-
B	5% sediment	473	1.4	<2	31.5	17	37.1	20	2100	0.036	-
20% Oxic A	-	405	2.3	<2	38	22.2	41.1	19	2000	0.025	-
B	-	860	1.6	<2	32.9	19.1	37.1	17.7	1900	0.036	-

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
LOS ANGELES												
Los Angeles - Site 1												
EPA RSW		-	8	<2	0.8	8.3	2.6	24.7	11.7	<5	<0.005	<2
20% Oxic	A	RSW used	24	<2	0.6	3.2	1.2	13.1	7	12	0.006	<2
	B	RSW used	32	<2	0.6	3.4	0.8	18.6	7	13	<0.005	<2
Los Angeles - Site 1 (Bioassay Elutriates)												
EPA RSW	A	-	58	1.5	<2	10.7	8.6	1.3	5.3	0.29	0.004	-
	B	-	27	1.5	<2	11.5	9.6	1.3	6.5	0.16	0.007	-
5% Oxic	A	5% sediment	160	1.2	<2	40.1	10.8	21	38	1500	0.002	-
	B	5% sediment	37	1.2	<2	44.2	11.1	22	38	1500	0.03	-
20% Oxic		-	5	1.1	<2	44.1	9.9	26	37	1500	0.03	-
Los Angeles - Site 2 (Bioassay Elutriates)												
EPA RSW	A	-	202	1.8	<2	14.2	15.7	16.5	6.3	27	0.023	-
	B	-	180	2	<2	29	16.1	19.2	8.3	17	0.008	-
5% Oxic	A	5% sediment	202	1.6	<2	45.4	17.4	32.9	23.3	2800	0.040	-
	B	5% sediment	225	1.3	<2	35.7	19.6	30.1	25.3	3100	0.040	-
20% Oxic	A	-	248	1.6	<2	43.7	21.3	26.1	18.3	3600	0.150	-
	B	-	293	1.9	<2	25.4	13	19.2	14	1000	0.020	-
NEW ENGLAND												
Newport												
Site Water		-	17	0.9	<2	29	3.5	7.9	5.3	72	<0.001	<2
20% Oxic	A	-	183	0.9	<2	11.4	3.6	9.1	2.1	18	0.003	<2
	B	-	209	1	<2	11.9	3.2	7.9	0.8	<0.5	<0.001	<2
20% Anoxic	A	N_2 mixing	187	0.8	<2	26	3.6	6.3	2.4	14	0.003	<2
	B	N_2 mixing	213	0.9	<2	14.4	2.9	7.1	2.2	26	0.026	<2

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Newport (Bioassay Elutriates)												
EPA RSW	A	-	48	1.3	<2	21.7	8.0	3.8	25.3	34	-	-
	B	-	48	1.2	<2	10.4	8.6	7.6	19.5	65	-	-
5% Oxic	A	5% sediment	37	0.7	<2	10.3	6.1	2.5	1.4	700	-	-
	B	5% sediment	27	0.7	<2	9.5	5.8	2.5	1.4	900	-	-
20% Oxic	A	-	5	0.6	<2	8.4	6.1	3.8	3.2	400	-	-
	B	-	37	0.6	<2	8.0	6.1	2.5	3.8	500	-	-
Bridgeport												
Site Water			60	2.3	-	10.5	-	4.5	6.4	32	-	-
5% Oxic	(1 hr)	5% sediment	82	1.8	-	6.8	-	3.8	4.2	32	-	-
	(24 hr)	5% sediment, 24 hr. settle	78	2.4	-	11.5	-	3.8	5.9	45	-	-
10% Oxic	(1 hr)	10% sediment	92	2	-	7.5	-	3	5.5	26	-	-
	(24 hr)	10% sediment, 24 hr. settle	90	2	-	5.8	-	4.8	4.9	16	-	-
20% Oxic	(1 hr)	-	100	2	-	3.5	-	4.5	2.8	38	-	-
	(24 hr)	24 hr. settle	120	2.1	-	5	-	4.8	9.0	33	-	-
Stamford												
Disposal Site Water		-	9	1	<2	18	6.9	7.5	8.9	101	<0.001	<2
20% Oxic	A	-	541	0.8	<2	11.7	7	8.7	2.9	82	<0.001	<2
	B	-	528	0.9	<2	18.4	6.6	9.5	1.6	17	<0.001	<2
20% Anoxic	A	N_2 mixing	230	0.8	<2	11.5	4.8	5.5	2.5	92	0.017	<2
	B	N_2 mixing	247	0.8	<2	13.4	5	10.3	3.1	149	0.016	<2
Stamford (Bioassay Elutriates)												
EPA RSW	A	-	48	1.3	<2	21.7	8	3.8	25.3	34	-	-
	B	-	48	1.2	<2	10.4	8.6	7.6	19.5	65	-	-
5% Oxic	A	5% sediment	48	0.9	<2	21.3	11.1	27.7	36.5	700	-	-
	B	5% sediment	48	0.9	<2	19.8	10.8	21.4	34.9	900	-	-
20% Oxic	A	-	69	2.3	<2	39.8	18.4	26.4	42.5	2200	-	-
	B	-	37	0.7	<2	12.9	8	13.8	18.3	1000	-	-

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Norwalk - Site 1												
Site Water		-	88	1.2	<2	3.7	3.3	<1	3.0	<5	<0.001	<2
20% Oxic	A	-	49	0.9	<2	1.9	3.9	<1	0.7	466	0.004	<2
	B	-	39	0.8	<2	1.3	3.6	<1	1.3	33	0.003	<2
20% Anoxic	A	N_2 mixing	39	0.8	<2	2.4	3.6	<1	3.8	493	0.003	<2
	B	N_2 mixing	39	0.8	<2	3.9	3.9	<1	3.7	424	0.001	<2
Norwalk - Site 1 (Bioassay Elutriates)												
EPA RSW	A	-	25	2.4	<2	25.5	10.4	2.5	29.4	-	-	-
	B	-	63	2.3	<2	16.5	8.7	2.2	26.1	-	-	-
5% Oxic	A	5% sediment	275	2.8	<2	75.8	13	26.3	41.3	-	-	-
	B	5% sediment	413	2.9	<2	70	13.4	23.8	50.6	-	-	-
20% Oxic	A	-	925	2.6	<2	59.4	12.5	20.6	34.3	-	-	-
	B	-	1225	3.4	<2	58.1	12.1	20.8	31.9	-	-	-
Norwalk - Site 2												
Site Water		-	88	1.2	<2	3.7	3.3	<1	3	<5	<0.001	<2
20% Oxic		-	19	0.9	<2	1.7	3.9	<1	0.8	16	0.003	<2
20% Anoxic		N_2 mixing	19	0.8	<2	2.9	3.3	<1	1.3	303	0.003	<2
Norwalk - Site 2 (Bioassay Elutriates)												
EPA RSW	A	-	19	4.7	<2	47.1	58.9	28.1	55.4	33	<0.001	-
	B	-	10	5	<2	76.5	84.6	28.8	45.3	49	<0.001	-
5% Oxic	A	5% sediment	354	2.1	<2	76.5	14.7	22.2	61.2	3747	0.200	-
	B	5% sediment	380	2.3	<2	70.6	16.7	22.9	61.2	3544	0.200	-
20% Oxic	A	-	532	1.7	<2	47.1	11.1	13.3	33.4	3342	0.150	-
	B	-	582	1.7	<2	52.9	11.5	13.2	28.2	3139	0.096	-
5% ST		Stirred	88	1.8	<2	82.4	12.7	17.7	45.6	3291	0.188	-
20% ST		Stirred	380	1.9	<2	52.9	11.5	12.9	25.9	2127	0.200	-

(Continued)

Table A2 (Continued)

Sample Designation	Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
ASHTABULA											
Ashtabula											
Site Water	-	<20	1.8	-	9.1	-	4.2	10.6	7	-	-
5% Oxic (1 hr)	5% sediment	160	1.8	-	6.3	-	4	9.5	14	-	-
(24 hr)	5% sediment, 24 hr. settle	170	1.7	-	5.4	-	4.3	9.2	13	-	-
10% Oxic (1 hr)	10% sediment	180	1.7	-	5.5	-	3.7	8.8	23	-	-
(24 hr)	10% sediment, 24 hr. settle	200	1.4	-	6.4	-	3.5	9.3	14	-	-
20% Oxic (1 hr)		280	1.7	-	10.4	-	3.7	11.8	26	-	-
(24 hr)	24 hr. settle	340	1.7	-	6.3	-	4.3	11	35	-	-
LAKE SUPERIOR											
Lake Superior											
Site Water	-	1	<0.2	-	1.6	<1	<1	1.9	5	-	-
20% SH	Shaken	50	-	-	1.5	-	-	3.4	5	-	-
Duluth Harbor											
Site Water	-	1	<0.2	-	1.6	<1	<1	1.9	5	-	-
20% SH	Shaken	225	-	-	0.7	-	-	2	31	-	-
MENOMINEE RIVER											
Menominee River - Site 1											
Site Water	-	<10	<0.5	<1	8	7.8	<1	1	25	0.03	<40
20% Oxic A	-	44	0.6	<1	<1	4.7	8	5.1	452	0.06	860
B	-	28	<0.5	1.1	<1	6.5	3	2	102	0.04	800
20% Anoxic A	N_2 mixing	240	<0.5	<1	6	2.3	<1	<1	458	0.03	920
B	N_2 mixing	76	<0.5	1.3	19	5.9	<1	1.6	171	0.04	910
Plop A	Plop test procedure followed	<10	0.5	1	<1	5.4	<1	<1	98	0.03	460
B	Plop test procedure followed	27	<0.5	<1	7	8.7	2	1.9	83	0.03	373

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Menominee River - Site 2												
Site Water		-	<10	<0.5	<1	19	2.4	<1	2.3	16	0.03	50
20% Oxic	A	-	25	0.5	<1	<1	6	<1	2.3	131	0.04	640
	B	-	17	0.5	<1	<1	4.8	<1	1.4	28	0.05	1020
20% Anoxic	A	N_2 mixing	<10	0.5	<1	3	7.7	2	<1	144	0.03	850
	B	N_2 mixing	<10	0.6	<1	<1	8.2	<1	2.3	66	0.04	750
Plop	A	Plop test procedure followed	48	<0.5	<1	11	5	<1	1.3	92	0.07	600
	B	Plop test procedure followed	<10	<0.5	<1	16	5.5	<1	0.7	29	0.03	610
FOUNDRY COVE												
Foundry Cove												
Site Water		-	32	7.4	<2	126.1	8.7	1	3	12	0.050	23.8
20% Oxic	A	-	26	5	<2	19.4	17.4	1	1.3	<5	0.035	9.1
	B	-	35	5.4	<2	3.1	19.8	<1	1.7	4	0.035	2.7
20% Anoxic	A	N_2 mixing	17	2.8	<2	5.7	18.2	<1	2.7	6	0.015	0
	B	N_2 mixing	35	3.5	<2	<1	17.4	<1	2	10	0.035	0
WES LAKE												
Wes Lake												
Site Water		-	70	1.6	<2	13.9	5.6	5.2	4.4	21	0.029	<2
20% Oxic	A	-	2800	1.1	<2	26.8	7	4.6	3.1	79	0.029	<2
	B	-	7050	1.1	<2	24.4	7	5.6	3.4	94	0.127	<2
WILMINGTON												
Wilmington												
Site Water		-	172	1.5	<2	37.7	6.8	3.6	6.2	43	0.029	<2
20% Oxic	A	-	172	1.6	<2	25.2	9.2	2.6	4.5	39	0.029	<2
	B	-	142	1.6	<2	38.5	10.1	3.6	4.9	44	0.029	<2

(Continued)

Table A2 (Continued)

Sample Designation	Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
MOBILE BAY											
Mobile Bay (August 1974)											
Site Water	-	650	<0.5	-	7.2	-	1.5	3.7	13	-	-
20% SH	Shaken	1430	<0.5	-	4.4	-	1.9	4.2	27	-	-
15% SH	Shaken	1310	<0,5	-	4	-	1.7	3.6	21	-	-
10% SH	Shaken	1540	<0.5	-	4	-	1.5	4	14	-	-
5% SH	Shaken	860	<0.5	-	4.2	-	2.1	4.1	13	-	-
Site Water	-	15	<0.3	-	7.9	-	<2	1.2	3	-	-
20% SH	15 min. SH	1510	1.0	-	9.6	-	<2	0.6	97	-	-
20% SH	30 min. SH	1540	<0.3	-	4	-	<2	<0.5	25	-	-
20% SH	60 min. SH	1540	0.4	-	5.8	-	<2	<0.5	12	-	-
20% SH	90 min. SH	1370	0.3	-	7.8	-	<2	0.75	16	-	-
Site Water	-	110	0.3	-	7.2	-	1.8	1.1	3	-	-
20% ST	Stirred	710	<0.3	-	1.9	-	2	<0.5	2690	-	-
20% SH	Shaken	880	<0.3	-	1.8	-	2.3	<0.5	41	-	-
20% ST	Stirred	680	<0.3	-	1.5	-	1.7	<0.5	4500	-	-
Mobile Bay - Site 1											
Site Water	-	88	1.4	<2	12	5.5	4.3	2.6	60	0.03	<2
5% Oxic A	5% sediment	138	1	<2	13	5.5	3.7	1.8	9	0.03	<2
B	5% sediment	75	1.1	<2	11	4.3	3.9	1.8	5	0.03	<2
20% Oxic A	-	2900	1.3	<2	24	4.7	5.2	2	16	0.03	<2
B	-	9000	1.4	<2	20	5.2	5.2	1.9	29	0.03	<2
20% Anoxic A	N_2 mixing	9025	1.3	<2	20	4.8	5.7	2.3	23	0.03	<2
B	N_2 mixing	9050	1.1	<2	32	3.2	5.1	2	22	0.12	<2

(Continued)

Table A2 (Continued)

Sample Designation	Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Mobile Bay - Site 2											
Site Water	-	294	1.4	<2	182	6.8	6.4	3.2	44	0.05	-
5% Oxic A	5% sediment	507	1.3	<2	149	6.4	6	2.9	18	0.03	-
B	5% sediment	266	1.4	<2	176	6.1	6	3.6	16	0.03	-
Site Water	-	182	1.3	<2	17	6.2	5.1	3.3	63	0.03	-
20% Oxic A	-	1019	1.2	<2	15	7.7	4.8	2.4	25	0.03	-
B	-	2688	1.4	<2	12	7.1	6.1	2.1	23	0.03	-
20% Anoxic A	N_2 mixing	4675	1.3	<2	11	6.5	6.7	2.2	18	0.03	-
B	N_2 mixing	2132	1.2	<2	38	7.3	5.7	2.4	18		-
Mobile Bay - Site 2 (Bioassay Elutriates)											
EPA RSW A	-	114	1.8	<2	10	18.5	5.7	13.4	14	-	-
B	-	160	1.5	<2	10	17.5	7	13.1	12	-	-
5% Oxic A	5% sediment	251	2.6	<2	30.7	8.7	7	4.4	2400	-	-
B	5% sediment	327	1.9	<2	14.4	8.7	6.3	4.4	2023	-	-
20% Oxic A	-	6138	1.3	<2	7.7	5.8	5.7	2.5	874	-	-
B	-	6282	1.5	<2	37.7	6.9	8.3	3	1241	-	-
APALACHICOLA											
Apalachicola - Site 1											
EPA RSW	-	8	0.8	<2	24.7	8.3	2.6	11.7	3	<0.001	<2
20% Oxic A	RSW used	7000	<0.5	<2	5.2	2.6	1.6	6.5	26	0.022	<2
B	RSW used	3800	<0.5	<2	3.9	2.8	1.6	8.6	93	0.015	<2
20% Anoxic/Oxic A	RSW used; 15 min. N_2 + 15 min. air mixing	1800	0.6	<2	1.8	3.4	1.4	6.8	47	<0.001	<2
B	RSW used; 15 min. N_2 + 15 min. air mixing	2450	0.6	<2	10	2.6	0.7	7.2	95	<0.001	<2

(Continued)

Table A2 (Continued)

Sample Designation	Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Apalachicola - Site 1 (Bioassay Elutriates)											
EPA RSW A	-	58	1.5	<2	10.7	8.6	1.3	5.3	0.29	-	-
B	-	27	1.5	<2	11.5	9.6	1.3	6.5	0.16	-	-
5% Oxic A	5% sediment	1900	<0.5	<2	7.9	6.1	1.9	3.2	2100	-	-
B	5% sediment	2100	<0.5	<2	7.4	6.1	6.3	3.8	2200	-	-
20% Oxic A	-	4400	0.7	<2	5.2	6.7	2.5	1.4	1000	-	-
B	-	4500	0.6	<2	5.6	5.8	8.2	1.4	1600	-	-
Apalachicola - Site 2											
EPA RSW	-	7	1.6	<2	12.2	12	2.3	4.6	8	0.042	<2
20% Oxic	RSW used	78	1.1	<2	13.8	4.2	2.2	2.8	181	0.028	<2
20% Anoxic/oxic A	RSW used; 15 min. N_2 + 15 min. air mixing	78	1.1	<2	10	4.7	1.8	3.4	22	0.055	<2
B	RSW used; 15 min. N_2 + 15 min. air mixing	88	1.1	<2	10.5	4.8	1.5	5.9	21	0.057	<2
Apalachicola - Site 3											
Surface Site Water	-	49	1	<2	20	5.8	1.3	2.6	491	0.020	<2
20% Oxic	-	76	1.1	<2	2.3	3.8	1	1.9	43	0.020	<2
20% Anoxic	N_2 mixing	217	1.1	<2	17.8	5.8	1.5	1.8	73	0.030	<2
Bottom Site Water	-	133	3.1	<2	6.2	3.5	1.4	2.6	12	0.020	<2
20% Oxic	-	1249	1.8	<2	7.1	4.1	2	2.9	18	0.020	<2
20% Anoxic	N_2 mixing	1166	1.6	<2	5.8	4.5	2.1	4.8	41	0.020	<2
Apalachicola - Site 4											
Surface Site Water	-	49	1	<2	20	5.8	1.3	2.6	491	0.020	<2
20% Oxic	-	163	1.2	<2	16.8	3.7	1.4	2.1	91	0.020	<2
20% Anoxic	N_2 mixing	170	1.2	<2	7.7	5.6	1.4	2.7	47	0.020	<2
Bottom Site Water	-	133	3.1	<2	6.2	3.5	1.4	2.6	12	0.020	<2
20% Oxic	-	583	1.5	<2	5.8	2.9	1.2	4.4	21	0.020	<2
20% Anoxic	N_2 mixing	512	1.6	<2	4.7	2.9	1.5	2.9	31	0.020	<2

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd*	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Apalachicola - Site 5												
Surface Site Water		-	49	1	<2	20	5.8	1.3	2.6	491	0.020	<2
20% Oxic		-	223	1.1	<2	4.5	5	1.3	2.3	49	0.020	<2
20% Anoxic		N_2 mixing	284	1.2	<2	17.3	5.1	1.3	3	75	0.020	<2
Bottom Site Water		-	133	3.1	<2	6.2	3.5	1.4	2.6	12	0.020	<2
20% Oxic		-	849	1.9	<2	13.8	3.8	1.2	4	22	0.020	<2
20% Anoxic		N_2 mixing	450	1.9	<2	6.8	4.1	1.4	3.5	25	0.020	<2
Apalachicola - Site 5 (Bioassay Elutriates)												
EPA RSW	A	-	19	4.7	<2	47.1	58.9	28.1	55.4	33	<0.001	-
	B	-	10	5	<2	76.5	84.6	28.8	45.3	49	<0.001	-
5% Oxic	A	5% sediment	1696	1.6	<2	12.4	8.8	5.2	5.6	3747	<0.001	-
	B	5% sediment	1722	1.8	<2	12.9	7.6	4.4	8.3	2987	<0.001	-
20% Oxic	A	-	2759	1.6	<2	10	7.2	4.4	3.7	2987	<0.001	-
	B	-	2810	1.6	<2	15.4	7.2	4.8	4.3	3089	<0.001	-
UPPER MISSISSIPPI												
Upper Mississippi Site Water		-	84	0.8	<2	7.1	7.6	3.2	3	15	0.025	<2
5% Oxic	A	5% sediment	236	0.8	<2	11.1	9	3.5	3.2	25	0.035	3.7
	B	5% sediment	175	0.8	<2	12.3	9	3.8	3.4	190	0.015	2.1
20% Oxic	A	-	548	0.8	<2	6.6	8.6	3.4	3.6	195	0.035	<2
	B	-	1134	0.8	<2	7.2	8.9	3.8	2.8	70	0.025	4.2
Upper Mississippi (Bioassay Elutriates)												
EPA RSW	A	-	<10	<0.5	<2	4	7	5	5	262	-	-
	B	-	<10	<0.5	<2	4	7	5	5	263	-	-
5% Oxic	A	5% sediment	55	11.8	<2	52	18	50	66	1137	-	-
	B	5% sediment	56	12.8	<2	52	19	49	66	1138	-	-
20% Oxic	A	-	168	21.1	<2	69	25.6	76	87	1175	-	-
	B	-	167	21.5	<2	69	25.9	77	87	1175	-	-

(Continued)

Table A2 (Continued)

Sample Designation	Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
JAMES RIVER											
James River											
Site Water	-	40	<0.05	<2	5.4	<2	2.1	<1	29	<0.005	2.1
5% Oxic A	5% sediment	44	<0.05	<2	21	3.9	2.9	<1	112	<0.005	3.2
B	5% sediment	177	<0.05	3.1	6.4	3.4	2.3	<1	36	<0.005	4.2
20% Oxic A	-	78	<0.05	<2	4.4	5.7	4.8	<1	54	0.009	3.7
B	-	150	<0.05	2.8	52.8	6.5	4.8	1.0	75	<0.005	4.7
20% Anoxic A	N_2 mixing	358	<0.05	<2	<1	16.5	5	1.6	578	<0.005	5.3
B	N_2 mixing	140	<0.05	<2	1.7	3.4	2.5	1.3	685	0.010	3.2
James River (Bioassay Elutriates)											
EPA RSW A	-	145	0.7	<2	59	6.5	3.1	4	10	-	-
B	-	145	0.7	<2	68	6.9	3.9	4	8	-	-
5% Oxic A	5% sediment	99	1.1	<2	522	11.2	55.9	176	982	-	-
B	5% sediment	129	0.9	<2	438	9.5	54.6	153	798	-	-
20% Oxic A	-	129	1.4	<2	799	13.4	96.9	192	1196	-	-
B	-	145	1.7	<2	859	15.5	94.6	193	1089	-	-
BAILEY'S CREEK											
Bailey's Creek											
Site Water	-	40	<0.05	<2	5.4	<2	2.1	<1	29	<0.005	2.1
5% Oxic A	5% sediment	37	<0.05	3.8	4.3	2.9	3.9	1.3	67	0.007	5.3
B	5% sediment	64	<0.05	<2	1.4	<2	3.3	2.9	81	<0.005	4.2
20% Oxic A	-	44	<0.05	<2	<1	2.3	2.5	<1	91	<0.005	4.2
B	-	100	<0.05	<2	2.9	2.2	3.1	1.5	19	<0.005	<2
20% Anoxic A	N_2 mixing	109	<0.05	<2	1.5	3.4	13.1	7.1	204	0.007	2.4
B	N_2 mixing	41	<0.05	<2	4.2	3.6	6.4	2	160	0.007	5.3

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Bailey's Creek (Bioassay Elutriates)												
EPA RSW	A	-	8	<0.5	<2	2.2	2.9	3.1	2.7	263	-	-
	B	-	8	<0.5	<2	2.9	2.6	1.9	2.6	252	-	-
5% Oxic	A	5% sediment	192	1.9	3.5	103	17	44.4	33.7	1202	-	-
	B	5% sediment	205	1.7	3.6	102	17.6	43.2	31	1110	-	-
20% Oxic	A	-	305	2.8	6.1	100	22.3	43.2	31.8	1095	-	-
	B	-	305	2.5	6.5	117	23.5	40.8	28.2	1024	-	-
TEXAS GULF COAST												
GBEC - Buoy 1												
Buoy 1 Site Water		-	2.6	1.2	1.6	3.5	2.9	<1	1.9	5	0.006	-
5% Oxic		5% sediment	1165	1.2	2.3	9.2	4.6	<1	2.5	7	0.0018	-
20% Oxic		-	370	1.2	2.6	7.3	5.8	<1	2.9	103	0.026	-
GBEC - Buoy 1 (Bioassay Elutriates)												
EPA RSW	A	-	0.1	0.2	2	8	13.5	1.6	3.4	10	<0.001	-
	B	-	0.1	0.2	2	8	13.4	1.7	3.4	10	<0.001	-
5% Oxic	A	5% sediment	140	0.3	2.1	17	13.7	1.6	7.2	162	0.004	-
	B	5% sediment	140	0.5	2.1	17	13.6	1.6	5	229	0.002	-
10% Oxic	A	10% sediment	190	0.7	<2	9	13	1.6	3.8	144	<0.001	-
	B	10% sediment	190	0.7	<2	9	13	1.6	3.4	143	<0.001	-
20% Oxic	A	-	2310	1.3	2.1	9	13	1.6	2.6	117	<0.001	-
	B	-	2310	1.0	2.1	12	13.8	1.6	3	175	<0.001	-
GBEC Disposal Site Water			40.3	<0.3	8	12.4	-	4	3.7	<1	0.041	-
GBEC - Buoy 9												
Buoy 9 Site Water		-	17.2	<0.3	<2	12.4	-	2.1	2.5	<1	0.069	-
5% Oxic (Buoy 9 Water)		5% sediment	1010	<0.3	10	5	-	2	<1	<1	0.034	-
5% Oxic (Disposal Site Water)		5% sediment	467	<0.3	5.1	13.7	-	3.1	<1	<1	0.034	-
20% Oxic (Disposal Site Water)		-	1780	<0.3	5.9	15.4	-	1.1	<1	<1	0.028	-

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
GBEC - Buoy 9 (Bioassay Elutriates)												
EPA RSW	A	-	< 0.1	0.2	2	8	13.5	1.6	3.4	10	<0.001	-
	B	-	< 0.1	0.2	2	8	13.5	1.6	3.4	10	<0.001	-
10% Oxic	A	10% sediment	60	1.5	0.6	17	13.7	1.8	9.8	370	0.004	-
	B	10% sediment	60	1.8	0.6	16	13.8	1.8	9.3	405	0.002	-
GBEC - Buoy 11												
Buoy 11 Site Water		-	48	<0.3	<2	16	-	4	1.4	<1	0.021	-
5% Oxic (Buoy 11 Water)		5% sediment	1730	<0.3	6	8.3	-	5	<1	<1	0.020	-
5% Oxic (Disposal Site Water)		5% sediment	1840	<0.3	4	12.8	-	4	1	<1	0.021	-
20% Oxic (Disposal Site Water)		-	5050	<0.3	4.5	13.7	-	5	<1	<1	0.014	-
GBEC - Buoy 11 (Bioassay Elutriates)												
EPA RSW	A	-	410	<0.1	4.9	17.9	16.5	<1	3.8	5	-	-
	B	-	410	<0.1	4.7	18.1	16.5	<1	3.8	6	-	-
5% Oxic	A	5% sediment	1000	<0.1	10.2	9.1	9.5	<1	2.2	22	-	-
	B	5% sediment	1000	<0.1	10.2	8.5	8.4	<1	2.7	31	-	-
10% Oxic	A	10% sediment	1100	<0.1	10.2	12.5	13.5	<1	4.3	28	-	-
	B	10% sediment	1100	<0.1	10.2	9	10.5	<1	3.2	28	-	-
20% Oxic	A	-	4600	<0.1	10.2	11.6	10.5	<1	3.8	41	-	-
	B	-	3800	<0.1	10.2	15.8	10.5	<1	4.3	94	-	-
Texas City Channel-Site 1												
Site Water		-	52	1.8	11.1	13.3	4.1	3.6	5.4	5	0.27	-
5% Oxic	(1 hr)	5% sediment	262	5.5	108.4	4.8	11.4	3.9	4.6	18	0.08	-
	(24 hr)	5% sediment, 24 hr. settle	551	3.8	41.4	5.1	9.5	4.1	4.2	15	0.01	-
20% Oxic	(1 hr)	-	626	5.9	159	4.7	14.7	4.5	2.2	20	0.22	-
	(24 hr)	24 hr. settle	1063	3.9	134.9	3.2	9.5	5.2	1.8	27	0.10	-

(Continued)

Table A2 (Continued)

Sample Designation	Modifications[**]	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Texas City Channel-Site 1 (Storage Study)											
Site Water	70 day storage	2	0.9	2.9	8.2	5.3	<1	3	8	0.009	-
5% Oxic	5% sediment, 70 day storage	269	1.1	2.6	9.5	6.3	<1	3.3	35	0.026	-
20% Oxic	70 day storage	539	1	3.7	10.9	9.5	<1	3.5	20	0.035	-
Site Water	72 day storage	209	1.8	<1	13.5	4.3	8.3	2.5	6	-	-
20% Anoxic/Oxic	20 min. N_2 + 10 min. air mixing, 72 day storage	1551	2.3	<1	9.1	18.6	11.2	2.6	15	-	-
25% Anoxic/Oxic	25% sediment, 20 min. N_2 + 10 min. air mixing, 72 day storage	475	2.1	<1	14	18.5	9.8	1.7	19	-	-
Texas City Channel-Site 1 (Bioassay Elutriates)											
EPA RSW A	-	400	<0.1	4.7	17.9	16.5	<1	3.8	5	-	-
B	-	400	<0.1	4.7	18	16.5	<1	3.7	5	-	-
5% Oxic A	5% sediment	2400	<0.1	6.1	5.9	8.4	<1	2.16	29	-	-
B	5% sediment	2500	<0.1	7.4	5.7	8.4	<1	2.16	56	-	-
10% Oxic A	10% sediment	6200	<0.1	7.4	5.6	10.5	<1	2.7	33	-	-
B	10% sediment	6200	<0.1	8.8	4.6	10.5	<1	2.7	45	-	-
20% Oxic A	-	8600	<0.1	8.8	5.6	11.5	<1	3.2	75	-	-
B	-	8300	<0.1	8.8	4.6	12.5	<1	3.7	91	-	-
Texas City Channel-Site 2											
Site Water	-	5	<0.3	<2	8.7	7.8	<1	<1	<1	0.028	-
5% Oxic	5% sediment	396	<0.3	10.3	14.3	12.9	2	<1	2	0.048	-
10% Oxic	-	472	<0.3	3.7	10.7	9.8	2	<1	2	0.014	-

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Texas City Channel-Site 2 (Bioassay Elutriates)												
EPA RSW	A	-	1100	0.9	0.6	11.6	7.5	6.3	5.8	1	2	-
	B	-	1100	0.9	0.6	11.6	7.5	6.3	5.8	1	2	-
5% Oxic	A	5% sediment	79000	0.8	0.3	9.5	<0.1	5.2	0.9	9	0.4	-
	B	5% sediment	82000	0.4	<0.1	9.2	<0.1	5.7	0.9	8	0.4	-
10% Oxic	A	10% sediment	94000	0.8	0.3	9.7	<0.1	6.8	2.9	3	0.5	-
	B	10% sediment	86000	0.4	0.1	8.9	0.8	6.5	2.9	4	0.5	-
20% Oxic	A	-	19000	0.7	<0.1	18.5	6	3	2.9	10	0.5	-
	B	-	22000	0.7	0.3	7.8	6	4.6	2.6	8	0.5	-
Texas City Channel-Site 3												
Site Water		-	12	<0.3	<2	19.3	4.6	1	1.7	<1	0.099	-
5% Oxic		5% sediment	3970	<0.3	<2	15.7	6.9	3.2	<1	<1	0.034	-
20% Oxic		-	5760	<0.3	4	17	8.1	1.6	1.2	<1	0.034	-
Texas City Channel-Site 3 (Bioassay Elutriates)												
EPA RSW	A	-	1100	0.9	0.6	11.6	7.5	6.3	5.8	1	2	-
	B	-	1100	0.9	0.6	11.6	7.5	6.3	5.8	1	2	-
5% Oxic	A	5% sediment	107000	0.2	<0.1	13.4	4.5	7.3	4.6	27	0.4	-
	B	5% sediment	100000	0.3	<0.1	12.6	5.3	7.9	4.1	5	0.4	-
10% Oxic	A	10% sediment	22000	0.5	<0.1	12.7	<0.1	6.5	4.1	4	0.4	-
	B	10% sediment	23000	0.3	<0.1	12.3	1.5	6.8	3.8	4	0.4	-
20% Oxic	A	-	37000	0.7	<0.1	8.9	8.3	5.5	3.5	4	0.6	-
	B	-	48000	0.6	<0.1	9.1	3.8	6.3	4.1	5	0.6	-
Texas City Channel-Site 4												
Site Water		-	74	1.1	<2	1.3	4.9	7.2	5.7	5	0.007	<2
5% Oxic		5% sediment	315	1.5	<2	1.4	6.8	8.6	4.2	12	0.009	<2
20% Oxic		-	1200	1.5	<2	<1	5.4	8.6	3.8	10	0.007	<2

(Continued)

Table A2(Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Texas City Channel-Site 4 (Storage Study)												
Site Water		40 day storage	67	1	<2	21.1	43	5.6	5.9	8	0.010	<2
20% Oxic	A	40 day storage	1650	1.3	<2	16.8	35	5.6	6.6	15	0.010	<2
	B	40 day storage	3000	<0.5	<2	20.6	45	6.2	7.1	17	0.015	<2
Texas City Channel-Site 4 (Bioassay Elutriates)												
EPA RSW	A	-	84	<2.7	2	16.3	8.2	13.8	12.1	11	0.018	-
	B	-	68	<2.7	2	19.5	8.5	16.3	15.3	11	0.028	-
5% Oxic	A	5% sediment	700	<2.5	2	39	8.8	27.5	14.7	2500	0.018	-
	B	5% sediment	700	<3	2	53.2	9.7	31.3	24.7	3000	0.025	-
10% Oxic	A	10% sediment	700	<2.6	2	40.8	9.1	30	18.2	3300	0.018	-
	B	10% sediment	800	<2.6	2	66	9.4	31.3	22	3300	0.023	-
20% Oxic	A	-	1000	<2.7	2	32.3	9.1	30	14.2	2800	0.075	-
	B	-	1200	<2.6	2	44	9.1	31.3	15.3	2600	0.015	-
Texas City Channel-Site 5												
Site Water		-	68	0.8	<2	1.3	5.1	15	5.7	10	0.003	<2
5% Oxic		5% sediment	2750	0.6	<2	<1	8.2	13.9	3.6	10	0.003	<2
20% Oxic		-	1200	0.6	<2	<1	7.1	16.4	6.4	16	0.009	<2
Texas City Channel-Site 5 (Storage Study)												
Site Water		40 day storage	50	1	<2	20.1	40	20	9.3	6	0.013	<2
20% Oxic		40 day storage	1950	0.7	<2	22.8	37.5	5.9	8.1	10	0.025	<2

(Continued)

Table A2 (Continued)

Sample Designation	Modifications[**]	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Texas City Channel-Site 5 (Bioassay Elutriates)											
EPA RSW A	-	84	2.7	<2	16.3	8.2	13.8	12.5	11	0.018	-
B	-	68	2.7	<2	19.5	8.5	16.3	15.3	11	0.028	-
5% Oxic A	5% sediment	1700	3.2	<2	34.8	7	20	10.9	7000	0.028	-
B	5% sediment	600	2.7	<2	35.8	7.9	20	14.9	3000	0.015	-
10% Oxic A	10% sediment	1000	2.6	<2	41.9	9.7	20	14	2500	0.013	-
B	10% sediment	1000	2.5	<2	47.5	8.5	20	13.6	2700	0.018	-
20% Oxic A	-	1700	2.5	<2	60	7	25	16.2	2500	0.018	-
B	-	1700	2.6	<2	57.8	7.6	25	21.3	3900	0.018	-
Texas City Channel-Site 6											
Site Water	-	51	1.4	<2	3	4.8	9.7	5.4	5	0.007	<2
5% Water	5% sediment	905	0.7	<2	<1	5.3	10.7	4.1	8	0.007	<2
20% Oxic	-	1600	0.5	<2	<1	5.7	12.1	3.7	20	0.007	<2
Texas City Channel-Site 6 (Storage Study)											
Site Water	40 day storage	50	1	<2	19.9	40	5	12.6	5	0.007	<2
20% Oxic	40 day storage	6100	1.3	<2	17	30	5.9	7.1	63	0.018	<2
Texas City Channel-Site 6 (Bioassay Elutriates)											
EPA RSW A	-	270	2.8	3	23.7	19.1	17.8	12.7	27	0.023	-
B	-	248	2.3	3	22.4	19.1	17.8	14	17	0.013	-
20% Oxic A	-	3700	1.1	<2	24.9	14.3	12.4	12	227	0.018	-
B	-	3500	2.4	<2	26.9	17.4	16.5	6.7	143	0.015	-

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Houston Ship Channel (August 1974)												
Site Water		-	47	<0.5	-	13.8	-	1.5	1.8	35	-	-
20% Oxic	A	-	179	<0.5	-	5.5	-	1.3	1.9	55	-	-
	B	-	263	0.5	-	3.8	-	1.3	1.9	45	-	-
	C	-	255	0.5	-	3.7	-	1.8	1.9	32	-	-
	D	-	242	0.7	-	3.1	-	1.7	1.9	30	-	-
Site Water		-	230	1.8	-	22	-	2.2	8	40	-	-
20% Anoxic	A	N_2 mixing	710	2.0	-	10	-	10.8	11	7000	-	-
	B	N_2 mixing; air contamination	360	1.3	-	15.3	-	1.3	21.6	68	-	-
20% Oxic		-	330	0.9	-	8.6	-	2.	4.6	48	-	-
Houston Ship Channel-Site 1												
Site Water		-	76	1.1	<2	35.2	13.6	4.8	5	339	0.047	3.6
5% Oxic	A	5% sediment	130	0.7	<2	20.1	10.2	4.8	1.4	29	0.047	2.1
	B	5% sediment	121	0.7	<2	19.3	10.3	4.8	1.6	23	0.047	<2
20% Oxic	A	-	57	<0.5	<2	14.5	12	4.3	3.3	274	0.047	2.3
	B	-	43	<0.5	<2	48.2	12.7	5.1	1.6	210	0.047	3
20% Anoxic	A	N_2 mixing	179	<0.5	<2	14.6	12	4.3	1.4	64	0.047	2.8
	B	N_2 mixing	195	<0.5	<2	17.1	12.5	4.8	1.5	52	0.047	2.8
Houston Ship Channel-Site 2												
Site Water		-	352	0.6	<2	182	8.6	4.3	4.4	39	0.047	2.3
5% Oxic	A	5% sediment	646	0.6	<2	27.5	11.3	3.1	4	26	0.047	2.3
	B	5% sediment	92	0.6	<2	16.4	8.6	5.4	2.5	786	0.057	2.3
20% Anoxic	A	N_2 mixing	218	0.7	<2	17.5	7.5	4.3	2.4	181	0.047	3.9
	B	N_2 mixing	345	1.0	<2	16.6	8.3	4.9	2.2	144	0.047	3.8
20% Oxic	A	5 replicate tests made	1340	0.7	<2	143.9	8.4	3.6	2.3	42	0.248	2.5
	B	-	929	0.6	<2	28.2	13.8	3.8	4.6	68	0.248	2.7
	C	-	943	0.9	<2	45.5	12.9	4	7.2	48	0.191	2.5
	D	-	1301	0.8	<2	44.8	7.4	3.8	5.5	37	0.048	2.6
	E	-	1260	1.0	<2	38.1	6.8	4.3	5	31	0.048	2

(Continued)

Table A2 (Continued)

Sample Designation	Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Houston Ship Channel-Site 2 (continued)											
Site Water	-	78	1.2	<2	52.6	8	12.2	4	22	0.047	<2
Plop Test (2ℓ)	Plop test procedure followed, 2ℓ tot. vol.; 1 hr. mixing; 30 min. settle	346	1	<2	20.9	12.3	14.5	3	43	0.047	<2
(4ℓ)	Plop test procedure followed, 4ℓ tot. vol.; 1 hr. mixing; 30 min. settle	188	1.3	<2	23.9	11.6	10.7	2.2	29	0.154	<2
(8ℓ)	Plop test procedure followed, 8ℓ tot. vol.; 1 hr. mixing; 30 min. settle	362	1.2	<2	21.6	13.6	12.2	2.5	33	0.047	<2
Houston Ship Channel (Bioassay Elutriates)											
EPA RSW A	-	132	2.7	<2	158	23.3	24.5	32.3	39	0.004	-
B	-	152	2.8	<2	178	25	24.5	40.3	41	0.004	-
5% Oxic A	5% sediment	192	1.8	<2	112	16.8	17.3	13	899	0.004	-
B	5% sediment	192	2	<2	160	21.8	18	14.2	1105	0.004	-
20% Oxic A	-	233	3.1	<2	142	18.8	16.6	13.7	1825	0.004	-
B	-	213	1.6	<2	124	15.3	12.2	10.3	2504	0.004	-
Houston Ship Channel-Site 3											
Site Water	-	33	1	<2	14.8	10.5	3	3.1	6	0.047	2
5% Oxic A	5% sediment	1857	1	<2	27.6	10.8	2.7	2.6	<5	0.047	2
B	5% sediment	1654	0.9	<2	11.1	11	2.4	2.2	<5	0.154	2.2
20% Oxic A	-	3914	1	<2	14.9	12.6	3.4	2.9	33	0.047	<2
B	-	4284	0.9	<2	17.8	10	4.6	1.9	50	0.047	2.2
20% Anoxic A	N_2 mixing	4432	0.8	<2	10.7	6.9	4	1.8	15	0.047	2.6
B	N_2 mixing	4951	0.6	<2	10.7	7.6	4	1.7	429	0.154	2.3

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Houston Ship Channel-Site 3 (Bioassay Elutriates)												
EPA RSW	A	-	111	2.4	<2	141	32.5	12.2	22.1	38	0.004	-
	B	-	132	2.5	<2	90	34.2	14.4	22.6	39	0.004	-
5% Oxic	A	5% sediment	11366	1.8	<2	73	19.5	7.2	8.4	1023	0.004	-
	B	5% sediment	11342	2.1	<2	150	20.8	10.1	9.9	1064	0.004	-
20% Oxic	A	-	9563	3.1	<2	169	14.3	5.8	9.5	406	0.004	-
	B	-	9648	2.4	<2	137	16	7.2	8.8	961	0.004	-
Morgan's Point												
Site Water		-	4.1	0.3	2	14.9	2.6	<1	1.3	12	0.041	-
5% Oxic		5% sediment	1920	<0.3	5	15.6	4	<1	<1	3	0.028	-
20% Oxic		-	507	<0.3	6	14.8	1.6	1.3	<1	2	0.086	-
Morgan's Point (Bioassay Elutriates)												
EPA RSW	A	-	5	-	-	-	-	-	-	-	-	-
	B	-	5	-	-	-	-	-	-	-	-	-
5% Oxic	A	5% sediment	747	-	-	-	-	-	-	-	-	-
	B	5% sediment	300	-	-	-	-	-	-	-	-	-
10% Oxic	A	10% sediment	470	-	-	-	-	-	-	-	-	-
	B	10% sediment	731	-	-	-	-	-	-	-	-	-
20% Oxic	A	-	1059	-	-	-	-	-	-	-	-	-
	B	-	959	-	-	-	-	-	-	-	-	-
Galveston Channel												
Site Water		-	72	1.3	<2	10.7	9.3	6.3	7.3	4	0.007	<2
20% Oxic		-	1900	1.2	<2	69.3	12.2	7.5	3.9	3	0.071	<2

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Trinity River												
Site Water		-	17.	1.1	-	15.9	-	2.2	2.9	<5	-	-
20% SH	A	Shaken	87	0.9	-	17.5	-	3.6	7.2	8	-	-
	B	Shaken	91	1.6	-	20	-	3.8	2.9	74	-	-
Port Lavaca												
Site Water		-	90	1	<2	17.7	2.6	2.4	4	25	0.016	-
5% Oxic		5% sediment	450	1	<2	19.4	4.7	5.9	3.7	74	0.019	-
20% Oxic		-	550	0.8	<2	18	4.5	6.6	3.2	22	0.037	-
5% Anoxic/Oxic	A	5% sediment; 20 min. N_2+ 10 min. air mixing	450	1.2	<2	20.4	4.7	5	3.7	189	0.033	-
	B	5% sediment; 20 min. N_2+ 10 min. air mixing	600	1	<2	17.3	3.7	5.6	2.6	40	0.007	-
20% Anoxic/Oxic	A	20 min. N_2 + 10 min. air mixing	610	1	<2	16.2	5.6	5.7	3.6	38	0.017	-
	B	20 min. N_2 + 10 min. air mixing	980	0.8	<2	18.5	2.6	3.9	1.9	21	0.015	-
Port Lavaca (Bioassay Elutriates)												
EPA RSW	A	-	50	<0.3	<2	18	12.2	1.2	6.2	4	<0.001	-
	B	-	50	<0.3	<2	18	12.2	1.2	6.2	4	<0.001	-
5% Oxic	A	5% sediment	580	0.7	<2	29	8.1	4.6	4.7	102	0.011	-
	B	5% sediment	600	1.2	<2	17	11.4	3.7	4.8	93	0.004	-
10% Oxic	A	10% sediment	1030	1.1	<2	27	20.7	7.3	4.9	81	0.005	-
	B	10% sediment	1090	0.8	<2	25	7.2	5.7	6.1	93	0.021	-
20% Oxic	A	-	1770	1.3	<2	28	14	4.8	4.9	178	0.015	-
	B	-	1550	1.2	<2	28	9.3	3.8	3.9	107	0.007	-

(Continued)

Table A2 (Continued)

Sample Designation	Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Corpus Christi-Site 1											
Site 1 Water	-	110	1.2	-	14	-	3.5	2.5	-	-	-
5% Oxic A	5% sediment	115	0.8	-	7.6	-	4.8	1.5	-	-	-
B	5% sediment	178	0.5	-	7.1	-	5.9	2.5	-	-	-
Corpus Christi-Site 2											
Deionized Water	-	75	0.8	-	2.5	-	8	2	3	-	-
5% Oxic A	Deionized water used, 5% sediment	48	0.9	-	2.2	-	7.8	2.5	2	-	-
B	Deionized water used, 5% sediment	53	0.6	-	2.5	-	7.7	2.8	2	-	-
Site 2 Water	-	300	1	-	19	-	10	3	10	-	-
5% Oxic A	5% sediment	767	0.6	-	5	-	7	2.8	<10	-	-
B	5% sediment	658	0.6	-	4.8	-	8.7	2.5	<10	-	-
Site 4 Water	-	250	1.2	-	7	-	7.5	3	25	-	-
5% Oxic A	5% sediment	767	0.8	-	3.2	-	7.5	2.5	<10	-	-
B	5% sediment	658	0.8	-	3.5	-	7.5	2.9	11	-	-
Corpus Christi-Site 3											
Site 3 Water	-	170	0.9	-	19	-	2	6.2	6	-	-
5% Oxic	5% sediment	458	0.9	-	14.8	-	1.8	5.8	22	-	-
10% Oxic	10% sediment	462	0.7	-	10.8	-	2.1	6.5	16	-	-
20% Oxic	-	713	0.9	-	13.5	-	2.5	6.9	14	-	-
Site 4 Water	-	145	0.8	-	22	-	2	10	68	-	-
5% Oxic	5% sediment	318	0.9	-	19.5	-	2	7.9	51	-	-
10% Oxic	10% sediment	553	0.8	-	21.5	-	2.5	7.2	60	-	-
20% Oxic	-	563	0.6	-	13	-	2	7.8	40	-	-

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Corpus Christi-Site 4												
Site 4 Water		-	35	1.5	-	11.6	-	4.4	2	-	-	-
5% Oxic	A	5% sediment	50	<0.5	-	8.2	-	4.8	2.2	-	-	-
	B	5% sediment	50	<0.5	-	7.8	-	5.4	3.8	-	-	-
Port Aransas												
Site Water		-	20	1.3	-	7.4	-	1.9	1	12	-	-
5% Oxic (1 hr)		5% sediment	420	1.2	-	5.8	-	1.6	1	12	-	-
(24 hr)		6% sediment, 24 hr. settle	460	1.3	-	7.9	-	1	0.7	8	-	-
10% Oxic (1 hr)		10% sediment.	540	1.2	-	2.4	-	1.4	0.5	15	-	-
(24 hr)		10% sediment, 24 hr. settle	700	1.3	-	4.4	-	2	1.9	15	-	-
20% Oxic (1 hr)		-	900	1.4	-	1.5	-	1.8	0.8	8	-	-
(24 hr)		24 hr. settle	900	1.5	-	2.6	-	1.8	1	5	-	-
SEATTLE												
Duwamish River-Site 1												
Site Water		-	143	1.8	<2	20.9	6.6	1.5	4.5	12	0.055	<2
5% Oxic	A	5% sediment	244	1	<2	21.3	3.6	1	2.4	14	0.020	<2
	B	5% sediment	141	1.1	<2	9.2	3.7	1.2	2	17	0.060	<2
20% Oxic	A	-	52	1.1	<2	14.3	6.5	0.7	1.8	97	0.020	<2
	B	-	116	1.1	<2	11.5	7.5	0.9	2.7	54	0.060	<2
20% Anoxic	A	N_2 mixing	77	0.9	<2	4	5.5	0.8	2.1	10400	0.020	<2
	B	N_2 mixing	26	0.9	<2	3.8	5.8	1	3.2	5770	0.020	<2
Duwamish River-Site 2												
Site Water		-	143	1.8	<2	20.9	6.6	1.5	4.5	12	0.055	<2
5% Oxic	A	5% sediment	131	0.9	<2	15.4	5.1	0.9	1.8	52	0.090	<2
	B	5% sediment	131	0.9	<2	13.9	5.4	2.3	1.8	18	0.055	<2
20% Oxic	A	-	251	1.8	<2	18.4	8.1	2.9	1.9	489	0.095	<2
	B	-	211	1.1	<2	15	7.9	2.2	2.1	606	0.055	<2
20% Anoxic	A	N_2 mixing	100	0.9	<2	1.7	7.3	1.5	2.2	13910	0.055	<2
	B	N_2 mixing	111	1	<2	1.5	7.3	1.5	2.5	14050	0.068	<2

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Duwamish River-Site 3												
Site Water		-	143	1.8	<2	20.9	6.6	1.5	4.5	12	0.055	<2
5% Oxic		5% sediment	151	0.9	<2	11.4	5.4	2	2.3	55	0.015	<2
20% Oxic		-	191	0.9	<2	13.3	7.3	2.2	2	1165	0.020	<2
20% Anoxic		N_2 mixing	90	<0.5	<2	1.6	5.6	1.3	1.9	22660	0.068	<2
Plop Test		Plop test procedure followed	205	1.6	<2	9	8.3	2.2	3.7	1300	0.028	<2
Duwamish River-Site 3 (Bioassay Elutriates)												
EPA RSW	A	-	49	3.1	<2	76	11.9	47.8	13.3	11	0.060	-
	B	-	29	3.3	<2	141	13.9	49.3	14.8	13	0.060	-
5% Oxic	A	5% sediment	2430	33.8	<2	94	15.1	12.9	21.3	8506	<0.001	-
	B	5% sediment	2177	2.6	<2	82	14.7	12.9	19.8	9772	<0.001	-
20% Oxic	A	-	6962	6.1	<2	188	21.5	33.6	71.6	22177	0.096	-
	B	-	6354	6.1	<2	106	13.9	14	34.3	23038	0.128	-
NEW YORK												
Perth Amboy Channel												
Site Water		-	<10	<0.5	<2	6.1	<2	<1	3.6	8	0.03	<2
5% Oxic	A	5% sediment	52	<0.5	<2	3	<2	<1	<1	<5	0.03	<2
	B	5% sediment	44	<0.5	<2	<1	2.5	<1	<1	31	0.03	3
20% Oxic	A	-	28	<0.5	<2	<1	<2	<1	<1	689	0.03	<2
	B	-	19	<0.5	<2	<1	2.6	<1	<1	31	0.03	<2
20% Anoxic	A	N_2 mixing	28	<0.5	<2	<1	<2	<1	<1	32	0.03	16
	B	N_2 mixing	44	<0.5	<2	<1	<2	<1	1.3	248	0.03	13

(Continued)

Table A2 (Continued)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Perth Amboy Channel (Bioassay Elutriates)												
EPA RSW	A	-	<10	1.4	<2	62	40	6.6	270	33	-	-
	B	-	15.4	1.7	<2	51.9	41	8.4	261	46	-	-
	C	-	12.5	2	<2	41	39	9.6	252	71	-	-
Site 1												
20% Oxic	A	3 replicates run	<10	0.7	<2	9.7	10.9	12.3	15.6	1537	-	-
	B		<10	0.6	<2	5.1	8.8	11.1	12	1170	-	-
	C		<10	0.6	<2	8	11.8	8.4	13.8	1974	-	-
Site 3												
20% Oxic	A	3 replicates run	12.5	0.6	<2	6.5	11.5	8.1	14.1	2637	-	-
	B		-	-	<2	-	-	-	-	2568	-	-
	C		<10	0.5	<2	8	12.5	9.3	14.9	2838	-	-
Site 2												
20% Oxic	A	3 replicates run	<10	0.6	<2	20.7	11.5	17.7	25.4	1536	-	-
	B		12.5	0.9	<2	18.6	16.3	14.7	21.1	1546	-	-
	C		<10	0.6	<2	14.3	17.8	12	20.8	1464	-	-
Perth Amboy Anchorage												
Site Water		-	28	<0.5	<2	<1	<2	<1	3.9	22	0.03	4
5% Oxic	A	5% sediment	36	0.7	<2	<1	<2	<1	<1	25	0.03	12
	B	5% sediment	28	0.6	<2	<1	<2	<1	<1	11	0.05	<2
20% Oxic	A	-	<10	0.6	7	<1	<2	<1	1	161	0.03	19
	B	-	<10	<0.5	<2	<1	4.1	<1	<1	48	0.03	13
20% Anoxic	A	N_2 mixing	20	<0.5	<2	<1	<2	<1	2.8	71	0.07	44
	B	N_2 mixing	84	<0.5	<2	<1	<2	<1	<1	65	0.03	44
Plop Test	A	Plop test procedure followed	52	<0.5	7.7	<1	<2	<1	<1	82	0.03	14
	B	Plop test procedures followed	12	0.7	2	<1	<2	<1	1.2	45	0.07	6

(Continued)

Table A2 (Concluded)

Sample Designation		Modifications**	Mn	Cd	Cr	Zn	Ni	Pb	Cu	Fe	Hg	As
Bay Ridge Channel												
Site Water		-	44	<0.5	2.6	7.4	<2	1.8	4.9	8	0.02	<2
5% Oxic	A	5% sediment	<10	<0.5	2.8	1.8	3.2	2.3	<1	11	0.02	<2
	B	5% sediment	<10	<0.5	2.2	1.2	<2	4.5	<1	23	0.02	<2
20% Oxic	A	-	28	<0.5	2.6	<1	2.8	<1	<1	1081	0.18	2
	B	-	<10	<0.5	2.7	<1	3.9	1.6	<1	679	0.02	<2
20% Anoxic	A	N_2 mixing	93	<0.5	2.6	<1	3.2	6	<1	1100	0.02	6
	B	N_2 mixing	76	<0,5	3.6	<1	<2	3.1	<1	1286	0.02	<2

*For further information regarding these data, numbers of replicate analyses, etc., consult Lee et al.

**Any modifications of the standard 20% oxic elutriate test as described in Lee et al. are presented here. These include variations in the percent sediment in total elutriate volume used (sed), agitation by shaking (SH), or stirring (ST), using N_2 (N_2 mixing) or N_2 followed by air mixing (20 min N_2 + 10 min air mixing) instead of using compressed air mixing, duration of mixing period (15 min SH), variations in duration of settling period (24 hr settle), test water used when not collected at the site (deionized water used), duration of storage of sediment and water samples prior to elutriation when storage studies were conducted (70 da stor), or when additional replicate tests were run.

Table A3

Summary of Nitrogen Compound Concentrations in the Elutriates and Site Water for All Sites Studied

(mg N/l)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Trinity River						
Site water	-	-	0.09	0.02	7.07	0.06
20% A	-	-	2.96	0.25	6.42	0.08
B	-	-	1.15	0.21	6.50	0.17
C	-	-	2.20	0	6.97	0.12
Mobile Bay						
Site water	-	-	0.40	0.04	0.15	0.02
20%	-	-	2.62	0.25	0.14	0.02
15%	-	-	1.87	0.17	0.12	0.02
10%	-	-	1.25	0.09	0.13	0.03
5%	-	-	0.83	0.09	0.15	0.02
Site water	-	-	<0.03	0.01	0.08	0.02
20%-15 min agitation A	-	-	3.38	0	0.03	0.02
B	-	-	1.82	1.02	0.33	0.09
30 min agitation	-	-	3.58	0.65	<0.04	∿0.03
60 min agitation	-	-	3.34	0.35	0.07	0.01
90 min agitation	-	-	3.51	0.51	0.10	0.02
Site water	-	-	0.06	0.01	0.08	0.01
1 liter shaken	-	-	4.5	0.2	0.06	0
1 liter stirred A	-	-	4.4	0.2	0.06	0.01
B	-	-	-	-	0.13	0.01
3 liter stirred	-	-	4.5	0.1	0.12	0.01
Houston Ship Channel						
Site water	4.12**	0.07**	-	-	<0.02	∿0.04
20% oxic A	63.8	0.80	-	-	0.02	0.01
B	79.8	4.80	-	-	0.03	0.01
C	57.6	1.70	-	-	0.04	0.01
D	75.5	1.00	-	-	0.04	0.01

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium *		Nitrate *	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Site water	2.7	0.3	1.8	0.02	-	-
20% anoxic A	25.1	1.7	52.5	0.6	-	-
B	26.6	4.1	45.7	0.6	-	-
20% oxic	14.1	1.9	35.5	0.1	-	-
Port Aransas						
Site water	0.22	0.04	0.06	0	0.01	0
5%-1 hr settling	0.25	0.06	0.07	0	< 0.01	0
24 hr settling	0.25	0.08	0.07	0.01	< 0.01	0
10%-1 hr settling	0.32	0.05	0.08	0	< 0.01	0
24 hr settling	0.22	0.05	0.08	0.01	< 0.01	0
20% 1 hr settling	-	-	0.10	0.01	< 0.01	0
24 hr settling	0.39	0.12	0.08	0.01	< 0.01	0
Ashtabula						
Site water	-	-	0.03	0.01	1.28	0.10
5%-1 hr	-	-	5.01	0.26	1.28	0.05
10%-1 hr	-	-	7.73	0.19	1.13	0.09
24 hr	-	-	9.40	0.30	1.05	0.02
20%-1 hr	-	-	14.12	0.14	1.17	0.04
24 hr	-	-	16.07	0.18	0.84	0.04
Bridgeport - Eaton's Neck						
Site water	-	-	0.02	0	0.12	0
5%-1 hr	-	-	6.26	0.23	0.11	0.01
24 hr A	-	-	5.49	0.23	0.06	0.02
B	-	-	5.24	0.14	0.06	0.01
10%-1 hr	-	-	10.83	0.09	0.08	0.01
24 hr	-	-	10.93	0.18	< 0.06	0.01
20%-1 hr	-	-	15.45	0.32	0.08	0.01
24 hr	-	-	19.0	0.48	0.05	0

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Corpus Christi Site 3						
Site water	0.54	0.07	0.34	0.02	0.23	0.04
5%	0.9	0.3	4.0	0.08	0.21	0.02
10%	0.9	0.3	5.29	0.02	0.6	0
20%	1.8	0.4	8.8	0.2	2.37	0.51
Corpus Christi Site 4						
Site water	0.54	0.03	0.08	0.01	0.03	0.02
5%	0.8	0.2	2.8	0.04	< 0.03	∿ 0.01
10%	1.1	0.2	4.81	0.02	< 0.03	∿ 0.01
20%	1.6	0.1	7.13	0.02	< 0.02	∿ 0
Galveston offshore						
Disposal site water	0.55	0.16	0.25	0	0.18	0.01
Buoy 9:						
Site water	0.27	0.09	0.13	0.01	0.20	0.01
5% oxic with Buoy 9 water	0.07	0.09	0.69	0.01	0.22	0.01
5% oxic with disposal site water	0.44	0.21	0.80	0.01	0.13	0.01
20% oxic with disposal site water	0.58	0.26	1.58	0.02	0.14	0.01
Buoy 11:						
Site water	0.38	0.08	0.10	0	0.24	0.01
5% oxic with Buoy 11 water	0.07	0.29	0.93	0.01	0.21	0.02
5% oxic with disposal site water	0.24	0.28	0.92	0	0.13	0.01
20% oxic with disposal site water	0.42	0.17	1.74	0.01	0.13	0.01
Buoy 1:						
Site water	0.17	0.04	< 0.05	0	< 0.04	0
5% oxic	< 0.05	0	0.41	0	< 0.04	0
20% oxic	< 0.05	0	1.20	0.1	0.05	0

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium *		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Texas City Channel 1						
Site water	5.0	1.0	0.59	0.04	0.08	0.01
5% oxic-1 hr settling	3.2	1.3	3.06	0.18	0.05	0.01
24 hr	2.2	0.7	3.84	0.2	0.06	0.02
20% oxic-1 hr settling	0.2	1.5	7.5	0.1	0.05	0
24 hr	0.6	0.6	10.3	0	0.07	0.02
Texas City Channel 1						
Site water	0.23	0.02	< 0.05	0	0.08	0.01
5% oxic	< 0.05	0.06	2.42	0.02	0.10	0.01
20% oxic	< 0.05	0.53	8.34	0.17	0.43	0.02
Texas City Channel 1						
Site water	0.27	0.04	< 0.05	0	0.10	0.01
20% - N_2 + air	0.05	0.50	12.3	0.1	0.12	0.01
25% - N_2 + air	1.01	0.29	12.8	0.07	0.50	0.01
Texas City Channel 2						
Site water	0.25	0.02	0.15	0.01	0.11	0.01
5% oxic	< 0.05	0.08	1.59	0	0.11	0.01
20% oxic	< 0.05	0.68	6.02	0.08	0.09	0.01
Texas City Channel 3						
Site water	0.30	0.21	0.22	0.01	0.11	0.01
5% oxic	< 0.05	0.21	2.98	0.01	0.09	0.01
20% oxic	< 0.05	0.23	4.66	0.23	0.11	0.01
Morgan's Point						
Site water	0.84	0.81	1.2	0.01	0.38	0.01
5% oxic	0.98	0.66	1.86	0.06	0.27	0.01
20% oxic	< 0.05	0.91	4.23	0.11	0.26	0.01

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Texas City Channel 4						
Site water	0.84	0.30	0.12	0	0.10	0.01
5% oxic	< 0.05	∿ 0	4.50	0.09	0.42	0.01
20% oxic	< 0.05	∿ 0	10.25	0.25	0.74	0.02
Texas City Channel 4						
Site water	1.89	0.30	0.24	-	0.19	0.01
20% oxic A	< 0.05	∿ 0	9.38	-	0.10	0.02
20% oxic B	0.32	0	15.1	-	< 0.04	0
Texas City Channel 5						
Site water	0.79	0.22	0.12	0	0.11	0.01
5% oxic	0.93	0.16	1.63	0.01	0.05	0.01
20% oxic	< 0.05	∿ 0	2.95	0.02	0.09	0.01
Texas City Channel 5						
Site water	0.31	0	0.22	-	0.09	0.01
20% oxic	1.17	0.52	4.31	-	0.05	0.01
Texas City Channel 6						
Site water	0.15	0.09	0.12	0.01	0.04	0.01
5% oxic	0.91	0.01	4.41	0.01	0.50	0.01
20% oxic	2.65	2.30	15.70	0.42	0.31	0.02
Texas City Channel 6						
Site water	0.85	0	0.21	-	0.08	0.01
20% oxic	< 0.05	∿ 0	11.3	-	0.07	0.01
Galveston Channel						
Site water	0.30	0.08	0.08	0	0.10	0.02
20% oxic	< 0.05	∿ 0	7.36	0.10	0.09	0.01

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Port Lavaca						
Site water	0.46	0.02	<0.05	∿0	<0.04	∿0
5% oxic	0.23	0.06	0.81	0.02	0.38	0.01
5% oxic/anoxic A	0.34	0.09	0.78	0.01	0.28	0.01
B	0.67	0.07	1.53	0.07	0.07	0.01
20% oxic	0.98	0.13	1.98	0.05	<0.04	0.01
20% oxic/anoxic A	0.66	0	1.54	0	0.05	0.01
B	0.80	0.03	3.28	0.01	<0.04	∿0
Port Lavaca						
Site water	0.86	0.15	0.10	-	<0.04	∿0
20% oxic	< 0.05	∿0	1.95	-	0.12	0.01
Rodeo Flats						
Site water	0.05	0.08	0.19	0.01	0.27	0.06
5% oxic	0.19	0.09	0.60	0.03	0.28	0.01
20% oxic	0.48	0.06	1.64	0.02	0.17	0.02
Mare Island						
Site water	0.68	0	0.20	0	0.88	0.02
5% oxic-20 hr settling	1.27	0.15	0.41	0.02	0.73	0.03
20% oxic-20 hr settling	1.81	0.07	0.90	0.01	0.39	0.04
Mare Island						
Site water	4.6	1.4	0.22	0.01	1.04	0.07
5% oxic	4.2	1.0	0.36	0	0.91	0.04
Mare Island						
Site water	0.87	0.11	0.29	0.01	0.35	0.01
20% anoxic A	1.62	0.26	0.54	0.02	0.44	0
B	1.73	0.10	0.59	0.02	0.44	0.005
C	1.72	0.10	0.56	0.02	0.44	0.01
Oakland Harbor						
Site water	0.05	0.02	0.16	0.02	0.23	0.02
20% oxic A	0.80	0.35	1.70	0.12	0.21	0.02
B	2.62	1.02	4.73	0.41	0.18	0.01
20% N_2 + air A	-	-	-	-	0.19	0.01
B	0.34	0.30	1.42	0.07	0.11	0.01

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Oakland Harbor						
Site water	0.42	0.02	0.20	0.01	0.26	0.01
20% oxic	<0.05	∿0	4.28	0.05	0.20	0.02
20% anoxic/oxic	<0.05	∿0	5.96	0.06	0.19	0.04
Brenton's Reef						
Site water	0.40	0	<0.05	∿0	0.09	0.01
20% oxic A	0.16	0.09	0.86	0.01	0.22	0
B	0.55	0.03	0.77	0	0.09	0.01
20% anoxic/oxic A	0.05	0	2.70	0.03	0.20	0.01
B	0.05	0	2.93	0	0.16	0.01
Norwalk Harbor (south)						
Site water	<0.05	∿0	0.07	0.01	0.26	0.01
20% oxic	<0.05	∿0	3.16	0.05	0.15	0.01
20% anoxic/oxic	0.06	0.02	1.46	0.02	0.17	0.02
Norwalk Harbor (north)						
Site water	<0.05	∿0	0.07	0.01	0.26	0.01
20% oxic A	<0.05	∿0	6.12	0.08	0.11	0.01
B	<0.05	∿0	5.33	0.14	0.16	0.01
20% anoxic/oxic A	<0.05	∿0	5.05	0.04	0.11	0.01
B	<0.05	∿0	5.51	0.07	0.09	0.01
Los Angeles						
Site water	<0.05	∿0	0.18	-	0.08	0
20% oxic A	<0.05	∿0	34.5	-	0.12	0.01
B	<0.05	∿0	37.7	-	0.12	0.01
Apalachicola 1						
EPARSW	<0.05	0	0.18	-	0.08	0
20% oxic A	0.90	0.40	11.1	-	0.13	0
B	0.22	0.24	8.8	-	<0.04	∿0
20% anoxic/oxic A	<0.05	∿0	6.54	0.06	0.08	0.01
B	0.28	0.34	7.56	0	0.09	0.01
Apalachicola 2						
EPARSW	0.10	0.07	<0.05	∿0	0.17	0.01
20% oxic A	0.17	0.30	5.51	0.07	0.08	0.01
B	<0.05	0	6.37	0	0.10	0.01

(continued)

Table A3 (continued)

Sample Designation	Organic N *		Ammonium *		Nitrate *	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
20% anoxic/oxic A	< 0.05	∿0	5.58	0.03	0.10	0.01
B	0.05	0	4.24	0.12	0.11	0.01
Apalachicola 3						
Site water (No. 411)	0.25	0.03	< 0.05	∿0	0.19	0.02
20% oxic	1.28	0.34	4.24	0.12	0.20	0.01
20% anoxic	3.01	0.23	3.39	0.17	0.15	0.03
Site water (No. 412)	< 0.05	∿0	0.34	0.03	0.07	0.02
20% oxic	0.18	0.13	4.70	0.06	0.04	∿0
20% anoxic	< 0.05	∿0	3.70	0.10	< 0.04	∿0
Apalachicola 4						
Site water (No. 411)	0.25	0.03	< 0.05	∿0	0.19	0.02
20% oxic	1.67	0.23	2.91	0.09	0.15	0.02
20% anoxic	3.53	0.19	2.45	0.11	0.16	0.02
Site water (No. 412)	< 0.05	∿0	0.34	0.03	0.07	0.02
20% oxic	0.21	0.11	3.33	0.08	< 0.04	∿0
20% anoxic	< 0.05	∿0	3.39	0	< 0.04	∿0
Apalachicola 5						
Site water (No. 411)	0.25	0.03	< 0.05	∿0	0.19	0.02
20% oxic	1.84	0.34	5.62	0.09	0.15	0.01
20% anoxic	2.24	0.06	4.36	0.06	0.13	0.02
Site water (No. 412)	< 0.05	∿0	0.34	0.03	0.07	0.02
20% oxic	0.38	0.11	6.10	0.05	< 0.04	∿0
20% anoxic	< 0.05	∿0	4.31	0.13	0.04	∿0
Duwamish River 1						
Site water	0.17	0.03	< 0.05	0	0.39	0.01
20% oxic A	1.58	0.14	2.70	0.03	0.37	0.01
B	1.22	0.23	1.90	0.06	0.41	0.01

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
20% Anoxic A	2.40	0.14	2.38	0.06	0.36	0.01
B	2.56	0.22	2.26	0.06	0.33	0.01
5% oxic A	0.53	0.09	0.50	0.01	0.46	0.01
B	0.18	0.15	0.68	0.01	0.37	0.02
Duwamish River 2						
Site water	0.17	0.03	<0.05	~ 0	0.39	0.01
5% oxci A	1.86	0.06	1.54	0	0.33	0.01
B	0.20	0.13	1.54	0.01	0.37	0.01
20% oxic A	1.26	0.32	5.74	0.15	0.33	0.01
B	2.92	0.56	5.82	0.08	0.31	0.01
20% anoxic A	3.71	0.25	4.15	0.11	0.32	0.01
B	1.02	0.19	3.92	0.11	0.34	0.01
Duwamish River 3						
Site water	0.17	0.03	<0.05	~ 0	0.39	0.01
5% oxic	0.36	0.15	1.38	0.01	0.42	0.01
20% oxic	1.96	0.16	3.80	0.05	0.36	0.01
20% anoxic	3.19	0.33	3.77	0.10	0.48	0.03
Plop	0.38	0.01	0.86	0.01	0.45	0.02
Houston Ship Channel 1						
Site water	0.60	0.04	3.60	0.04	0.32	0.01
5% oxic A	-	-	6.78	0.04	0.26	0.01
B	1.73	0.14	6.85	0	0.28	0.02
20% oxic A	-	-	13.65	0.07	0.16	0.01
B	1.40	0.34	14.72	0.11	0.10	0.01
20% anoxic A	1.26	0.60	11.22	0.04	0.12	0.01
B	-	-	10.09	0.06	0.13	0.01

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Houston Ship Channel 2						
Site water	-	-	3.99	0	0.21	0.01
5% oxic A	-	-	8.02	0.04	0.16	0.02
B	-	-	8.31	0.04	0.16	0
20% oxic A	-	-	17.7	0.42	0.07	0
B	-	-	17.0	0.14	0.17	0.10
C	-	-	16.3	0.07	0.07	0
D	-	-	16.08	0.17	0.08	0
E	-	-	15.95	0.07	0.08	0.01
20% anoxic A	-	-	13.1	0.14	0.08	0.01
B	-	-	12.3	0	0.10	0.01
Plop 2 liter	-	-	-	-	0.46	0.02
4 liter	-	-	2.76	0	0.43	0.02
8 liter	-	-	6.28	0.13	0.38	0
Houston Ship Channel 3						
Site water	0.54	0.14	1.08	0	0.28	0.01
5% oxic A	-	-	2.95	0.10	0.32	0.01
B	0.38	0.07	3.04	0.04	0.24	0.02
20% oxic A	< 0.05	∿ 0	8.11	0	0.11	0
B	-	-	8.62	0.05	0.06	0.01
20% anoxic A	0.12	0.26	6.50	0.06	0.10	0
B	-	-	7.16	0.13	0.09	0.03
Mobile Bay 1						
Site water	0.05	0	0.05	0	0.25	0.02
5% oxic A	0.51	0.02	2.25	0	0.22	0.01
B	0.60	0.03	2.38	0.02	0.24	0.01
20% oxic A	0.13	0.10	4.76	0.08	0.14	0.01
B	0.94	0.21	6.13	0.06	0.08	0
20% anoxic A	1.14	0.08	4.61	0	0.10	0
B	0.75	0.17	5.71	0.03	0.10	0

(continued)

Table A3 **(continued)**

Sample Designation	**Organic N** *		**Ammonium** *		**Nitrate** *	
	$\bar{X}$	**SD**	$\bar{X}$	**SD**	$\bar{X}$	**SD**
Mobile Bay 2						
Site water (No. 412)	< 0.05	∿0	0.08	0.01	0.28	0.01
Site water (No. 418)	< 0.05	∿0	< 0.05	∿0	0.30	0.01
5% oxic A	< 0.05	∿0	1.04	0.01	0.37	0
B	< 0.05	∿0	0.98	0.01	0.37	0.01
20% oxic A	0.44	0.09	3.42	0.04	0.27	0
B	0.28	0.10	3.54	0.04	0.25	0.01
20% anoxic A	0.52	0.05	2.60	0	0.20	0.01
B	0.71	0.03	2.28	0	0.24	0.01
Upper Mississippi						
Site water	0.13	0.04	1.18	0.01	0.14	0.01
5% oxic A	0.58	0.08	1.24	0.02	0.14	0
B	0.72	0.02	1.44	0.01	0.12	0
20% oxic A	0.75	0.06	1.96	0.04	0.14	0.01
B	0.78	0.09	2.41	0.02	0.12	0
Bailey Creek						
Site water	< 0.05	∿0	< 0.05	0	0.89	0.03
5% oxic A	< 0.05	∿0	1.59	0.02	0.86	0.01
B	< 0.05	∿0	2.14	0.07	0.96	0.04
20% oxic A	0.58	0.12	4.37	0.06	0.89	0
B	0.34	0.28	5.71	0.21	0.84	0.04
20% anoxic A	< 0.05	∿0	3.92	0.10	0.93	0.01
B	< 0.05	∿0	4.56	0.07	0.79	0.01
James River						
Site water	< 0.05	∿0	< 0.05	∿0	0.86	0.01
5% oxic A	< 0.05	∿0	2.08	0.01	0.73	0.03
B	< 0.05	∿0	2.25	0.04	0.54	0.01

(continued)

Table A3 (continued)

Sample Designation	Organic N *		Ammonium *		Nitrate *	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
20% oxic A	1.36	0.42	5.77	0.31	0.73	0.03
B	0.81	0.20	5.50	0.19	0.66	0.01
20% anoxic A	0.63	0.21	4.61	0.07	0.56	0
B	< 0.05	∿0	4.35	0	0.70	0.04
Menominee 1						
Site water	< 0.05	∿0	0.06	0.01	0.04	0
20% oxic A	< 0.05	∿0	10.24	0.38	0.05	0
B	< 0.05	∿0	10.70	0.14	0.04	0
20% anoxic A	< 0.05	∿0	9.17	0.10	< 0.04	∿0
B	< 0.05	∿0	8.34	0.17	< 0.04	∿0
Plop A	< 0.05	∿0	2.82	0.10	0.05	0
B	< 0.05	∿0	2.47	0.09	< 0.04	∿0
Menominee 2						
Site water	0.08	0.02	0.05	0	0.08	0.01
20% oxic A	0.21	0.19	14.4	0.14	< 0.04	∿0
B	0.33	0.15	13.6	0.14	< 0.04	∿0
20% anoxic A	< 0.05	∿0	13.0	0	0.04	0
B	0.10	0.93	12.8	0.71	0.05	0.01
Plop A	< 0.05	∿0	4.61	0.07	< 0.04	∿0
B	< 0.05	∿0	4.80	0.04	< 0.04	∿0
Perth Amboy Hopper						
Site water A	0.28	0.13	0.60	0.08	0.10	0
B	0.35	0.04	0.54	0.02	0.08	0
5% oxic A	2.37	0.72	10.28	0.44	0.12	0.01
B	1.18	0.31	10.05	0.11	0.08	0.01
20% oxic A	5.61	1.74	33.05	1.20	0.08	0.01
B	3.30	1.59	24.85	1.20	< 0.04	∿0
20% anoxic A	2.75	0.53	15.55	0.21	< 0.04	∿0
B	2.78	0.86	18.55	0.64	< 0.04	∿0

(continued)

Table A3 (continued)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Perth Amboy Mechanical						
Site water A	0.15	0.04	0.63	0.02	0.14	0.01
5% oxic A	0.80	0.14	4.66	0.07	0.12	0
B	0.69	0.19	4.40	0.13	0.11	0
20% oxic A	1.53	0.41	13.85	0.21	0.06	0.01
B	1.78	0.90	14.40	0.57	0.06	0.01
20% anoxic A	0.31	0.39	9.90	0.32	0.10	0
A	0.59	0.18	8.78	0.09	0.10	0
Site water B	0.22	0.03	0.68	0.01	0.15	0
Plop A	0.67	0.07	3.04	0.04	0.15	0
B	0.55	0.06	3.24	0.02	0.14	0.01
Bay Ridge Hopper						
Site water	0.37	0.09	0.56	0	0.13	0
5% oxic A	0.15	0.10	2.43	0.08	0.12	0
B	0.39	0.16	2.96	0.11	0.10	0.01
20% oxic A	0.21	0.08	8.40	0.08	0.05	0
B	0.20	0.17	7.46	0.14	0.04	0
20% anoxic A	0.73	0.15	5.24	0.09	0.04	0.01
B	0.55	0.36	5.50	0.09	0.06	0
Wilmington						
Site water	< 0.05	∿0	0.06	0.01	< 0.04	∿0
20% oxic A	< 0.05	∿0	0.56	0.01	0.04	0.01
B	< 0.05	∿0	0.56	0.01	0.08	0.01
Vicksburg						
Site water	0.15	0.03	0.20	0	0.10	0.01
20% oxic A	0.42	0.06	2.82	0.03	< 0.04	∿0
B	0.27	0.08	2.97	0	< 0.04	∿0

(continued)

Table A3 (concluded)

Sample Designation	Organic N*		Ammonium*		Nitrate*	
	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Foundry Cove						
Site water	<0.05	∿ 0	< 0.05	∿ 0	0.52	0.04
20% oxic A	0.82	0.16	3.13	0.04	0.18	0.03
B	0.47	0.19	3.18	0.02	0.82	0.02
20% anoxic A	< 0.05	∿ 0	3.06	0.02	0.69	0.04
B	< 0.05	∿ 0	3.12	0.02	0.71	0.03

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*Mean and standard deviation calculated from triplicate analyses.

**Mean and standard deviation calculated from five replicate analyses.

Table A4

Summary of Elutriate Test Results for Organic Analyses

Study Site	Test Ratio	Oil and Grease			Total Fe	TOC	Total PCBs			Total Pesticides		
	Percent Sediment	Sediment (mg/kg)	Water (mg/l)	Elutriate (mg/l)	Sediment (mg/kg)	Sediment (mg/kg)	Sediment (µg/kg)	Water (ng/l)	Elutriate (ng/l)	Sediment (µg/kg)	Water (ng/l)	Elutriate (ng/l)
Rodeo Flats	5	2227	NT	NT	1,350	NT	90	85	25	B	B	B
Mare Island	5	3120	NT	NT	1,380	NT	95	225	21	B	B	B
Oakland 1	20	1168	< 0.5	36.3	37,300	24,000	600	B	ND	148	3.3	20.2
Oakland 2	20	1436	< 0.5	95.0	37,300	25,000	360	B	20	135	3.7	B
TCC 1	5	304	NT	NT	23	NT	7426	130	150	15.5	1.4	5.1
TCC 4	20	533	0.9	14.7	NT	6,000	3010	52	9619	12.5	2.4	28.1
GBEC Buoy 1	5	24	NT	NT	41	NT	192	210	170	5.5	11.8	10.5
Galveston Channel	20	437	NT	NT	NT	9,200	33	171	261	6.7	10.3	49.4
Los Angeles A7*	20	2438	NT	NT	17,478	30,800	4430	B	36	675	15.6	110.7
Newport, R.I.	20	168	NT	NT	885	12,000	50	B	B	B	B	B
Stamford W	20	1877	< 0.5	0.9	9,531	32,000	2064	7	26	341.7	B	8.0
Norwalk S	20	2506	< 0.5	1.8	9,387	44,000	2820	B	104	320.9	4.2	94.8
Apalachicola 1*	20	345	NT	NT	18,800	29,000	13	24	14	8.4	15.6	4.4
Apalachicola 3*	20	423	NT	NT	9,648	49,000	454	38	B	10.6	4.7	14.0
Duwamish River	20	702	< 0.5	< 0.5	16,079	36,000	406	12	33	32.2	2.8	6.8
Perth Amboy Channel	20	4044	< 0.5	< 0.5	13,900	35,000	1078	B	69	104.2	B	B
Perth Amboy Anchorage	20	2091	0.7	1.3	14,800	31,500	1760	17	81	71.5	B	34.9
Bay Ridge Channel	20	826	< 0.5	3.6	15,900	31,800	1820	8	86	184.4	14.2	15
HSC 1	20	5868	< 0.5	54.7	19,045	6,000	5325	B	11,755	578.5	32.6	417.2
HSC 2	20	3634	< 0.5	47.0	20,055	4,900	7918	64	872	695.8	33.0	170.1
Upper Mississippi R.	20	60	0.5	2.0	11,820	3,300	7	B	11	3.3	5.7	10.4
James River	20	471	< 0.5	0.6	NT	27,000	45	B	B	23.6	B	37.3
Bailey Creek	20	1059	< 0.5	0.9	NT	21,000	235	B	34	1093.5	17.6	34.5

Study Site	Test Ratio	Total DDT			Total Aldrin-Dieldrin			Heptachlor			Lindane		
	Percent Sediment	Sediment (µg/kg)	Water (ng/l)	Elutriate (ng/l)	Sediment (µg/kg)	Water (ng/l)	Elutriate (ng/l)	Sediment (µg/kg)	Water (ng/l)	Elutriate (ng/l)	Sediment (µg/kg)	Water (ng/l)	Elutriate (ng/l)
Rodeo Flats	5	B	B	B	B	B	B	B	B	B	B	B	B
Mare Island	5	B	B	B	B	B	B	B	B	B	B	B	B
Oakland 1	20	148	B	13.7	B	B	B	B	B	6.5	B	3.3	B
Oakland 2	20	135	B	B	B	B	B	B	1.6	B	B	2.1	B
TCC 1	5	8.4	B	B	6.1	1.4	3.6	B	B	B	1.0	B	1.5
TCC 4	20	9.5	B	17.6	2.6	1.4	8.9	B	B	B	0.4	1.0	1.6
GBEC Buoy 1	5	2.2	5.2	5.2	2.5	4.1	2.8	B	B	B	0.8	2.5	2.5
Galveston Channel	20	5.4	B	B	0.9	6.9	35.1	B	B	B	0.4	3.4	14.3
Los Angeles A7*	20	675	B	6.2	B	B	B	B	B	B	B	15.6	2.5
Newport, R.I.	20	B	B	B	B	B	B	B	B	B	B	B	B
Stamford W	20	322.2	B	8.0	7.6	B	B	11.9	B	B	B	B	B
Norwalk S	20	215.3	B	67	70.3	B	12.6	23.1	B	9.0	12.2	4.2	6.2
Apalachicola 1*	20	5.6	B	B	B	B	B	B	B	B	2.8	15.6	4.4
Apalachicola 3*	20	6.6	B	B	B	B	B	3.2	2.4	7.7	0.8	2.3	7.0
Duwamish River	20	30.4	B	2.8	B	B	B	1.8	2.8	4.0	B	B	B
Perth Amboy Channel	20	66.1	B	B	26.4	B	B	5.9	B	B	5.8	B	B
Perth Amboy Anchorage	20	34	B	25.8	20.3	B	B	9.9	B	4.5	7.3	B	4.6
Bay Ridge Channel	20	130	B	5.2	22.7	9.2	B	14.6	B	5.3	16.7	5.0	4.5
HSC 1	20	413.4	B	316.2	132.2	B	78.0	B	B	B	32.9	32.6	23.0
HSC 2	20	515.8	B	117.1	110.6	7.8	31.7	50.8	5.8	13.2	18.6	19.4	8.0
Upper Mississippi R.	20	1.6	B	6.0	0.7	3.4	1.3	B	B	B	1.0	2.3	3.1
James River	20	20	B	17.3	B	B	B	B	B	B	3.6	17.6	20.0
Bailey Creek	20	1065	B	22.8	B	B	B	B	B	B	8.5	17.6	11.7

NT = Not Tested
ND = No Data
B = Below Detection Limit
*Indicates elutriate test run with Reconstituted Sea Water.

Table A5

Summary of Acute Toxicity for Selected US Waterway Sediments*

Sediment Location	Percent Surviving at 96 Hours Sediment Percentage					
	5%		10%		20%	
	(A B)**	$\bar{X}$	(A B)	$\bar{X}$	(A B)	$\bar{X}$
Mobile Bay, Alabama		100		-		100
San Francisco Bay, California						
Rodeo Flats	(90,100)	95	(70,70)	70	(60,80)	70
Mare Island	(100,90)	95	(80,90)	85	(80,80)	80
Oakland Inner Harbor		100		-		100
Los Angeles Harbor, California						
Buoy A-7 (Site 1)	(70,80)	75		-	(40, -)	40
Buoy C-2 (Site 2)	(90,100)	95		-	(90,100)	95
Bridgeport, Connecticut	(80,70)	75	(50,40)	45		-
Norwalk River, Connecticut						
North Site		100		-		100
South Site		100		-	(90,90)	90
Stamford, Connecticut						
West Branch		100		-	(90,100)	95
Apalachicola, Florida						
Site 1		100		-	(80,100)	90
Site 5	(90,90)	90		-	(80,80)	80
Menominee River, Michigan		100		-	(40,30)	35
Upper Mississippi River, St. Paul, Minn.		100		-		100
Hudson River, New York						
Foundry Cove - *P. pugio*		-		-	(100,90)	95
- *D. magna*		-		-	(90,80)	85

(Continued)

Table A5 (Continued)

Sediment Location	Percent Surviving at 96 Hours Sediment Percentage 5% (A B)**	 5% $\bar{X}$	 10% (A B)	 10% $\bar{X}$	 20% (A B)	 20% $\bar{X}$
New York-New Jersey Harbors						
Perth Amboy Anchorage	(80,70)	75		-	(80,70)	75
Perth Amboy Channel	(60,80)	70		-	(40,40)	40
Bay Ridge Channel	(100,50)	75		-	(100,90)	95
Perth Amboy Channel - Site 1		-		-	(90,80,90)	87
- Site 2		-		-	(60,60,60)	60
- Site 3		-		-	(80,80,90)	83
Ashtabula Harbor, Lake Erie, Ohio	Reproduction occurred		Reproduction occurred			100
Newport, Rhode Island						
Offshore		100		-		100
Corpus Christi, Texas						
Site 3	(90,80)	85	(80,100)	90	(90,90)	90
Houston Ship Channel, Texas						
Site 2		100		-	(90,90)	90
Site 3		100		-		100
Morgan's Point, Texas	(90,90)	90	(80,80)	80	(70,60)	65
Port Lavaca, Texas		100		100		100
Galveston Bay Entrance Channel, Texas						
Buoy 1	(90,90)	90		100	(98,80)	85
Buoy 9		-		100		-
Buoy 11		100		100	(90,90)	90

(Continued)

Table A5 (Concluded)

Sediment Location	Percent Surviving at 96 Hours Sediment Percentage					
	5%		10%		20%	
	(A B)**	$\bar{X}$	(A B)	$\bar{X}$	(A B)	$\bar{X}$
Texas City Channel, Texas						
Site 1		100		100		80
Site 2	(90,90)	90	(90,90)	90	(80,80)	80
Site 3	(90,90)	90	(90,90)	90	(80,80)	80
Site 4	(80,100)	90	(90,70)	80	(90,80)	85
Site 5		100		100		100
Site 6		100		100	(90,100)	95
Bailey Creek, Virginia (4/76)		100		-	(80,100)	90
Bailey Creek, Virginia (7/76)	(80,60)	70		-	(80,100)	90
James River, Virginia	(90,60)	75		-	(80,100)	90
Near Windmill Point dredge discharge		-		-		100
Duwamish River, Seattle, Wash.	(100,90)	95		-		100

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*Freshwater - _Daphnia magna_

Marine water - _Palaemonetes pugio_

**A and B are replicate bioassays.

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