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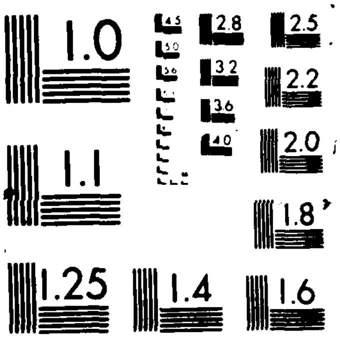
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US Army Corps of Engineers
Water Resources
Support Center

Management of Bottom Sediments Containing Toxic Substances

Proceedings of the 11th U.S./Japan
Experts Meeting

4-6 November 1985
Seattle, Washington

Thomas R. Patin, Editor

April 1987

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The 11th U.S./Japan Meeting on Management of Bottom Sediments Containing Toxic Substances was held 4-6 November 1985 in Seattle, Washington. The meeting is held annually through an agreement with the US Army Corps of Engineers and the Japan Ministry of Transport to provide a forum for presentation of papers and in-depth discussions on dredging and disposal of contaminated sediment.

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PREFACE

The 11th U.S./Japan Experts Meeting on Management of Bottom Sediments Containing Toxic Substances was held 4-6 November 1985 in Seattle, Washington. The meeting is held annually through an agreement with the U.S. Army Corps of Engineers and the Japan Ministry of Transport to provide a forum for presentation of papers and in-depth discussions on dredging and disposal of contaminated sediment.

COL George R. Kleb, Commander and Director of the Water Resources Support Center, was the U.S. Chairman. Mr. Kenichi Fujimori, Ministry of Transport, Tokyo, Japan, was the Japanese Chairman.

Coordinator of the organizational activities and editor of this report was Mr. Thomas R. Patin, program assistant, Dredging Operations Technical Support Program (DOTS), U.S. Army Engineer Waterways Experiment Station (WES), Vicksburg, Miss. Ms. Jamie W. Leach and Ms. Betty W. Watson, WES Information Products Division, were instrumental in editing, coordinating, and preparing the text. Dr. Robert M. Engler is the Manager of the DOTS Program.

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DELEGATES

11th U.S./JAPAN EXPERTS MEETING ON MANAGEMENT OF BOTTOM SEDIMENTS CONTAINING TOXIC SUBSTANCES

Japanese Delegation

Kenichi Fujimori Co-chairman	Director, Environmental Protection Division, Ports and Harbours Bureau, Ministry of Transport
Motoo Fujiki	Institute of Community Medicine, University of Tsukuba
Shinzo Kimura	Bureau of Port and Harbour, Tokyo Metropolitan Government
Hiroshi Kojima	Japan Dredging and Reclamation Engineering Association
Nobumasa Morii	Japan Bottom Sediments Management Association
Shinichi Kojima	Japan Bottom Sediments Management Association
Hiroaki Mori	Japan Bottom Sediments Management Association
Makoto Natori	Executive Director, Japan Bottom Sediments Management Association
Yasuo Abe	Japan Bottom Sediments Management Association
Yutaka Tagunchi	Japan Bottom Sediments Management Association
Naoshi Ishimatsu	Japan Work Vessel Association

US Delegation

COL George R. Kleb Co-chairman	Commander/Director, Water Resources Support Center, US Army Corps of Engineers
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D. A. Hotchkiss	Port of Seattle
K. E. Phillips	US Army Corps of Engineers, Seattle District
Bob Parker	US Army Corps of Engineers, Seattle District

Tommy E. Myers	US Army Engineer Waterways Experiment Station
John W. Barko	US Army Engineer Waterways Experiment Station
M. H. Salazar	Environmental Sciences Division, Naval Ocean Systems Center
Richard P. Leonard	US Army Corps of Engineers, Buffalo District
Jan A. Miller	US Army Corps of Engineers, Chicago District
Robert M. Engler	US Army Engineer Waterways Experiment Station
Edward B. Hands	US Army Engineer Waterways Experiment Station
Thomas R. Patin	US Army Engineer Waterways Experiment Station
David Mathis	Water Resources Support Center, US Army Corps of Engineers
Bill Murden	Water Resources Support Center, US Army Corps of Engineers
Art Hurme	Water Resources Support Center, US Army Corps of Engineers

AGENDA

11th U.S./JAPAN EXPERTS MEETING ON MANAGEMENT OF BOTTOM SEDIMENTS CONTAINING TOXIC SUBSTANCES

Seattle, Washington
4-6 November 1985

Co-Chairmen

Mr. Kenichi Fujimori
Director, Environmental Protection Division
Ports and Harbours Bureau, Ministry of Transport

Colonel George R. Kleb
Commander/Director, Water Resources Support Center
U.S. Army Corps of Engineers

Monday, November 4, 1985

0830-0930 Registration

0930-1000 Opening Remarks

1000-1030 Mr. Kenichi Fujimori, "Study on Improvement of the Marine Environment"

1030-1100 Break

1100-1130 Dr. Spencer A. Peterson, "Lake Restoration Methods: Some Work, Some Don't"

1130-1200 Mr. Nobumasa Morii, "Sedimentation and Effluent Concentration of Soft Clay on Reclamation Works"

1200-1330 Luncheon (Ticket Required)

1330-1400 Mr. James D. Krull, "Commencement Bay - A Remedial Investigation of Contaminated Sediments in a Marine Environment"

1400-1430 Mr. Douglas A. Hotchkiss, "Contaminant Mobility and Engineering Design at a Nearshore Confined Disposal Site--Port of Seattle Terminal 91 Short Fill"

1430-1500 Mr. Makoto Natori, "Test Results of Sediment Removal in Ise Bay"

1500-1530 Break

1530-1600 Mr. Keith Phillips, "Evaluation of Dredging as a Remedial Technology for the Commencement Bay Superfund Site"

1600-1630 Mr. Shinichi Kojima, "Effects of Sediment Removal in Tokyo Bay"

1800-	U.S. Reception (Madison Hotel - Ticket Required)

Tuesday, November 5, 1985

0830-0900	Mr. Tommy E. Myers, "Development of Leachate Testing Protocols for Disposal of Contaminated Sediments"
0900-0930	Dr. Motoo Fujiki, "Method for Estimation of Methylmercury Dissolved into Water from Dredged Material by Using Marine Copepod, *Acartia clausi*"
0930-1000	Dr. John W. Barko, "Water Quality Manipulations in a Eutrophic Impoundment"
1000-1030	Break
1030-1100	Dr. Michael H. Salazar, "The Effects of Sediment on the Survival of Mysids Exposed to Organotins"
1100-1130	Mr. Richard P. Leonard, "Food Chain Studies at Times Beach Confined Dredged Material Disposal Site, Buffalo, New York"
1130-1200	Mr. Jan Miller, "The Indiana Harbor Experience"
1200-	Travel to Port of Seattle Vessel
1300-1500	Port of Seattle Harbor Technical Tour and Luncheon (By Invitation - Ticket Required)
1830	Japan Reception (Westin Hotel) (By Invitation - Ticket Required)

Wednesday, November 6, 1985

0830-0930	Mr. Yasuo Abe, "Use of Anti-Turbidity Curtains at a Sand Compaction Piling Area in Yokohama Harbour"
0900-0930	Mr. Michio Gomyoh, "The Movement of Sediment Under a Progressive Wave"
0930-1000	Dr. Robert M. Engler, "The London Dumping Convention and Its Role in Regulatory Dredged Material: An Update"
1000-1030	Break
1030-1100	Mr. Shinzo Kimura, "Analytical Survey of Accumulations of Sediments in Canals of Tokyo Port"
1100-1130	Mr. Edward B. Hands, "Techniques for Monitoring Stability of Offshore Submerged Disposal Mounds"
1130-1200	Closing Remarks, Signing of Joint Communique

JOINT COMMUNIQUE

The eleventh meeting of experts on bottom sediments containing toxic substances pursuant to the Agreement Between the Government of the United States of America and the Government of Japan on Cooperation in the Field of Environmental Protection was co-chaired by Colonel George R. Kleb, Commander and Director, Water Resources Support Center, U.S. Army Corps of Engineers, and Mr. Kenichi Fujimori, Director, Environmental Protection Division, Ports and Harbours Bureau, Ministry of Transport, Japan. The purpose of the meetings conducted under this agreement is the exchange of information in regulatory, technical, and operational areas relevant to management of bottom sediments and exploration of areas where joint effort appears fruitful.

Experts from both countries presented technical papers on a variety of subjects including legal and research programs to evaluate and protect the environment. More specifically, evaluating and predicting the behavior of nutrients, organics, and contaminants in dredged sediments and identifying and evaluating the many alternatives to dredging and disposal of contaminated sediments from: lakes, rivers, waterways, estuaries, and marine waters associated with these sediments. Management practices with the most promising long-term utility were emphasized.

This eleventh meeting was highly successful and fruitful. The information exchange was very effective. The conference generally agreed that the dredging technology and sediment management programs of both countries will significantly benefit. The Co-Chairmen agreed that the next meeting will be in Japan in 1986 and that that date will be decided jointly.

George R Kleb
Commander/Director, Water
Resources Support Center
US Army Corps of Engineers

COLONEL GEORGE R. KLEB
November 6, 1985

Kenichi Fujimori
Director, Environmental Protection
Division, Ports and Harbours Bureau
Ministry of Transport

KENICHI FUJIMORI
November 6, 1985

STUDY ON IMPROVEMENT OF THE MARINE ENVIRONMENT

Kenichi Fujimori
Director, Environmental Protection Division
Ports and Harbours Bureau
Ministry of Transport
Kasumigaseki, Chiyoda-ku, Tokyo 100

ABSTRACT

Although the marine environment in Japan has significantly improved, problems such as eutrophication in closed bays still remain. The Ports and Harbours Bureau of the Ministry of Transport administers improvement projects and has developed related technologies. The success of these projects requires the study of various technologies which promote the self-cleaning functions of natural seawater and the integration of relevant technologies into combinations of the highest efficiency.

INTRODUCTION

The Ministry of Transport (Department of Ports and Harbours) is designated to undertake "projects for the prevention and elimination of marine pollution." For this purpose, the Ministry of Transport has undertaken execution and investigation with regard to improving the marine environment through direct control projects (undertaken by the Ministry of Transport) and subsidized projects (undertaken by local public bodies with government subsidy). Special investigations are made on the prevention and elimination of organic pollution in closed bays. This paper describes future studies on improvement of the marine environment with reference to changes in the marine environment and results obtained in investigations already conducted.

CHANGES IN THE MARINE ENVIRONMENT

Seawater

Health and living environmental standards concerning pollution of seawater were established in 1971 based on the Basic Law for Environmental Pollution Control.

Health environmental standards apply to items such as cadmium, cyanide, organic phosphorus, lead, hexavalent chromium, arsenic, alkyl mercury, polychlorinated biphenyls (PCB), and total mercury, which are harmful to human health. Living environmental standards apply to organic pollution sources which are hazardous to usable water, and are measured in units pH, such as dissolved oxygen (DO), biochemical oxygen demand (BOD) or chemical oxygen demand (COD), suspended solids (SS), and a number of coliform groups.

As a regulating measure to fulfill environmental standards, the effluent regulations based on the Water Pollution Control Law were enforced the same year. Since then, considerable improvement has been recognized in health environments. Improvement of the living environment, however, has not been as successful, especially in enclosed bays such as Tokyo Bay, Ise Bay, and Seto Inland Sea (Figure 1).

The locations of Tokyo Bay, Ise Bay, and Seto Inland Sea are shown in Figure 2 and pertinent data given in Table 1. The seawater of these enclosed bays is still, and the waters are not exchanged as well as in the open sea. The bays are utilized for shipping and are surrounded by densely populated urban area. Therefore, the use of these waters is very dense and, at the same time, chronic organic pollution is created by the inflow of large amounts of industrial and household wastewater.

Under these circumstances, the reduction of the total quantity of the pollution loads of Tokyo Bay, Ise Bay, and Seto Inland Sea has proceeded since 1979. Due to the urgent need for environmental preservation in Seto Inland Sea, the Temporary Control Law for Preservation of Seto Inland Sea Environment was enacted in 1973, and was revised to a Special Control Law in 1978. Organic pollution of these areas is caused by direct inflows of pollutants and

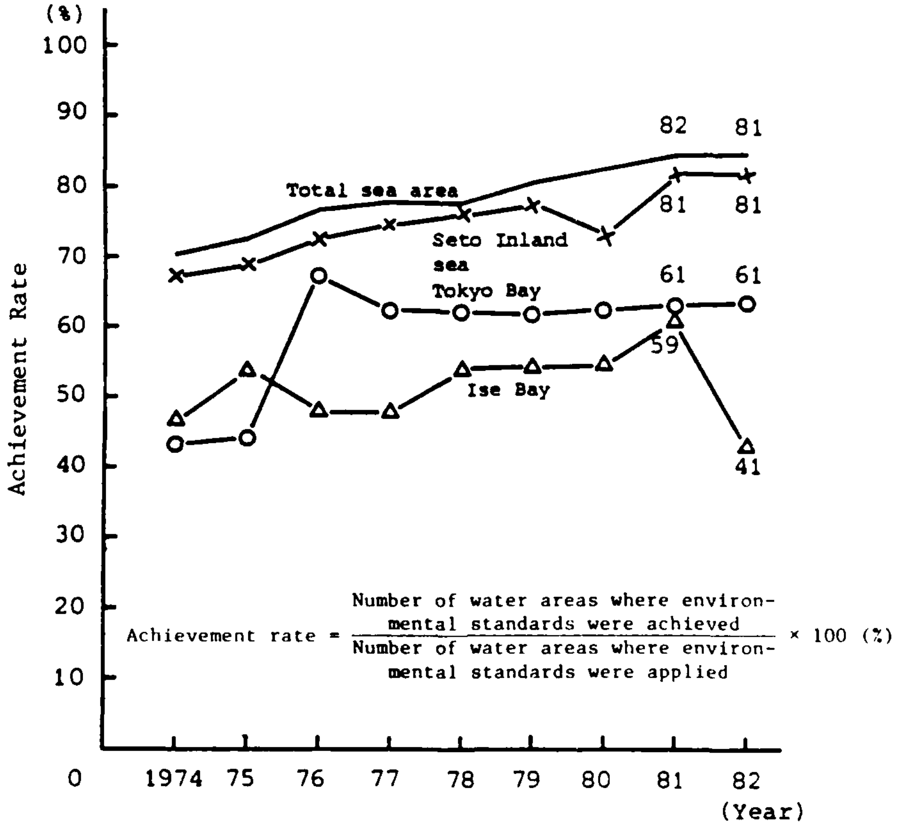


Figure 1. Changes in environmental standards achievements in wide, closed water areas (COD). Values from a survey by Environmental Agency.

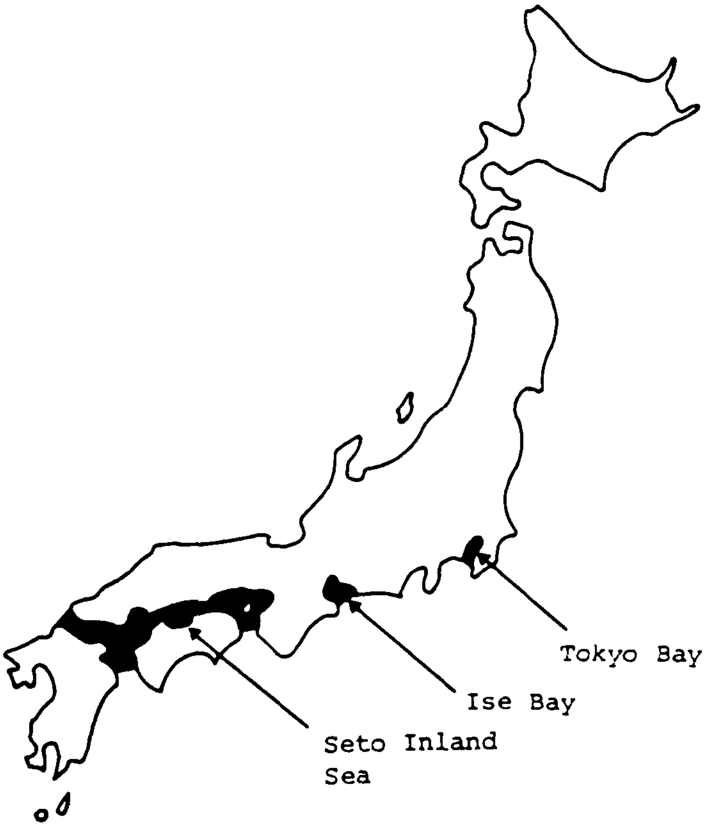


Figure 2. Location of Tokyo Bay, Ise Bay, and Seto Inland Sea

by organic matter generation (secondary pollution) such as phytoplankton propagation caused by increased nitrogen and phosphorus (eutrophication). In recent years eutrophication control of enclosed bays has been vigorously investigated.

Preservation of the sea environment in the enclosed bays is intended to protect fishing areas, prevent and eliminate red tides, and preserve recreation areas. These situations are discussed below.

TABLE 1. TOKYO BAY, ISE BAY, AND SETO ISLAND SEA DATA

	Tokyo Bay	Ise Bay	Seto Island Sea
Area, km^2	1,200	1,700	21,800
Volume, km^3	18	34	816
Average depth, m	15	20	37

Regarding protection of fishing areas, excessively eutrophic waters are unsuitable for the growth of marine animals. Marine resource exhaustion is thought to be caused by deteriorated water quality such as generation of poor oxygen water mass from excessive oxygen consumption in water through increased COD, and the decrease of benthos is thought to be caused by increased organic pollution and bottom sludge.

In the relationship between water quality and marine organisms, it should be noted that the optimum water quality differs depending on the species of fishes, and that objectives of fisheries vary according to the change in tastes. But, fundamentally, water quality preservation is a prerequisite for the preservation of fishing areas, and environmental control is now required for effective utilization of fishing areas.

Regarding the prevention and elimination of red tides, the causes of red tides were specified after much investigation. Increased nitrogen and phosphorus (eutrophication) is today regarded as the major cause. A red tide is a sudden development of phytoplankton on a mass scale. The number of red tide occurrences in Seto Inland Sea is shown in Figure 3. The frequency has decreased since 1976, the peak year, but still remains at a comparatively high level. Water quality has not sufficiently improved from the viewpoint of prevention and elimination of red tides. Red tides force people in the fishing industry to limit the quantity of fish cultures and to evacuate fish cultures (to limit or evacuate cultured fish to avoid the damage by red tides) which involves extra costs (which are not included in the damages shown in Figure 3). Types of damage by red tides are death of potential fish catch, escape of fishery resources from the sea areas, development of poisonous shellfish (poisonous matter accumulation in shellfish bodies), death of bottom organisms due to low oxygen levels, and adverse effects to recreation facilities (closing of bathing resorts).

Regarding the preservation of recreational areas, clean sea water is a prime requisite. Bathing resort water quality standards established by the Environment Agency are set under 2 ppm of COD. The future demands for recreation in sea areas will show a steady increase, and to accommodate that increase, it is desirable to create comfortable sea environments where people can enjoy various marine recreations. A recent survey determined that the level of bathing resort facilities is inadequate, and that seawater quality limits conditions for maintaining bathing resorts.

Bottom sludge

Environmental preservati[illegible] the viewpoint of water quality, but bott[illegible] considered, because of its close relationship with water [illegible] exchange and with the oceanic [illegible] some cases, the bottom sludge should be [illegible].

In general, [illegible] are COD, IL (ignition loss), and [illegible] the values of each index in enclosed [illegible] values. Standards for sludge dredging in polluted [illegible] harbors and rivers are said to be [illegible] = 0.8 - 1.5 mg [illegible] calculated that [illegible] in Japan cont[illegible]

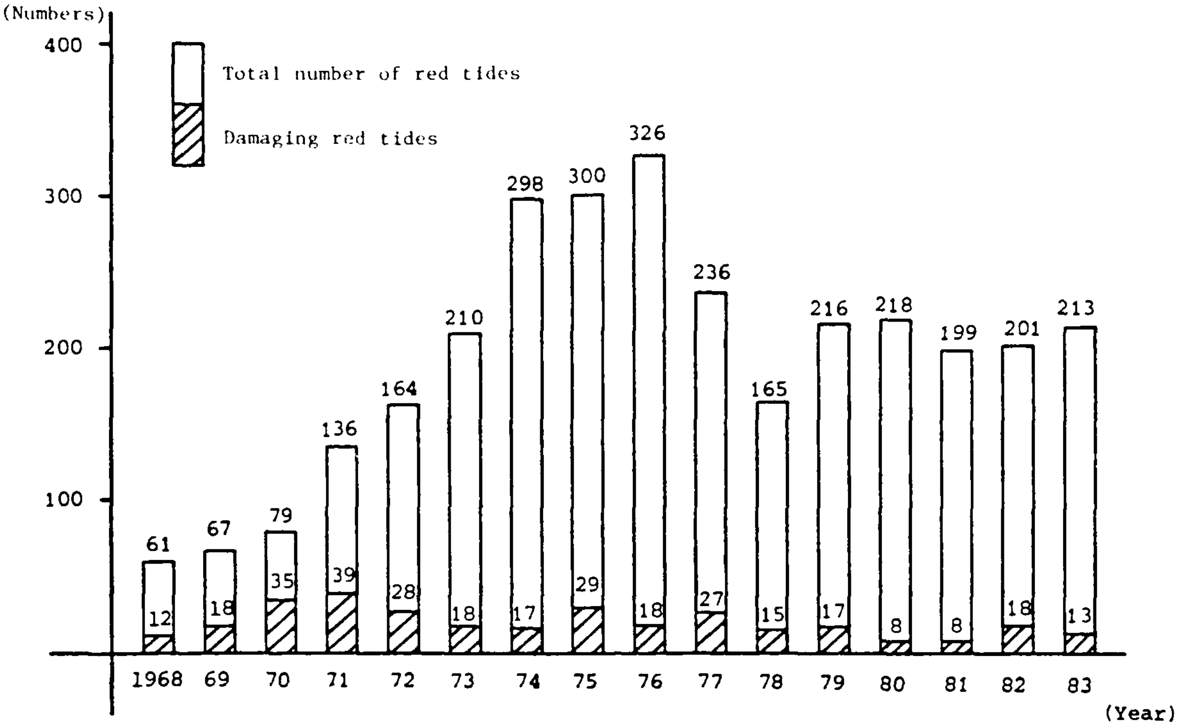


Figure 3. Total number of red tides and number of damaging red tides in Seto Inland Sea (survey by Fishery Control Office of Seto Inland Sea, Fisheries Agency).

700 million m^3 of sludge with over 30 mg/g COD in Tokyo Bay and others. Moreover, changes in bottom substances with the passage of time show no improvement. Table 2 shows the quantity of polluted bottom sediment in five bay areas.

TABLE 2. PRESENT QUANTITY OF POLLUTED BOTTOM SEDIMENT IN FIVE BAY AREAS (Over 30 mg/g COD)

Sediment	Tokyo Bay	Osaka Bay	Hiroshima Bay	Suo-Nada Sea	Ise Bay
Total Area, km^2	1,200	1,529	946	3,100	510
Polluted Bottom Sediment					
Area, km^2	441	670	530	16.4	112
Thickness, cm	53	34	42	5	20
Quantity, million m^3	235	226	221	0.8	22

NOTE: Osaka Bay, Hiroshima Bay, and Suo-Nada Sea are in Seto Inland Sea.

CIRCULATION IN THE SEA

As previously stated, the present condition of the enclosed bays is not satisfactory, and various efforts are under way to improve the sea environment. The Ministry of Transport is also conducting bottom sediment clarification projects as one sea environment cleaning measure. In such measures it is necessary to understand the balance of substances and the mechanism of seawater pollution. A model of the circulation mechanism in the sea is discussed in the following paragraphs.

Figure 4 shows a conceptional diagram of circulation patterns. Fundamentally, seawater pollution is caused by dilution and mixing by dispersion of inflow pollutants. This model, however, includes a eutrophication process which is a production of organic matters (COD) from inorganic phosphorus by phytoplankton. Phytoplankton production requires light. The model has double layers in a vertical direction in the seawater, one with light and the other without. Seawater pollutants exchange with the bottom substances by sedimentation and by redissolution of bottom substances back into the seawater.

The factors of each process in the model (production, decomposition, sedimentation, dissolution, oxygen consumption) were obtained from in situ experiments and laboratory experiments and represent the present state of pollution by adding an inflow load. In the pollution mechanism, a significant phytoplankton production effect is calculated and shows water pollution by eutrophication. Biological activities other than phytoplankton production and organic matter decomposition by bacteria (such as predation by higher animals), are excluded from this model.

Table 3 shows the balance of organic pollution substances of seawater obtained from the above model. These results reveal that internal production (COD production by the production activity of phytoplankton from nutritive

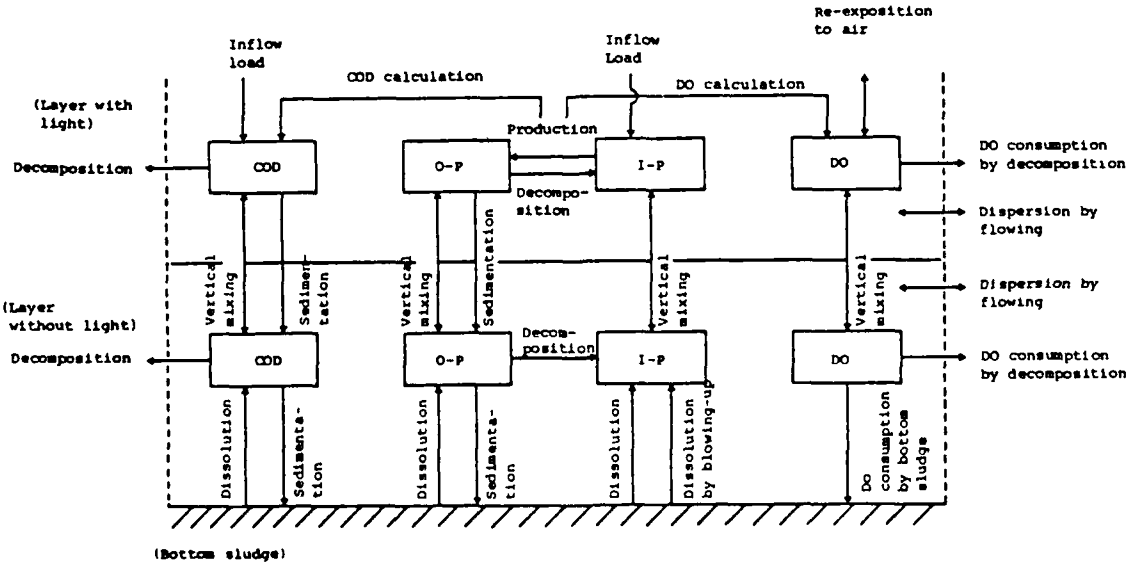


Figure 4. Conceptional diagram of water circulation

TABLE 3. BALANCE OF SUBSTANCES (PRESENT SITUATION)

				Tokyo Bay		Osaka Bay		Hiroshima Bay		Suo-Nada Sea		Ise Bay	
				(t/day)	Composition ratio (%)	(t/day)	Composition ratio (%)	(t/day)	Composition ratio (%)	(t/day)	Composition ratio (%)	(t/day)	Composition ratio (%)
Present situation	COD	Inflow		284.3	4.0	285.0	6.3	67.0	2.7	50.0	1.4	50.0	2.6
		Dissolution		96.2	1.3	131.0	2.9	42.0	1.7	614.0	16.3	70.0	3.6
		Internal production		6,813.0	$\boxed{94.7}$	4,121.0	$\boxed{90.8}$	2.414.0	$\boxed{95.6}$	3,095.0	$\boxed{82.3}$	1,815.0	$\boxed{93.8}$
		Total		7,193.5	100.0	4,537.0	100.0	2,523.0	100.0	3,759.0	100.0	1,935.0	100.0
	P	Inflow	PO4-P	15.0	53.0	25.5	55.9	2.2	13.9	0.9	6.8	2.3	25.8
			O-P	7.0	24.7	6.4	14.0	0.6	3.8	0.2	1.5	1.5	16.9
			Total	22.0	77.7	31.9	69.9	2.8	17.7	1.1	8.3	3.8	42.7
		Dissolution (PO4-P)		6.3	$\boxed{22.3}$	13.7	$\boxed{30.1}$	13.0	$\boxed{82.3}$	12.1	$\boxed{91.7}$	5.1	$\boxed{57.3}$
		Total		28.3	100.0	45.6	100.0	15.8	100.0	13.2	100.0	8.9	100.0

NOTE: Units are tons/day; Osaka Bay, Hiroshima Bay, and Suo-Nada Sea are in Seto Inland Sea.

salt) has a very large COD load ratio in the sea, and that the dissolving load from the bottom sludge has a considerable ratio in the load of nutritive salt, which is a major cause of internal production. Therefore, bottom sludge plays a significant role in organic seawater pollution.

SELF-CLEANING ACTION OF THE SEA AND ITS ENHANCEMENT

As the previously mentioned circulation mechanism indicates (Figure 4), nature is provided with a mechanism to clean inflow pollution load by dilution and mixing, sedimentation, or decomposition. Therefore, if the inflow pollution load is under the permissible level, the environment can be maintained by the sea's own self-cleaning action. Consequently, the concept of "environmental capacity" is proposed as maximum pollution load at permissible limits in a definite area (e.g. enclosed bays).

Thus, nature's self-cleaning action can be taken into consideration when environmental cleaning measures are adopted. Specifically, the self-cleaning action such as dilution and mixing in the above-mentioned substance circulation mechanism can be promoted, or actions which have been excluded (e.g. predation by higher animals) can be promoted. Development and research in this field are under way at various organizations. In the field of wastewater treatment, soil infiltration systems of small-scale wastewater treatment (by the adsorption, filtration, and decomposition actions of soil) and the stabilizing process of nitrogen and phosphorus with water plants (waterhyacinth, etc.) are especially being developed.

The Ministry of Transport has studied methods to promote the sea's own self-cleaning action.

The main factors of the sea's self-cleaning actions are exchange of seawater, exposure to air, predation, and the seawater cleaning action of dry beaches and sand beaches. These factors are discussed below.

Seawater exchange to prevent standing water in enclosed bays can be promoted by widening and deepening the mouths of the bays, forced water conveyance with electric power, excavation of new waterways, and construction of trenches (excavation of trenches in the sea bottom). Seawater exchange promotion processes for wide bays such as Tokyo Bay, Ise Bay, and Seto Inland Sea are now being researched with model tests, and value simulations are under way. It can be expected that seawater exchange can be promoted as an additional effect from excavation of new fairways at bay mouths. On the other hand, in small areas such as inside harbor breakwaters, a process to promote seawater exchange has been achieved in harbor improvement projects. In these cases, one measure is to construct a raceway in one part of a breakwater which will promote interchange with water from the open ocean. Figure 5 shows a raceway in the Kikuma Harbor breakwater, Ehime Prefecture, constructed in 1984.

Exposure to air is a process which supplies oxygen to seawater. This exposure promotes organic pollutant (COD) decomposition, one of the substance circulation mechanisms (Figure 4). This process adopts a method by which air bubbles are blown into the seawater by using electric power; therefore pollutant dilution by mixing with seawater is expected to result as an additional effect. In this respect, the Ministry of Transport carried out verification

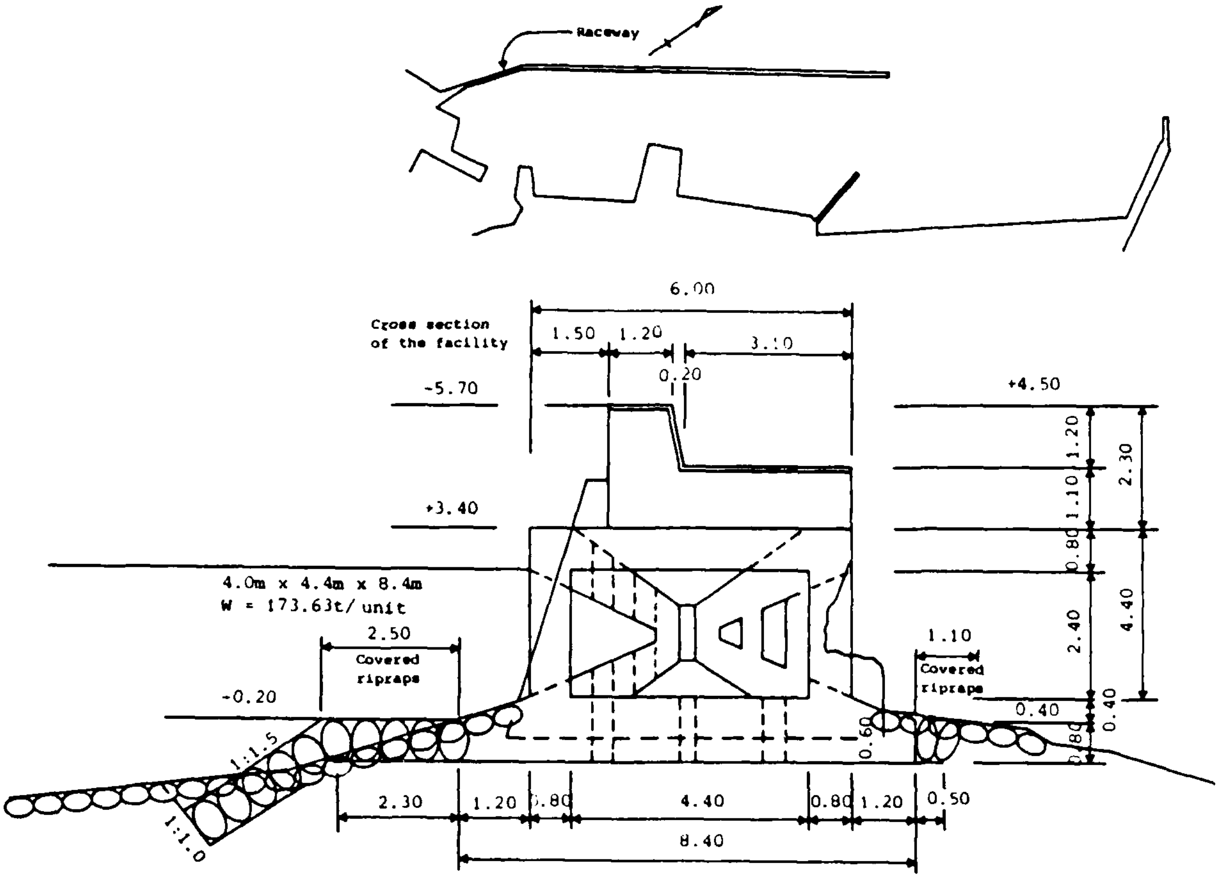


Figure 5. Kikuma Harbor raceway project

tests at Shimizu Harbor in 1982, the outline of which is shown in Figure 6. The result of the tests revealed that, in a definite sea area, exposure to air shows layer breakdown and air dissolution effects.

The increase of DO and the decrease of phosphorus were especially significant. Though this system requires maintenance, spectacular effects can be expected by using the system as an ornamental fountain. It is desirable to utilize the system in a total waterside environment improvement project.

Regarding predation by higher animals, the activities of seaweeds, fish, and shellfish to take in pollutants were observed in nature, and a concept to preserve the environment by controlling these activities was realized. Utilization of living creatures and plants presents certain problems such as the scale of effect, compared with total pollution load, and the maintenance of living conditions for the creatures and plants. It is, however, possible to use this method as a very reasonable environmental cleaning method by utilizing recovered living creatures and plants as resources. It was determined from laboratory tests that the maximum intake speed was 20 μg COD/piece/minute with filter feeders (shellfish). Nitrogen and phosphorus stabilization using large seaweeds has been studied by the Fisheries Agency, and the Ministry of

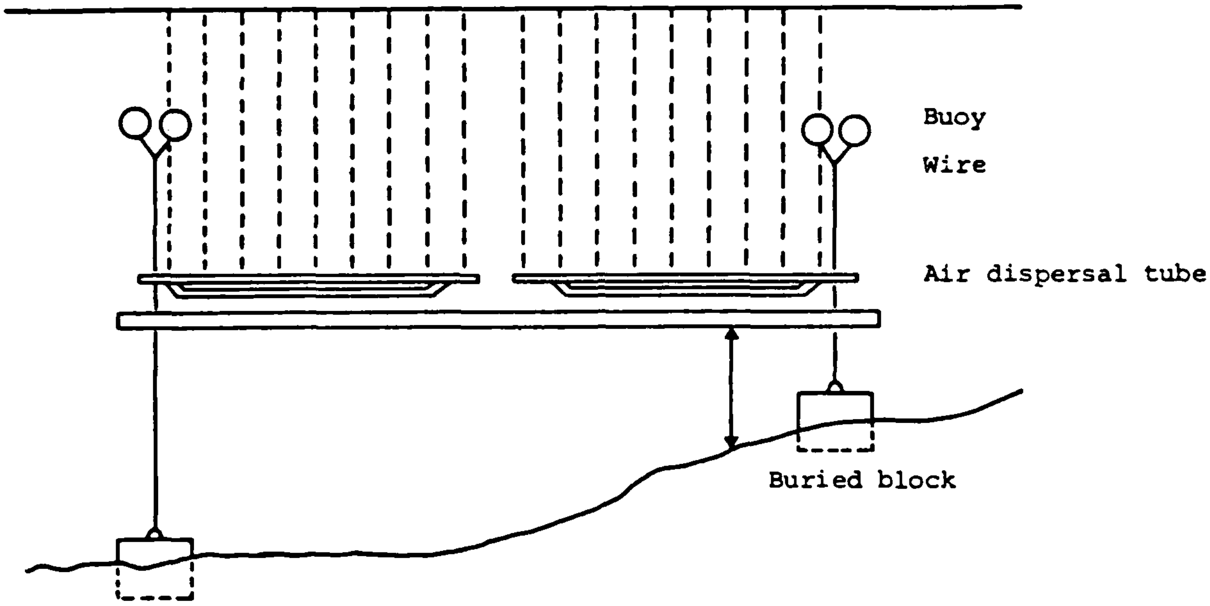


Figure 6. Test of exposition to air at Shimizu Harbor

Transport is conducting verification tests related to the use of a living filter of reeds to treat excess organic sludge water at Ube Harbor, 1982-1984. Figure 7 is a plan view of the site.

The main factors of the seawater cleaning action of dry beaches and sand beaches are filter feeder predation, exposure to air by wave breakdown which accelerates the decomposing action of bacteria in aerobic environments, and absorption of pollutants by sand particles. It is calculated that these provide significantly high cleaning functions. Therefore, the preservation and preparation of dry beaches and sand beaches are very effective sea environment preservation measures together with utilization as recreation places. Further studies are expected.

FUTURE SEA CLEANING MEASURES

As stated, there are various processes to promote self-cleaning actions of the sea. In adoption of cleaning methods for wide enclosed bays such as Tokyo Bay, Ise Bay, and Seto Inland Sea, these processes can be combined or arranged, but many problems still remain, for example, determining, the best arrangements for raceways, choice of air exposure equipment, construction of dry and sand beaches, adjusting living conditions for higher animals to maintain their predation activities, and arranging equipment for nitrogen and phosphorus stabilization by seaweeds.

However since 1979 the Ministry of Transport has carried out feasibility studies related to bottom sediment clarification (removal of organic sludge from the sea bottom) as part of a study concerning seawater cleaning measures in enclosed bays. The investigations were carried out on actual pollution situations; the scope of the research was broad and included simulation of

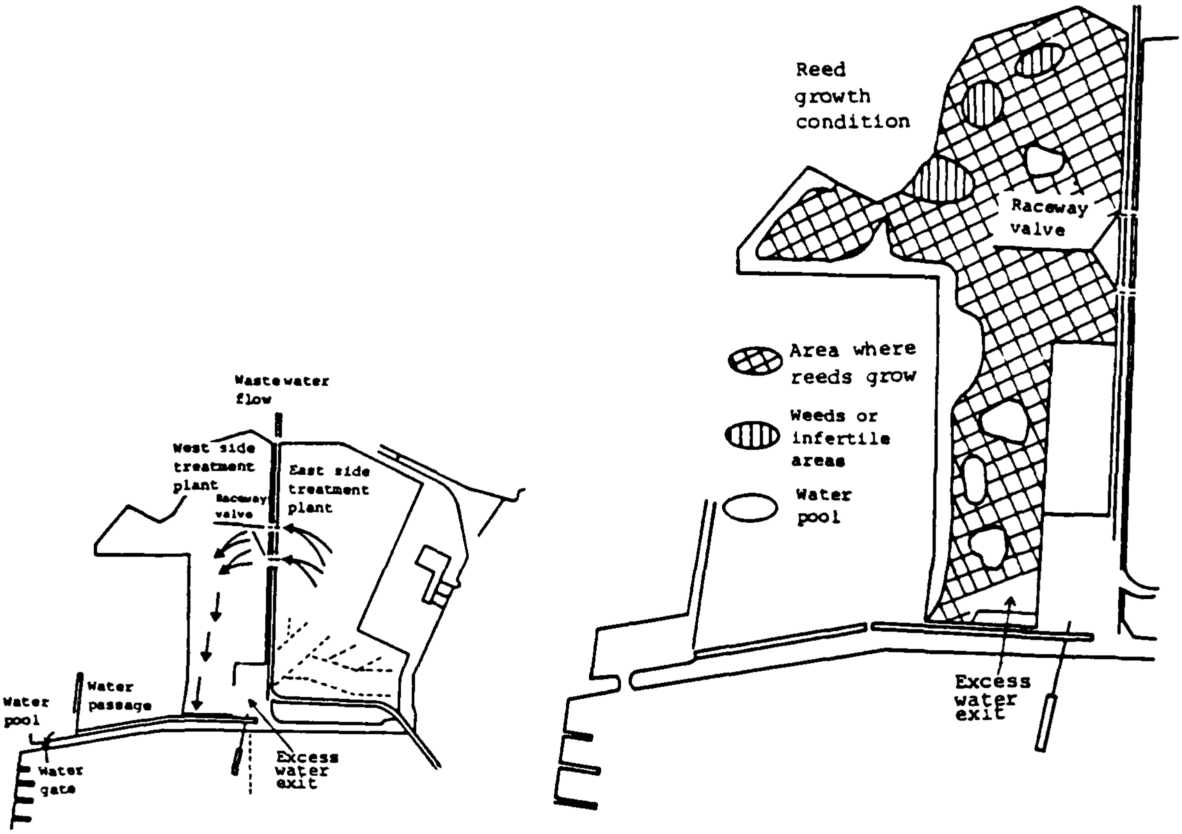


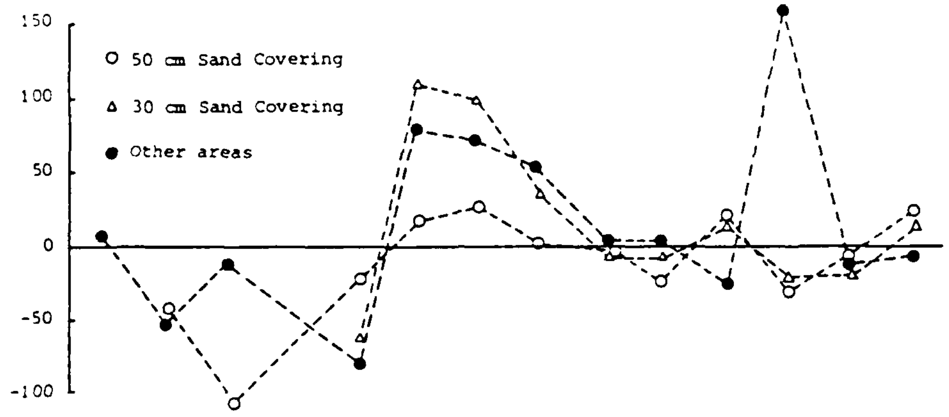
Figure 7. Sludge water treatment facility at Ube Harbor (a living filter)

seawater pollution and test engineering works. Dissolution of nutritive salts from the surface layer of the sea bottom, biological production in the sea, and other important data were obtained.

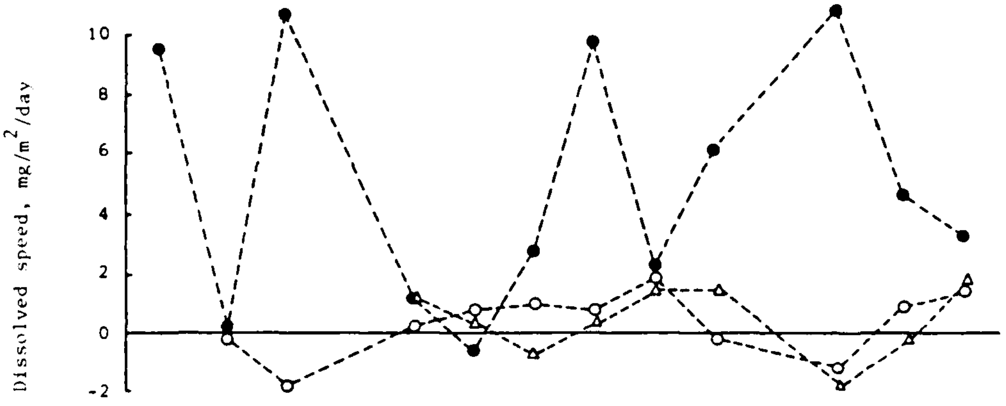
The effects of various bottom sediment clarification processes were confirmed through tests and follow-up studies in the sea areas. Dredging, sand covering, and trench digging were tested, and the results of follow-up studies were reported. Some are as discussed below.

Sand covering work was carried out in Hiroshima Bay in Seto Inland Sea. In November 1979, sand covering (1.92 ha) was done to a 50-cm thickness, and sand covering (4.48 ha) to a 30-cm thickness was done in August 1980.

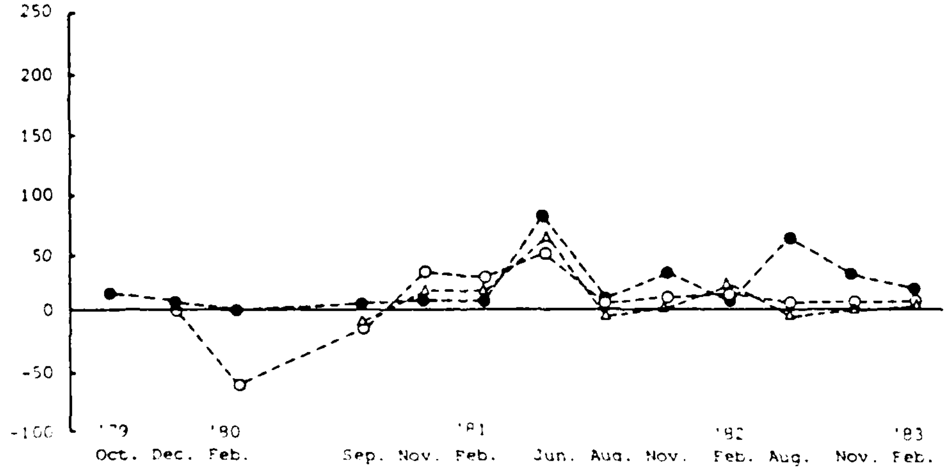
Figure 8 shows the results of follow-up studies of dissolution controlling effects. It was determined that pollutant dissolution (COD, T-P, and T-N) from the sea bottom was decreased by covering the bottom sludge with sand, and that the effects of the covering lasted. Dissolution of T-P was remarkably decreased by sand covering. It is assumed that the chemical adsorption of phosphorus by sand particles in the aerobic environment at the sea bottom in addition to physical restriction by covering served to decrease pollution in the bay.



a. COD



b. T-P



c. T-N

Figure 8. Sand covering effects in dissolution control, Hiroshima Bay

Dredging and sand covering were carried out in Ise Bay. In June 1980, dredging 60 cm in depth (1 ha), and in June 1981 dredging (1 ha) of 30 cm in depth was conducted. In June 1983, sand covering (5 m by 5 m each) was applied to a 10-cm and 30-cm thickness. Figure 9 shows the results of follow-up studies of benthic organism recovery. The average number of benthic organism species and the actual organism population increased. The increase of benthic organisms in cases of sand covering was especially remarkable.

Improvements in the sea environment are expected to result from bottom sediment clarification work such as dredging and sand covering. Further investigations and research will be continued to undertake bottom sediment clarification projects. It should be noted that, as stated, the selection of suitable cleaning methods should be made carefully because the sea's substance circulation is extremely complicated.

Therefore, overall sea environment cleaning measures will be established by combining bottom sediment clarification works and various self-cleaning promoting methods. Details of the processes will be established in the future. It is important to review closely the most suitable process combinations to preserve the environment of enclosed bays.

CONCLUSIONS

Tokyo Bay, Ise Bay, and Seto Inland Sea will become more important as valuable spaces and their resources will become more important with the progress of the development of the sea. Under these circumstances, the Ministry of Transport will continue in its efforts to establish total cleaning measures for the environment of the sea around Japan. Adequate cleaning projects will be carried out in accordance with changes in seawater quality and bottom substances, and with the demands of the time.

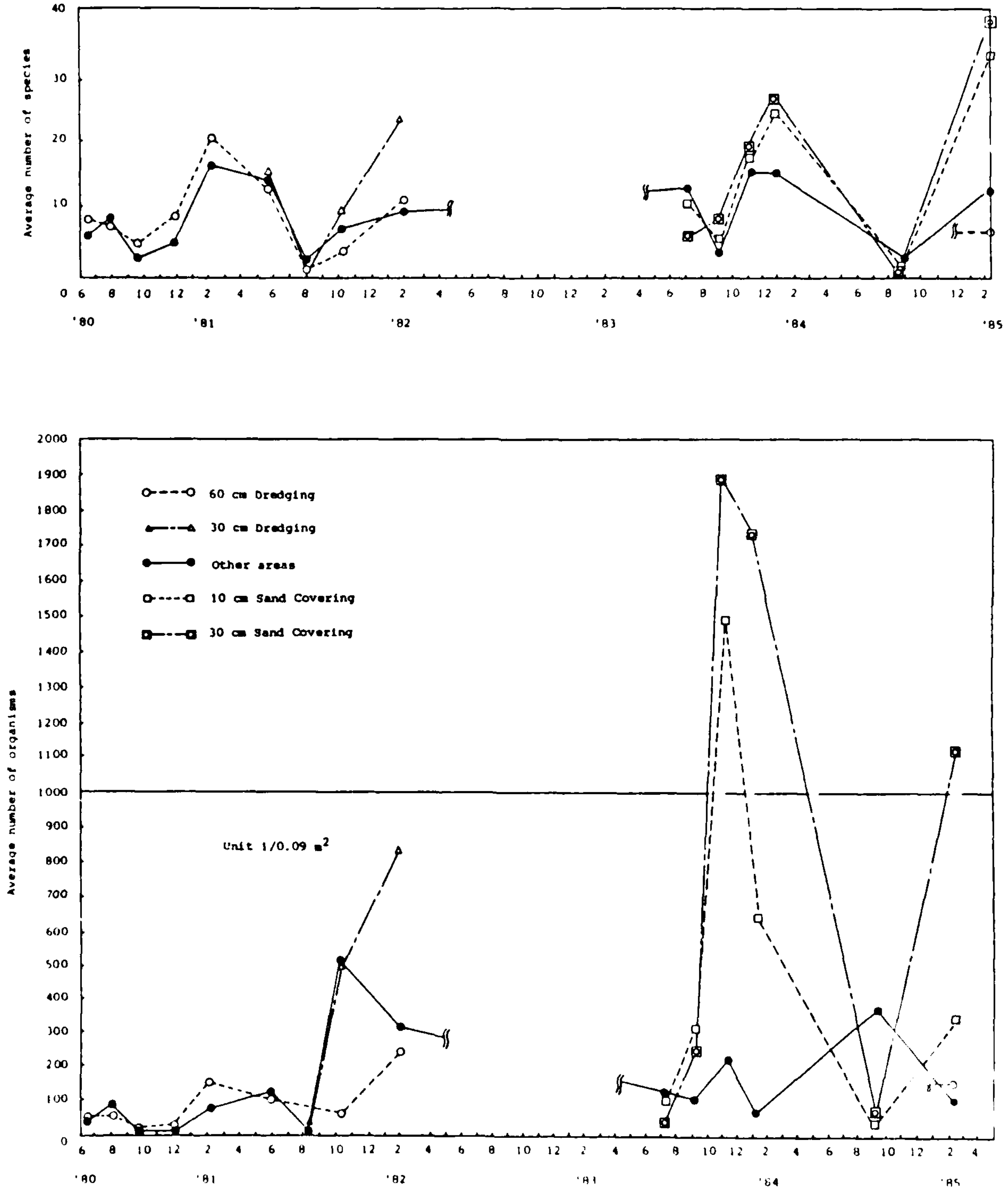


Figure 9. Recovery of benthic organisms after dredging and sand covering, Ise Bay

LAKE RESTORATION METHODS: SOME WORK, SOME DON'T

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ABSTRACT

Lake restoration has grown out of pilot and full-scale lake manipulation studies of the late 1960s and early 1970s. Some procedures have proven to be successful, and some have failed. This paper broadly divides the techniques into those that have proven to be successful, those that require more research, and combinations of treatments that are desirable together and those that should be avoided together. The procedures are divided further into those that reduce nutrient concentrations, and those that control algae and macrophyte biomasses directly. Both successes and failures are addressed with assessment of the reasons for either response. The procedures are summarized with assessment of the reaons for either response. The procedures are summarized briefly from the major reference on this topic: *Lake and Reservoir Restoration* (1986) by G. D. Cooke, E. B. Welch, S. A. Peterson, and P. R. Newroth, Butterworth Press. Boston, Massachusetts.

INTRODUCTION

Cultural eutrophication of lakes was thought for many years to be an irreversible process. Even those lakes which had wastewater discharges diverted from them showed little if any signs of recovery. This latter fact attests to the very slow process of lake recovery if the lakes are left to their own devices. However, a blooming of research in the late 1960s and early 1970s began to reveal some aspects of the trophic dynamics of lakes previously not understood. For example, in 1966, Sakamoto published his now famous paper depicting the relationship between total phosphorus concentrations in the water and algal biomass. Vollenweider (1975) began to publish papers in 1975 describing the relationships between phosphorus loading rates to lakes and their hydraulic residence times. The phosphorus discharge rates and internal loss rates were gradually refined throughout the early 1970s, thus producing the basis for lake restoration design criteria. The object of lake restoration became either phosphorus reduction in the water column or direct control of algae or macrophyte biomass. Many methods were developed to accomplish these goals. Some of the methods have proved successful and some have not. The purpose of this paper is to provide a brief summary on the success and failure of several different lake restoration procedures. The information in this paper is from a forthcoming book by Cooke et al. (1986).

SUCCESSFUL TECHNIQUES

Procedures to Reduce Nutrient Concentrations

Nutrient Diversion

Diversion of nutrient income is widely recognized as a highly necessary and usually essential first step to obtaining improvement in lake trophic state. Uttormark and Hutchins (1980) examined the responses of 23 lakes to diversion and found that while concentration of nutrients (P in this case) may have declined, only 30 percent of the lakes showed improvement to a lower trophic state. This may have been because diversion was insufficient to lower concentration to the level that would change the lakes from eutrophic to mesotrophic. Despite this problem, a reasonable reduction in algal biomass, absence of blue-green algal scums, and a reduction of fecal coliform counts to an acceptable range will make diversion a success.

Another reason diversion might fail to significantly improve trophic state is continued release of nutrients from sediments into the water column. This factor, along with morphometric and hydrologic features such as mean depth and flushing rate, is important in determining how much change in concentration will occur after diversion and how long it will take for the lake to reach a new, lower concentration equilibrium. Ultimately, internal P release should decline as older, richer sediments are depleted of nutrients and are covered with those of lower nutrient content, thus producing lower nutrient levels in the water column.

Silt income control may be very significant in preventing the rapid development of shallow water and the spread of nuisance matrophytes An extensive littoral area from long-term deposition might require sediment removal or other procedures before macrophyte growth is controlled. Control of sediment income from the watershed is thus again an essential first step to improvement in trophic state.

Dilution/Flushing

Dilution reduces the concentration of nutrients in the water column by adding nutrient-poor water. Flushing reduces algal biomass by increasing the loss rate of cells. Both effects will occur if large amounts of low nutrient water are added.

Dilution has been very effective in Moses Lake and Green Lake, Washington (Oglesby 1969). Welch (1981) concluded that the best plan is to have a continuous low-rate input of low-nutrient water, coupled with nutrient diversion. While this procedure will bring about lake improvement, its applicability is limited by availability of nutrient-poor waters for diversion into eutrophic lakes.

Phosphorus Inactivation/Precipitation

The purpose of this technique is to lower the concentration of P in the water column by either precipitating it or preventing its release from sediments (inactivation). Phosphorus in the water column is sorbed to salts of aluminum (aluminum sulfate or sodium aluminate) as they are added to the lake surface or to the hypolimnion. With inactivation, sufficient aluminum is added to create a barrier to P release at the sediment/water interface.

Phosphorus inactivation/precipitation has the potential to create aluminum toxicity in lakes. This situation could occur through excessive application of soluble aluminum (Al III) or through a decrease in pH following application of appropriate amounts of aluminum. Kennedy and Cooke (1981, 1982) have described a method for determining the maximum safe dose of aluminum sulfate for a lake, based on lake water alkalinity and toxicity of dissolved aluminum to trout. An alternative dose procedure for softwater lakes has been described by Dominic (1980). Following these procedures ensures maximum P inactivation while minimizing the potential for creating toxicity problems.

The P inactivation technique is particularly effective in long-term control of P concentration in the water column (Cooke and Kennedy 1981a, 1981b). The longest period of P control to date is 12 years (Garrison and Knauer 1984). Failures have been due to insufficient dose, polymixis, and insufficient nutrient diversion.

Sediment Removal

Sediment removal is one of the most commonly prescribed techniques for long-term lake improvement. Lake deepening and the removal of toxic materials, macrophytes, and nutrient-rich sediments are its main purposes. Until recently (Peterson 1979, 1981, 1982), little published documentation about its effectiveness, environmental impact, and costs was available.

Lake deepening projects are generally successful except in instances where sediment income remains high. Similarly, control of nutrient release from sediments can be achieved by removal, although it is often less costly to do so through P inactivation. There is little documentation of macrophyte control by dredging. The success or failure of macrophyte control appears to be associated with dredging beyond the light compensation depth.

There is widespread concern about the negative environmental impacts of dredging, but most negative effects (algal blooms, high turbidity) are of short duration. One common problem is inadequate sizing of disposal area. Diking in upland areas is the most frequently used means of containing the dredged material since the filling of wetland areas is strictly controlled. Fill permits are required for any areas in excess of 10 ha.

Dredging costs range from \$0.25 to \$14.00/m^3. Removal of contaminated material may exceed \$25.00/m^3. One way of lowering costs is to sell uncontaminated dredged material for topsoil dressing.

Procedures to Control Algal and Macrophyte Biomass

Drawdown

Lake level drawdown is used successfully to control the growth of certain macrophyte species, to consolidate flocculent lake sediments, to provide an opportunity to repair docks and dams, to remove sediment or to install sediment covers, and to manage fish (Cooke 1980a).

Not all aquatic plants are susceptible to the freezing or high temperature conditions of a winter or summer drawdown. Alligatorweed (*Alternanthera philoxeroides*) and brittle naiad (*Najas flexilis*) have been reported to increase in density after a drawdown. Waterlily (*Nuphar* sp.), waterhyacinth (*Eichhornia crassipes*), and musk grass (*Chara vulgaris*) usually decrease. For most macrophytes, the published data are too sparse to generalize. A danger may be that susceptible species will be replaced by tolerant ones, leading some investigators to suggest that the best control will be achieved by 2 to 3 years of drawdown, followed by 2 years of stable water levels (Lantz et al. 1964; Lantz 1974). Most drawdowns are done during the winter since this does not interfere with most recreational uses and there is no invasion by terrestrial plants. Also, there is greater assurance of refill during spring melt and rainfall.

Drawdown has been effective in fish management. In Louisiana reservoirs, populations of shad and sunfish are removed by winter drawdown, and their spawning is prevented by summer drawdown (Lantz et al. 1964). In reservoirs with 5 or more consecutive years of drawdown, game fish standing crops and reproductive rates have increased.

Negative effects of water level drawdown include algal blooms upon refilling. Fish kills have occurred in isolated pools of water during drawdown, and great changes have taken place in the macroinvertebrate communities.

This management technique can be effective in macrophtyte control and can be beneficial in the sense that it provides management personnel with access to the lake for the purpose of enacting other techniques.

Sediment Covers

Several sheeting materials have been tested for their effectiveness in preventing macrophyte growth (Cooke 1980b). Impermeable materials such as polyethylene have been found to be effective but troublesome because they accumulate gases underneath and balloon to the surface. One way to prevent "ballooning" is to cover the plastic with gravel and silt, but this practice provides new substrate for plant growth, thus defeating the purpose of the plastic cover. Burlap has been shown to be effective for one growing season, but the material usually decomposes and must be replaced annually.

Fiberglass screen coated with polyvinyl chloride (PVC) has been very effective in preventing macrophyte growth. It is easy to install and does not "balloon," but is very expensive (\$140 to \$170/ft^2; \$2.15/m^2). Perkins, Boston, and Curren (1980) suggested that control with screening was achieved by physical restriction of plant growth through the screen. In most instances, the screens must be repositioned annually to remove accumulated silt. Due to its cost, this material appears to be best suited for small areas around docks and bathing beaches.

TECHNIQUES REQUIRING MORE RESEARCH AND DEMONSTRATION

Procedures to Reduce Nutrient Concentration

Hypolimnetic Withdrawal

The object of this technique is to siphon or pump nutrient-rich hypolimnetic water from the lake. This technique removes nutrients from the lake and should reduce the impact of vertical entrainment of nutrients on the epilimnion. The siphon is sometimes called an "Olszewski tube" after its originator (Olszewski 1961).

Experiences with this device in small European lakes have been described by Bjork (1974) and Gachter (1976). Total P and internal P release decreased, transparency increased, and fewer blue-green algae were present.

This technique deserves further study since it is simple, dependable, and inexpensive. The high P, low dissolved oxygen content of the hypolimnion water could pose a threat to receiving waters. Other possible problems might include disruption of the thermocline, excessive water loss from the lake, and the requirement that the discharge end of the siphon be below the surface level of the lake.

Sediment Oxidation

Ripl (1976) described a procedure (discussed below) for oxidizing the top 15 to 20 cm of anaerobic lake sediment to reduce internal nutrient release. There are few published data regarding the success of this new procedure and its cost.

$Fe(Cl_3)$ is added first if Fe is naturally insufficient. Ferric hydroxide formation binds P and the sulfur in FeS is lost as H_2S. Calcium hydroxide is added next to increase pH to the optimum for denitrification. Finally, calcium nitrate is added to the sediments, oxidizing reduced carbon and acting as an alternative electron receptor. Reduction of iron (III) does not occur until the nitrate is exhausted. The chemicals are added using a harrow device. A positive result (lower pH, NH_4 decrease, 0_2 demand lowered) was found in Lake Lillesjon, Sweden (Ripl and Lindmark 1978), but costs of chemicals and equipment were high. Further laboratory and field evaluations of this technique are needed.

Procedures to Control Algae and Macrophyte Biomass

Biological Controls

Biological control methods may be the most promising because they offer the possibility of exerting control of nuisance plants without eradication and without the use of expensive machinery or toxic chemicals. Among the most controversial of the proposed biological control methods is the introduction of grass carp, or white amur (*Ctenopharyngodon idella* Val.), a voracious consumer of macrophytes. Successful introductions have been reported by Mitzner (1978) and by Shireman and Maceina (1981). The US Army Corps of Engineers and others are conducting large-scale feasibility studies in Florida. While new data are now being accumulated, there remains no certainty with regard to the effects of these animals on other species, their role in nutrient recycling, whether they transmit fish diseases, and whether or not they will reproduce and infest nontarget waters. Currently the fish is widespread in the southern Mississippi River system. It will be fortunate indeed if this animal proves to be useful for plant control. Many releases were made before many of the above questions were addressed.

A plant pathogen (*Cercospora rodmanii*) appears to be effective in control of waterhyacinth (*Eichhornia crassipes*) and is now being prepared for a large-scale operations test by the US Army Engineer Waterways Experiment Station (Freeman et al. 1981).

Since LaMarra (1975) reported the nutrient recycling ability of the common carp, others have further demonstrated eutrophying characteristics of benthivorous fish (Anderson et al. 1978; Keen and Gagliardi 1981). The word "biomanipulation" has been used to include lake improvement activities in which fish such as bullheads (Ichtalurids) are removed, or in which the pelagic food web is altered to decrease predation on the large herbivorous zooplankton (Shapiro, LaMarra, and Lynch, 1975; Shapiro 1979). These and other procedures deserve considerable further support and attention. These methods may, for example, be one way to improve lakes where significant nutrient diversion cannot be achieved.

Artificial Circulation

The purpose of this procedure is to prevent thermal stratification or to destratify the lake. The most effective and least expensive technique is to introduce compressed air at maximum depth, increasing a curtain of bubble and intensive mixing. An airflow rate of $9.4\ m^3/min/km^2$ should be adequate (Lorenzen and Fast 1977).

Artificial circulation is included as a "technique requiring more research" because the expected results often are not realized. The reason for this is not readily explained. However, Pastorak, Ginn, and Lorenzen (1981) found that in only 20 percent of the cases reviewed was the flow rate of $9.4\ m^3/min/km^2$ achieved, a factor that might account for some of the reported failures of the technique. The principal improvement is increased aeration of the water column and expanded fish habitat. Theoretically, aeration and circulation should inhibit P release from sediments, limit algal productivity by increasing cell residence time in dark waters, decrease pH, and perhaps shift the competitive balance from blue-green algae to greens and diatoms, limit the survival of blue-green pseudovacuoles (buoyancy), and favor the survival of herbivorous zooplankton. Phytoplantkon decreased in about half the cases examined by Pastorak, Lorenzen, and Ginn (1982). Transparency decreased more often than it increased. Circulation may introduce nutrients to the water column, and in nutrient-limited lakes an increase in biomass may occur. This technique might be most effective in lakes where nutrients are not limiting. However, additional research is needed.

Hypolimnetic Aeration

This is a technique to aerate the hypolimnion without achieving destratification (Pastorak, Lorenzen, and Ginn 1982). It is accomplished by an "air-lift" in which water is lifted up a cylinder by compressed air to the lake surface, aerated, and then returned to the hypolimnion.

Successful oxygenation is usually achieved, although metalimnetic oxygen minima may occur with an undersized aerator. Phosphorus concentration may be reduced following the introduction of oxygen; however, hypolimnetic levels tend to remain high. There are few data on biological effects, including those on phytoplankton, zooplankton, and fish (Pastorak, Lorenzen, and Ginn 1982). More documentation about the responses of lakes to this treatment is needed.

Harvesting

Harvesting is one of the more controversial procedures, in part because there is little scientific evidence to demonstrate long-term effectiveness. The technique has been incorrectly categorized with herbicides as a cosmetic treatment. Unlike the introduction of toxic materials to lakes, nutrients and oxygen-consuming biomass are removed during a harvest, thus protecting the lake from acute oxygen depletion as occurs with herbicides. There is some evidence that longer term control of macrophytes can be achieved by employing a proper cutting frequency and cutting technique. Since harvesting is usually less costly than herbicides, and because removal of nuisance plant biomass can have a protective effect for the lake, the use of this procedure deserves more careful attention and additional experimental work.

COMBINATIONS OF LAKE RESTORATION TECHNIQUES

For the most part, each lake or reservoir restoration or improvement technique has been addressed separately, as if the use of one precludes the simultaneous use of another. However, "integrated control methods," similar to those used in agriculture, may be far more effective and less costly than use of any single technique.

Some procedures, when used together, could counter each other or produce a negative effect. Some of these situations are summarized below.

- Flushing. Do not use with hypolimnetic withdrawal because the probability of destratification is greatly enhanced and would negate the effectiveness of withdrawal if it occurred. It is not compatible with dilution unless the water used for flushing is very low in nutrients, thus making the effects of the two essentially identical. If used with phosphorus inactivation or sediment oxidation, there is a risk that chemicals added to the sediments might be removed. Flushing in conjunction with sediment removal would have to be used with care since turbidity and nutrients generated by the dredge would quickly be transported through the lake. If they are used simultaneously, then silt curtains around the dredge site would be useful.

Flushing and lake level drawdown should not be used in the same season. The introduction of phytophagous fish must be done only where the fish cannot escape to downstream ecosystems.

- Phosphorus Inactivation. Flushing is not compatible with this procedure, as described earlier. Aluminum salts and the reagents used in sediment oxidation might be incompatible, especially if pH is affected and denitrification is impeded. Artificial circulation and hypolimnetic aeration could interfere with the effectiveness of phosphorus inactivation by mixing the treated sediments with overlying water.

- Sediment Oxidation. As noted above, there could be problems with the combined use of sediment oxidation and flushing or phosphorus inactivation.

- Biological Controls. Phytophagous fish should be introduced only where they cannot escape downstream. Hypolimnetic withdrawal, dilution, and flushing together with fish stocking would require barriers to prevent the fish from escaping.

Very little information has been published on the application of two or more techniques in the same lake or reservoir. This is due in part to high costs and also to the desire to learn about one procedure without the confounding effect of additional manipulations. The next major advances in lake restoration technology might well be in the area of combinations of lake restoration techniques.

The compatibility of some procedures is obvious. An example is lake drawdown with sediment removal and the application of sediment covers. It is also used in fish management, where it might result in production of larger sport fish. Harvesting can enhance the effectiveness of 2,4-D applications by allowing access of the chemical to root systems, and plants damaged by a harvester may be more susceptible to plant pathogens. Diversion of nutrient loading is obviously compatible, if not necessary, with every technique.

The entire area of integrated control of nuisance aquatic vegetation is in its infancy. Documentation of lake management via combined methods is strongly urged. Effectiveness of the techniques as well as a cost analysis is needed.

CONCLUSIONS

The goal here is to briefly review and analyze the current state of knowledge of the procedure to protect, improve, and restore eutrophic lakes and reservoirs. An obvious finding, while not always explicitly stated, is the small amount of actual knowledge about protection, management, and restoration of freshwater resources. At present, solutions seem to be based almost exclusively on the application of chemicals or the introduction of machinery. In many cases these approaches are effective, but they are also expensive. The developing interest in biological control may greatly add to the arsenal of effective and inexpensive techniques. This avenue may become extremely important in the near future as the demand for clean water increases.

There are ample reasons to believe that surface water resources of both developed and developing nations will continue to deteriorate with the rising demands of agriculture and the inability of governmental units to provide adequate sewage treatment as urban populations increase. There is a need to greatly increase our knowledge of freshwater ecosystems and of the methods to manage, protect, and restore them. This is vital, not only for recreational purposes, but for consideration of human health and economic development.

Several of the above procedures rely on chemical manipulations of the sediments. Far more knowledge is needed about this practice, including studies on the significance of internal loading. We have very little information about the long-term effectiveness and impacts of these restoration and improvement methods and, finally, we know next to nothing about the efficacy of combined uses of these procedures. An in-depth coverage of the above lake restoration practices is contained in the book entitled *Lake and Reservoir Restoration* by G. D. Cooke, E. B. Welch, S. A. Peterson, and P. R. Newroth. It is scheduled for publication by Ann Arbor Science, Butterworth Press, Boston, Massachusetts, in February 1986.

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SEDIMENTATION AND EFFLUENT CONCENTRATION OF SOFT CLAY ON RECLAMATION WORKS

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Japan Bottom Sediments Management Association

ABSTRACT

This paper establishes a basic design standard for the disposal of effluents in a containment area. Field investigations and model tests were conducted to clarify the flow patterns and settlements of the discharged material. By considering the results obtained by these investigations, we discuss the method of estimation for effluent suspended solids (SS) concentration during dredging and reclamation works.

The basin in front of a spillway is an ideal sedimentation basin. The settling tests that introduced the idea of overflow rate in an ideal sedimentation basin gave a good correlation between overflow rate and SS residual ratio. If settling tests of the bottom sediments are carried out in advance, from which correlation coefficients are obtained, an approximate estimation of effluent concentration can be determined by the regression equation corresponding to the reclamation progress.

INTRODUCTION

Dredging sea bottom sediment and placing it into diked containment areas, thus preventing water pollution, is important for the preservation of the environment. Recently in Japan, it has been very common to dredge soft clay that contains a large amount of oil or organic substances. A very important problem is management of the effluent SS concentration--this management must be incorporated in the design and execution of dredging and reclamation work. This paper describes the behavior of effluents with the progress of sedimentation at the pump dredging and reclamation site, based on field investigations and the model tests of the containment area.

This paper also describes a simplified method of estimating the effluent concentration in the containment area by applying a settling test using a cylinder. Because settlement and sedimentation change in the containment area in a very complicated fashion with the progress of reclamation, there is a limit in accuracy of estimation from laboratory tests involving a number of assumptions. However, the method to estimate the approximate effluent SS concentration derives its utility from its simplicity and speed of test execution.

FIELD INVESTIGATION

Outline of the Project

The project for the investigation is the dredging and reclamation works of about 400,000 m^3 in dredged volume by a cutter suction pump dredger of 4,000 ps in horsepower. The site is located in Imari Port in Kyushu, Japan. Figure 1 shows the site location of the dredging and reclamation area and Table 1 gives the specifications of the project.

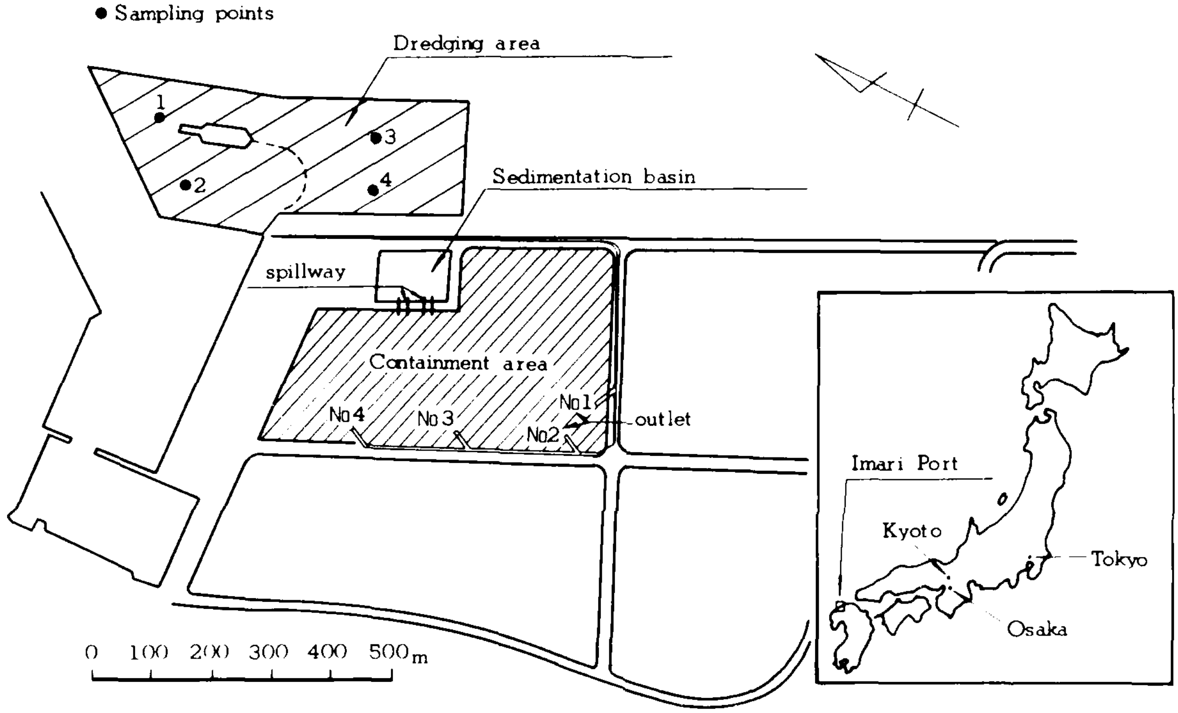

Figure 1. Plan of the dredging and reclamation area

Method of Investigation

To determine the soil characteristics, the sea bottom sediments around the dredging area were sampled by penetration of an acrylic tube, 75 mm in diameter and 850 mm in length. The sediments were also sampled at the outlet of the conduit pipe to determine wet density and unit weight. Turbidity in the containment area was measured by three submersible turbidimeters. Measurement points were 50 cm below the water surface and 1 m above the bottom. Effluents at the spillway were sampled three times a day throughout the construction period and brought into the laboratory for suspended solids test. Flow pattern in the containment area was surveyed by tracing the locus of a buoy.

TABLE 1. PROJECT SPECIFICATIONS

Item	Specification
Dredged volume	389,000 m^3
Area of containment	272,000 m^2
Design thickness of reclamation	3.25 m (+3.20 m from water level)
Area of sedimentation basin	16,960 m^2 (160 m × 106 m)
Volume of sedimentation basin	61,000 m^3

Investigation Results

A field survey was carried out in the middle and latter periods of construction. The reclamation progress ratio was 30 to 40 percent at the middle period and 80 to 90 percent at the latter period. Physical properties of the dredged material are shown in Table 2.

Figure 2 shows the contour lines of the sedimentation and the flow pattern of disposed material during the first observation (reclamation progress ratio about 40 percent). The thick lines in Figure 2 are the depths at which the turbidimeter (Tb) indicated more than 500 ppm (the limit of detection) and the dotted lines are the depths measured by use of a lead. The difference between the two lines, calculated to be about 1 to 2 m, is considered to be the height of unconsolidated sedimentation of the dredged material. Water content of this layer was about 150 to 230 percent. The dredged mixture flowed toward both sides due to the higher sedimentation in front of the outlet. The extremes of the oblique lines in Figure 2 indicate dead zones where flow was very low.

Figure 3 shows the depth measured by a ruler, during the second observation (reclamation progress ratio about 90 percent). The sedimentation progresses through almost the entire region and shows a leveled configuration. Water content at 50 cm below the sedimentation surface was 200 to 450 percent. From Figure 2, it can be seen that the inclination of the sedimentation surface varies from 1 in 250 to 1 in 200 in the most steeply inclined regions. On the other hand, in Figure 3, the inclination in the latter period of the reclamation becomes less than 1 in 1,000. It is presumed that the sedimentation liquefied in the containment area due to the high water content and, as a necessary consequence, the inclination became less steep. Actually, it was observed that the sedimentation surface in the detention area located farthest from the outlet ascended slowly with the progress of reclamation.

TABLE 2. PHYSICAL PROPERTIES OF THE DREDGED MATERIAL

Specific Gravity	Natural Water Content %	Grain-Size Distribution, %			Ignition Loss %
		Sand	Silt	Clay	
2.620	97	3 - 5	48 - 57	40 - 47	10

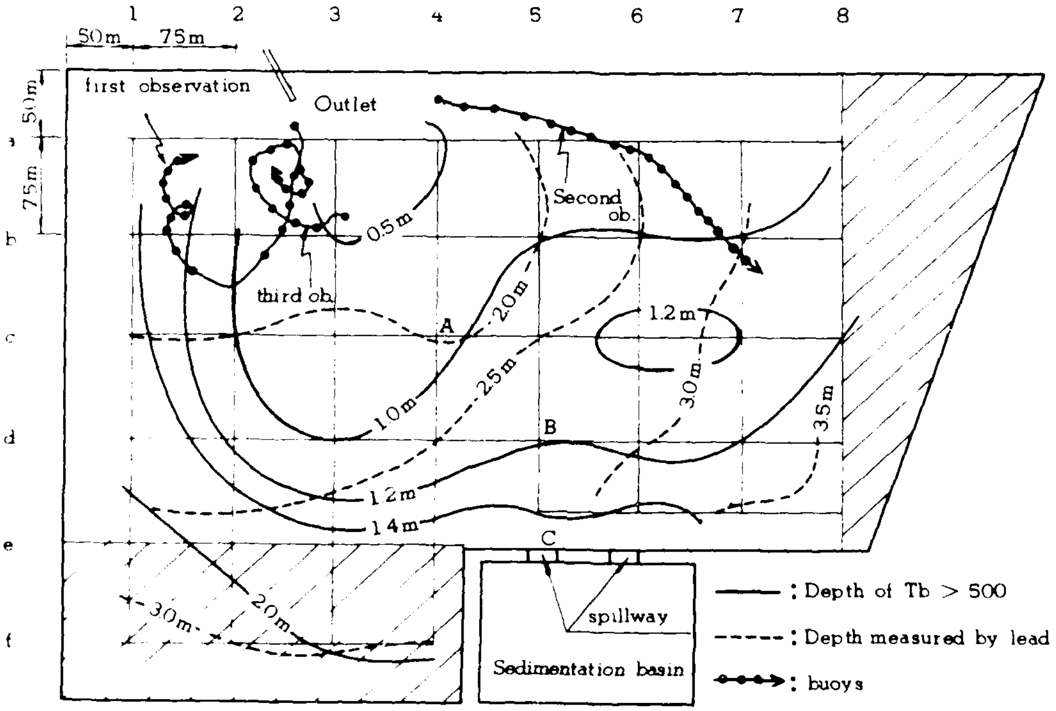

Figure 2. Contour lines and flow patterns in the containment area

Figure 4 shows the result of continuous turbidity measurements at 50 cm below the water surface. Dredging was performed between about 7:00 to 17:00. Therefore the turbidity of 10 to 20 ppm during the first stage is the effluent concentration after about 13 hr of settlement. Turbidity at points A, B, and C in Figure 4 shows marked increases after 15, 45, and 80 min from the beginning of the measurement. These are presumed to be the times that the disposed material reached three points. The propagation velocities of turbidity are calculated to be 23 cm/sec from the outlet to point A, 6 cm/sec from point A to point B, and 3 cm/sec from point B to point C.

Figure 5 shows the plane distribution of turbidity at 50 cm below the water surface; contour lines are also included. Equivalent turbidity lines spread at right angles with the discharge direction, indicating the effluent flowed easily toward both sides as compared with the discharge direction, due to the sedimentation in front of the outlet.

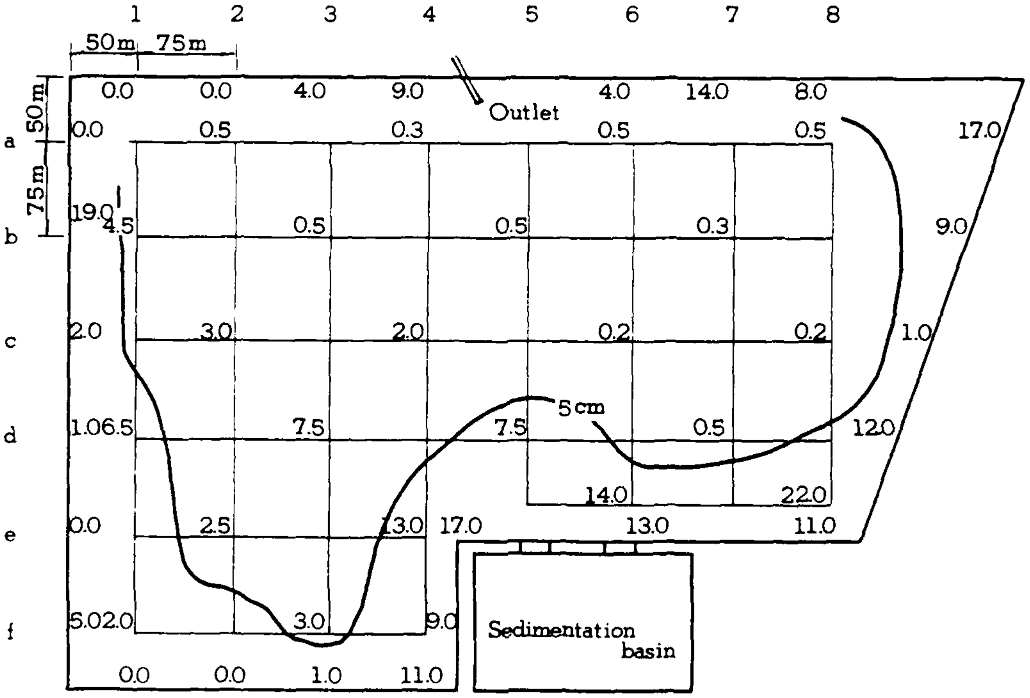

Figure 3. Depth measured by a ruler (March 2, p.m.)

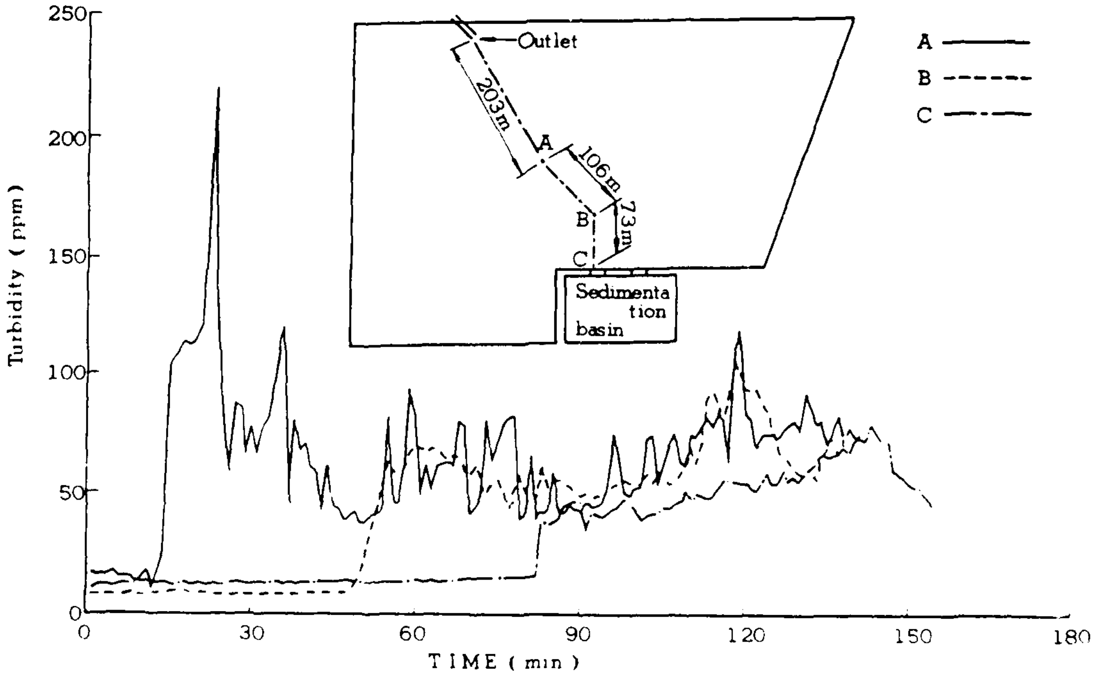

Figure 4. Result of continuous measurement of turbidity (February 17, a.m.)

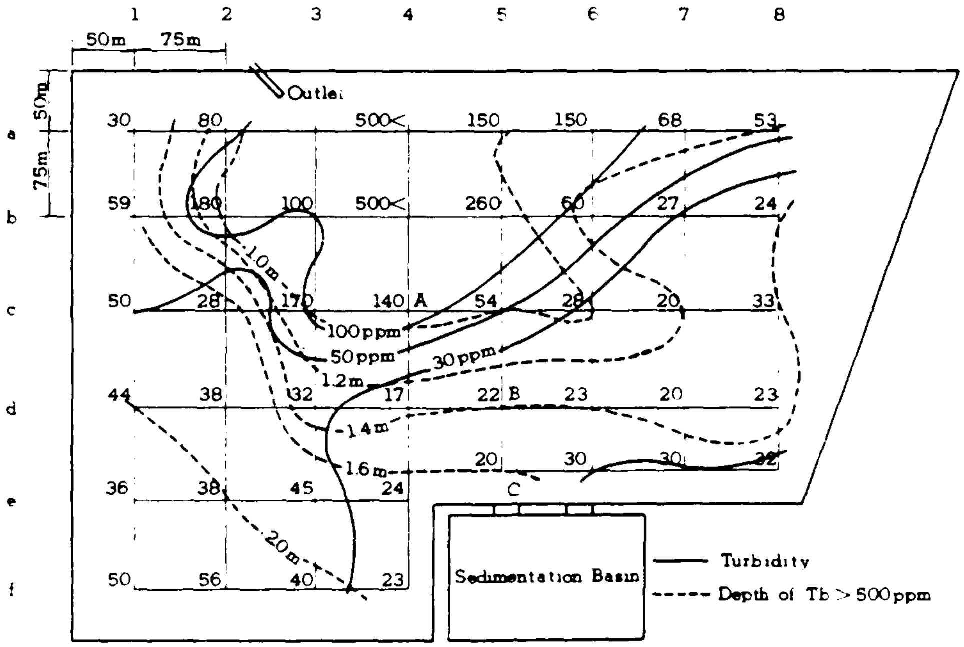

Figure 5. Plane distribution of turbidity at 50 cm below the water surface; contour lines are also included (February 17, a.m.)

Continuous turbidity measurements at points A to C in Figure 6 show a quite different tendency. In Figure 6, though the turbidity at point A, which is almost centrally located, increases with time, the turbidity at point B and C remains constant.

Figure 7 shows the plane distribution of turbidity and the contour lines measured during the afternoon of the same day. Dredged mixtures on that day spread in a radial dispersion as was shown in Figure 2; therefore, it is concluded that propagation of the dredged mixture increased the turbidity at point A, but not directly to the points B or C.

Although the dredging work began on February 7, it was February 15 when the effluent began to overflow at the spillway. Variations of the effluent's SS concentration with time show a sawtooth tendency as shown in Figure 8. The control limit of SS concentration at point Y in Figure 8 was 50 mg/ℓ to prevent water pollution. High molecular coagulants were added to the effluents to satisfy the standard since February 15. Some 10,050 kg of high molecular coagulants were used. Average dosage was 26 g/cum of dredged material.

MODEL TEST OF THE CONTAINMENT AREA

As the flow patterns and sedimentations in the containment area are considered to be a complicated phenomenon in which many parameters interact with each other, a simplified model test was carried out to predict the phenomena.

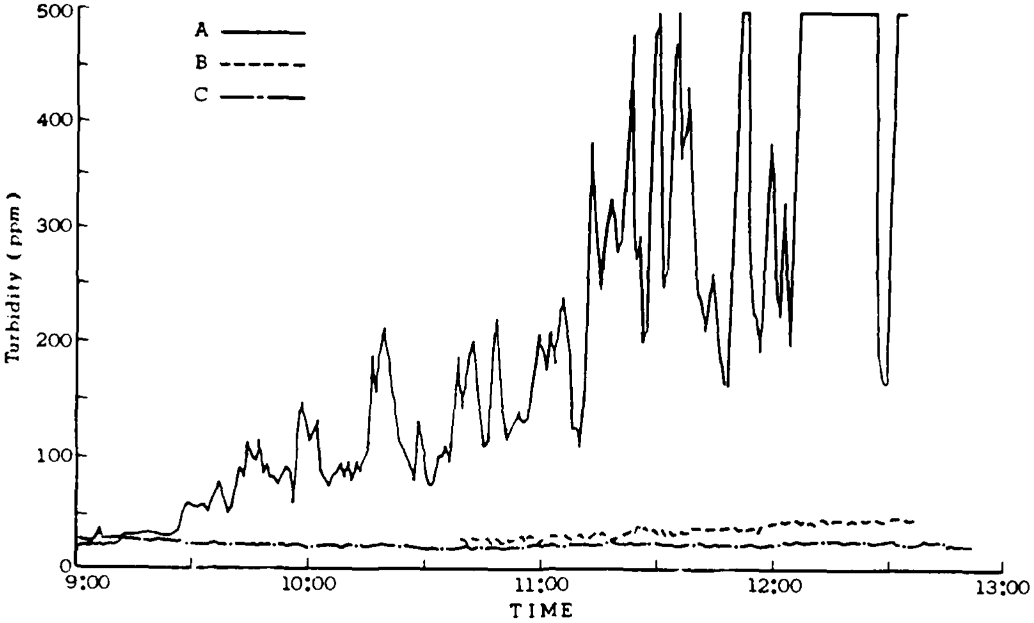

Figure 6. Result of continuous measurement of turbidity (February 18, a.m.)

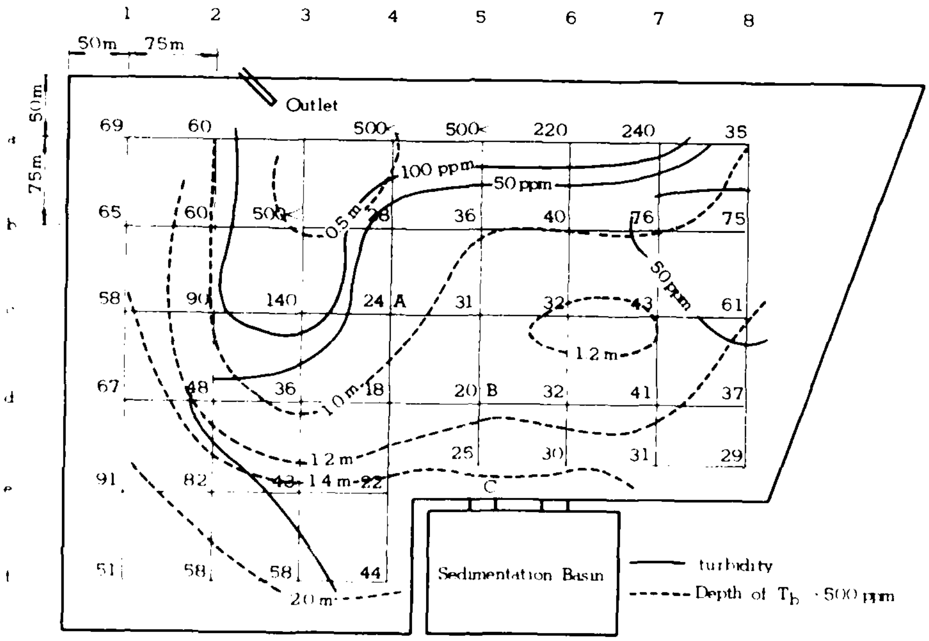

Figure 7. Plane distribution of turbidity at 50 cm below the water surface, contour lines are also included (February 18, a.m.).

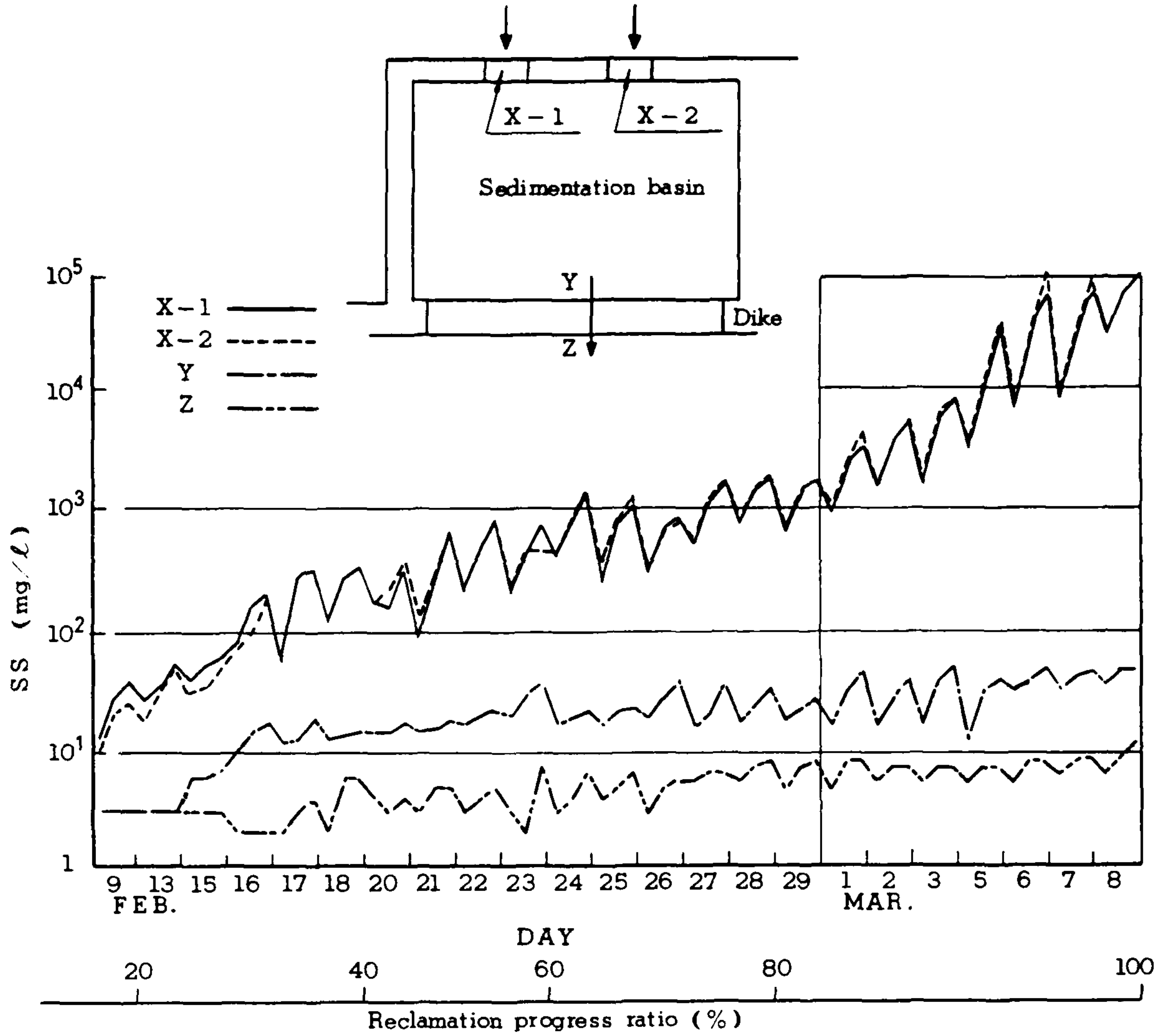

Figure 8. Elapsed time fluctuations of the effluent SS concentration in the sedimentation basin

Test Equipment

During the latter period of the reclamation, should the depth become shallow, the dredged mixture spreads radially and sedimentation progresses in a three-dimensional mode. However, during the early or middle period, at rather high depths, the sedimentation can be considered as a two-dimensional phenomenon, in the longitudinal and vertical directions. Thus a two-dimensional model test using water tanks was conducted. The test equipment is illustrated in Figure 9.

The prescribed concentrated slurry with the sea bottom sediments and artificial seawater, as described later, were poured from one side of the water tank keeping a constant water head by the siphon principle. On the other side of the water tank an effluent weir is established. Samples of the sedimentation were taken during the experiment at points BC, DE, and GH, shown in Figure 9, to measure the concentrations. The samples were taken at six points vertically using a sampling device as illustrated in Figure 10. After pouring the slurry, water content and grain-size analysis were performed at points A through H in Figure 9. Table 3 shows the test conditions.

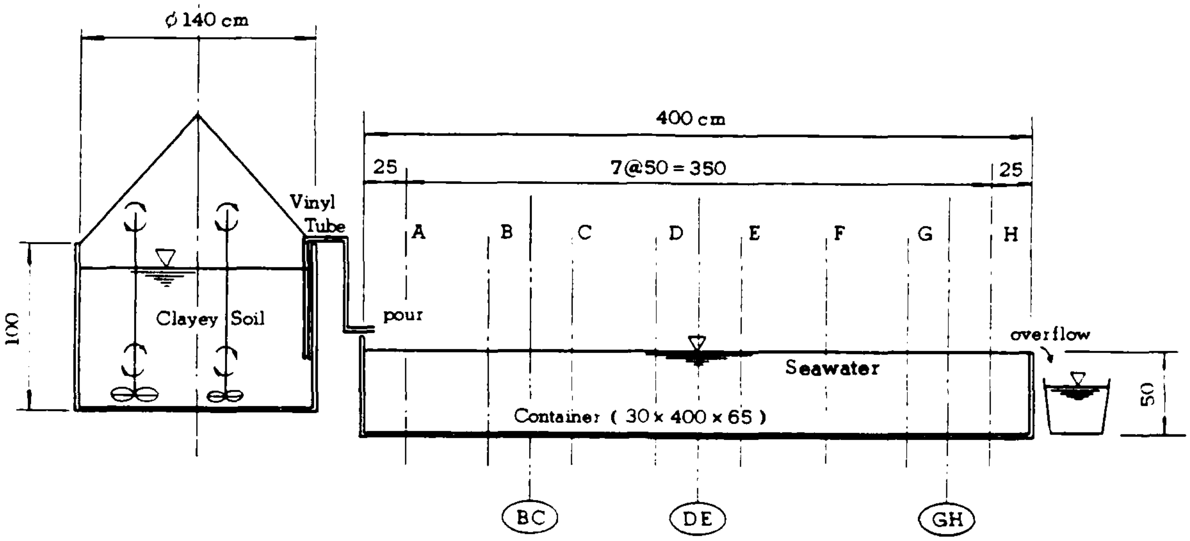

Figure 9. Outline of test equipment (all units in centimeters)

Results of Test

We attempted to clarify the effect of kinds of water on the settling velocity of particles. Figure 11 shows the decrease of the clarified water SS concentration with time using several different solutions. The variation of the clarified water SS concentration of artificial seawater (commercially called "Aquamarine" in Japan) shows a similar tendency to that of natural seawater. In saline solutions, the SS concentrations are larger than the other cases; the SS concentrations in fresh water are the largest. This means that the settling velocity of particles in seawater is greater than in fresh water due to flocculation of negatively charged sediment particles into Mg++ or Na+ in seawater.

Figure 12 shows the grain-size distributions in fresh water and in artificial seawater obtained by mechanical analysis of soil using a hydrometer. When seawater is applied to the analysis, the result shows that there are fewer particles smaller than 5 μm; the apparent grain size becomes larger. Therefore, for actual sedimentation in seawater, artificial seawater was selected for the model test due to its similar properties to natural seawater.

Figure 13 shows the outline of the sedimentation condition 10 min after the beginning of discharge. Discharged material reached the bottom of the tank as a forced plume, then spread slowly in the horizontal direction. A suspended layer in the upper region, a zone settling layer in the middle region, and a consolidated settling layer in the bottom region developed as shown in Figure 13.

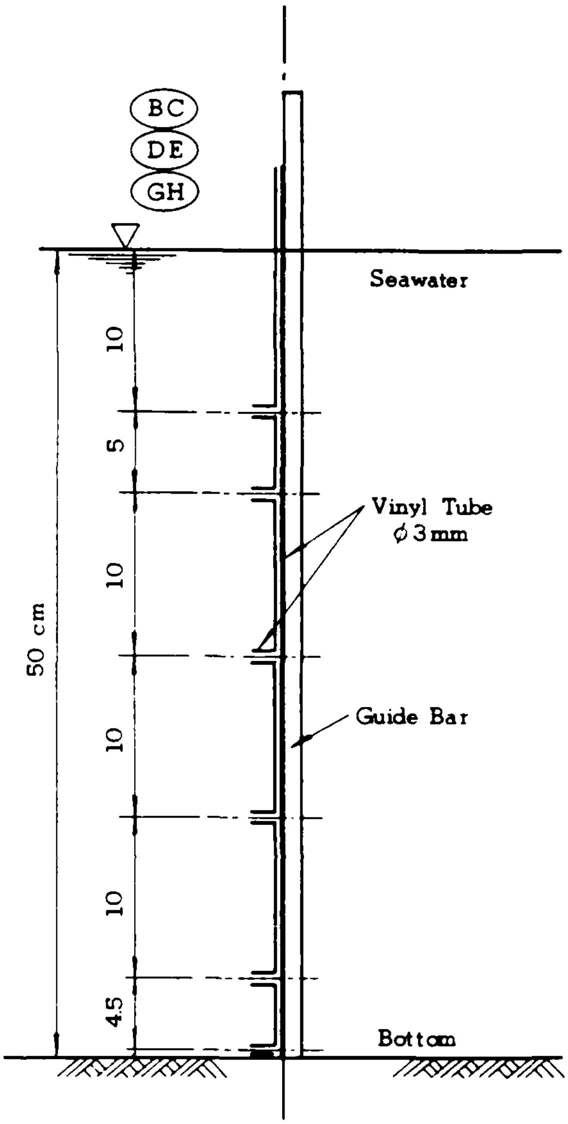

Figure 10. Sampling device of the mixture

TABLE 3. TEST CONDITIONS

Influent Concentration mg/ℓ	Diameter of Inlet Pipe mm	Influent Loading Rate ml/min	Velocity of Inlet Pipe m/sec	Elapsed Discharge Time min
96,300	4	250	0.33	240

Figure 14 shows the vertical distributions of the concentration at points BC, DE, and GH as a function of elapsed time. Though the water content in each layer changed only slightly with the passage of time, it tended to increase with distance from the outlet tube. The water content distributions in consolidated sedimentation shown in Figure 15 exhibited the same tendency.

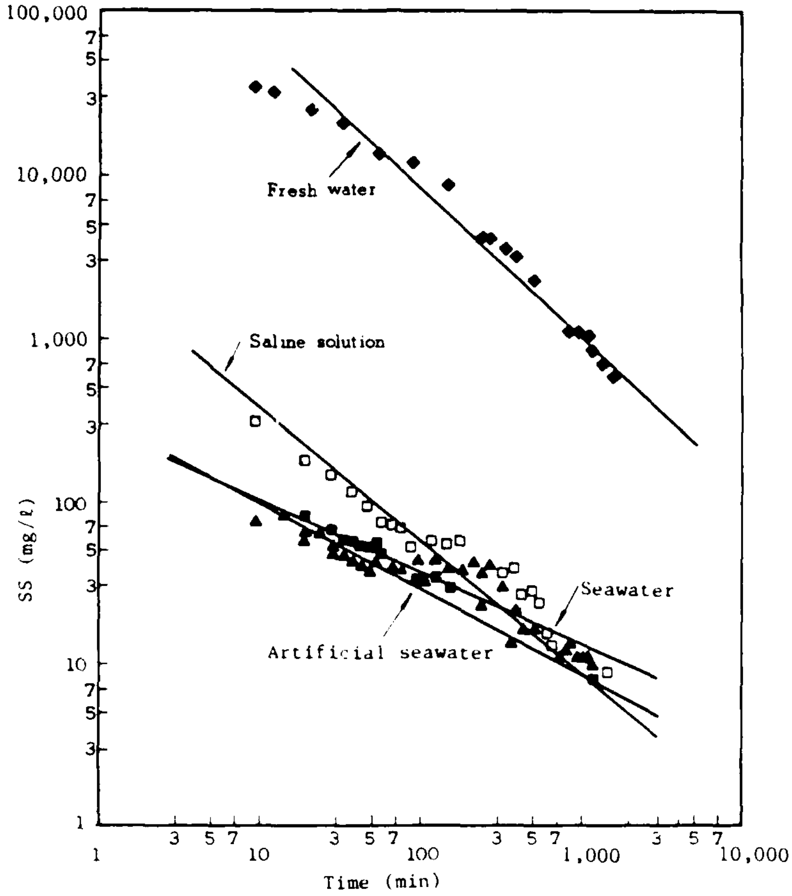

Figure 11. Variation of clarified water SS concentration using several different solutions (SS_o = 33,690 mg/ℓ)

This tendency corresponds to the grain-size distributions of each point. Smaller particles are accumulated in proportion to the distance from the outlet tube (Figure 16). It may be noticed that the content of colloids increased remarkably with distance from the outlet.

SETTLING TEST ESTIMATION OF EFFLUENT CONCENTRATION

Based on the findings of this study, we determined a simplified method for the estimation and management of effluent SS concentration.

Relation Between Overflow Rate and Effluent Concentration

Overflow rate is one of the most important indices in an ideal sedimentation basin in the field of water and wastewater works. Overflow rate

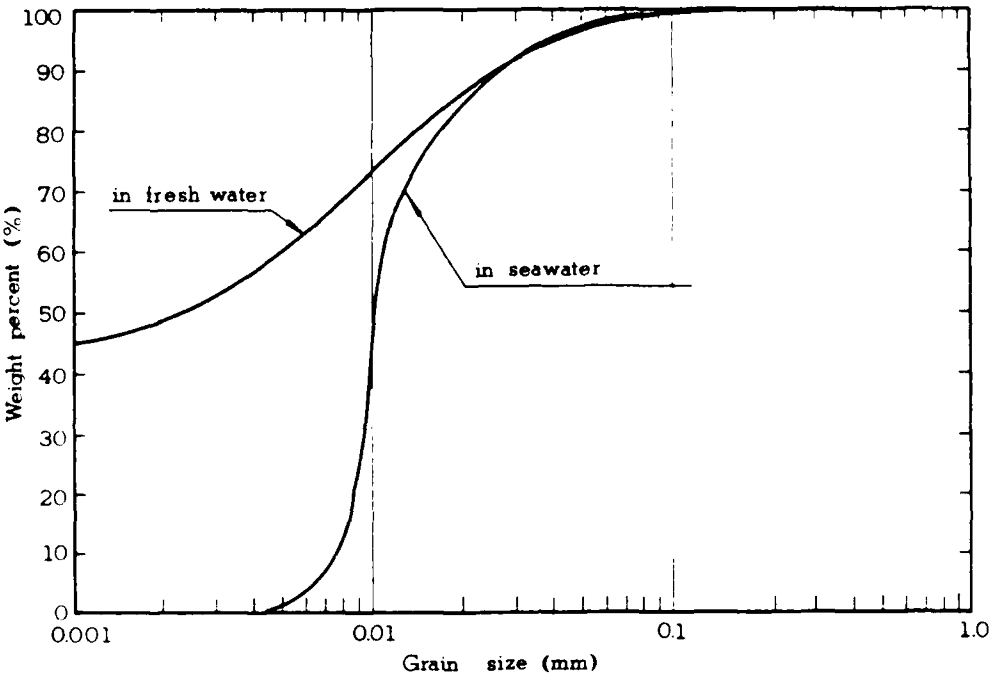

Figure 12. Grain-size distributions of tested sediments in fresh water and in artificial seawater

Figure 13. Outline of sedimentation

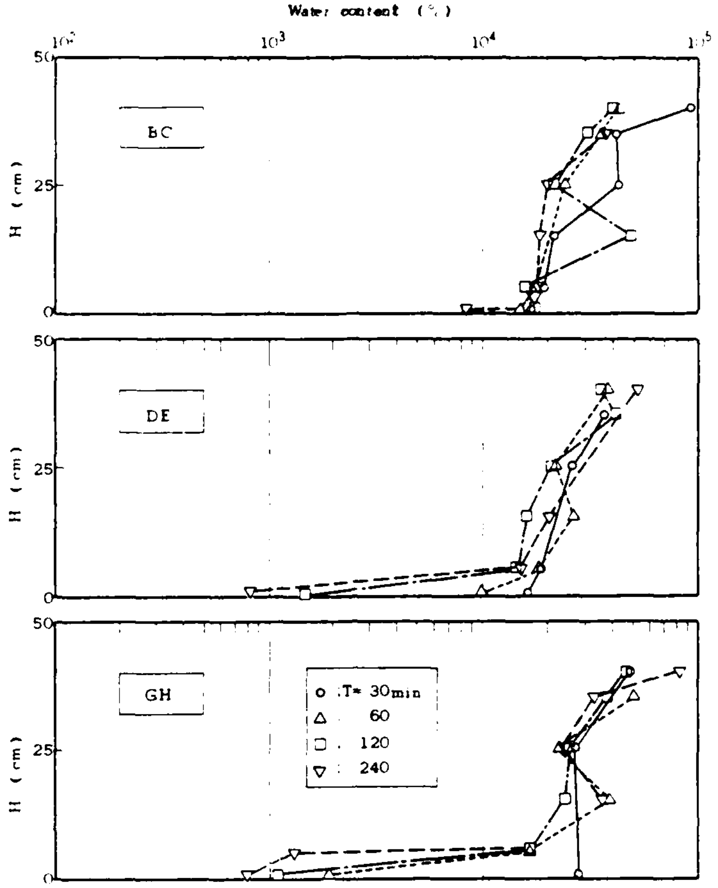

Figure 14. Density distribution of sedimentation

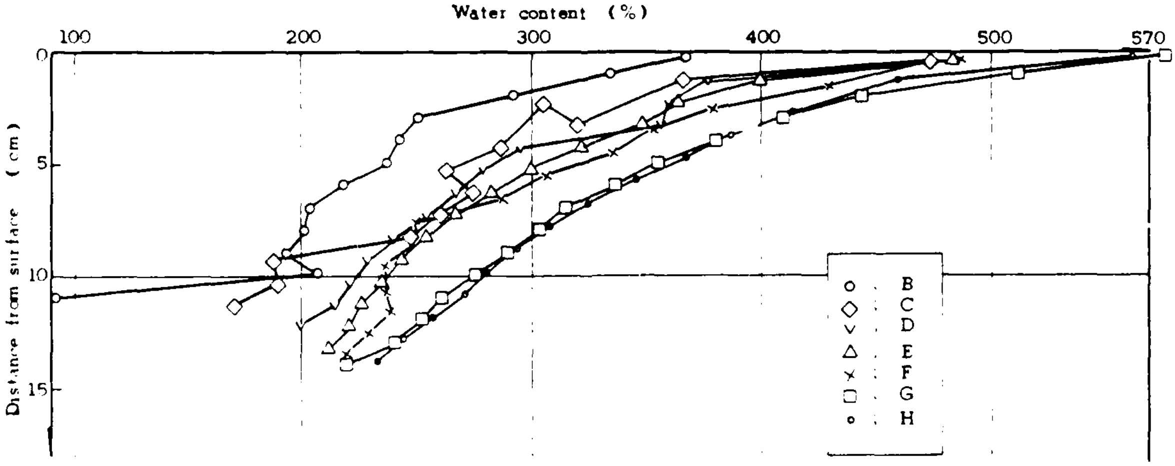

Figure 15. Water content distribution of consolidated sedimentation

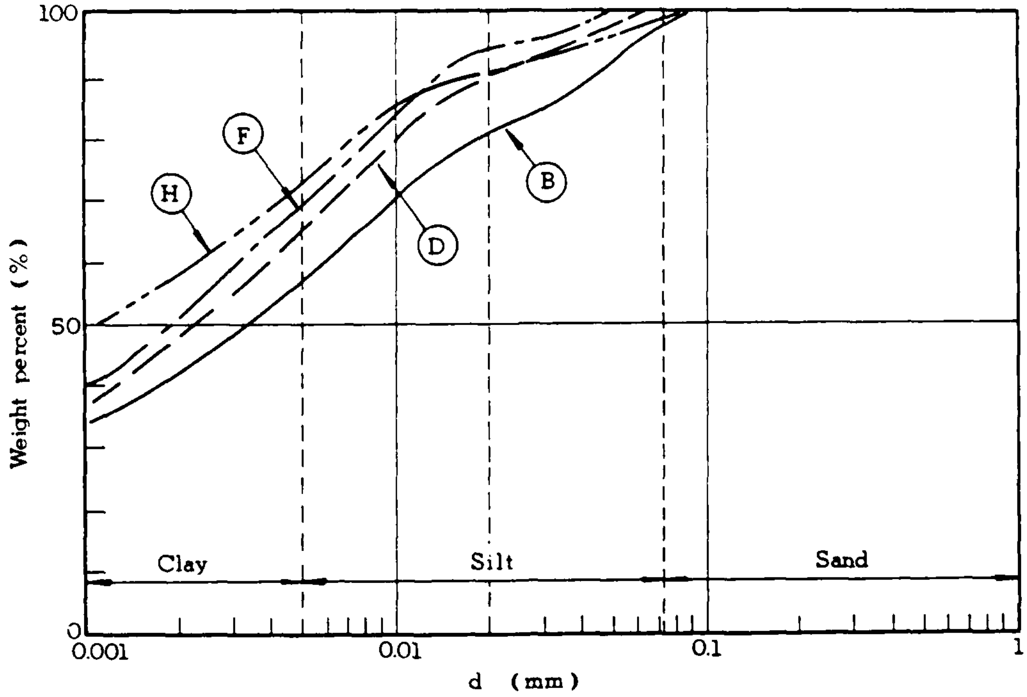

Figure 16. Grain-size distributions with distance from the outlet tube

is expressed as the dimensional quantity Q/A where Q is the influent loading rate and A is the surface area of the sedimentation basin. An ideal sedimentation basin is shown in Figure 17 based on basic assumptions, as follows:

a. Fluid flows only in the horizontal direction and has an equal velocity at all points.

b. Concentration of suspended solids is the same at all depths.

c. The particles, once settled on the bottom, are not resuspended.

Overflow rate is the rate of climb when fluid springs with uniform velocity from the entire floor; therefore all particles having a faster settling velocity than this spring velocity Q/A will be settled. This is a reasonable explanation if the basin has sufficient depth, but there is a limit in depth to adapt this assumption due to deflection or scouring effects. From the authors' experience, this limit is applicable up to about 50 cm in depth, at about 1 to 3 hr of detention.

Overflow depths are usually designed to be 10 to 15 cm in actual reclamation works. At a point 10 cm below the water surface, in front of the spillway, after t sec from dredged material propagation, the particles having a settling velocity greater than 10/t cm/sec would be deeper than 10 cm below the water surface. If the basin around the spillway in the containment area is assumed to be an ideal sedimentation basin, 10 cm divided by the time t

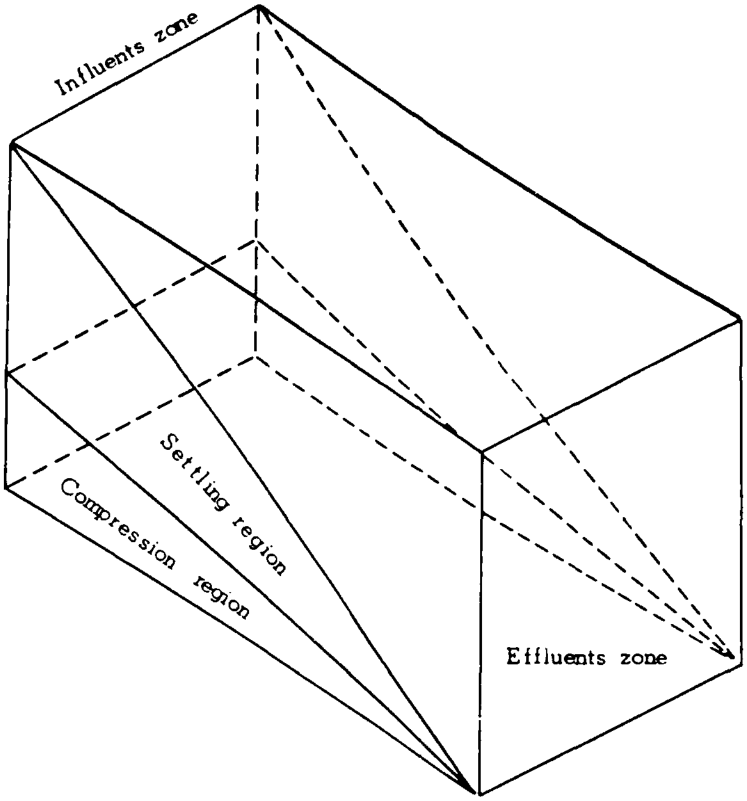

Figure 17. Schematic diagram of an ideal sedimentation basin

corresponds to an overflow rate of Q/A . Based on this point of view, we carried out a settling test and examined the method of estimation of the effluent concentration.

Test Equipment

The settling test utilizes a transparent cylinder with a height of 40 cm as shown in Figure 18. Two cylinders, 60 mm and 138 mm in diameter, were used to study the effect of diameter. The sea bottom sediments were diluted with artificial seawater to the specified concentration in this cylinder. A turbidimeter, shown in Figure 19, was placed 10 cm below the water surface. This turbidimeter is a transmitted light type using a luminous diode, and can measure up to 3,000 caolinite concentrations. The relation between SS concentration and turbidity has been investigated previously; therefore, the measured turbidity values were translated to SS concentrations using regression equations.

Results of Test and Discussion

The purpose of this cylinder test is to discuss a method for the management of effluent SS concentration from the relation between overflow rate Q/A

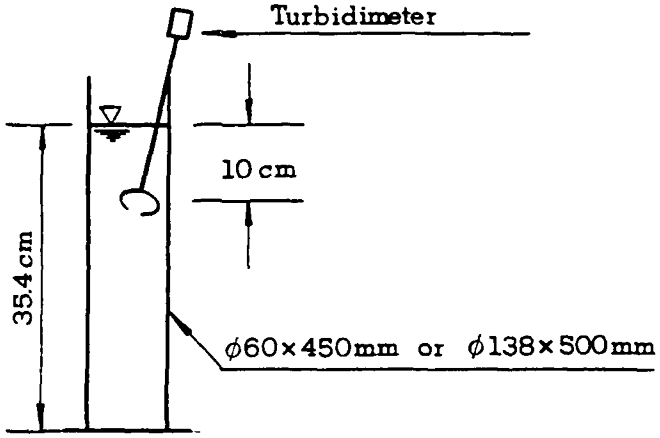

Figure 18. Test equipment

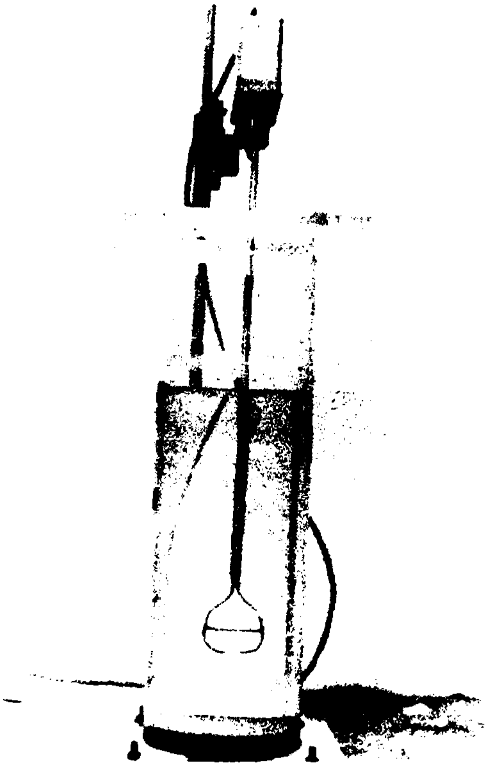

Figure 19. Transmitted light type turbidimeter

and SS residual ratio SS_i/SS_o obtained by the continuous measurement of clarified water at a point in a cylinder. For this purpose, it is important to measure SS concentration on a real-time basis.

Tested sediments were sea bottom silty clays as shown in Figure 12. The relationship between SS concentration and turbidity shows a good correlation as shown in Figure 20. Regression equations obtained were as follows:

$$0 < T_b < 150 \text{ mg}/\ell : \quad SS = 1.51T_b - 5.5 \qquad (1)$$

$$150 < T_b \quad : \quad SS = 1.70T_b - 131 \qquad (2)$$

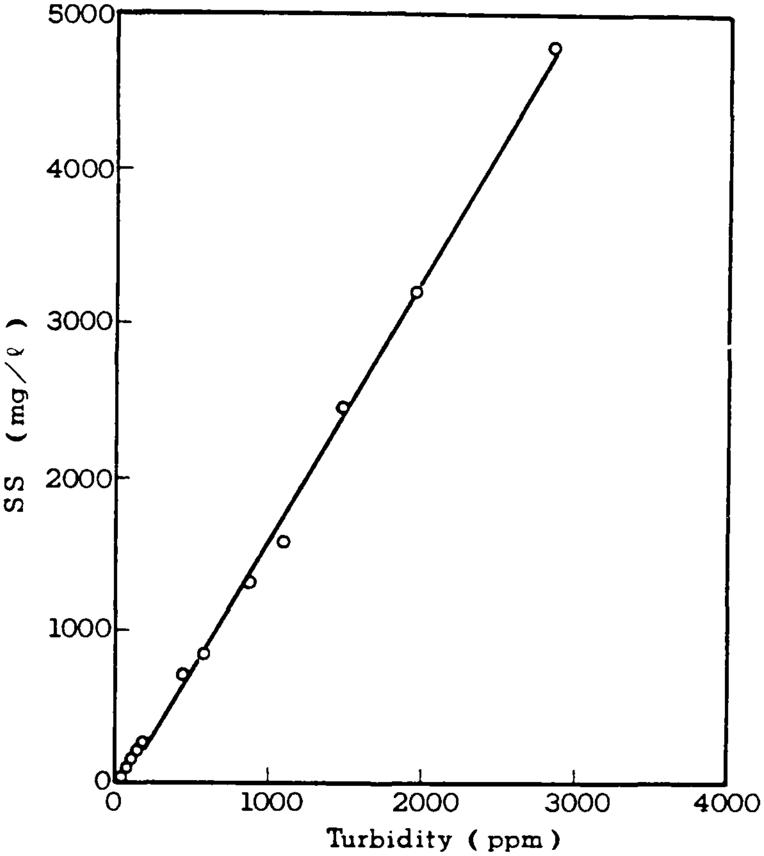

Figure 20. Relations between SS concentration and turbidity

Based on a slurry content of the pump dredged material of 10 to 15 percent, SS concentrations were supposed to be about 10,000 to 100,000 mg/ℓ. The field investigations and model tests showed that the effluent SS concentrations in the spillway became much lower than the influent concentration due to the removal of the sedimentation. Accordingly, tests were conducted over a wide range of influent concentrations, as shown in Table 4.

TABLE 4. RANGE OF INITIAL SS CONCENTRATIONS ON CYLINDER TESTS

Case	Initial Concentration mg/ℓ
1	48,800
2	29,900
3	8,510
4	7,730
5	5,250
6	4,120
7	2,780
8	1,030
9	940
10	380

Figure 21 shows the concentrations after 8 min for typical examples. When the initial SS concentration was 48,800 mg/ℓ, zone settling can be observed, but in cases when SS_o is less than 10,000 mg/ℓ, the phenomenon of zone settling is not observed.

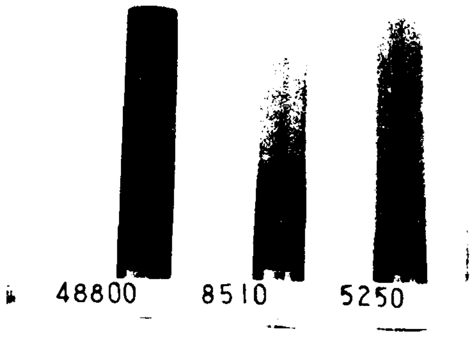

Figure 21. Concentrations after 8 min of settlement

From Figure 22, it can be seen that the clarified water SS concentrations seem to be nearly equal after about 30 min of settlement. A good correlation is found in the relation between SS concentration and elapsed time. Regression equations were obtained as follows:

$$SS_o = 48{,}800 \text{ mg}/\ell \quad : \quad SS = 875t^{-0.568} \tag{3}$$

$$22{,}900 - 8{,}510 \quad : \quad SS = 1{,}610t^{-0.664} \tag{4}$$

$$7{,}730 - 379 \quad : \quad SS = 4{,}440t^{-0.841} \tag{5}$$

In Equations 3 to 5, the regression coefficient a of the equation $SS = at^b$ indicates virtual suspended solids after 1 min of settlement (t = 1) and the greater the concentration, the smaller the value of a .

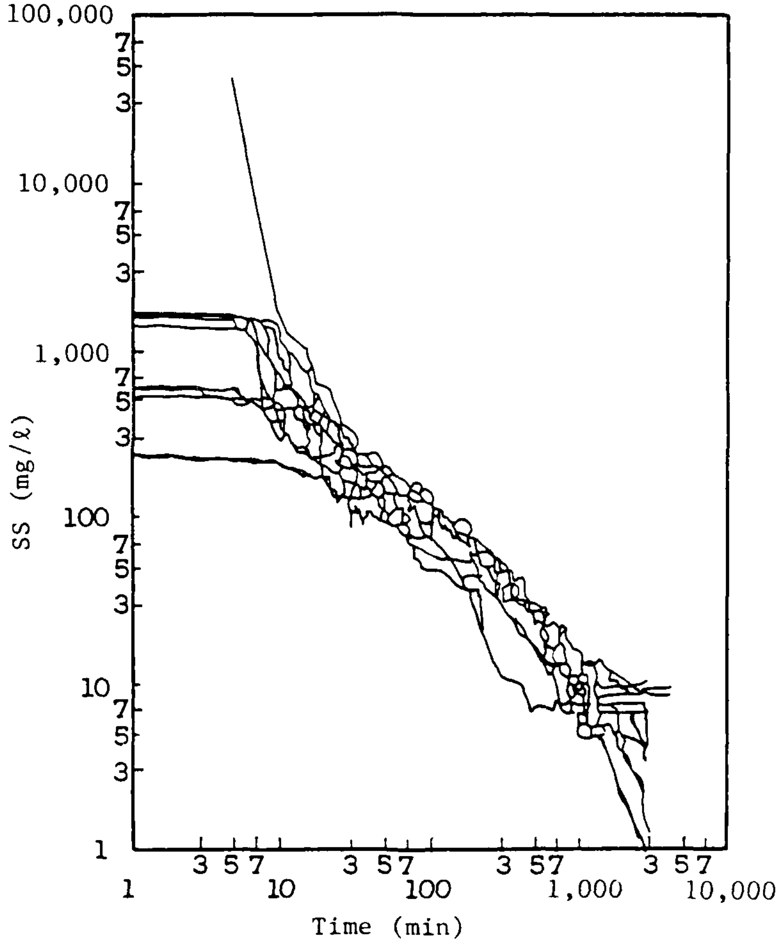

Figure 22. Clarified water SS concentration at 10 cm below water surface versus time

In ordinary pump dredging the SS concentration of the dredged mixture is considered to be about 10,000 mg/ℓ, the dredged mixture settles in a very short time, and the SS concentration is less than 10 percent of the influent concentration. Figure 23 shows the relationship between overflow rate Q/A , which is obtained as (10/t), and SS residual ratio (SS_i/SS_o), where t_1 to

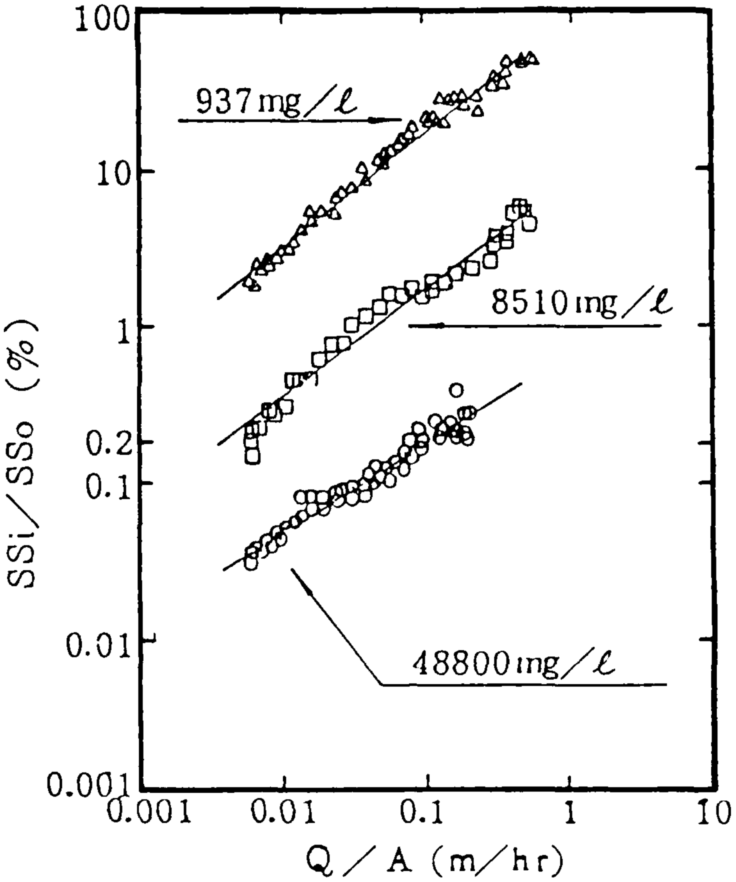

Figure 23. Relations between overflow rate Q/A, which is obtained as (10/t), and SS residual ratio SS_i/SS_o

t_i is the elapsed time and SS_i is the concentration at each time. These relationships have a significant influence on the initial concentration--the greater the initial concentration, the smaller the residual ratio becomes. These curves can be represented by regression equations in the form given below:

$$SS_i/SS_o = a(Q/A)^b \tag{6}$$

As the inclination b in Equation 6 is almost identical, they can be treated as essentially straight lines on a logarithmic graph paper. Thus, an approximate estimation of effluent SS concentration on reclamation work can be performed by the relation, Q/A - SS_i/SS_o as shown in Figure 23 and in Equaation 6, by substituting the influent loading rate Q, influent SS concentration SS_o, and effective area A_i.

Figure 24 shows the result of calculations of the significant differences in the residual ratio (during 10 to 1,000 min) by the wholly significant difference method. In Figure 24, the symbols a through g represent temporary group names. Groups a and b are absolutely different from each other, but group c is not recognized to be different from group cd. In Figure 24, although significant differences can be observed in the case of different initial concentrations (ex. Case 1, Case 2), there is no difference between the case of a small cylinder and a large cylinder. Therefore, for the settling test, we concluded that it is sufficient to use a 60-mm diam-cylinder.

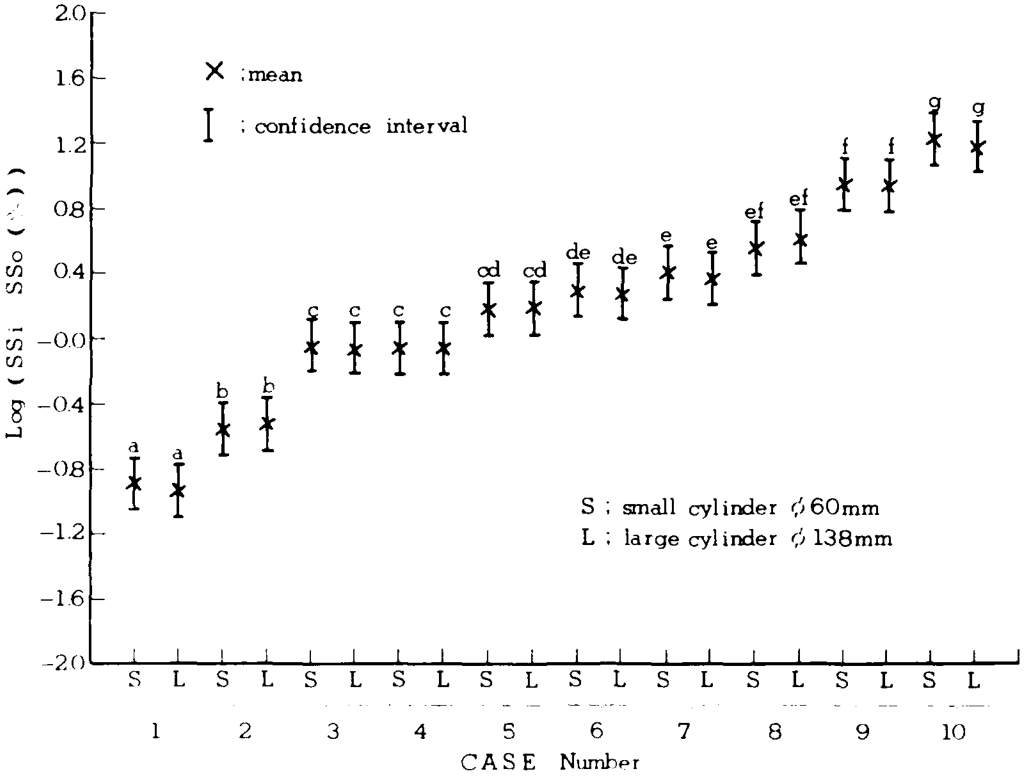

Figure 24. Calculated result on significant difference of the SS residual ratio by wholly significant difference method

CONCLUSIONS

Various findings concerning the sedimentation and effluent SS concentrations were obtained from the reclamation work by pump dredging from field investigations and a model test. Based on these findings, a simplified method has been determined for the estimation of effluent SS concentration. Results were as follows:

Sedimentation and Flow Pattern of Dredged Mixture

- The settling velocity of particles in seawater is greater than in fresh water due to the flocculation of negatively charged sediment particles into Mg^{++} or Na^{+} in seawater. It is therefore better to apply seawater or artificial seawater to investigate prototypes.

- Flow patterns of the dredged mixture are considerably affected by the sedimentation of large-sized particles. Smaller particles accumulate in proportion to the distance from the outlet.

- Clarified water concentration at the spillway is affected by variations in the flow pattern.

- Water content at 50 cm below the sedimentation surface was 200 to 450 percent. Test results indicate that the water content on sedimentation during consolidation is 200 to 600 percent. Hence, the sedimentation at 50 cm below the sedimentation surface is considered to result from consolidation.

- There are significant differences in the water content and the grain-size distributions of the sedimentation with the distance from the outlet.

Estimation of Effluent Concentration by a Cylinder Test

- Zone settling can be observed when influent concentrations are more than 10,000 mg/ℓ, but in the case of an initial concentration less than 10,000 mg/ℓ, flocculent settling can be observed.

- The clarified water SS concentrations become nearly equal after about 30 min of settlement, independent of the initial concentration.

- A good correlation can be found in relationships between SS concentration and elapsed time as follows:

  $$SS = at^b$$

  Coefficient a changes with the initial concentration.

- Relationships between overflow rate Q/A and SS residual ratio SS_i/SS_o can be represented by the regression equations in the form given below:

  $$SS_i/SS_o = a(Q/A)^b$$

  The regression coefficient a also changes with the initial concentration SS_o and the greater the SS_o, the smaller the value of a.

- The diameter of the cylinder has no effect on the settling pattern or the clarified water SS concentration within the range of tested diameters.

COMMENCEMENT BAY - A REMEDIAL INVESTIGATION OF CONTAMINATED SEDIMENTS IN A MARINE ENVIRONMENT

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ABSTRACT

Contamination of sediments in Commencement Bay by toxic pollutants has caused considerable concern about degradation of marine habitats, induction of disease in indigenous organisms, and contamination of organism tissues. The approach to and results of an intensive 2-year, $2 million investigation are presented. The goals of the investigation were to determine the extent of contamination, determine if the contamination resulted in adverse effects to public health or the environment, identify problem contaminants and their sources, and identify potential remedial actions. An extensive synoptic data set was obtained during the investigation. These chemical, toxicity (bioassays), and biological effects (benthic community structure, bioaccumulation, fish histopathology) indicators were used along with historical data to identify problem areas and problem contaminants. Sources of problem contaminants were identified to the extent possible using horizontal and vertical patterns of chemical contamination, available source loading data, information on dredging history, knowledge of environmental fate processes, and knowledge of past and present industrial activities in the area. Problem areas, problem contaminants, and contaminant sources were prioritized such that resources could be effectively allocated for corrective actions.

INTRODUCTION

Commencement Bay is an urban-industrial embayment in south-central Puget Sound (Washington). Urbanization and industrial development began in the area during the final quarter of the 19th century. Dredge and fill activities have transformed the broad tideflat that once existed at the mouth of the Puyallup River into a series of industrialized waterways and an active port facility. Past and present industrial and commercial operations located in the area include pulp and lumber, ship building, metal plating, chlorine and other chemical production, aluminum smelting, oil refining, copper smelting and refining, concrete production, food processing, shipping, ship dismantling, marinas, scrap yards, railroads, and a number of other storage, transportation, and manufacturing facilities.

The presence of these operations and the urban growth of the surrounding area have resulted in the release of contaminants into the marine environment. The contaminants include metals (e.g., arsenic, lead, zinc, copper, and mercury) and organic compounds [e.g., polycholorinated biphenyls (PCB), dibenzofurans, chlorinated pesticides, plasticizers (pthalates), and polynuclear aromatic hydrocarbons (PAH)]. Over 425 point and nonpoint discharges have been identified in the area, each of which is a potential source of contamination.

Concern about the potential effects of sediment contamination in Commencement Bay (e.g., toxicity to marine life, induction of disease in fish, and potential human health impacts from consumption of fish and shellfish) resulted in designation of the area as a National Priorities List site by the US Environmental Protection Agency (EPA) under the Comprehensive Environmental Response, Compensation, and Liability Act. In April 1983, the Washington State Department of Ecology (WDOE) and the EPA entered into a Cooperative Agreement. EPA provided funds for the investigation and WDOE was designated lead agency.

GOALS AND OBJECTIVES

The goals of the investigation were to determine the extent of contamination, determine if contamination resulted in adverse effects to public health or the environment, identify problem contaminants and their sources, and identify potential remedial actions. The specific objectives set were:

a. Define a problem sediment.

b. Apply definition of problem sediment to delineate problem areas.

c. For problem areas, determine problem chemicals.

d. For problem chemicals, determine problem sources.

e. Prioritize problem areas, problem chemicals, and problem sources.

f. Assess impacts of fish and crab consumption on human health.

g. Document alternative methods of dredging, handling, and disposing of contaminated sediments.

h. Initiate a decisionmaking framework for managing the disposal of contaminated sediments.

i. Identify potential remedial alternatives.

This paper will deal mainly with objectives a through e. Discussion and results of objective f through i may be found in Tetra Tech (1985), US Army Engineer District, Seattle (1985), US Army Engineer Waterways Experiment Station (1985), and Versar, Inc. (1985).

APPROACH

The major elements of the decisionmaking approach developed for the investigation are presented in Figure 1. The first step in the process was to determine the types of information needed to best meet the objectives of the project (i.e., determine problem areas, problem chemicals, problem sources, and a means of ranking these in priority order). Because definitive cause-effect information and sediment quality criteria were not available for the marine environment, it was necessary to collect information on both chemistry and biological effects. The indicators chosen were chemical analyses of sediments, toxicity tests of field-collected sediments (i.e., amphipod bioassays as a lethal indicator and oyster larvae bioassays as a sublethal indicator), bioaccumulation of contaminants in English sole (*Parophrys vetulus*) and cancrid crabs, and biological effects (i.e., alterations of benthic invertebrate assemblages and prevalences of liver lesions in English sole). Ancillary information determined necessary included physical characteristics of the sediments (e.g., grain size), organic content of the sediments, and a measure of the oxidation state of the sediments. Chemical analysis of water column particulates was also determined to be useful in making judgments on contaminant movement in the project area.

The next step in the process was to review and evaluate the existing data base. A sampling and analysis plan was then developed. Stations were located to fill gaps in the historical data base, to define more precisely areas of known contamination, and to evaluate gradients of contamination and effects relative to suspected sources of contamination.

A project reference area was selected so that Commencement Bay data could be compared with data from an embayment judged to have relatively uncontaminated sediments. The project reference area is distant from the industrialized areas of Puget Sound. Concentrations of chemicals in the sediments sampled from the project reference area were comparable or lower than those in other Puget Sound reference areas.

The sampling plan was implemented by an intensive field survey. Results of the field survey were used to characterize the project area (Figure 2) on several degrees of spatial resolution:

a. Overall project area.

b. Study areas (the eight waterways and the Ruston-Point Defiance shoreline).

c. Problem areas (area within segments determined to be a problem).

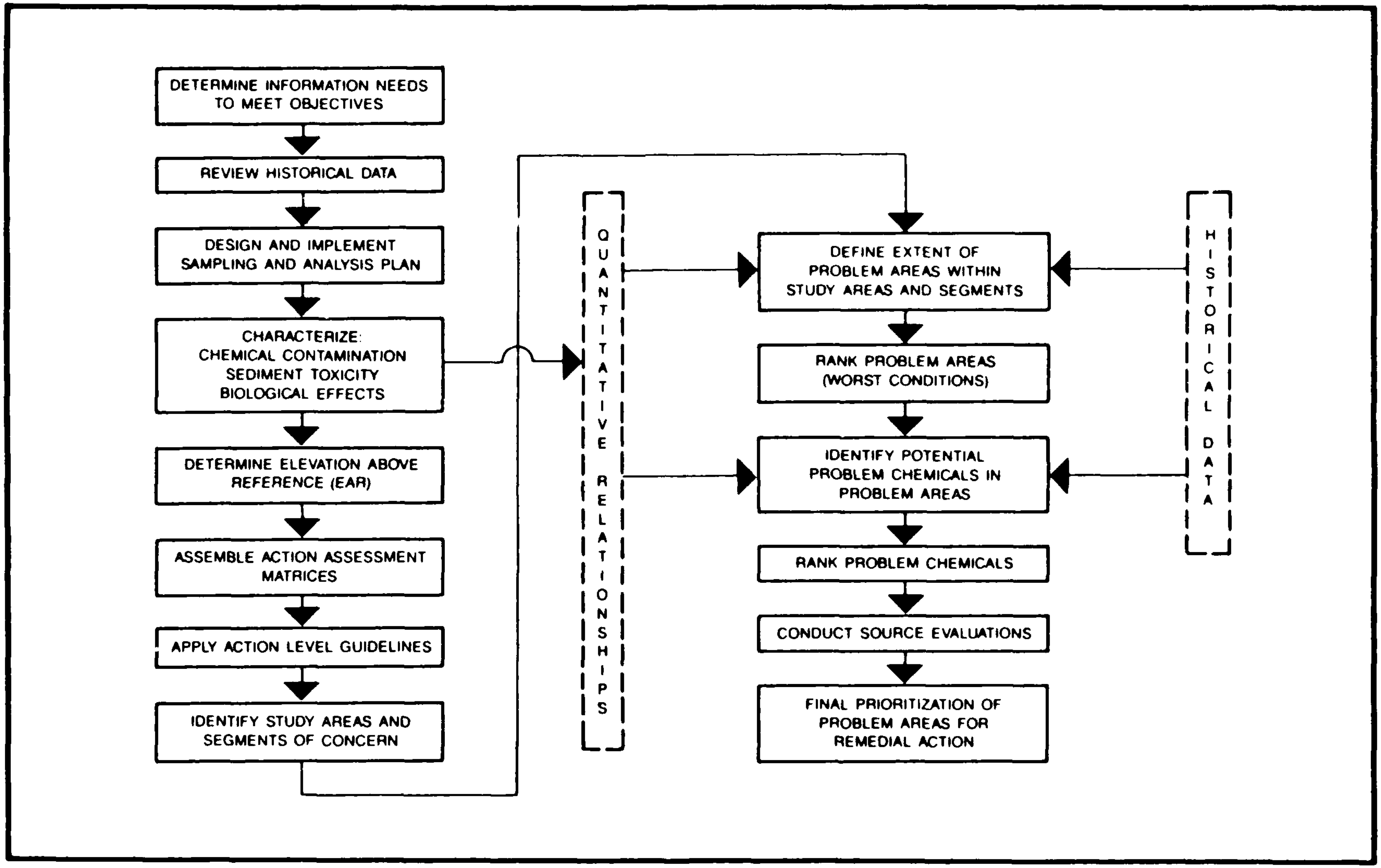

Figure 1. Decisionmaking approach to the Commencement Bay nearshore/tideflats remedial investigation

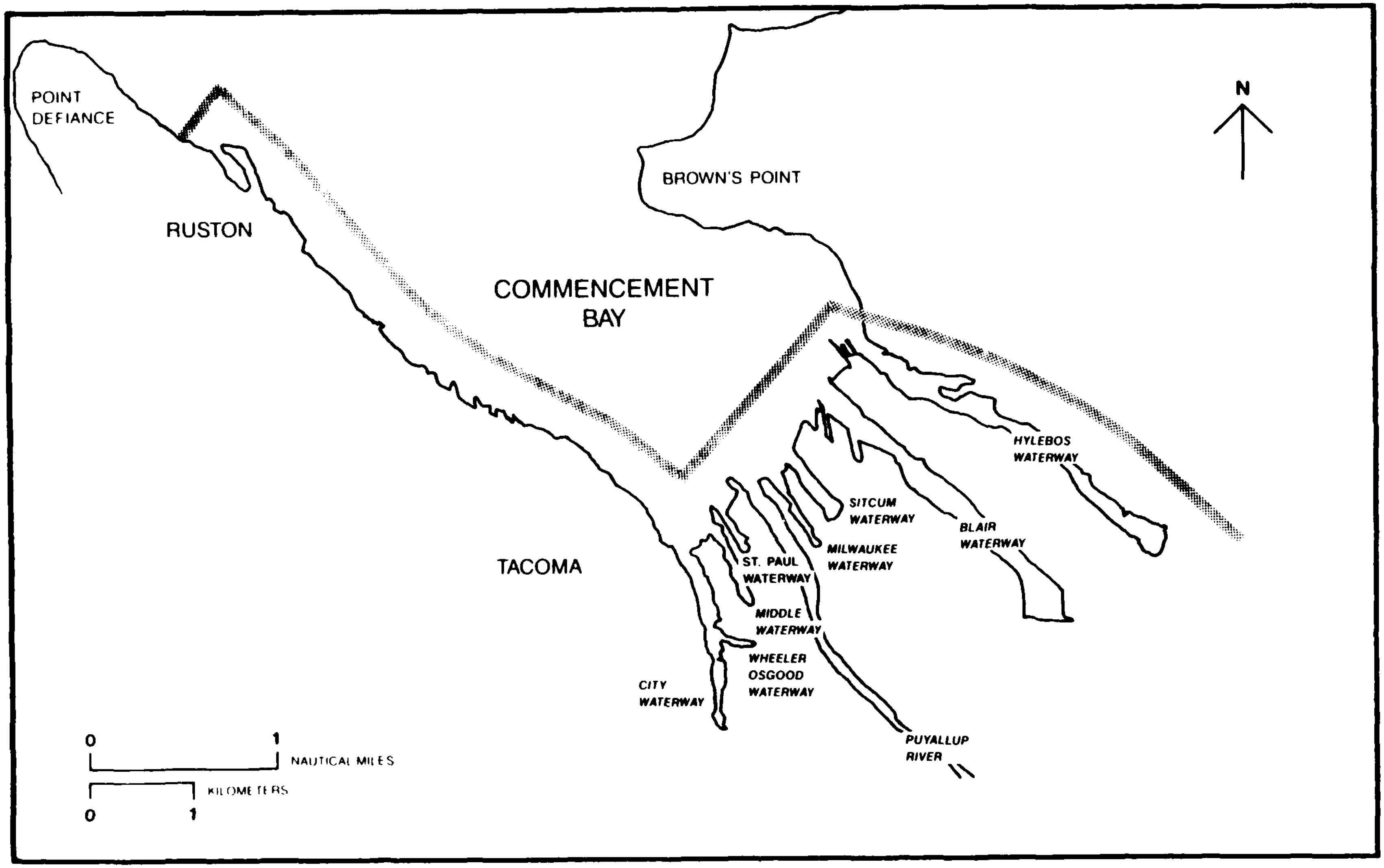

Figure 2. Commencement Bay nearshore/tideflats study area

Sediment contamination, toxicity, bioaccumulation, [illegible] were described by establishing an index for each environmental [illegible] indicator investigated, calculating "elevations above reference" [illegible] each indicator, and testing each EAR for significance. EAR are [illegible] indicator variable at a Commencement Bay area and at the reference [illegible] example, an EAR = 5 indicates that the variable was five times higher in Commencement Bay than in the reference area. For toxicity, bioaccumulation, and biological effects, a significant EAR was one that was statistically different ($P < 0.05$) from the project reference area (except for benthic alterations within the waterways, where Blair Waterway was used in lieu of the project reference area). For sediment chemistry, the EAR was determined to be significant if the contaminant value exceeded the maximum value in any of several reference areas throughout Puget Sound.

In addition to characterization of the project area as described above, results of the field survey were also used to develop quantitative relationships among sediment contamination, toxicity, and biological effects. These analyses were conducted primarily to determine levels of sediment contamination above which significant toxicity or biological effects would be expected to occur, and to identify contaminants suspected of causing the observed effects. Results of the quantitative relationships analyses were used to help define boundaries of problem areas and to identify potential problem chemicals.

EAR for study areas and segments were assembled into action assessment matrices. These matrices display the indices (EAR) for each environmental variable or indicator for each study area or segment under consideration. Also displayed is whether the EAR is significant or not significant and the reference value from which the EAR was determined. Action level guidelines were then applied to identify study areas and segments of concern. The action level guidelines defined the minimum levels and specific combination of environmental indicators required before problem area definition could proceed. This threshold for action was always exceeded if at least three of the five indicators of sediment contamination, toxicity, and biological effects were determined to be significantly elevated above reference conditions in an area. Each study area or segment of concern included either a problem area that extended over most of the study area or segment, or a "hot spot" of major significance within the study area or segment.

After areas of concern were defined, problem areas were identified, and their spatial extents were determined. Problem area definition incorporated all available data, including historical data and use of results of the quantitative relationships among indicators developed as part of the present study. Problem areas were then ranked according to their most extreme levels of contamination, toxicity, and biological effects. Priority for evaluation of potential sources of contamination was based on this ranking. Potential problem chemicals were also identified and ranked at this point.

After problem areas and potential problem chemicals were identified and ranked, potential sources of contamination were evaluated. Final prioritization of problem areas recommended for remedial action was then determined on the basis of the relative magnitude of problems in each area, the spatial extent of each area, and the level of confidence that sources of potential problem chemicals were accurately identified.

The efforts of source identification integrated a large data base on potential contaminant sources, observed contaminant levels in water and sediment, and ancillary information. Some of the most valuable information utilized in source identification included:

a. Spatial gradients of contamination in surficial sediments.

b. Vertical gradients of contamination in sediment cores.

c. Analyses for the contaminant(s) in discharges.

d. Dredging history.

e. Environmental fate processes.

f. Industrial activities.

APPLICATION OF APPROACH AND RESULTS

Action Assessment Matrices

An example of an action assessment matrix is shown in Table 1. These matrices were prepared on both a waterway (area) basis and on a segment basis in which major waterways were divided into three to five smaller areas. The matrices display the magnitude of the indicators (elevations above reference) and whether or not the EAR value is significantly different from reference. Reference values are also displayed so that absolute values of indicators may be determined. The indicators are:

a. Sediment chemistry
  - Selected metals and organic compounds

b. Sediment toxicity
  - Amphipod bioassay
  - Oyster larvae bioassay

c. Benthic infauna
  - Total benthos abundance
  - Polychaete abundance
  - Mollusc abundance
  - Crustacean abundance

d. Fish pathology
  - Hepatic neoplasms, preneoplastic nodules, megalocytic hepatosis, nuclear pleomorphism

e. Fish bioaccumulation
  - Selected metals and organic compounds

In all Commencement Bay study areas, average concentrations of one or more chemicals exceeded the range of concentrations found in Puget Sound reference sediments (Table 1). However, surface sediment contamination throughout Commencement Bay was variable both in concentration and composition. Lowest concentrations of chemicals were found in Blair and Milwaukee

TABLE 1. ACTION ASSESSMENT MATRIX OF SEDIMENT CONTAMINATION, SEDIMENT TOXICITY, AND BIOLOGICAL EFFECTS INDICES FOR COMMENCEMENT BAY STUDY AREAS

	STUDY AREA ELEVATIONS[a]								
VARIABLE	Hylebos	Blair	Sitcum	Milwaukee	St. Paul	Middle	City	Ruston	REFERENCE VALUE[b]
SEDIMENT CHEMISTRY									
Sb	$\boxed{10.}$	4.0	8.0	3.6	4.2	9.3	7.0	$\boxed{510.}$	110. ppb
As	$\boxed{12.}$	$\boxed{7.6}$	$\boxed{11.}$	3.6	2.2	$\boxed{9.6}$	$\boxed{7.5}$	$\boxed{620.}$	3370. ppb
Cd	$\boxed{2.4}$	1.9	$\boxed{2.8}$	1.7	1.7	$\boxed{2.8}$	$\boxed{5.5}$	$\boxed{27.}$	950. ppb
Cu+Pb+Zn	$\boxed{10.}$	4.8	$\boxed{24.}$	$\boxed{7.3}$	5.5	$\boxed{18.}$	$\boxed{22.}$	$\boxed{120.}$	35000. ppb
Hg	$\boxed{8.1}$	< 3.7	5.0	3.8	5.1	$\boxed{26.}$	$\boxed{10.}$	$\boxed{160.}$	40. ppb
Ni	1.4	0.7	0.6	0.8	0.8	0.7	1.4	$\boxed{2.8}$	1740. ppb
Phenol	$\boxed{< 6.4}$	$\boxed{< 5.2}$	$\boxed{4.3}$	$\boxed{< 2.1}$	$\boxed{12.}$	$\boxed{11.}$	$\boxed{9.4}$	$\boxed{4.5}$	< 33 ppb
Pentachlorophenol	$\boxed{1.7}$	$\boxed{< 2.3}$	$\boxed{< 2.1}$	< 0.90	U 1.9	$\boxed{5.6}$	$\boxed{< 1.9}$	< 1.0	U 33. ppb
LPAH	$\boxed{<45.}$	$\boxed{<28.}$	$\boxed{<68.}$	$\boxed{<60.}$	$\boxed{<73.}$	$\boxed{<110.}$	$\boxed{<120.}$	$\boxed{<87.}$	< 41. ppb
HPAH	$\boxed{<120.}$	$\boxed{<42.}$	$\boxed{<65.}$	$\boxed{<68.}$	$\boxed{<27.}$	$\boxed{< 97.}$	$\boxed{<140.}$	$\boxed{<85.}$	< 79. ppb
Chlor. benzenes	$\boxed{9.9}$	< 4.4	2.6	< 2.5	< 1.8	< 6.1	$\boxed{< 9.0}$	< 3.3	U 21. ppb
Chlor. butadienes	$\boxed{130.}$	$\boxed{< 2.7}$	$\boxed{< 2.4}$	< 1.2	< 1.3	$\boxed{< 6.8}$	$\boxed{< 1.9}$	$\boxed{< 1.7}$	U 62. ppb
Phthalates	$\boxed{4.0}$	< 2.6	< 0.58	< 0.66	< 0.56	$\boxed{< 5.1}$	$\boxed{< 7.1}$	$\boxed{4.5}$	< 280. ppb
PCBs	$\boxed{<48.}$	$\boxed{< 6.3}$	$\boxed{7.6}$	$\boxed{6.6}$	$\boxed{<17.}$	$\boxed{8.5}$	$\boxed{<12.}$	$\boxed{19.}$	< 6.0 ppb
4-Methylphenol	$\boxed{<7.3}$	$\boxed{<12.}$	$\boxed{10.}$	$\boxed{13.}$	$\boxed{1300.}$	$\boxed{< 33.}$	$\boxed{30.}$	$\boxed{<10.}$	< 13. ppb
Benzyl alcohol	$\boxed{5.0}$	$\boxed{< 2.2}$	$\boxed{2.4}$	$\boxed{3.4}$	$\boxed{< 6.7}$	$\boxed{3.3}$	$\boxed{4.7}$	$\boxed{< 1.2}$	U 10. ppb
Benzoic acid	< 0.7	$\boxed{< 3.2}$	< 0.5	< 0.7	< 1.0	U 0.1	< 1.6	< 0.5	< 140. ppb
Dibenzofuran	$\boxed{29.}$	$\boxed{25.}$	$\boxed{73.}$	$\boxed{59.}$	$\boxed{52.}$	$\boxed{80.}$	$\boxed{58.}$	$\boxed{<160.}$	U 3.7 ppb
Nitrosodiphenylamine	$\boxed{2.1}$	$\boxed{< 2.4}$	$\boxed{< 7.3}$	U 1.0	U 1.2	< 1.0	$\boxed{14.}$	$\boxed{<22.}$	U 4.1 ppb
Tetrachloroethene	$\boxed{12.}$	< 0.6	U 1.0	---	U 1.0	---	U 1.0	---	U 10. ppb
SEDIMENT TOXICITY									
Amphipod bioassay	$\boxed{2.1}$	1.9	$\boxed{2.9}$	$\boxed{2.4}$	$\boxed{4.8}$	1.4	$\boxed{2.7}$	$\boxed{3.9}$	9.3 %
Oyster bioassay	$\boxed{2.2}$	$\boxed{1.6}$	1.3	1.4	3.8	1.8	$\boxed{2.6}$	2.2	13.0 %
INFAUNA[c]									
Total benthos	1.2	1.0	0.7	0.8	1.9	1.5	0.7	0.6	d
Polychaetes	0.6	1.0	0.4	0.7	1.5	0.7	0.8	0.5	d
Molluscs	$\boxed{3.4}$	1.0	1.4	1.1	$\boxed{6.8}$	$\boxed{5.4}$	$\boxed{2.5}$	1.2	d
Crustaceans	1.5	1.0	$\boxed{3.8}$	0.4	1.0	$\boxed{4.6}$	1.2	0.7	d
FISH PATHOLOGY									
Lesion prevalence	$\boxed{3.6}$	$\boxed{2.5}$	$\boxed{3.5}$	$\boxed{3.7}$	2.7	$\boxed{5.7}$	1.7	2.1	6.7 %
FISH BIOACCUMULATION									
Copper	5.6	1.0	$\boxed{4.0}$	2.3	$\boxed{9.1}$	1.0	3.8	$\boxed{2.5}$	U 38. ppb
Mercury	$\boxed{1.5}$	0.93	0.80	1.6	0.76	1.3	0.82	0.96	U 55. ppb
Naphthalene	0.67	0.41	0.33	$\boxed{24.}$	0.19	0.19	4.1	0.19	< 54. ppb
Phthalates	$\boxed{21.}$	$\boxed{11.}$	0.53	$\boxed{3.6}$	0.41	0.41	6.7	5.6	< 74. ppb
PCBs[e]	$\boxed{9.2}$	$\boxed{7.0}$	$\boxed{4.8}$	2.8	1.1	4.7	$\boxed{9.8}$	1.9	< 36. ppb
DDE	$\boxed{3.8}$	$\boxed{5.1}$	3.3	3.4	1.7	1.7	$\boxed{6.2}$	2.9	< 1.8 ppb

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[a] Boxed numbers represent elevations of chemical concentrations that exceed all Puget Sound reference area values, and statistically significant toxicity and biological effects at the $P<0.05$ significance level compared with reference conditions. The "U" qualifier indicates the chemical was undetected and the detection limit is shown. The "<" qualifier indicates the chemical was undetected at one or more stations. The detection limit is used in the calculations.

[b] Elevation above reference (EAR) values shown for each area are based on Carr Inlet reference values for each variable except for benthos (see footnote d).

[c] Infauna EAR are based on the elevation in biological effects represented by reductions in infaunal abundances relative to reference conditions. EAR for all other variables reflect an increase in the value of the variable at Commencement Bay compared with reference conditions.

[d] Different benthic reference values were used depending on sediment grain size.

[e] Locations where PCB concentrations are significantly elevated also pose a significant health risk to the exposed population.

Waterways, and at stations located in deeper waters outside of the waterways. Highest chemical concentrations were typically found adjacent to point source discharges. An exception was PCBs, which displayed a more patchy distribution not always associated with a particular point source. Highest PCB concentrations were found in Hylebos Waterway [2,000 μg/kg dry weight (DW)]. At some stations, concentrations of selected chemicals exceeded reference conditions by several orders of magnitude. The highest concentration of PAH (35,000 μg/kg DW) was found in sediments near the head of Hylebos Waterway. At the mouth of the same waterway, chlorinated butadienes were found at concentrations up to 65,000 μg/kg DW. Concentrations of alkylated phenols exceeded 96,000 μg/kg DW in sediments adjacent to the main outfall of a pulp and paper mill at the mouth of St. Paul Waterway. Metals contamination was greatest in sediments adjacent to the main outfalls of the now closed ASARCO copper smelter on the Ruston-Point Defiance shoreline. Maximum concentrations of four metals [arsenic (12,000 mg/kg DW), antimony (420 mg/kg DW), copper (14,000 mg/kg DW), and mercury (52 mg/kg DW)] all exceeded 1,000 times reference conditions in these sediments.

As is indicated in Table 1, there are also distinct differences among the study areas in the kinds and magnitudes of biological effects. Lowest sediment toxicity and effects on infauna were observed in Blair and Milwaukee Waterways. Overall, the waterway benthic assemblages were dominated by high abundances of polychaetes. However, areas of extreme biological effects (i.e., almost 100-percent response in bioassays or nearly complete absence of benthic infauna) were found in isolated areas adjacent to industrial facilities near the mouth of St. Paul Waterway and along the Ruston-Point Defiance shoreline. English sole from five of the eight study areas had significantly elevated prevalences of one or more serious liver lesions. The highest liver lesion prevalence of about 40 percent was measured in Middle Waterway (compared with 6.7 percent in the reference area). Relatively few of the sediment contaminants detected in Commencement Bay were measured at elevated concentrations in English sole muscle tissue. PCBs were detected in most fish tissue samples and were measured at highest concentrations (approximately 10 times reference) in City and Hylebos Waterways. PCBs were also the contaminants of greatest concern relative to potential effects on human health from seafood consumption.

Problem Chemicals

Quantitative relationships among the independent contaminant, toxicity, and biological effects variables were evaluated to meet two objectives:

a. Determine levels of sediment contamination above which significant toxicity or biological effects would be expected.

b. Identify problem contaminants from the numerous contaminants detected in Commencement Bay sediments.

Both statistical and nonstatistical approaches were used to evaluate whether toxicity or biological effects increased with increasing sediment contaminant concentrations. In this study, it was assumed that contaminants displaying monotonically increasing relationships with toxic or biological effects have a higher relative priority (i.e., a higher potential for being a causative agent) than do contaminants displaying no discernible relationship with biological effects.

Where synoptic biological and chemical data were collected, significant toxicity in both the amphipod mortality and oyster larvae abnormality bioassays as well as benthic effects (i.e., depressions of abundance of total taxa, Polychaeta, Mollusca, or Crustacea) were observed at all but one station where the DW concentration of at least one contaminant exceeded 1,000 times reference conditions. The exception was at one station where trichlorinated butadiene concentrations were nearly 2,000 times reference conditions, but neither bioassay nor any benthic effect was significant. In other sediments without significant toxic responses or benthic effects, concentrations of organic compounds (other than chlorinated butadienes) ranged from 1 to <400 times reference conditions, and concentrations of metals were <50 times reference conditions.

Sediment toxicity and the number of significant benthic effects were highest in the most chemically contaminated study areas. Typically, toxicity increased and abundances of major taxa decreased with increasing concentrations of some contaminants over the entire study area. A common characteristic of these relationships was that at lower chemical concentrations, there was considerable scatter in the magnitude of sediment toxicity and taxon abundances. When trends were observed, the minimum toxicity observed at a given concentration of a chemical increased and the maximum abundances decreased at higher contaminant concentrations. When data from all study areas were plotted together for a given contaminant, there was no clear trend in the values of maximum toxicity or minimum abundances over the concentration range of the contaminant. Thus, it is concluded that no one contaminant or contaminant group correlated with the effects observed in all areas.

In some cases, there was random scatter in the values up to a certain contaminant concentration. Above that concentration, there was a rapid change to uniformly high toxicity or low abundances at the few most contaminated sites. If the high contaminant levels were associated with the effects observed, the abrupt change in the scatter suggested an "effect threshold" for the contaminant.

The synoptic chemical, toxicity, and benthic infaunal data for 52 Commencement Bay stations were examined for each contaminant of concern to evaluate effect thresholds. An example of this approach using lead data is shown in Figure 3. In this case, the available data indicated that significant toxicity or benthic effects did not occur when sediment lead concentrations were below 11 mg/kg DW (EAR = 1.2). This level defined a "potential effect threshold." This threshold was termed "potential" because toxicity or benthic effects were found at some, but not all, of the stations with higher lead concentrations. The effects observed at these stations could have resulted from other contaminants or conditions.

Toxicity "apparent effect threshold" (AET) was defined as the lowest contaminant concentration above which significant toxicity was observed at all stations. An analogous benthic AET was defined as the lowest contaminant concentration above which significant benthic effects occurred at all stations. For lead, the toxicity AET was 660 mg/kg DW (EAR = 72) and the

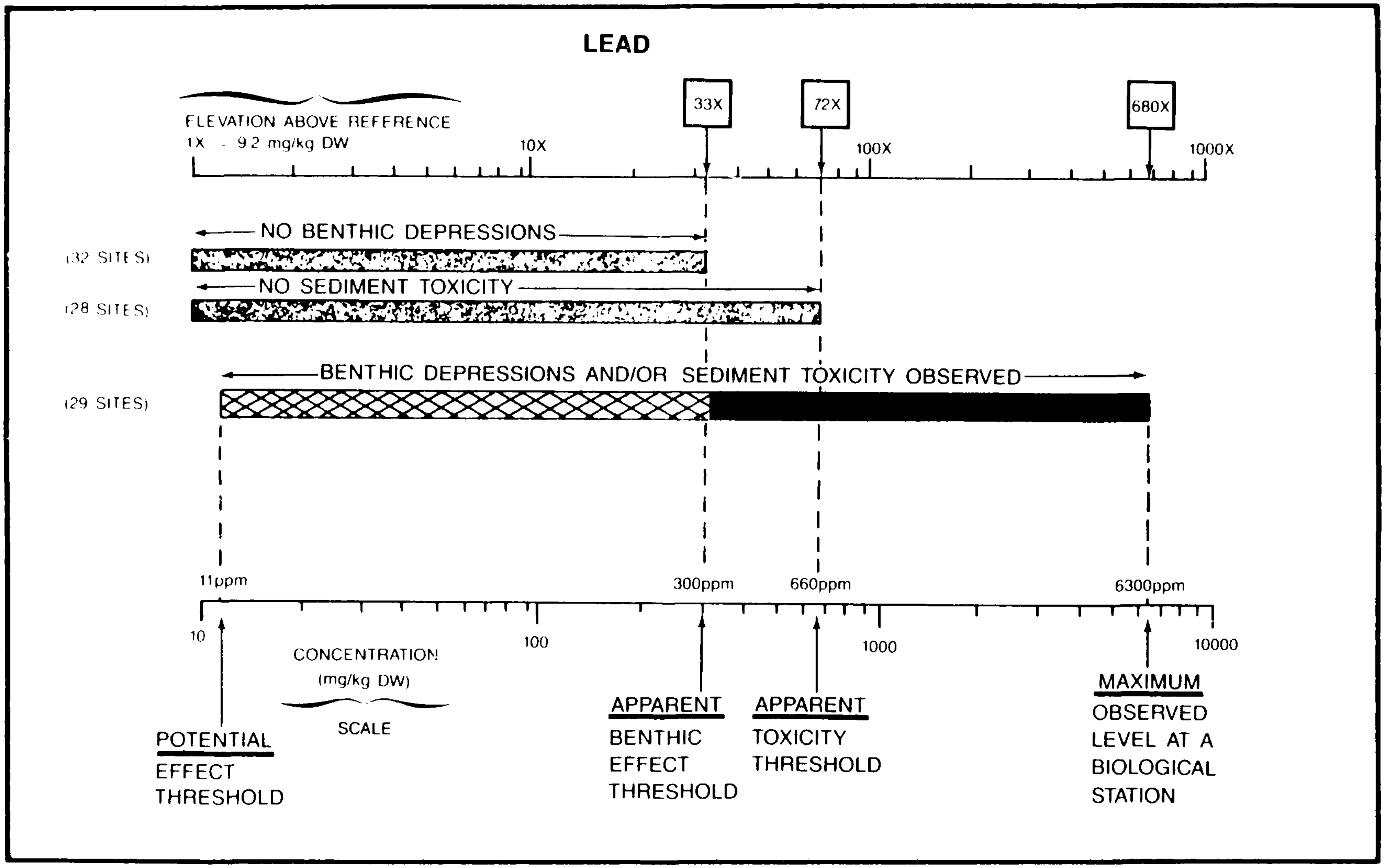

Figure 3. Example use of synoptic benthic effects and sediment toxicity data to determine apparent chemical effect thresholds

benthic AET was 300 mg/kg DW (EAR = 33) (Figure 3). The effect thresholds were termed "apparent" because significant toxicity or benthic effects were not found at some stations with equal or lower lead concentrations, while significant sediment toxicity or benthic effects were found at all stations with higher concentrations. These empirical relationships do not prove that contaminants found above an AET were responsible for the observed toxicity or benthic effects. However, within the limits of this data set, chemicals present above these concentrations were associated exclusively with problem sediments having significant toxicity or depressed benthic infaunal abundances (or both). Because of this association, all chemicals present above toxicity or benthic AET were defined as problem chemicals requiring further evaluation.

The approach shown in Figure 3 was used to identify toxicity and benthic AET for all chemicals of concern. AET expressed on a DW basis are summarized for metals, organic compounds, and conventional sediment variables in Table 2.

AET were exceeded by a number of chemicals at most of the 29 stations exhibiting statistically significant biological effects. All 6 of the 29 stations where neither AET was exceeded were unusual in that only one of the biological indicators showed a response. Most of these six stations exhibited toxicity by the amphipod bioassay only, and the toxicity may have been related to the high percentage of fine-grained material (>80 percent) at each station. The difference in thresholds for toxicity and benthic effects for several chemicals suggests that both bioassays were more sensitive to organic compound contamination, and that benthic infauna depressions were more sensitive to metals contamination.

AET were also calculated for normalizations to organic carbon and percent fine-grained material. For most sediments with multiple toxicity and benthic effects, chemicals exceeded an AET regardless of normalization.

Gradients of effects were analyzed for study areas having a sufficient number of stations with biological measurements to allow such analyses. Five such areas were analyzed. Strong relationships were observed for a few chemicals that were present at concentrations above an AET. Exposure-response relationships were found at selected sites for the following chemicals: PCBs, 4-methylphenol, PAH, mercury, lead, zinc, and arsenic.

Problem chemicals are defined as those exceeding an AET. Problem chemicals were further prioritized into three categories:

a. Priority 1 - Present above an AET, with distribution corresponding to observed toxicity or benthic effects gradients.

b. Priority 2 - Present above an AET at more than one station in the problem area, with no apparent relationship to toxicity or benthic effects gradients, or insufficient effects data were available for evaluation of gradients.

TABLE 2. APPARENT EFFECT THRESHOLDS (AET) FOR SEDIMENT CONTAMINANTS AND CONVENTIONAL VARIABLES

Metals	Toxicity, AET, mg/kg DW	Benthic Effects, AET, mg/kg DW
Antimony	5.3	3.1
Arsenic	93	85
Cadmium	5.8	5.8
Copper	310	310
Lead	660	300
Mercury	0.59	0.52
Nickel	39	39
Zinc	490	260

Organic Compounds	Toxicity, AET, μg/kg DW	Benthic Effects, AET, μg/kg DW
Phenol	420	1,200
2-methylphenol	63	72
4-methylphenol	670	670
LMW aromatic hydrocarbons	5,200	5,200
HMW aromatic hydrocarbons	12,000	17,000
Chlorinated benzenes	270	400
Chlorinated butadienes*	>47,000	>47,000
Total phthalates	3,400	5,200
Total PCBs	420	1,100
Benzyl alcohol	130	130
Dibenzofuran	540	540
n-Nitrosodiphenylamine	28	28
Tetrachlorethene	140	140
Ethylbenzene	37	37
Total xylenes	120	120

Conventional Variables	Toxicity, AET	Benthic, AET
Total volatile solids, %	22.2	22.2
Total organic carbon, %	15.1	15.1
Nitrogen	0.28	0.28
Oil and grease, mg/kg	2,200	4,300

* No station exhibiting toxicity or benthic effects had concentrations exceeding these levels.

c. Priority 3 - Present above an AET at only one station within the problem area.

Priority of Problem Areas

After source evaluations (which are not presented in this paper but may be found in Tetra Tech (1985)), problem areas were ranked so that resources could be effectively allocated to remedial actions. The ranking was based on a quantitative evaluation of:

a. The environmental hazard indicated by the problem area contamination, toxicity, and biological effects.

b. The spatial extent of each problem area.

c. The confidence that sources of problem chemicals in each problem area had been accurately identified.

The spatial extent and general priority for source evaluation of all problem areas identified in Commencement Bay are summarized in Figure 4. At the highest priority sites, all three site-specific indicators were significant. Of the twenty-one problem areas, eight received the highest priority for source evaluation, including three within Hylebos Waterway, two within City Waterway, one within Sitcum Waterway, one within St. Paul Waterway, and one along the Ruston-Point Defiance shoreline. The second priority sites are "hot spots" where chemical contaminants exceeded an AET, and both bioassays were significant or multiple benthic depressions were observed within the problem area. Four problem areas received second priority for source evaluation, including one within each of Hylebos, Middle, and City Waterways, and at a single station along the Ruston-Point Defiance shoreline. Third priority sites include those where chemical contaminants exceeded an AET, and one of the bioassays was significant or a single benthic taxon was significantly depressed. The lowest priority sites for source evaluation include those areas where no sediment toxicity or benthic effects were observed, but where AET applied to available chemical data suggested that toxicity or benthic effects would have been found had biological data been collected.

CONCLUSIONS

The complex task of dealing with chemical contamination of sediments is complicated by the lack of criteria for sediments and the lack of scientific cause-effect relationships between chemical contamination and effects on marine biota. The approach used for the Commencement Bay project was successful in defining problem areas and potential problem chemicals. Determination of problem sources was partially successful. The approach also generated a starting point for development of numerical sediment criteria, i.e., the "apparent effects thresholds."

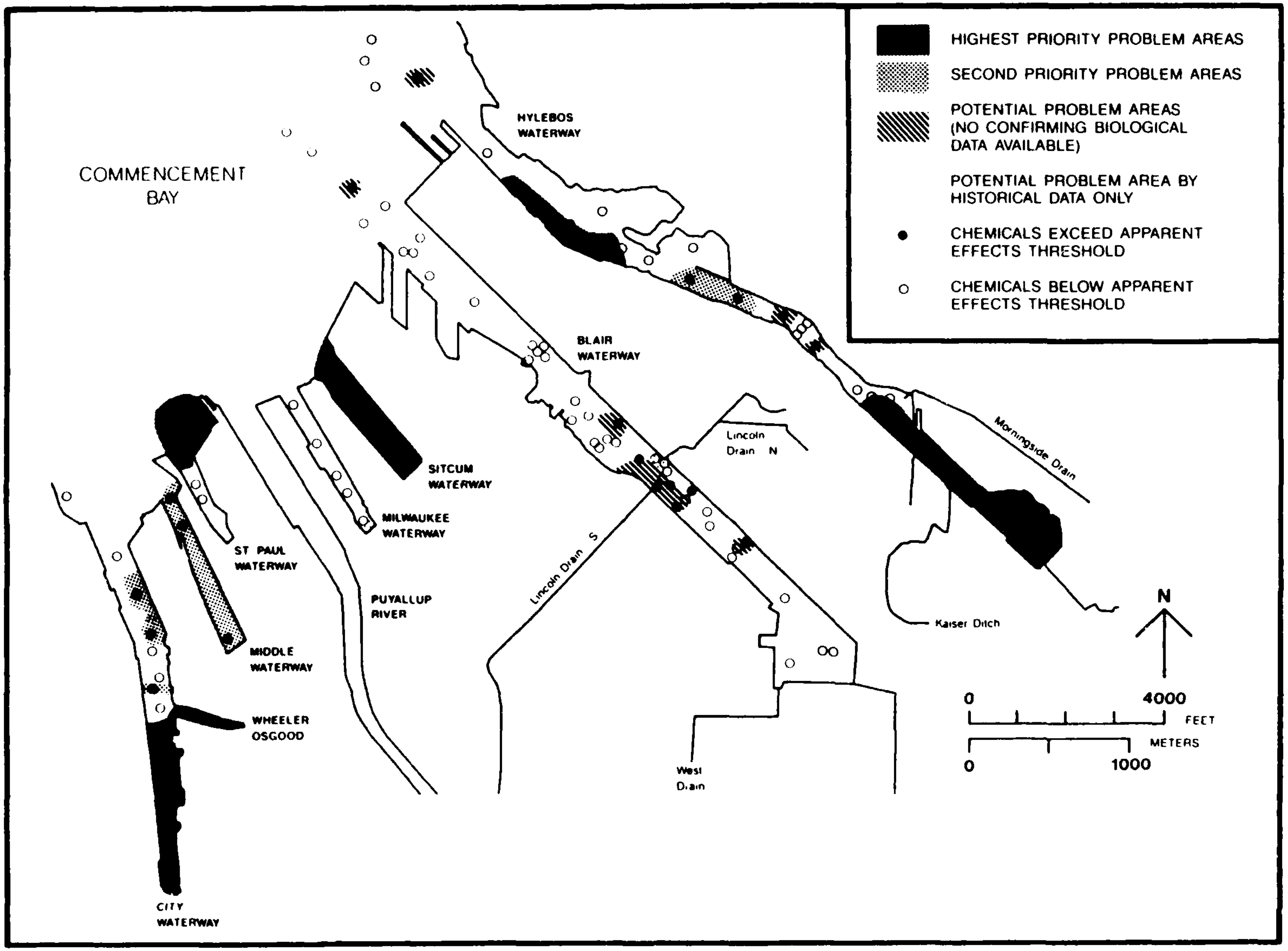

Figure 4. Definition and prioritization of Commencement Bay problem areas (Continued)

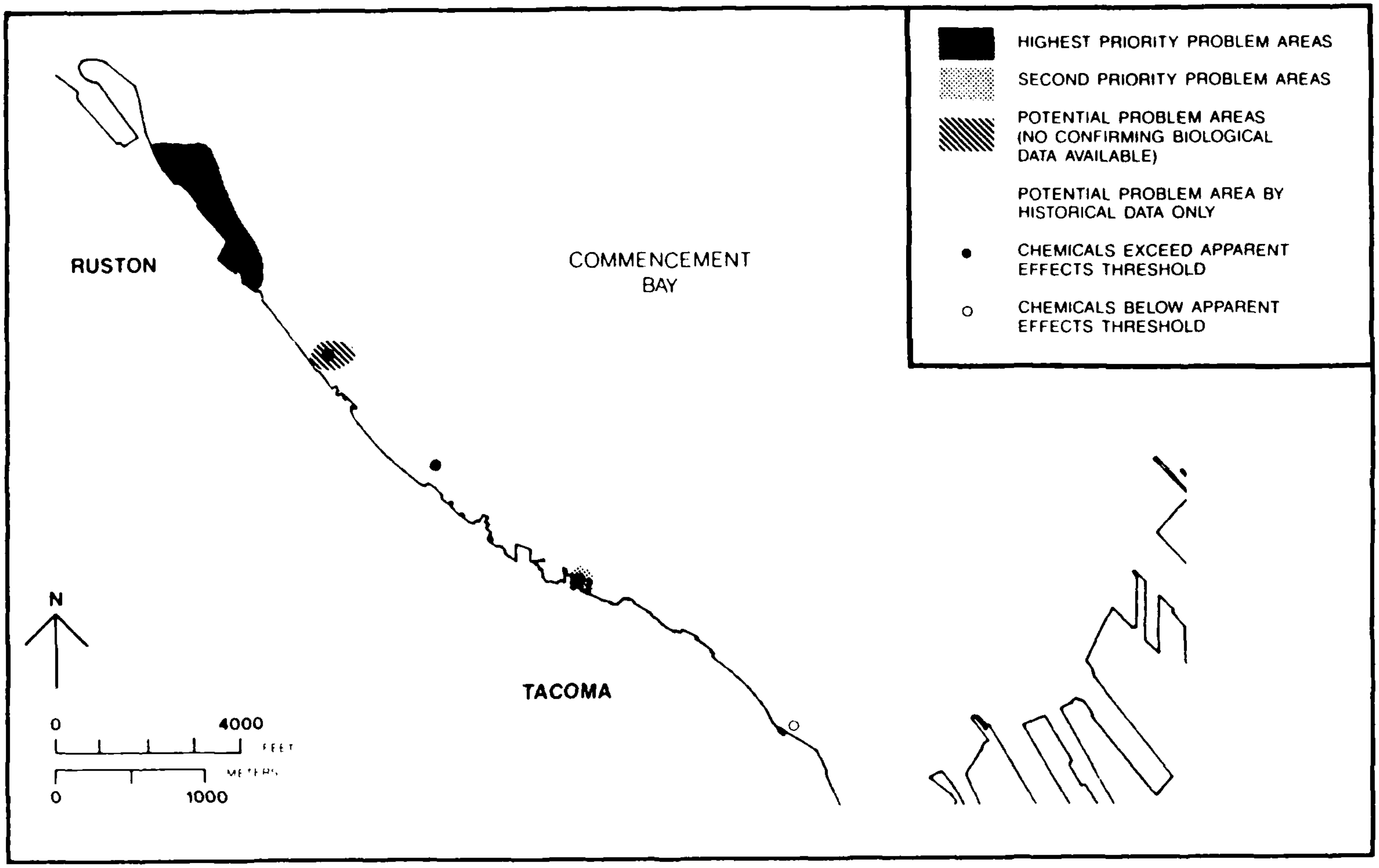

Figure 4. (Concluded)

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CONTAMINANT MOBILITY AND ENGINEERING DESIGN AT A NEARSHORE CONFINED DISPOSAL SITE-- PORT OF SEATTLE TERMINAL 91 SHORT FILL

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ABSTRACT

Though the use of a slip fill as a containment site for mildly contaminated dredged material disposal is not unprecedented, the agreements between the Port of Seattle and the regulatory agencies concerning the long-range monitoring and the remedial actions plan are unique. The Port of Seattle's Terminal 91 Short Fill must serve a dual purpose. The fill provides the additional Terminal 91 area required by the major tenant. The fill also serves as a disposal site for mildly contaminated dredged material unsuitable for open-water disposal from other Port terminal expansion projects. Both goals are complicated by the existing elevated street structure across the head end of the slip, which has necessitated a berm at both ends of this slip fill. As a result a pond is created between the new fill and the old end of the slip. The Port's preferred construction option is disposal of mechanically dredged sediments behind berms of clean structural grade, well sorted, sandy gravel. This will contain the sediments while allowing dewatering of the dredged material through the relatively permeable berms.

The Port was initially faced with regulatory agencies unable to appraise the Port's preferred construction option in light of heightened concern over Puget Sound and the lack of field data on contaminant mobility in nearshore disposal sites. A researchlike approach was worked out where the Port did an extensive computer model analysis necessary to apply the existing information to the particulars of the Terminal 91 site. After this showed the option acceptable, the Port, in coordination with the regulatory agencies, designed the Criteria, Thresholds, Monitoring, and Remedial Actions Plan. In addition to the monitoring, the Plan defines the criteria under which containment will be judged, the trigger thresholds, and the remedial actions to be taken if the site does not perform as designed and modeled. This Plan will, through extensive monitoring, study the effectiveness

of the fill at minimizing contaminant mobility. It will verify the preproject modeling and reevaluate long-range predictions. This example provides a reasonable approach to pursuing a project when faced with environmental concerns and a lack of directly applicable data.

INTRODUCTION

The project, known locally as the Terminal 91 Short Fill, is the Port of Seattle's plan to combine the construction of additional terminal area for the major tenant (Nissan) and the need for a disposal site for mildly contaminated dredged materials. Terminal 91 is a mixed-use terminal owned by the Port of Seattle. It has an upland area of approximately 100 acres and has a unique configuration including two parallel piers, each with a length of approximately one-half mile (Figure 1). The piers have an earth-fill central core with timber, piling-supported aprons around the perimeter. The Short Fill project consists of an approximate 5-surface-acre earth fill at the north end of the waterway between the piers. The purpose of this project is to provide both the additional space at Terminal 91 and a disposal site for mildly contaminated dredged materials from other terminal expansions, along the Duwamish Waterway.

BACKGROUND

Recent history of dredged material disposal in Seattle may be helpful in understanding the project. Dredging at the Port of Seattle has been done predominantly as a part of terminal construction. This dredging has been necessary to provide adequate berth depth and to provide structural stability along the shoreline. Maintenance dredging requirements have been minimal. Historically, dredged materials have been disposed in deep water in Elliott Bay, or used as fill in terminal construction. In recent years we have become more concerned with the water quality impacts of dredging. Initially, that concern was limited to water column effects during dredging and disposal. We monitored dredging and disposal operations for changes in water quality such as pH, dissolved oxygen, turbidity, and release of metals and polychlorinated biphenyls (PCBs). We did not detect significant degradation of the existing water quality regulations during those operations. The discovery that the beds of our waterways have accumulated chemical contamination has led to increased concern for proper disposal of dredged materials and the long-term impacts involved. The sediment contamination is caused by a multitude of sources (industries, highways, farms, and residences) and carried to the waterways by direct discharge, combined sewer overflow, and storm runoff. When we proceed with construction of marine facilities, we are forced to deal with these mildly contaminated sediments. A major difficulty that we have encountered is that there are no generally accepted standards for proper disposal of mildly contaminated dredged materials. We have experienced a rapid evolution in what is considered acceptable for disposal of dredged material. Less than 4 years ago we were disposing of mildly contaminated dredged materials within a shoreside fill with little chemical testing and no ongoing postdisposal monitoring. Today, for the Terminal 91 Short Fill, we are subjecting the proposed dredged materials to extensive chemical testing and

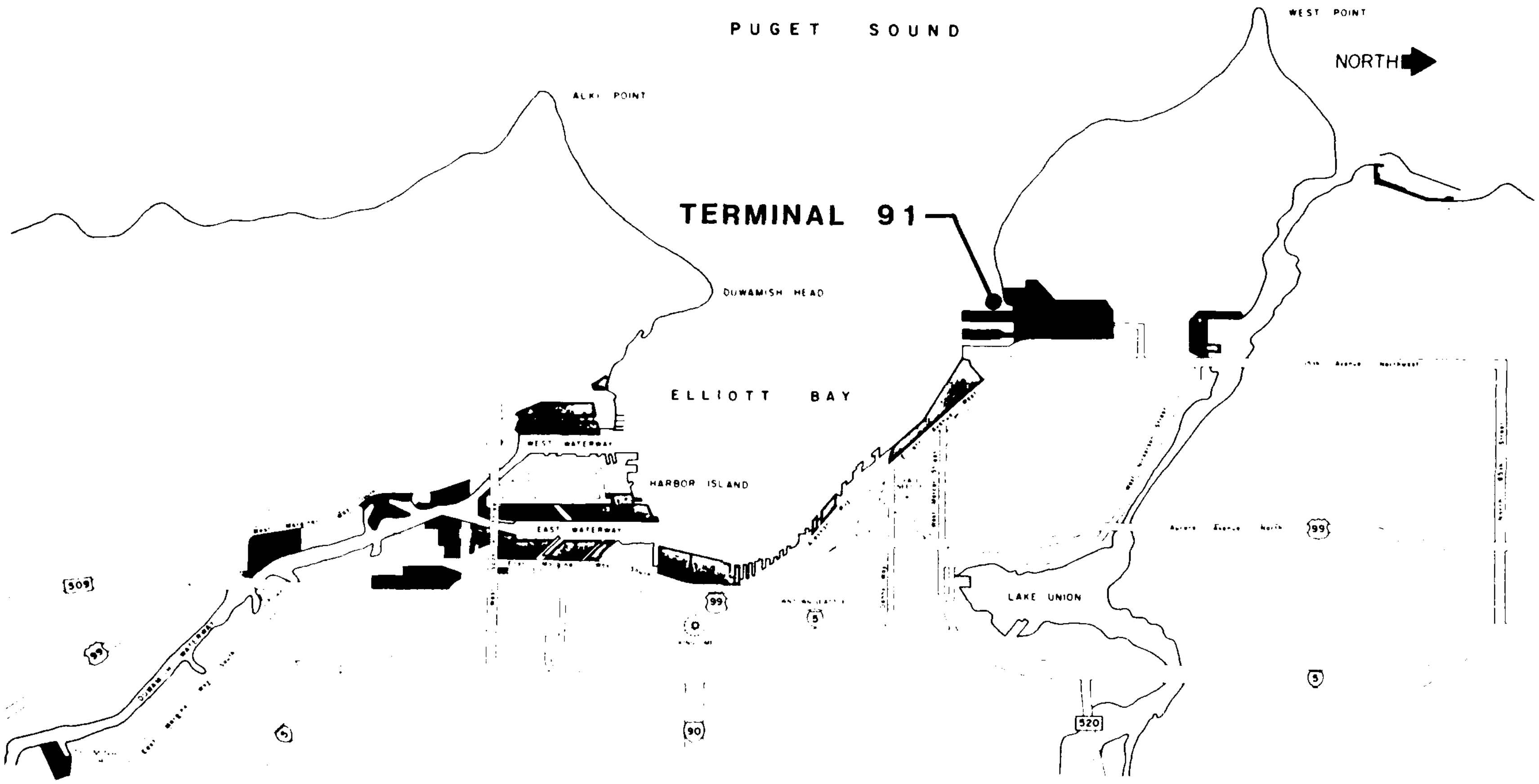


Figure 1. Terminal 91 location

predisposal, site-specific contaminant mobility analysis, as well as preparing the substantial monitoring and remedial action plan which is the subject of the second portion of this paper.

ENGINEERING DESIGN

The following is a brief description of the engineering and construction of the Terminal 91 Short Fill (Figure 2). Two containment berms are under construction between the piers at each end of the fill. The berms are approximately 55 ft high (18 ft above mean lower low water (mllw)) and are constructed of well-sorted sandy gravel material. The slopes are 2H:1V and are covered with riprap. The berms are constructed in lifts. The riprap is placed first to help stabilize the sandy gravel material that is then deposited in the lower lifts by bottom dump barge.

It was necessary to locate the north berm 75 ft south of the elevated street structure at the head end of the slip to prevent downdrag on piling which supports the structure. Consequently, a pond will be created which will require surface aeration and possibly mixing to prevent stagnation. The south berm will be constructed initially with a 100-ft-wide notch in the center down to -6 ft mllw. This will allow the bottom dump barges to be floated between the berms for disposal. A turbidity curtain is to be drawn across the notch during the disposal. When the contaminated fill rises to within 4 ft of the notch elevation, the notch will be raised to +0 ft mllw, and then on to final grade +18 ft mllw. This will minimize the loss of contaminants by overtopping due to the surge from the disposal. All of the mildly contaminated dredged material placed at the site will be dredged and deposited by mechanical means. After the contaminated dredged materials are placed within the berms, they will be capped with a minimum layer of 10 ft of sandy gravel fill. This will serve to surcharge the fill and promote dewatering of the dredged material prior to paving the surface with asphalt concrete.

CONTAMINANT MOBILITY

Original Design

The proposed design's ability to restrict contaminant mobility was based on the tendency for the typical contaminants found in Duwamish sediments to remain adsorbed onto the sediments. Therefore, if the sediment was held in place by the well-sorted sandy gravel berms, so would the bulk of the contaminants. Secondarily, the negligible leaching expected through the berm should not cause any water quality problems. This was due in part to the large dilution factor of the tide flowing in and out of the berms. These ideas were based on results from several US Army Corps of Engineers studies and our own study at Terminal 105.

The previous studies that addressed leachate and contaminant mobility were primarily those from the Corps of Engineers Dredged Material Research Program. These studies (Brannon, Plumb, and Smith 1978; Chen et al. 1978; Hoeppel, Myers, and Engler 1978; Mang et al. 1978; and Yu et al. 1978) indicated that with adequate control of the solids and the dilutions expected by tidal mixing, there would be no significant impact and the project should meet the appropriate water quality criteria.

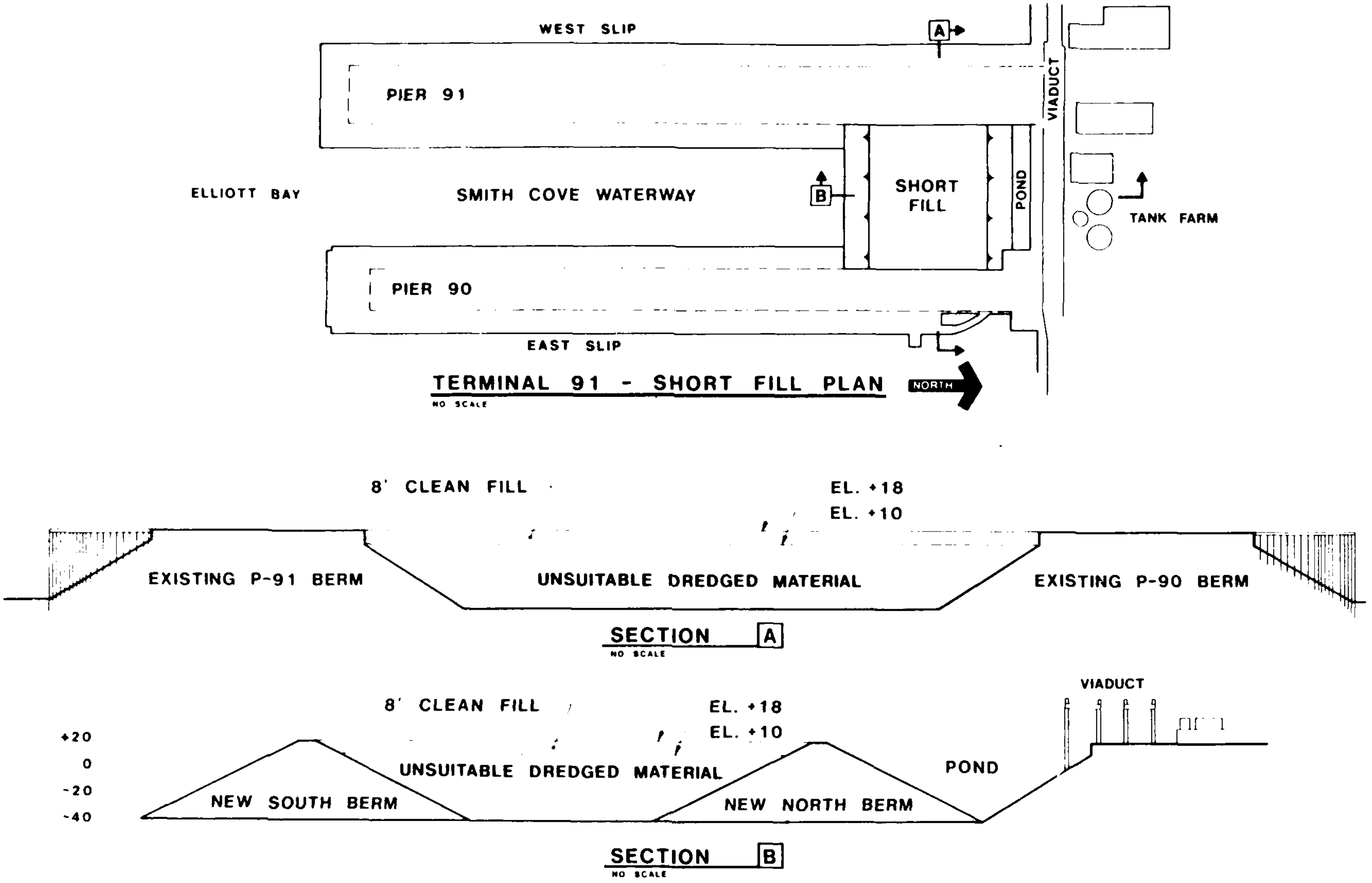


Figure 2. Plan view of Terminal 91

The Port of Seattle's previous study (Port of Seattle 1985) included a preproject survey and five postproject surveys for priority pollutants in the surrounding nearshore uplands disposal site of mildly contaminated dredged material. This site at Terminal 105 was a pit excavated in the nearshore uplands about 400 ft from the Duwamish Waterway. Brakish ground water at the site is tidally connected with the adjoining Duwamish Waterway through the sandy soils. This study clearly showed the initial dewatering stage with a small pulse of metals seen in the monitoring wells during the first few months. The organic compounds of the priority pollutants generally showed no movement except for phenols which were higher in the downgradient wells during months one through seven. This is not entirely unexpected due to the mobility of phenol and the mildly acid (pH 6.1) condition of the Terminal 105 ground water. It was concluded from this field study that disposal in partially saturated, tidally connected uplands with a fresh ground-water head "upstream" resulted in no loss of contaminants that would constitute a water quality problem in the adjacent waterway.

Climate of Concern

More recent studies in Puget Sound have been focusing on the mass flux of contaminants and the "ultimate fate" of contaminants in disposal sites.

Among these was the Metro Toxicant Protreatment Planning Study (TPPS) (Municipality of Metropolitan Seattle 1984) which looked at the fate and mass flux of toxicants in Puget Sound as a system and, in particular, at the contribution of municipal sewage effluent to the mass budgets. The TPPS brought to the forefront the realization that, due to its glacier carved bathymetry with a shallow inlet sill, Puget Sound has restricted flushing. The deeper sediments become the final sink for much of the contaminant load. These findings led to concern about the carrying capacity of the Puget Sound sediments and to the development of new contaminant criteria for the uncontained open-water disposal of dredged materials, and greater amounts of dredged material requiring contained disposal.

The "ultimate fate" of materials placed in a contained disposal site was also being reviewed by the Corps of Engineers' work (Javandel, Doughty, and Tsang 1984) which focused on dredging as a means of disposal for the Commencement Bay Superfund site (sediments). This work was done as a part of a Superfund site remedial investigation requiring the consideration of "ultimate fate." Based on more recent studies and approaches, the conclusion was that dredged material kept in its original state, saturated and anaerobic, will have maximum retention percent of contaminants. If unsaturated and oxygenated, it will lose more contaminants. Contaminant losses are proportional to water moving through the system.

Based on this more recent information, the regulatory agencies have required the Port to provide a more site-specific study of the ultimate fate of the materials to be disposed of in the Terminal 91 fill. This was requested in order for the agencies to assess the proposed construction, even though the analysis of the sediments scheduled for disposal at the site showed that contamination levels were well below hazardous waste classification (Table 1). As a result, the Port contracted for a modeling study (Hart-Crowser and Associates, Inc., and URS Corporation 1985) to combine data from previous studies,

TABLE 1. CONTAMINANT LEVELS IN TERMINAL 32 SEDIMENTS FOR DISPOSAL IN TERMINAL 91 SHORT FILL AND OPEN WATER*

Contaminant	Terminal 91 Disposal Terminal 32 Sediments Failing 4-Mile Rock Criteria	Open Water (4-Mile Rock) Criteria	Open-Water Disposal Terminal 32 Sediments Passing 4-Mile Rock Criteria
Metals, ppm			
As	17.4 ± 3.3	19	3.7 ± 1.1
Cd	5.0 ± 2.9	0.9	0.68 ± 0.25
Cu	119.0 ± 48.7	115	17.6 ± 7.2
Pb	88.1 ± 95.5	158	11.3 ± 15.5
Hg	1.0 ± 0.44	1.4	0.05 ± 0.06
Zn	435.9 ± 244	450	39.1 ± 11.3
Organics, ppb			
PCBs	2,763 ± 2,498	760	5.9 ± 24
DDTs	72 ± 74	9	N.D.
High PAHs	8,485 ± 7,206	14,000	292 ± 547
Low PAHs	3,778 ± 3,620	855	236 ± 202

* Values are total sediment concentration, reported on a dry weight basis and expressed as mean ± one standard deviation.

sediment chemistry, chemical theory, site permeability information, and ground-water information. The goal was to produce site-specific recommendations for the design and monitoring of the proposed project.

Modeling

This effort was conducted in three interdependent areas: site permeability, two-dimensional modeled flow rates, and final predictions of the concentrations of contaminants throughout the berm.

Permeability

The permeability study (Hart-Crowser and Associates, Inc. 1985) was based on previous boring logs, existing ground-water wells, and laboratory studies of proposed berm structural material. The in-place permeability of the dredged material fill was estimated from grain-size measurements of the material. This study (Figure 3) showed solid filled existing piers with approximately 10 ft of slightly more porous recent hydraulic fill overlying relic silty sands, which are also the floor of the slip.

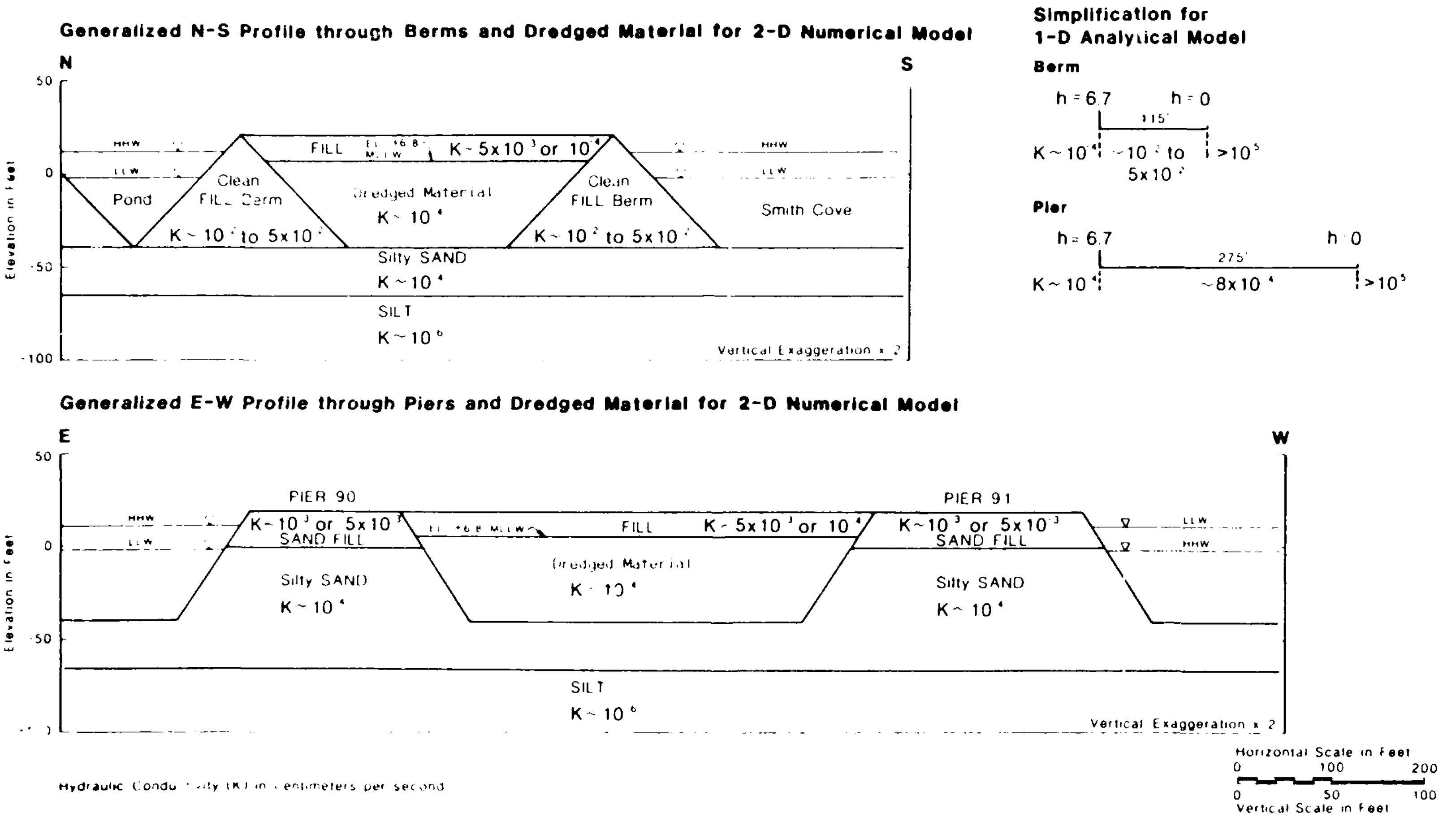


Figure 3. Generalized hydraulic conductivity profiles

The ground water inland from the head end of the slip, though being 4 ft above mean tide (+6.6), was found to be only minimally connected to the tides. Approximately 2 percent of the measured tidal movement was reflected in the ground water.

The permeability of the project berms is much lower than the existing filled piers which are also much wider. The net result is that dewatering and hydraulic tidal connection between the central fill and the open waterway is expected to occur through the project berms. The permeability of the fill is predicted to be orders of magnitude lower than the berms, and therefore controls water movement through the site.

Ground-Water Flow

The hydraulic response of the system was modeled using the FPM Finite Element Model (Golder 1983) in a vertical two-dimensional slice configuration with output (flow velocities and directions) used as input to the transport model. It was run for both initial dewatering and steady-state conditions in a variety of situations representing possible construction alterations and possible remedial actions.

The model used a two-dimensional cross section, north-south through the berms and fill. The boundaries were the pond north berm face, the low permeability silt layer 20 to 30 ft below the silty sand of the existing waterway, and the south berm waterway face. This cross section (Figure 4) was divided into 214 elements defined by 240 nodes (corners). The output was tables of hydraulic heads for each node, Darcy flow velocities (magnitude and direction) for each element, and plots of head contours. The dewatering phase assumed instantaneous construction; therefore, we used the 30 days postconstruction as an environmentally conservative highest level. The idealized 25-hr tidal component was simulated by holding boundary condition water levels on the south berm face at eight constant steps throughout the period. The head and duration of each step were based on mean tidal values for Seattle.

The water flow modeling shows that the movement of water through this fill is mainly dependent on the ground-water head established in the upstream pond and the permeability of the fill material. It also shows that artificially elevating the pond level to +13 during dewatering (to minimize flow into the pond) has no significant benefit. However, after dewatering, artificially lowering the pond to mean tide level would eliminate through flow through the fill and therefore minimize contaminant losses.

The cover fill, with high permeability equal to that of the berms, acts as a top drain during the dewatering phase, and does not act as a pathway allowing significant cycling of pond tidal water or tidal cycling through the fill as was initially suspected. Manipulating the permeability of the cover fill, therefore, does not provide a significant environmental advantage. Infiltration through the asphalt paving cap does not create a significant flow through the system as compared with tidal action or advective flow.

The hydraulic head contours (Figure 5) showed that most of the tidal action occurs in the berm and not in the fill. The tidally induced velocities (Table 2) are one to two orders of magnitude higher in the upper portion of the berm.

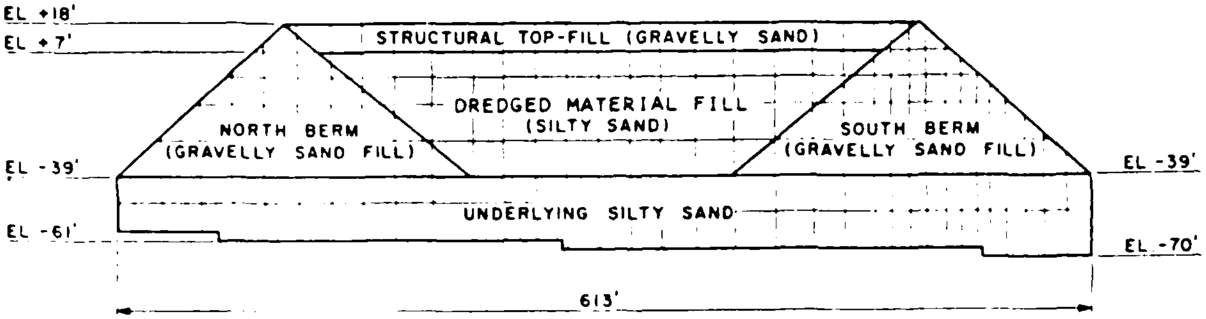


Figure 4. Finite element mesh for berm-fill system

TABLE 2. DARCY VELOCITIES AND MIXING AT THE BERM FACE (POND AT +9)

Elevation ft, msl	Tidal Velocity m/day	Advective Velocity m/day	Mixing Ratio
+7 to -3	5.7×10^{-1}	3.8×10^{-3}	150:1
-8 to -18	1.5×10^{-1}	6.7×10^{-4}	225:1
-24 to -39	1.3×10^{-2}	3.8×10^{-5}	350:1

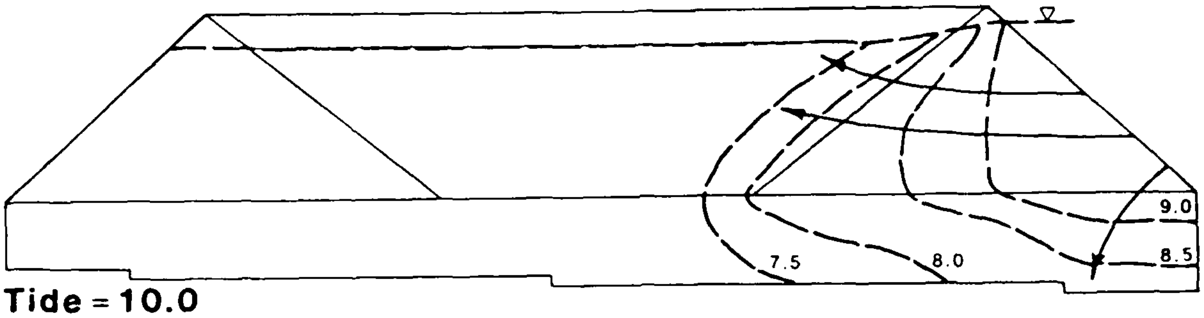
Contaminant Transport

The contaminant mass transported through the system was modeled using the output of the hydraulic model as input to the LEACH 1-D (curvilinear) analytical transport model (a modified form of ODAST, Javandel, Doughty, and Tsang 1984). The contaminant mass transport model was run using the various hydraulic model outputs with a range of contaminant absorption/desorption constants from d(k' = 0.1) to PCB (k' - 2900). This modeling showed that:

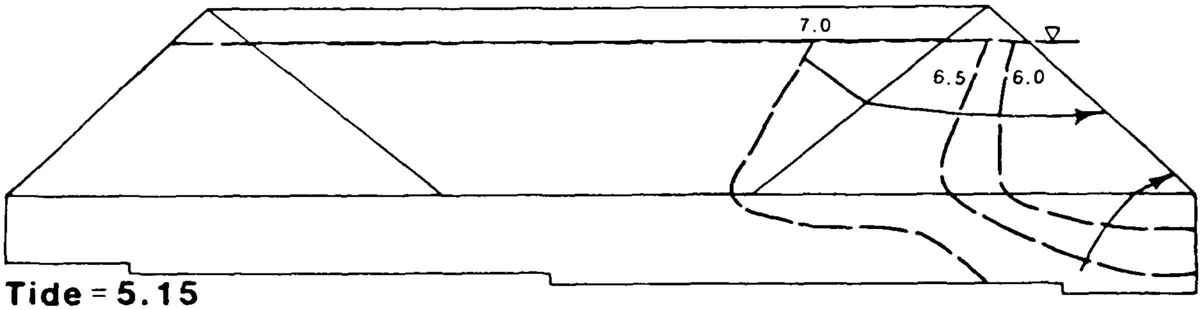
a. Implementation of the proposed Terminal 91 Short Fill as a dredged material containment facility using the as-planned construction is unlikely to produce concentrations of deleterious contaminants at the waterway berm face which exceed chronic saltwater exposure criteria. Cadmium and mercury concentrations within the south berm may begin to exceed these standards at some time approximately 40 years in the future. The maximum concentration should never be more than 10 times above the criteria. Mixing at the berm face with dilution ratios of more than 100 to 1 is likely to reduce these concentrations below chronic saltwater exposure criteria.

b. Allowing the pond to remain at its natural level (estimated at elevation +9 ft) over the long term is unlikely to cause water flow to increase to the point that waterway berm face concentration levels would exceed chronic saltwater exposure criteria. Concentrations within the berm may reach these levels within 20 years if the dispersion in the berm is very low (0.1 m^2/day).

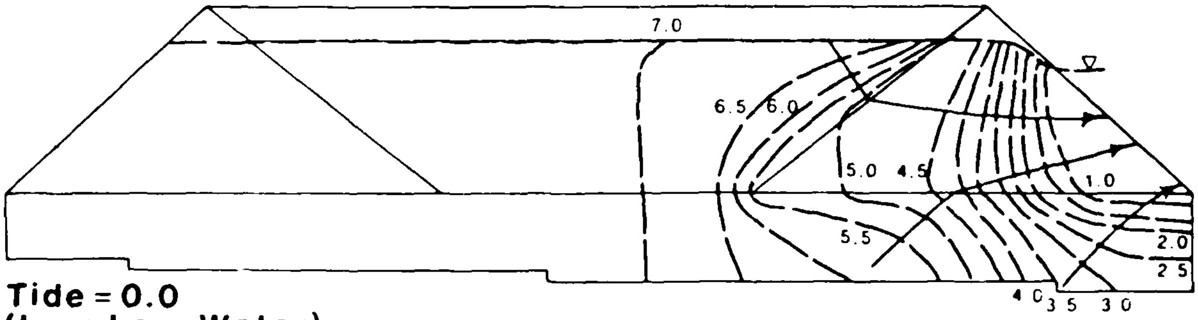
Low Pond with High Permeability Top Fill



Tide = 10.0
(Rising)
180 Min



Tide = 5.15
(Falling)
490 Min



Tide = 0.0
(Low Low Water)
642 Min

12 Head Contour

Flow Direction

HART-CROWSER & associates inc.

Figure 5. Hydraulic head contours during tidal cycle

c. Contamination will be removed from the fill through two dominant mechanisms: advective flow (caused both by dewatering of the dredged material during consolidation and by flow from the pond during later periods) and dispersion of contaminants (caused by tidal action). Dewatering advection will dominate during the early stages (less than 1 year) while tidal dispersion will dominate after this period.

d. Cadmium and mercury are the contaminants most likely to reach concentration levels in the berm which could exceed chronic saltwater exposure criteria. Absorption is likely to retard the movement of most organic contaminants.

e. Mixing at the waterway berm face is likely to cause more than a 100-to-1 average dilution in comparison with concentrations in the middle of the berm. The dilution of the berm is the result of mixing of clean seawater with ground-water flow from the berm during each tidal cycle. The bulk mixing ratio of more than 100 to 1 is based on the ratio of the relative volumes moved by tidal action, to the volumes moved by advection from the elevated pond (+9 ft), and infiltrated precipitation. The actual concentration at any given time during the tidal cycle is a function of dispersion. High dispersion results in actual concentrations during most of the tidal cycle close to the average produced by bulk mixing. Low dispersion will result in actual concentration near seawater levels during most of the cycle with short periods equal to in-berm concentrations.

f. Monitoring wells should be placed at the intertidal elevation in order to first detect contaminant movement. Flow velocities are highest and flow paths from the fill are shortest in this intertidal zone. The first contaminants to discharge from the system are likely to travel through this zone. Deeper wells should also be placed in order to verify long-term discharge and to compute average discharge rates and concentrations. Wells should be placed at the waterway berm face to measure the actual concentration of discharges. Wells in the middle of the berm area are also recommended to monitor in-berm concentrations.

g. Sensitivity analyses indicate that the system is sensitive to variations in the value of: dispersivity, hydraulic conductivity of the fill, and pond elevation. Sensitivities were such that:

(1) A one-order-of-magnitude variation in dispersion could cause a variation of one order of magnitude in concentration at certain distances and times.

(2) A one-order-of-magnitude variation in dredged material fill hydraulic conductivity could cause a variation in flow velocity of one order of magnitude within the berm.

(3) A 3-ft increase in pond height could cause a fivefold increase in berm velocity through the berm.

h. The system was not sensitive to hydraulic conductivity of the berm, rate of infiltrated precipitation through the asphalt-covered top, storativity of the dewatered fill, or specific storage of the whole system. An order-of-magnitude variation in these parameters (except specific storage) caused less than a 20 percent variation in berm flow velocities. A 50 percent variation in specific storage caused no measurable variation in berm flow velocity.

In summary, the modeling shows that the performance of the project is expected to meet the water quality criteria and the general goals of the containment. It shows that the probability of excessive contaminant loss is small and that even if it occurs, the impact is minimal on the specific site. The minimal impact conclusion is due to the low level, small quantity aspect of the leachate, and existing site contamination. The sediments of the existing site are as contaminated as those requiring containment in the fill. The site receives a large contaminant from several major storm drains and combined sewer overflows emptying into the immediate and nearby arch.

On the strength of these conclusions, the Port has been allowed to proceed, provided that an acceptable monitoring and remedial actions plan could be prepared.

Criteria, Thresholds, Monitoring, and Remedial Action Plan

The following is a detailed summary of the plan (Port of Seattle 1985) designed by the Port in cooperation with the Washington Department of Ecology and the other regulatory and resource agencies. The goals of this plan, and the subsequent monitoring program, are:

a. Establish the basic criteria against which performance of the disposal site will be measured.

b. Set up the monitoring program to measure the performance and follow the movement of contaminants at the site.

c. Establish threshold levels for initiation of remedial actions.

d. Set up the remedial actions that will be undertaken if the system does not meet the criteria and should prove to be a long-term water quality problem.

e. Propose research and modeling verification that will make these results more applicable to other dredged material disposal projects.

f. Prepare a monitoring plan to ensure the protection of local water quality during the filling operation.

The containment performance monitoring program is based upon accepted marine water quality criteria and consists of berm well samples and water column samples that will be or are in the threshold/remedial action phase. Additional well and pond samples will be collected to provide research information on movement of contaminants in the confined disposal site. All samples will be analyzed for heavy metals and some for full priority pollutants.

There will also be a monitoring program for the pond and a water quality monitoring program for the disposal operation. These are standard water quality type programs and will not be elaborated on in this paper.

The thresholds for remedial action apply only to the wells in south berm, Pier 90 and Pier 91. The dilution factors for calculating the threshold levels from the well data will be derived from the field sampling of the corresponding sets of berm face wells, and vertical berm wells located at the center of the south berm. Exceeding a threshold value will initiate notification of the agencies and a series of confirming samples. Action decisions will be based on these sample sets.

The sampling frequency will be high during the first 6 months to document the dewatering stage and to establish dilution factors. The frequency will then be reduced. The monitoring program will be reviewed annually with the objective of reducing unproductive sampling. The long-term monitoring will be conducted quarterly and then semiannually. It will last for a total of 4 years after capping. At this point, the entire program will be reviewed and any future work decided upon.

The data will be reported to Washington Department of Ecology (WDOE), the Corps of Engineers (CE), and the Environmental Protection Agency (EPA) after each sample series. Water quality data from the disposal operation will be reported within 48 hr. Any decision regarding the performance of the fill containment and the need to institute remedial action will include WDOE, EPA, CE, and the Port of Seattle.

The main focus of the remedial action is to repair the site if a chronic water quality problem should exist. It is understood that the extensive construction necessary for the more extreme remedial actions and the time delays in confirming any real problem make it unrealistic to consider it applicable to the initial dewatering plan.

After the first year of monitoring and research, the data from the field studies will be used to update and reevaluate the long-term projections from computer modeling.

Criteria

The criteria used to judge the acceptable operation of the confined disposal site will be the EPA marine water quality criteria applied at the south berm face or side pier faces. The chronic criteria will be applied at the berm or pier face "discharge area." Where there are no EPA-accepted criteria, "background" seawater and 10 times background seawater will be used in a two-tier criteria. Exceeding 10 times (10X) background will initiate the same response as the EPA chronic criteria. Above background will initiate notification of WDOE, EPA, and CE and meetings with the Port and these regulatory agencies to determine the need for confirming samples, the impact of the compound detected, and the appropriate remedial actions, if any. These discussions will rely heavily on the EPA list of "Lowest Reported Toxic Concentrations" for priority pollutants. Any decision to call for remedial action will be based on a confirmed water quality problem. Though the WDOE, EPA, and CE have independent authority under the Clean Water Act that could apply to this situation, any decision to require remedial action will be made by consultation among these regulatory agencies.

Action Thresholds

The action threshold levels are those levels in the monitoring wells which would indicate a high probability of exceeding chronic saltwater criteria at the berm faces. This would indicate a failure of the disposal site to adequately contain the contaminants of concern, and could justify initiating remedial action. Initially, data from samples taken in the berm face wells will be compared against the established criteria. If criteria are exceeded, discussions between the Port of Seattle and the WDOE will be initiated. These discussions will first establish an appropriate dilution ratio. Secondly, there will be an analysis of the contaminant at the dilution ratio established to determine if there is an adverse impact to the aquatic environment.

The site-specific information used to establish the dilution factors between the berm face, the berm face wells, and the vertical berm wells will include both the modeling information and the field sampling data from the south berm. When finalized, these factors will be applied to all the south, east, and west berm pier vertical wells.

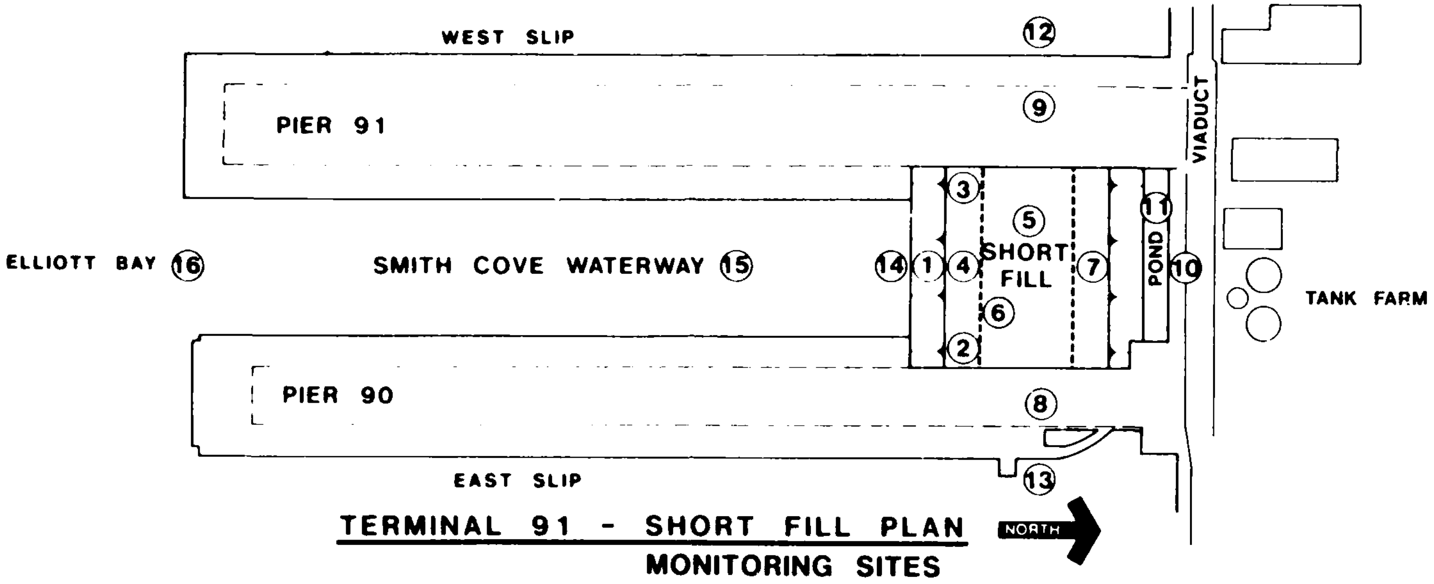
Decisions to begin remedial action to reduce the mobility of contaminants will be made based on the wells installed in the south, east, and west berms, and the seawater sample stations. Samples exceeding the threshold levels will initiate a more intense sampling sequence of three additional sample sets, taken at the affected sites every other week for 5 weeks and analyzed for the contaminants of concern. If the mean of these four samples is statistically greater than the threshold level, then discussions of the appropriate remedial action will be initiated with WDOE, EPA, and CE.

The exception to this sequence will be during the initial 6 months dewatering/consolidation phase. The dewatering is an extension of the construction phase and therefore is an accepted short-term impact (6-month) that may affect water quality at the berm face, but not beyond the dilution zone in the center waterway slip. During this period a problem with the disposal operation would be indicated by exceeding acute water quality criteria at the berm face or by exceeding chronic water quality criteria in the center slip beyond the dilution zone (100 yards south of the south crest of the south berm).

Monitoring Program

The monitoring program consists of ten well sites (Figure 6), three of which are three level well nests, and six water stations.

The frequency of sampling (Table 3) will vary during initial construction, the early consolidation phase, and the long-term monitoring phase. A suite of six metals (Cd, Hg, As, Zn, Pb, and Cu) and total organic carbon (TOC) will be sampled on a regular basis. There will also be several sampling series for the compounds on the priority pollutant list, an hourly sampling across a tidal cycle, and other research sampling.



A. WELLS:
- 1 SOUTH BERM, CENTER FACE
  - 1a INTERTIDAL (+7-3)
  - 1b MIDDLE
  - 1c BOTTOM (-30-40)
- SOUTH BERM SIDES
  - 2 EAST INTERTIDAL (+7-3)
  - 3 WEST INTERTIDAL (+7-3)
- 4 SOUTH BERM CENTER NEST
  - 4a INTERTIDAL (+7-3)
  - 4b MIDDLE
  - 4c BOTTOM (-30-40)
- 5 FILL, CENTER NEST
  - 5a INTERTIDAL (+7-3)
  - 5b MIDDLE
  - 5c BOTTOM (-30-40)
- 6 FILL, SOUTH
  - 6 INTERTIDAL (+7-3)
- 7 NORTH BERM, CENTER NEST
  - 7a INTERTIDAL (+7-3)
  - 7b MIDDLE
  - 7c BOTTOM (-30-40)
- 8. EAST (90) BERM
- 9. WEST (91) BERM
- 10 UPSTREAM

B. WATER STATIONS:
- 11. POND
- 12. 91 WEST SLIP
- 13. 89/90 SLIP
  - 90/91 SLIP
- 14 BERM FACE
- 15. DILUTION ZONE
- 16. MOUTH

Figure 6. Monitoring locations

TABLE 3. MONITORING SCHEDULE

												1 Yr												2 Yr			3 Yr			4 Yr	5 Yr
Time (Months)	1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	7	8	9	10	11	12							
	Construction Schedule																														
	Berms Fill Close Cap Dewatering																														
																		Data Review						Data Review		Data Review			Data Review	Data Review	
Sampling Sites													Monthly											Quarterly					Semi-Annually		
Wells																															
Upland	P		M			P		M	WL	P	M	O_H	M	P	M			P			M			P	M	P	M	P	M	P	M
East (P 90) Berm	P		M			P		M		P	M		M	P				P			M			P	M	P	M	P	M	P	M
West (P-91) Berm	P		M			P		M		P	M		M	P				P			M			P	M	P	M	P	M	P	M
North Berm																															
Intertidal						P		M		P	M			M							M				M		M		M		M
Middle						P		M		P	M	O_H	M	M	M			P			M			P	M	P	M	P	M	P	M
Bottom								M			M			M							M				M		M		M		M
South Berm																															
Face																															
Intertidal										P	M	O_H	M	P	26			P			M			P	M	P	M	P	M	P	M
Middle										P	M	O_H	M	P	26			P			M			P	M	P	M	P	M	P	M
Bottom										P	M	O_H	M	P	26			P			M			P	M	P	M	P	M	P	M
Vertical																															
Intertidal									WL	P	M	O_H	M	P	26			P			M			P	M	P	M	P	M	P	M
Middle										P	M	O_H	M	P	26			P			M			P	M	P	M	P	M	P	M
Bottom										P	M	O_H	M	P	26			P			M			P	M	P	M	P	M	P	M
Sides																															
E Intertidal								M		P	M	M	M	P				M			M			M	M	M	M	Discontinued if the center well is statistically similar			
W Intertidal								M		P	M	M	M	P				M			M			M	M	M	M				
Fill																															
Center																															
Intertidal										P	M	O_H	M	P	6			P			M			P	M	P	M	P	M	P	M
Middle										P	M	O_H	M	P	6			P			M			P	M	P	M	P	M	P	M
Bottom										P	M	O_H	M	P	6			P			M			P	M	P	M	P	M	P	M
South																															
Intertidal										P	M	O_H	M	P	26			M			M			M	M	M	M	M	M	P	M
Water Stations																															
Pond						P		M	WL	P	M	O_H			4			P			M			P	M	P	M	M	M	P	M
91 W	P		M			P		M		P	M							P			M			M	M	M	M	M	M	P	M
89	P		M			P		M	WL	P	M							P			M			M	M	M	M	M	M	P	M
90/91																															
WQ 1 Berm Face						WQ				P	M				13			P			M			P	M	P	M	M	M	M	M
						P																									
WQ 2 Dillution						WQ		M																							
Zone 8						P		M																							
WQ 3 Mouth	P		M			WQ		M		P	M	O_H						P			M			M	M	M	M	M	M	M	M
WQ 4 Open Water						WQ																									

P = Priority Pollutants
M = Metals + Services (Cd, Hg, AS, Zn, Pb, Cu, TOC)
WQ = Fill Operations Water Quality Service
WL = Water Levels hourly 26 hours Tidal Height/Pond Height
26 = Hourly 26 hours contaminant mobility with tide
O_H = High resolution Priority Organics (PCB's, DDT's, PAH's) and high resolution mobile metals (Cd, Hg, Zn)

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The construction period sampling will consist of two samples from the upstream and east and west pier wells. The other berm wells will be drilled as soon as possible and sampled twice during the fill operation along with the east and west pier wells.

Sampling will begin when the south berm is in place with vertical well nest and berm face well nest installed and in operation. This will be about the same time the fill is capped. This standard sampling will begin with monthly sampling for 6 months, then change to quarterly for the next 1-1/2 years, and finally to semiannually for the last 2 years. At this time there will be a major program review. If there are no significant water quality problems, the stations and analyses will be greatly reduced based on the knowledge gained.

Monitoring samples are to be taken during the last half of an outgoing tide. The water station samples will be taken on the same day, preferably on the same tide. All samples will be processed for the dissolved contaminants and not particulates. Standard field observations of time, tide well water, height, etc., will be taken at all stations. Standard analysis will be by the EPA-approved methods and to the standard minimum detection limits for seawater.

Remedial Actions

The following remedial actions represent those techniques found to be feasible in a site-specific engineering study of the possible options. Remedial actions will be implemented incrementally with follow-up testing to determine the effectiveness of each increment. The appropriate initial increment will be chosen by mutual agreement between WDOE, EPA, and CE in discussions with the Port. The actions include:

a. Maintain the pond water at mean tidal level (+6.6).

b. Pump interstitial water out of the fill and discharge to a METRO treatment plant (an interim measure). This would minimize water flow seaward through the south berm.

c. Ensure in-place chemical stabilization. This is a new technique, and would be used only if available information shows it to be acceptable for a specific contaminant.

d. Install a slurry wall in the north and south berms.

e. Install a slurry wall in the east and west piers.

f. Remove the contaminated dredged material from between the berms.

CONCLUSIONS

The Terminal 91 Short Fill project is providing a demonstration that nearshore unlined contained disposal of mildly contaminated dredged material is a viable option, even with increased concern over contaminant mobility. Because of the factors affecting mobility, a site-specific analysis must be performed.

The approach taken by the Port is a reasonable route to obtaining the permits to begin construction when generic information indicates, but does not give specific assurance, that the project will meet newer, more advanced water quality concerns. Using computer modeling in a site-specific preproject analysis in conjunction with monitoring and agreed-upon remedial action steps gives the greatest assurances that containment goals will be met and the environment protected. This is an expensive approach and only cost-effective where the site-specific analysis shows a high probability that the water quality criteria will be met.

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TEST RESULTS OF SEDIMENT REMOVAL IN ISE BAY

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ABSTRACT

The 5th District Port Construction Bureau, Ministry of Transport, has been conducting studies to obtain a feasible method for sediment removal in Ise Bay. These studies were conducted in Mikawa Bay to ascertain the effect and method of sediment removal since 1980. These studies were conducted prior to carrying out the sediment removal in Ise Bay, one of the major works for improvement of the ocean environment.

This paper summarizes a study of the effects of sediment removal, in terms of the amount of released organics and nutrients, and recovery of biotas and other findings in dredging tests performed in 1980 in highly contaminated sediments.

INTRODUCTION

Mikawa Bay, a part of Ise Bay, is backed by the City of Nagoya and is situated at the center of the Japanese Archipelago. Mikawa Bay is a closed inland bay bounded by the Chita Peninsula and the Atsumi Peninsula (Figure 1).

The 600-sq-km Mikawa Bay is shallow with an average depth of 9.2 m and a very narrow (about 10 km) mouth. These conditions do not favor the interchange of seawater, but do favor eutrophication.

First class rivers such as the Yahagi River and the Toyo River flow into the sea. The inflow loads from these rivers to the whole area of the Mikawa Bay included approximately 69 tons/day of chemical oxygen demand (COD), 35 tons/day of total nitrogen (T-N), and 3.0 tons/day of total phosphorus (T-P) in 1979. The Aichi Prefectural Government established the total amount regulation program with regard to COD in Ise Bay and Mikawa Bay in 1980, and has been implementing the program to reduce the inflow loads.

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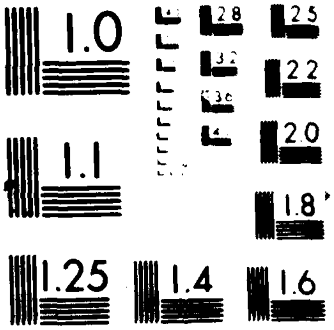
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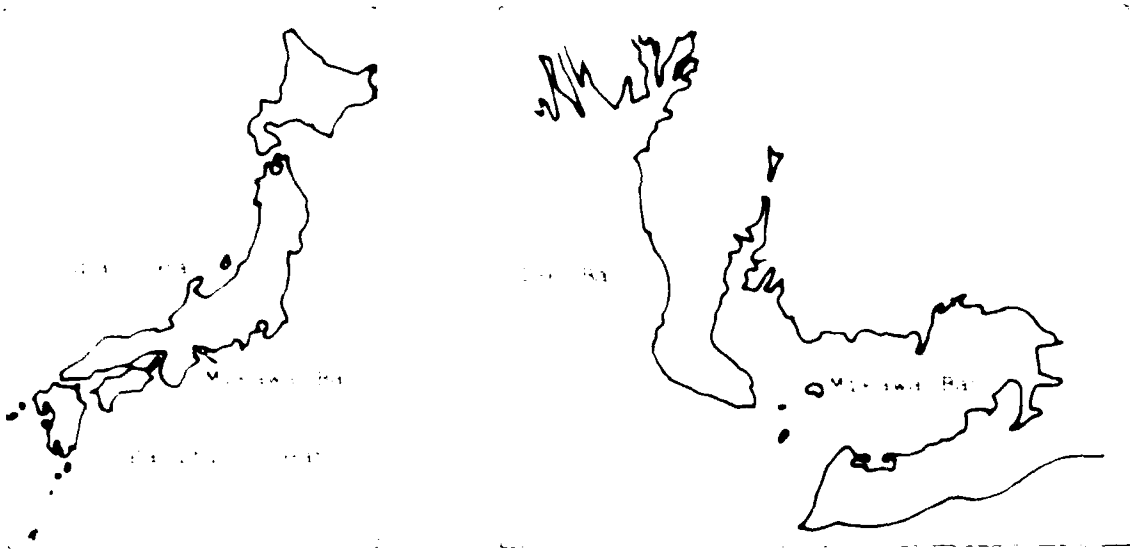


Figure 1. Location of Mikawa Bay

However, as seen in Figure 2, in spite of the program to reduce COD, seawater quality has not improved much since the environmental quality standard is continually exceeded.

The frequency of red tide is shown in Figure 3, and the fishery is being affected badly. Figure 4 indicates that organic sediments with a concentration of over 40 mg g•D.S. of COD is accumulating at the central part of Mikawa bay. Since sediment COD affects the occurrence of red tides, sediment removal was considered.

Flow conditions in Mikawa Bay are clockwise and steady in the inner part of the bay and counterclockwise and steady in the central part of the bay. At the central part of these two steady flows, highly concentrated organic sediment accumulates and is coincident with the retention area.

In order to improve water quality in Mikawa Bay where eutrophication has advanced considerably, it is necessary to reduce inflow loads and to minimize nutrient release. To do this, The 5th District Port Construction Bureau, Ministry of Transportation, is presently examining sediment removal projects.

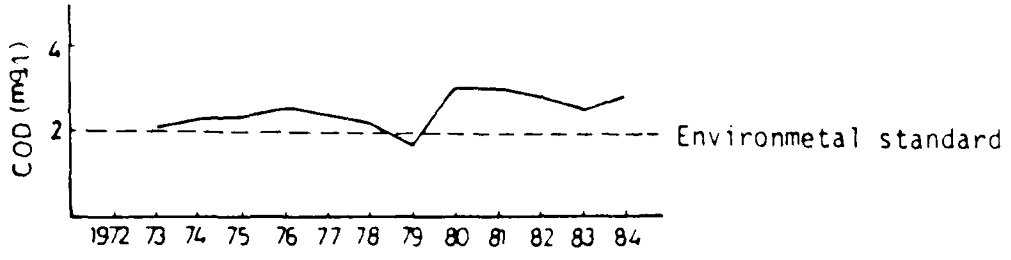
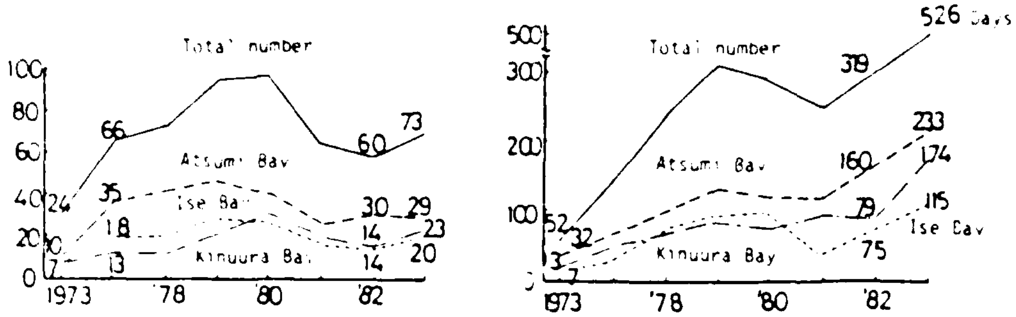


Figure 2. Annual variation of COD



a. Variations of red tide occurrence

b. Numbers of days of occurrence

Figure 3. Red tide frequency of occurrence and numbers of days of occurrence in Ise and Mikawa Bays (Note: Total number in 1979, 1981, 1982, and 1983 includes one case that occurred in the open sea of Atsumi)

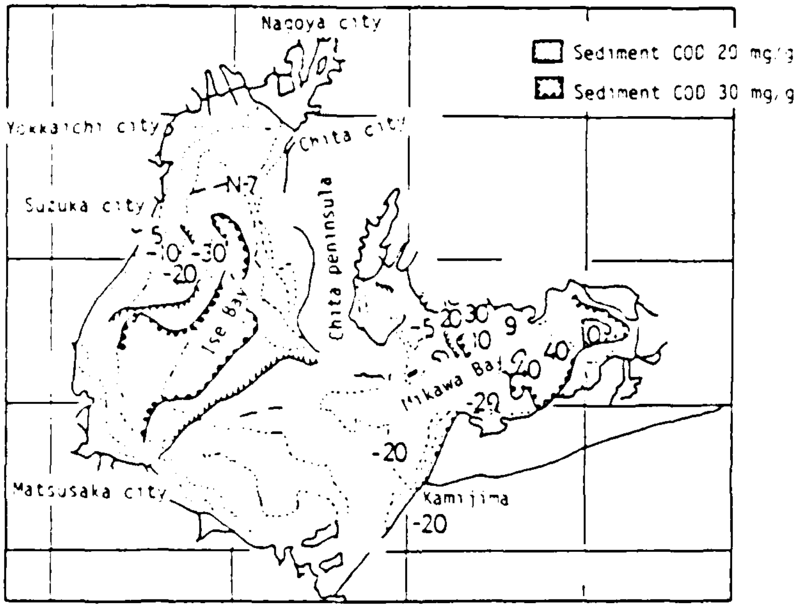


Figure 4. COD distribution of sediment in Ise Bay

One of the projects involved sediment removal at the center of Mikawa Bay, where substantial organic sediment accumulates. A general description of this part of the study is given below.

SYSTEM OF STUDY

The system of study is shown in Figure 5.

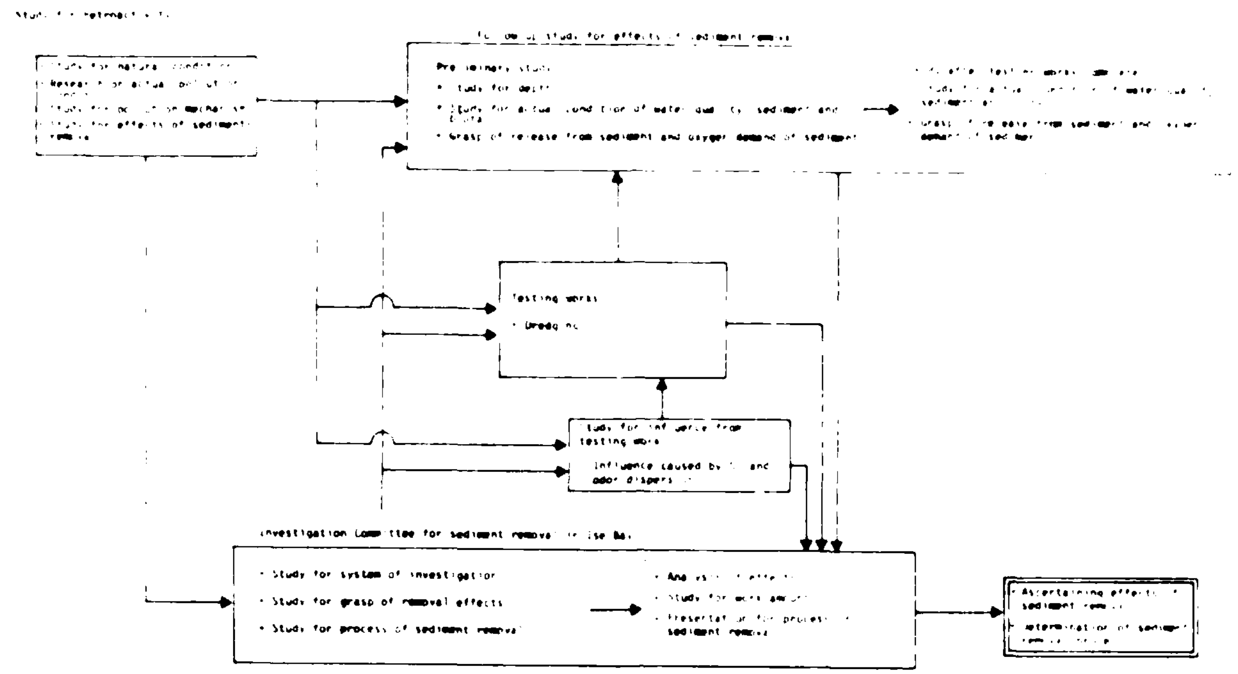


Figure 5. System of study

TEST SITES

The sites are located as shown in Figure 6. At a station 6 km away from Higashi-hazu Harbor in Mikawa Bay, sediment removal was performed by pump-type dredge in June 1980. Sediment removal was conducted in the area extending 100 m south and north, 100 m east and west, and with a depth of 0.6 m. The outer circumference was dredged in a deep ditch in order to prevent settling back of sediment from surroundings. Plane and cross-sectional views are shown in Figure 7.

See Appendix A for details of the study.

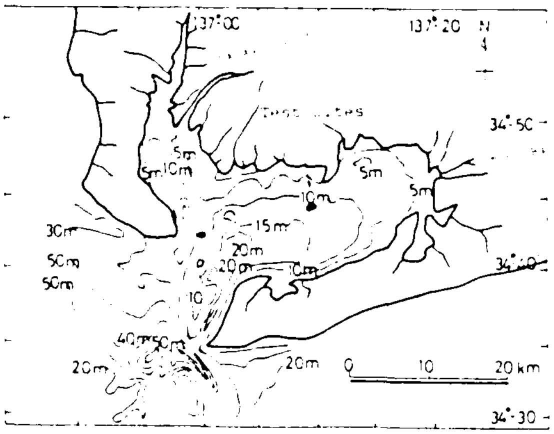


Figure 6. Location of test sites

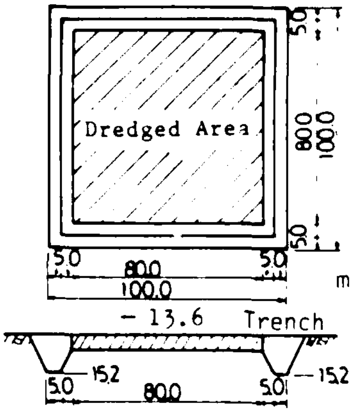


Figure 7. Plane and cross-sectional view of dredged area

FOLLOW-UP STUDY OF SEDIMENT-REMOVING EFFECTS

The study conducted in 1980 and 1981 is discussed below. Some items of the study have been continued.

Purpose of Study

The study was performed to determine the effects of sediment removal on the conditions of organic and nutrient release from sediment, oxygen demand, and recovery of biota.

Sampling Stations

Seven sampling stations were set up inside and outside the dredged area (Figure 8); stations No. 1 to No. 3 are located within the dredged area, and No. 4 to No. 7 are outside the area of dredging.

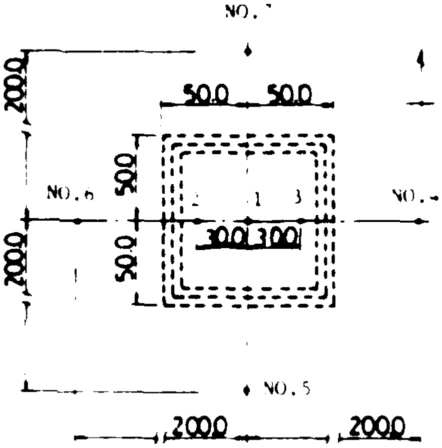


Figure 8. Sampling stations

Contents of Study

A list of the items studied is shown in Table 1.

Results of Study

Results of the study are itemized below:

a. Conditions of flow.

(1) Tidal current. The tidal current of the upper and lower water layers runs toward the inner part of the bay during rising tide and toward the mouth of bay during ebb tide.

(2) Steady flow. Steady flow (daily average flow) varies widely with flow direction and velocity. The maximum velocity in the upper water layer reached 20 cm/sec during both examinations and can temporarily run faster than the tidal current.

TABLE 1. STUDY ITEMS

[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]		[illegible]
[illegible]	[illegible]	[illegible]		● [illegible] sediment [illegible] ● [illegible]
[illegible]	[illegible]	[illegible]		● [illegible] sampler
[illegible]	[illegible]	[illegible]	Sediment [illegible] water [illegible] site [illegible] days	● [illegible] sampler
[illegible]	[illegible]	[illegible]		● [illegible] sampling [illegible] sediment ● [illegible]
[illegible]		[illegible]	● [illegible] ● Measuring [illegible] times [illegible] days	● Release test equipment Test site
[illegible]	[illegible]	[illegible]	● [illegible] test[illegible] ● Measuring [illegible] times [illegible] days	● Release test equipment Test site
[illegible]		[illegible]	● [illegible] conditions. Test started at saturation ● Measuring [illegible] times [illegible] days	● [illegible] test equipment
[illegible]	[illegible] Intermediate layer Bottom [illegible]	[illegible]	[illegible]	● [illegible] sampler
[illegible]	[illegible]	[illegible] water temperature	[illegible] water	Aanderaa current meter, Bergen [illegible]

[illegible] release test equipment [illegible] water [illegible] sediment.

b. Sediment (Figure 9).

(1) Organic pollutant index. The examination (in June) prior to the test indicated that the values of sediment between the outside and inside area were similar; however, after sediment removal a reduced amount of organics and nutrients inside the area was noted. Subsequently, the sediment indicated almost constant values until the third study in December after the test. The fourth study in February indicated a slight increase, probably a seasonal variation, [illegible] the difference between the outside and inside area was ma[illegible].

(2) Oxidation - reduction potential of sediment. The sediments outside the area were highly reduced when compared with the dredged inside area. Inside the area the oxidation reduction potential (ORP) is reduced during periods from the first study in July to the fourth study, and a tendency of weak reduction of sediment is shown. However, seasonal variations of T-S and sulfate-reducing bacteria are small.

(3) Physical properties. The sediment consisted of fine silt and clay, but little difference is seen in unit volume weight and moisture content ratio.

c. Benthos (Figure 10).

(1) Seasonal variations. Macrobenthos showed a noticeable reduction in wet weight, number of species, and number of individuals in the second study in September, compared with the study in June before the test and the first study in July. *Theora fragilis* belonging to Mollusca, predominant up to that time, disappeared and *Ancistrosyllis hanaokai* and *Prionoskio cirrifera* of Polychaeta now predominate. However, an increase in numbers of species and number of individuals occurred in the third study in December and the fourth study in February. This may indicate a recovery of the biota.

(2) Difference between the outside area and the dredged inside area. The first study in July after the test showed a noticeable reduction in microbenthos in the dredged inside area. This may be a result of the influence of testing. However, the second study in September, the third study in December, and the fourth study in February after the test showed a slight increase in the microbenthos of the inside area compared with the outside.

d. Pore water concentrations of P, N, and COD.

(1) Vertical distribution (Figure 11). Generally, the concentrations in the pore water were lower in the dredged area. Seasonal variations tend to be high in September and low in December; however, N and P in the area of sediment removal have been depressed since July.

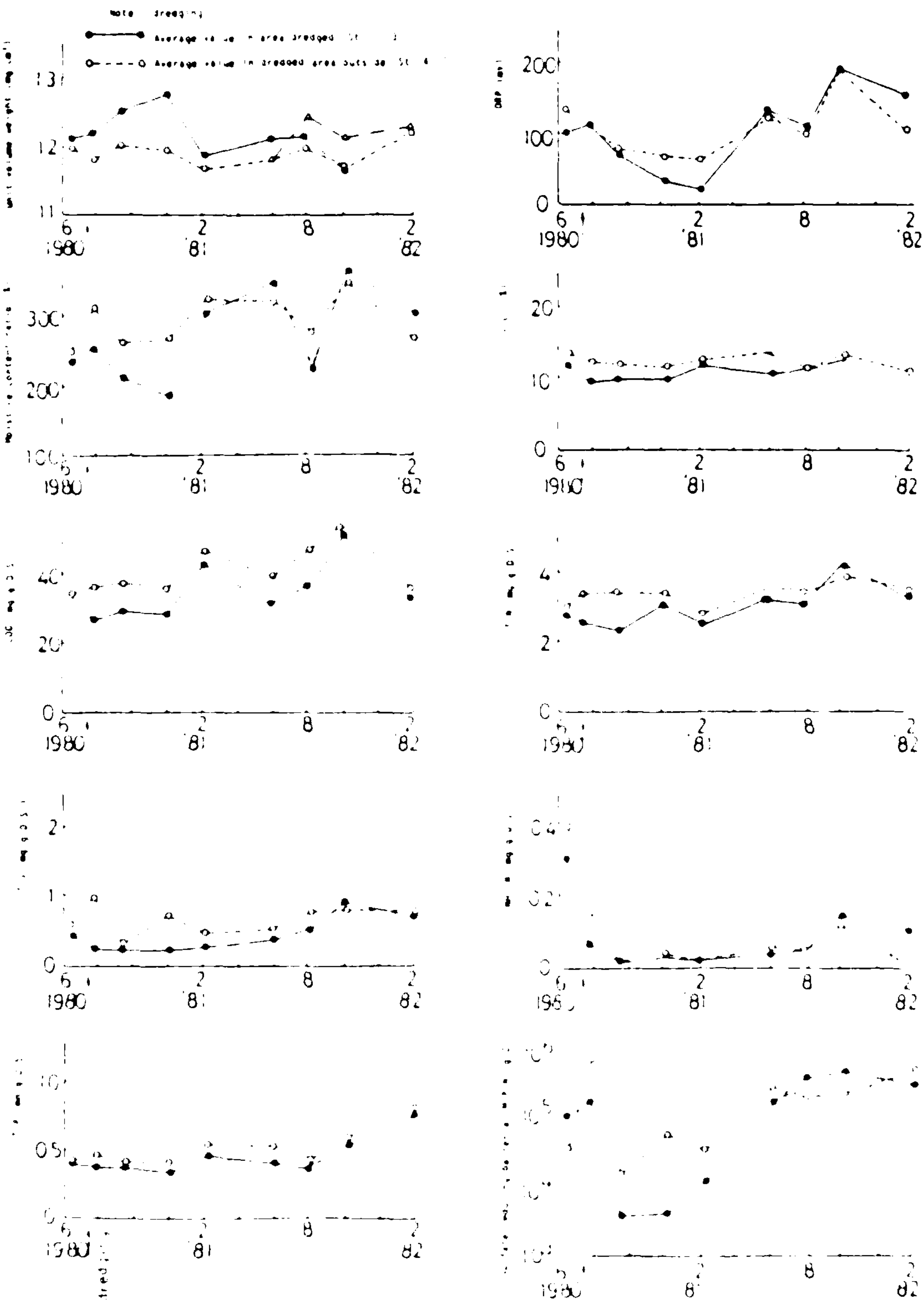


Figure 9. Seasonal variations in sediment composition

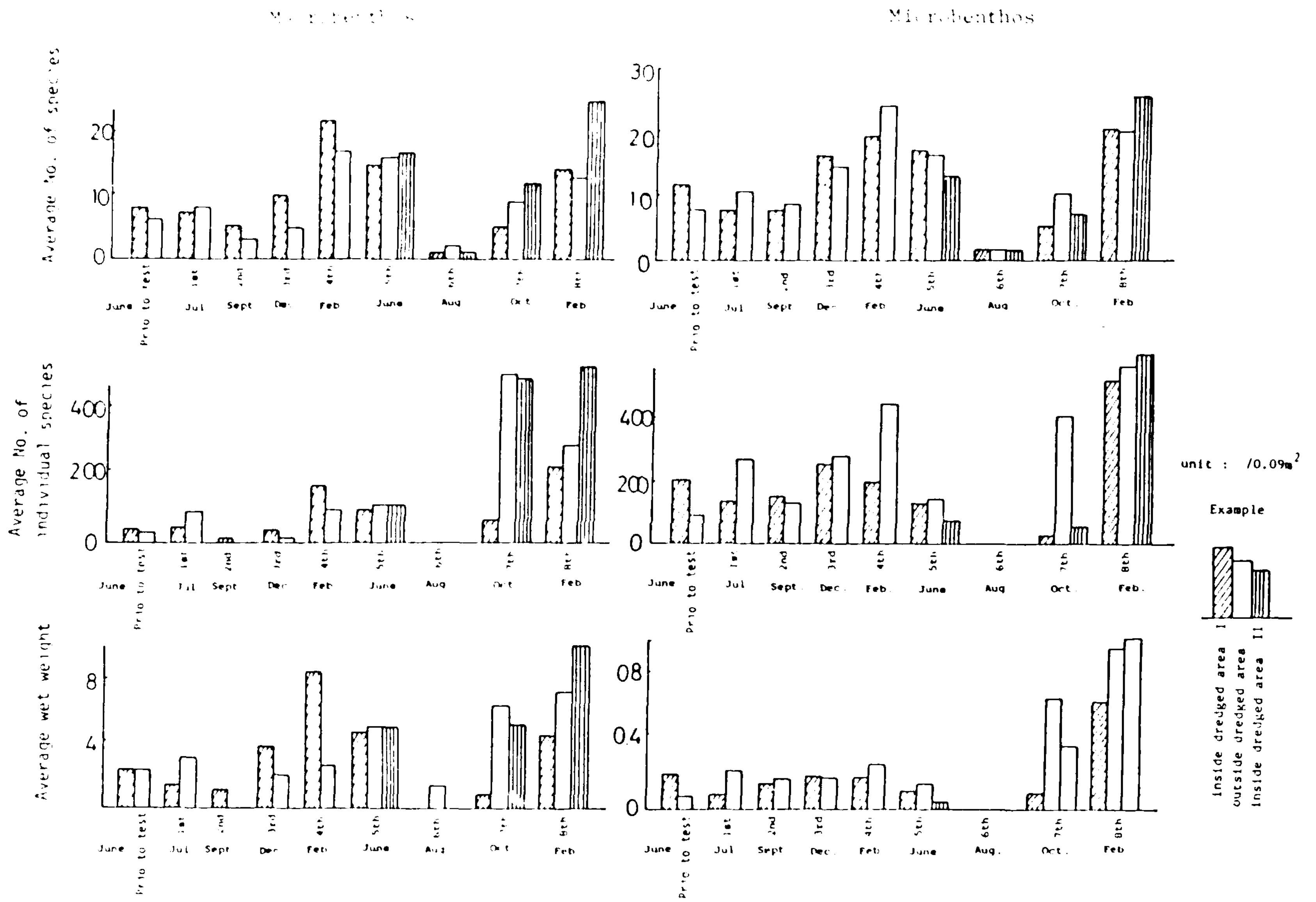


Figure 10. Seasonal variations in benthos

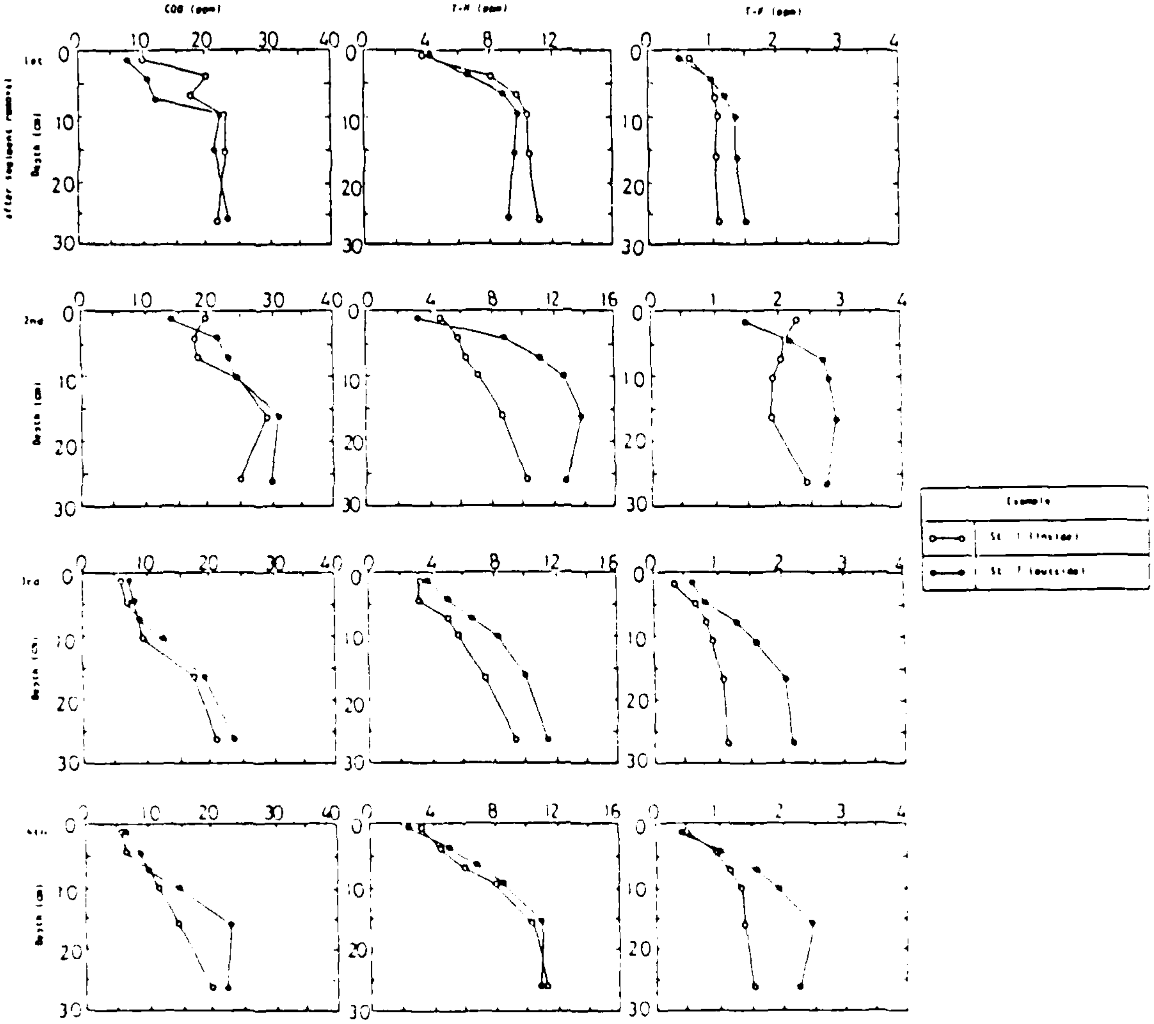


Figure 11. Vertical distribution of pore water

(2) Decomposition rate of organics in sediment (Figure 12). The pore water daily variation of organic concentrations was studied for 30 days by incubating the sediment (0-30 cm) with water temperature of the testing stations. The decomposition rate of organics inside the area of removal was slower than that outside the area in the second to fourth studies. This may indicate that hard-to-decompose organics are prevalent in those study areas.

e. Release test (Figure 13). Release test equipment was installed inside and outside the dredged area to determine variations in release flux. Both N and P showed higher values in the outside area in the first study in July and the third study in December. The second study in September indicated a very low release flux, and considerable differences between two areas were not seen.

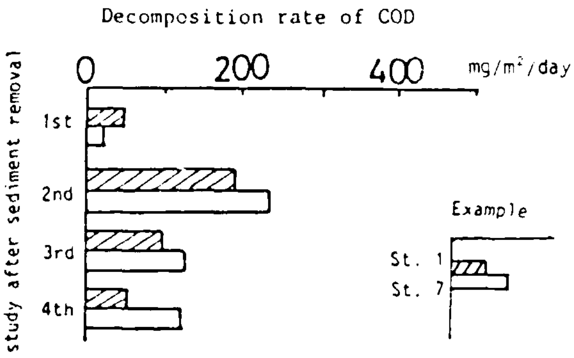


Figure 12. Sediment decomposition rate

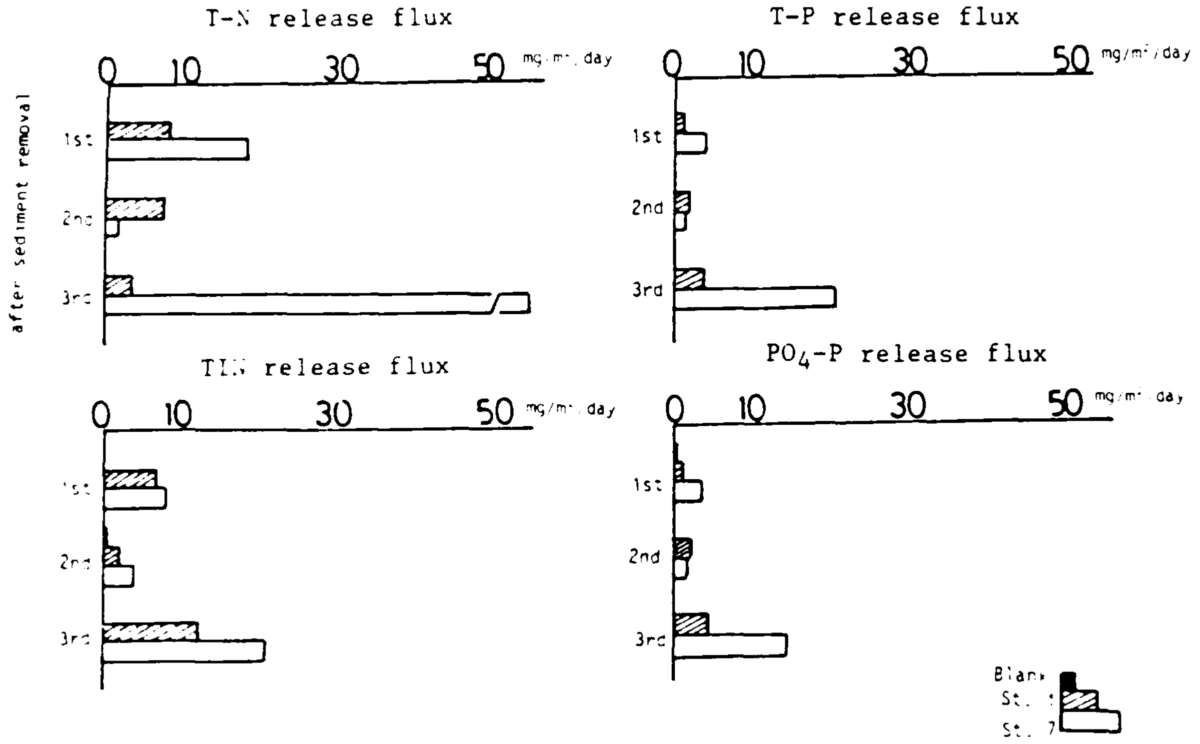


Figure 13. Release flux at stations No. 1 and No. 7

f. Dissolved oxygen demand test (Figure 14). The dissolved oxygen (DO) demand of seawater in the inside and outside area was determined on the seabed. Results obtained from each study showed the value in the inside area to be lower than the outside area.

g. New deposition. A sampler was installed in the water to determine deposition rate since the tests were conducted. The first study in July resulted in 0.2 to 0.3 cm/year at a layer 5 m above the seabed. Results of the third study in December showed the value to be one order greater than that of the first study. This resulted from the influence of scattering.

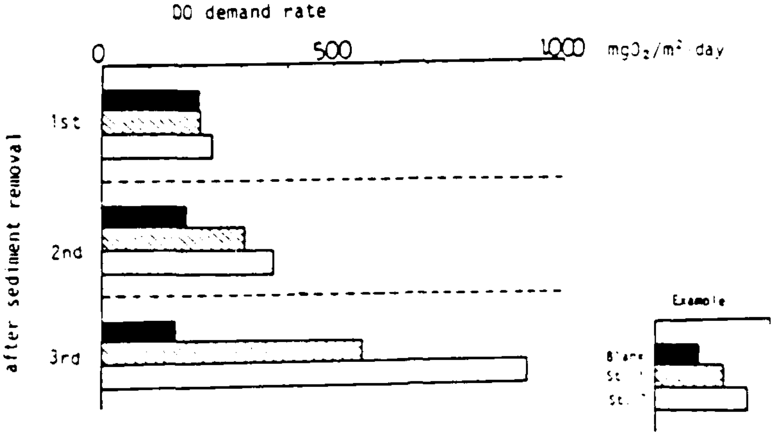


Figure 14. DO demand rate at stations No. 1 and No. 7

CONCLUSIONS

In order to understand the effects of sediment removal on ocean environmental improvement, a dredging project was conducted in Mikawa Bay, along with a follow-up study on the effects of the removal. Conclusions ascertained one year after the test was completed are as follows:

a. The organic and nutrient content in sediment was reduced.

b. The pore water index was low in the dredged inside area.

c. Decomposition rate of organics in sediment tended to be higher in the outside area.

d. For benthos, the numbers of individuals, number of species, and wet weight increased considerably inside the area as compared with the outside area.

However, the results of a longer term study show that the difference between the outside area and the dredged inside area is diminishing, and can be considered ineffective for long-term removal effects. This results from redeposition of sediment from surroundings areas.

From the test and follow-up study, it was determined that the dredging would be effective for 1 year; therefore, sediment removal would be considered one of the major methods for the ocean environmental improving processes.

APPENDIX A: STUDY DETAILS

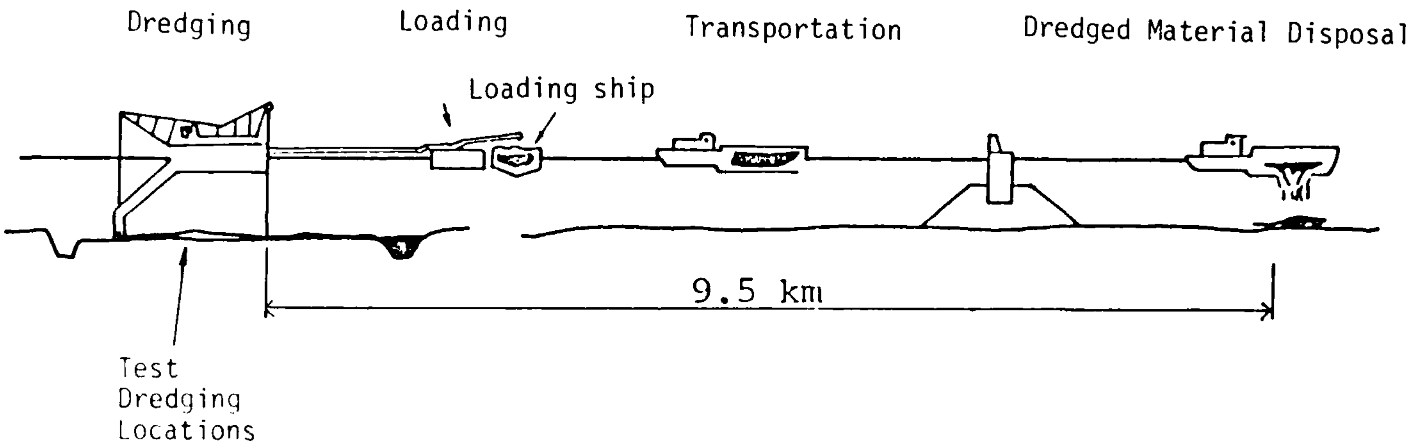


Figure Al. A general sketch for process

TABLE A1. ANALYTICAL RESULTS OF SEDIMENT

Items		Analytical Results		Criteria	
Cyanides	(mg/ℓ)	0.1	(mg/ℓ)	Less than 1	(mg/ℓ)
Cadmium	(mg/ℓ)	0.005	(mg/ℓ)	Less than 0.1	(mg/ℓ)
Arsenic	(mg/ℓ)	0.02	(mg/ℓ)	Less than 0.5	(mg/ℓ)
Alkaly mercury	(mg/ℓ)	Should not be detected		Should not be detected	
Total mercury	(mg/ℓ)	0.0005	(mg/ℓ)	Less than 0.005	(mg/ℓ)
Lead	(mg/ℓ)	0.05	(mg/ℓ)	Less than 1	(mg/ℓ)
Sexivalent chromium	(mg/ℓ)	0.02	(mg/ℓ)	Less than 0.5	(mg/ℓ)
Phosphorus	(mg/ℓ)	0.1	(mg/ℓ)	Less than 1	(mg/ℓ)
PCB	(mg/ℓ)	0.0005	(mg/ℓ)	Less than 0.003	(mg/ℓ)

- Construction period and volume of dredged soil

Construction: June 5 - July 31, 1980

Volume of soil: 11,990 m^3 (Area 10,000 m^2)

- Main capacity of dredge

Depth of dredging: 3.5 - 20.0 m

Distance for transportation: Barge loading - 3,000 m

Dredging capacity: 150 to 400 m^3/hr

Measure of ship (L × B × D): 45.60 m × 13.50 m × 3.30 m

Load displacement: 1,500 tons

Full draught: 2.20 m

Main generator: Diesel [illegible] KVA [illegible] units

Sub generator: Diesel [illegible] KVA

Swing winch: 30 [illegible] kw

Full speed: [illegible] m/min

[illegible] winch: [illegible] kw

Sludge collector

Sludge collector driving unit:	DC 110 kW Bearing revolution: 3 - 15 rpm
Open/shut shutter:	Oil pressure, remote control type
Ladder control:	Oil pressure, remote control type
Gas purge:	Ejector

Dredge pump

	Ladder Pump	In-Ship Pump
Type:	Volute	
Rated power:	150 PS	
Pumping capacity:	800 - 2,000 m^3/h	800 - 2,000 m^3/h
Head:	10 - 20 m	20 - 50 m
Revolution:	490 - 700 rpm	200 - 310 rpm
Motor:	AC 110 kW	D 2,000 PS
Pipe size:	Suction/delivery	400 mm/400 mm
Check valve:	Oil pressure type	

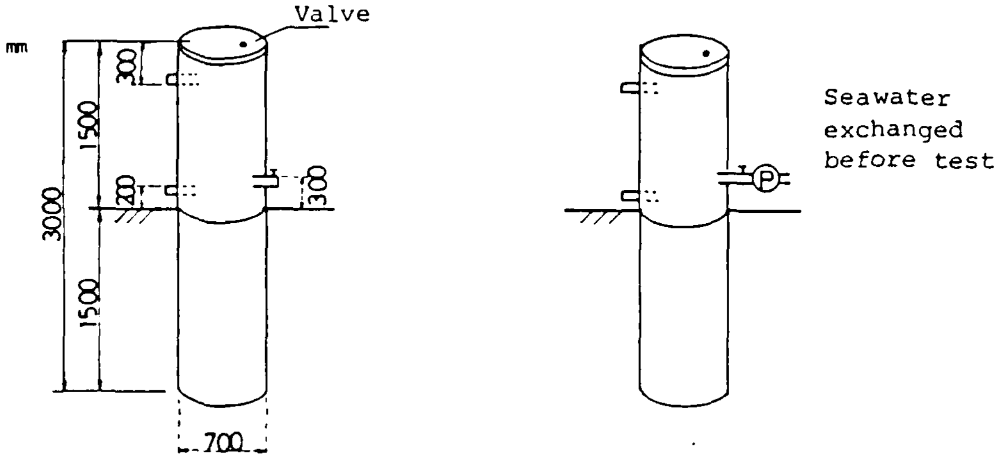
- Operation monitoring

TV monitoring system:	Submerged TV camera, Monitoring video unit, Lighting unit
Turbidity monitoring system:	Analog meter with recorder
Depth monitoring system:	Analog meter with recorder

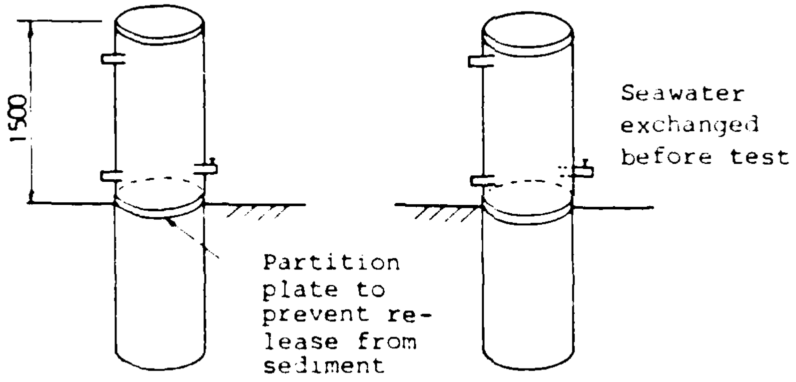
- Composition of ships

Dredge	D 2,250 PS	1 vessel
Wincher	15 tons	1 vessel
Loader	23 m × 11 m × 2.25 m	1 vessel
Transportation	2,200 m^3	1 vessel
Pusher	D 2,600 PS	1 vessel

(1) With Caps



(2) With Caps (blank)



(3) Without Caps

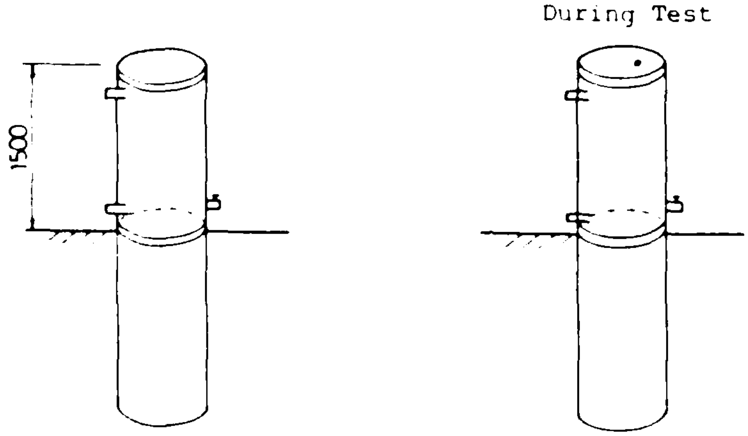
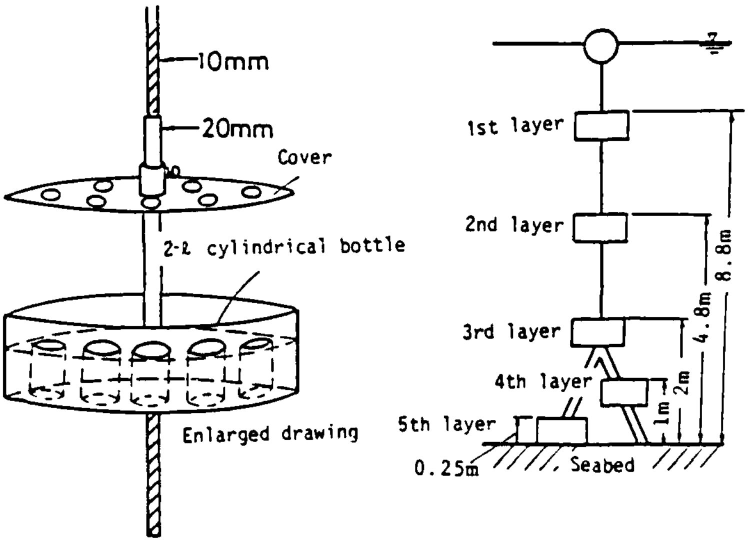
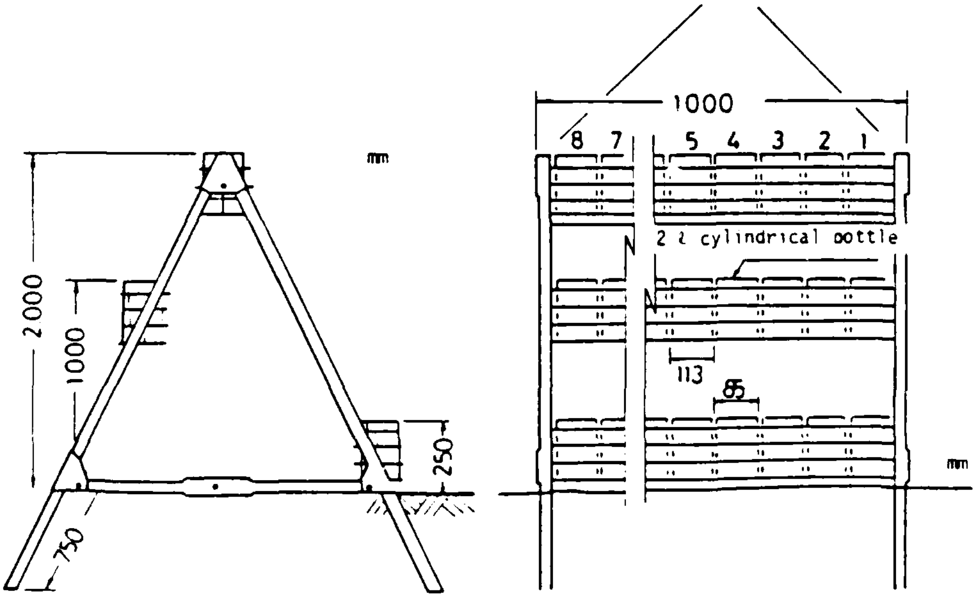


Figure A2. Release test pipes



a. Upper and intermediate layer (1 - 2 layer)



b. Lower layer (3 - 5 layer)

Figure A3. Installation of newly precipitating matter sampler

EVALUATION OF DREDGING AS A REMEDIAL TECHNOLOGY FOR THE COMMENCEMENT BAY SUPERFUND SITE

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ABSTRACT

Recent studies in Puget Sound have demonstrated a link between elevated concentrations of chemicals in bottom sediments and serious diseases (e.g., liver carcinomas) in bottom-dwelling fish. These and other studies have led to intensified efforts to identify contaminant sources and to clean up problem sediments. Remedial action at the Commencement Bay Nearshore/Tideflats Superfund Site, in the State of Washington, USA, is expected to include dredging and disposal of contaminated sediments from the bay's industrialized waterways. In addition, intensive navigation activities often require handling contaminated sediments. In response to this close interaction between Superfund cleanup investigations and navigation needs, the Corps of Engineers assisted the State of Washington by conducting an analysis of alternative dredging methods and equipment, disposal methods and sites, and site control and treatment practices for contaminated sediments derived from the bay.

This paper reviews some of the conclusions of the alternatives study regarding preferred dredging and disposal practices for sediment contamination found in Commencement Bay. Factors influencing contaminant mobility and an example application of chemical partitioning coefficients to a screening assessment of contaminated sediments are discussed. Key considerations in selecting appropriate equipment and methods, and the preferred methods for various classes of contaminants, are presented. Subsequent work by the Corps to develop decisionmaking guidance for dredged material management is briefly reviewed.

The Commencement Bay Superfund evaluations served to highlight a number of issues related to navigation project planning, acceptable disposal practices, and long-term management of contaminated sediments. The influence and consequences of these issues to dredged material management in Puget Sound are discussed.

INTRODUCTION

In the late 1970s, Puget Sound (Figure 1) was generally perceived as an ecologically rich and diverse marine water body nestled between two snow-capped mountain ranges to the west and east. The region's abundant precipitation and snowmelt swelled the numerous rivers that were the source and destination of the region's other famous resource, salmon. "Water quality" problems of the 1950s were improving. "Pollution" in Puget Sound referred to the aroma emanating from the occasional pulp mill and smelter stacks of the Northwest.

This perception abruptly shifted with the identification of chemical pollutants concentrated in sediments and biota at numerous locations in the Sound. While highest concentrations of these chemicals have been identified near major population and industrial centers, contaminants also have been identified in marine sediments and organisms in areas previously thought to be clean. Many of the chemicals accumulated in bottom-dwelling fish, and high levels of certain toxic chemicals in the urban bays were linked to serious diseases (e.g., liver carcinomas) of English sole (*Parophrys vetulus*) and other demersal fish species (Malins et al. 1984). These discoveries increasingly focused agency and public attention on the health of the Sound, raising questions regarding past practices and decisionmaking criteria.

In October 1981, the US Environmental Protection Agency (EPA) identified the Commencement Bay area of Tacoma, Washington, as one of the top priority sites on the national list of "Superfund" (Comprehensive Environmental Response, Compensation and Liability Act of 1980) hazardous waste sites. Site priority was based on widespread contamination in the soils, sediments, and biota of the bay resulting from airborne emissions of arsenic and cadmium from a local copper smelter, leaching of toxic metals from the large quantities of smelter slag deposited along the marine shoreline at the plant and as fill in the intertidal areas of the bay, and organic contaminants discharged by historical and current industrial activities (including four chemical manufacturing plants, two petroleum refineries, a pulp mill, a municipal treatment plant, and numerous waste disposal sites). Over 900 individual organic compounds have been detected in the bay sediments (Malins et al. 1984).

Remedial investigations of the site, performed with the State of Washington Department of Ecology (WDOE) as lead agency, focused on defining the extent of contamination in water, sediments, and biota; determining the contamination threat to health, welfare, and the environment; and identifying contamination sources and feasible remedial actions.

A major output of the remedial investigations has been the identification of a "problem" sediment that is sufficiently hazardous in place to warrant Superfund remedial action. A "problem" sediment was defined by an action threshold level that considers variables such as degree of sediment chemical contamination, sediment toxicity, contaminant bioaccumulation, fish pathology, and benthic macroinvertebrate community structure. Areas exceeding predetermined action levels were evaluated in terms of appropriate remedial response: (a) contaminant source control only, (b) sediment removal/containment only, (c) source control and sediment removal/containment, and (d) areas that are expected to recover to levels below threshold by natural processes. Dredging is a likely remedial method for problem groups (b) and (c). In addition,

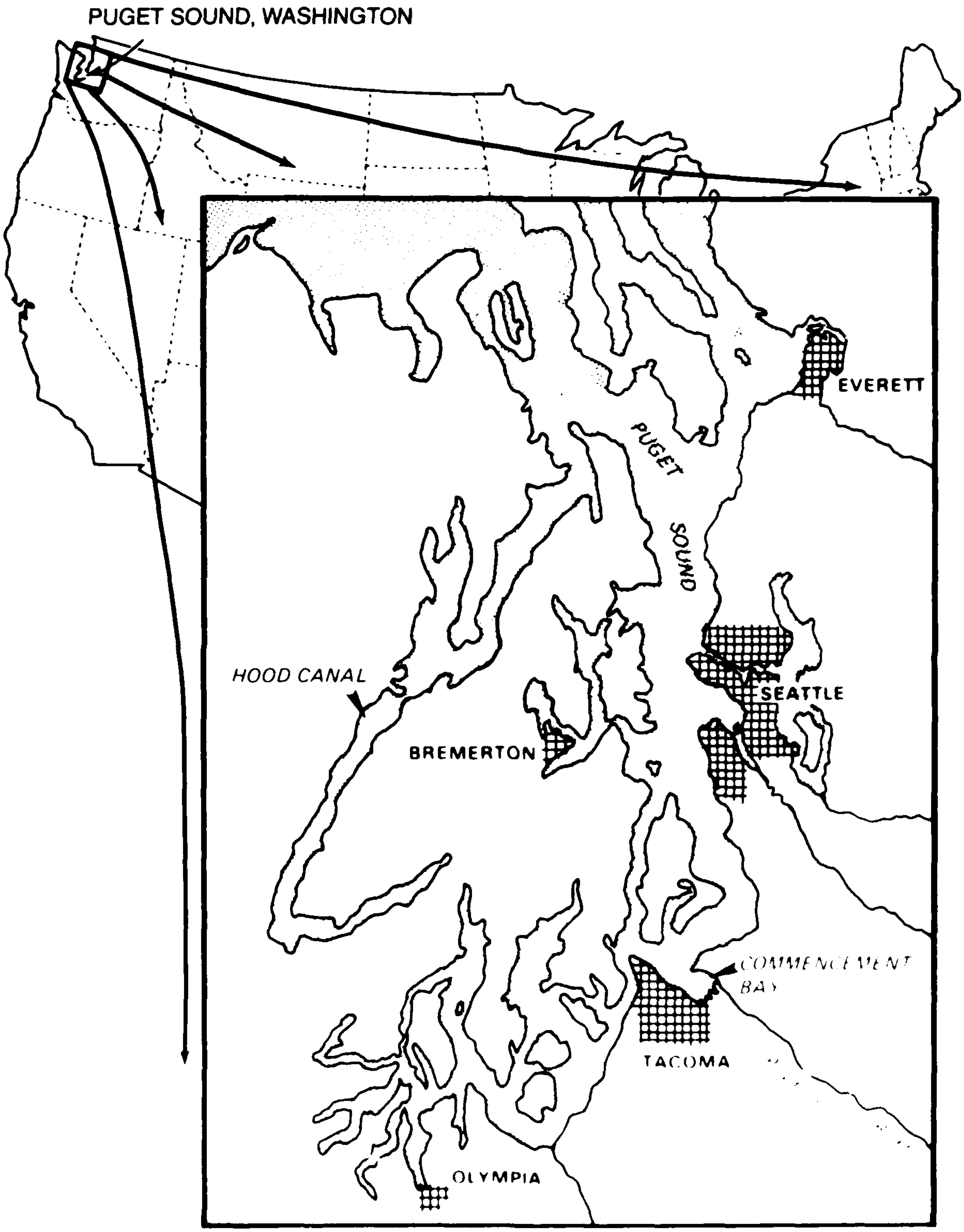


Figure 1. Puget Sound

intensive navigation in the Bay's industrial waterways routinely requires removal of contaminated sediments that may fall below (and occasionally above) the action threshold level.

By agreement with WDOE, the Seattle District, Corps of Engineers, analyzed alternative dredging methods and equipment, disposal methods and sites, and site control and treatment practices that may be applied to Commencement Bay sediments (Phillips, Malek, and Hamner 1985). Concurrently, the US Army Engineer Waterways Experiment Station (WES), working with the Seattle District, developed a decisionmaking framework to specify testing requirements and methods and provide guidance on interpreting and evaluating test results to determine whether management restrictions are needed to dredge or dispose of the material (Lee et al. 1985). This paper discusses select results of the alternatives analysis, briefly overviews the decisionmaking framework, and identifies a number of management issues associated with dredging and disposal of contaminated sediments.

ASSESSMENT OF DREDGING AND DISPOSAL METHODS

Primarily an assessment of existing information, the alternatives analysis for Commencement Bay was based on three evaluation factors: (a) cost of each alternative, (b) degree of contaminant confinement and release resulting with each alternative, and (c) considerations and limitations specific to each alternative (e.g., equipment and site availability, method efficiency, equipment depth limitations, sociopolitical concerns, and other indicators of practicability). Characteristics, operational considerations and control, and equipment considerations and modifications for dredging contaminated sediments were described for each dredging alternative.

The two basic types of dredges addressed, hydraulic and mechanical, were evaluated in terms of contaminant loss during dredging. Special-purpose dredges (many developed in Japan) that have been designed for contaminated sediments were included in the hydraulic dredge category. Three generic disposal methods for contaminated sediments, confined aquatic, upland, and nearshore, were examined in terms of their strengths and weaknesses in successfully containing various contaminant classes.

Chemical constituents associated with sediments are unequally distributed among different chemical forms and sediment phases depending on the physical-chemical conditions in the sediments and the overlying water. When contaminants introduced into the water column become fixed into the underlying sediments, they rarely if ever become part of the mineral structure of the sediment. Instead, these contaminants remain dissolved in the sediment interstitial water (pore water), become sorbed to the sediment ion exchange portion as ionized constituents, form organic complexes, and/or become involved in complex sediment oxidation-reduction reactions and precipitations. Dredging of contaminated sediments causes short-term loss of contaminants from gas, interstitial water, or solid phases. The dredging method also influences long-term contaminant losses at the disposal site.

Contaminated confinement and release during dredging and disposal were addressed in terms of three generalized contaminant classes: volatile, soluble, and sediment-bound. These classes refer to the phase (gas, liquid, solid) for which a contaminant has greatest affinity and in which relatively higher concentrations might be expected during the dredging and disposal process. The authors recognized that contaminants will partition and move between these phases to differing extents based on a number of factors. The classification was used solely as a basis for comparing dredging and disposal alternatives using existing data and information. No specific list of contaminants was developed.

Dredging

In Puget Sound, dredging normally occurs for one of three objectives: (a) maintenance or construction for navigation purposes (the most common); (b) projects where the dredged material is sought as fill; and (c) for sediment cleanup efforts. While much of the marine sediments in Puget Sound are relatively clean, dredging of contaminated sediments is common in the urban embayments. Dredging objectives for Commencement Bay include: (a) Federal channel dredging about every 10 years; (b) a large (2 million cubic yards) waterway fill project proposed for port development; and (c) Superfund contaminated sediments removal.

Characteristics of the dredges evaluated are summarized in Table 1. Evaluated on the basis of sediment resuspension at the dredge site, special-purpose hydraulic dredges (e.g., the Oozer dredge) produce less resuspension than conventional hydraulic dredges, and, with the exception of hopper dredge overflow, conventional hydraulic dredges produce less resuspension than conventional mechanical dredges. While mechanical dredges do not produce a slurry, conventional hydraulic dredges produce abundant slurry water; special-purpose dredges fall somewhere in between. In addition, hydraulic dredges produce less resuspension of solids at the dredging site and have a higher removal efficiency for liquids and solids than do mechanical dredges. However, using a hydraulic dredge to obtain high removal efficiency at the dredging site involves a trade-off requiring consideration of effluent quality and potential treatment at the disposal site.

There is no hard and fast rule to selection of the "correct dredge." Different dredging methods appear more appropriate for certain contaminant classes. If concern exists for loss of volatile contaminants during dredging, mechanical dredges perform better than hydraulic dredges. For sediment-bound contaminants, removal efficiency is greater using hydraulic dredges than mechanical dredges and appropriate technology exists to control solids at the disposal site. Soluble contaminants can be removed more efficiently by a hydraulic dredge, but are difficult to control at the disposal site and may require treatment of the effluent. Mechanical dredges using a watertight bucket offer an alternative to the water treatment concern by eliminating or reducing the slurry, but could necessitate double-handling for some disposal options.

A variety of modifications to dredging equipment and operation are appropriate when dredging contaminated sediments. Dredge equipment modifications that appear most promising for hydraulic dredges include the walking spud, ladder pumps, in-line production meters, and degasser collection and treatment (in the dredge furnace) system. Use of large, watertight buckets appears promising for mechanical dredges. Operational modifications for hydraulic cutterhead dredges include minimizing cutter revolution speed, controlling swing speed, and not overdigging the maximum cut depth. For mechanical

TABLE 1. SUMMARY OF DREDGE OPERATING CHARACTERISTICS

Dredge Type	Percent Solids in Slurry or Transported Material by Weight*	Dredging Cost $/cu yd**	Approximate Range of Production Rates cu yd/hr	Dredging Depths ft Minimum	Dredging Depths ft Maximum	Vertical Dredging Accuracy ±ft	Horizontal Dredging Accuracy ±ft
Bucket	up to 100%	1.60	30 - 600	0†	150††	2	1
Suction	10 - 15%	1.50	25 - 5,000	5 - 6	50 - 60‡	1	2 - 3
Dustpan	10 - 20%	1.50	25 - 5,000	5 - 14	50 - 60‡	1/2	2 - 3
Cutterhead	10 - 20%	1.50	25 - 5,000	3 - 14	12 - 65‡	1	2 - 3
Hopper	10 - 20%	1.39	500 - 2,000	10 - 28	65‡	2	10
Mudcat	10 - 40%	1.50	60 - 150	1-1/2	15	1/2	1/2
Pneuma	up to 80%	1.05-3.05	60 - 290	0†	150††	1	1
Oozer	up to 80%	--	450 - 650	--	100 - 150	1	2 - 3
Cleanup	30 - 40%	1.23	500 - 2,000	5 - 10	60	1	2 - 3
Refresher	30 - 40%	2.05	200 - 1,300	20	60	1	2 - 3

---

* Percent solids shown are normal working ranges. Percent solids $= \frac{\text{weight of dry sediment}}{\text{weight of wet slurry}} \times 100.$

** Cost values shown are representative of Commencement Bay, Washington, for the cutterhead, bucket, and hopper dredges. Values for other dredges are derived by relation to conventional equipment. Variability may exceed ± a factor of 2 to 3.

† Zero if used alongside of waterway; otherwise, draft of vessel will determine.

†† Demonstrated depth; theoretically could be used much deeper.

‡ With submerged dredge pumps, dredging depths have been increased to 100 ft or more.

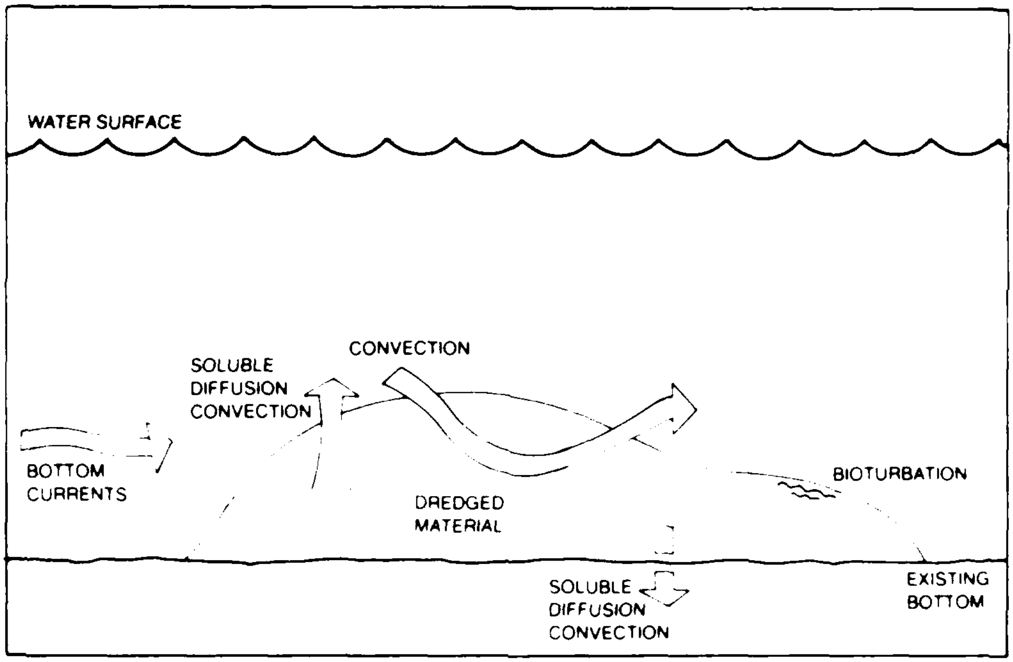
dredging, sweeping the bottom with the bucket, and digging fine-grained sediments from underneath (heavy buckets penetrating through soft surface materials) are practices to be avoided.

Short-term losses of soluble contaminants represent the key factor in selecting dredge type. These losses can be estimated by assuming a slow rate of contaminant transfer between phases during dredging and using a modified elutriate test. For hydraulic dredging, test results can predict weir concentrations (total and dissolved) expected for a given site, and predicted values can be compared against decisionmaking criteria with or without consideration of dilution in the receiving waters. It is more difficult to predict losses for mechanical dredging. Bucket size, sediment characteristics, and other job-specific factors will influence the actual losses in the field. As a usual rule, for dredging large volumes of low-level contaminated sediments where a low percentage of the contamination is soluble, hydraulic dredging with particulate control will provide greater confinement per given cost than will mechanical dredging with watertight equipment. As the percentage of soluble contaminants increases, the "confinement-per-cost" indicator will begin to favor the mechanical approach.

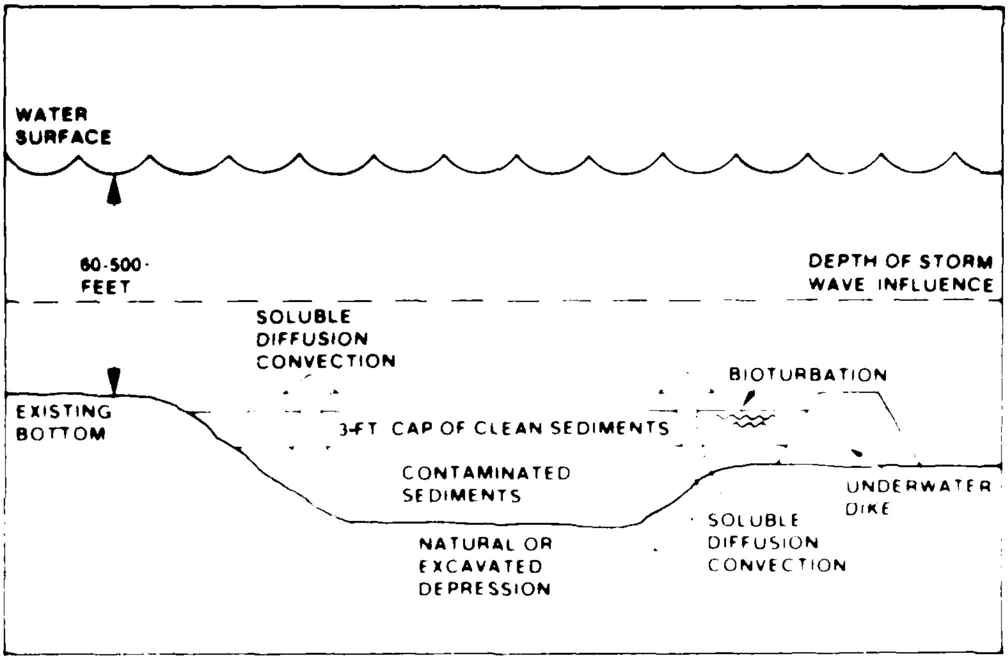
Most projects will contain all three classes of contamination. In terms of overall contamination, sediment-bound contaminants usually represent the bulk of the contamination, suggesting use of hydraulic equipment for maximum recovery and extraction efficiency. The amount of volatiles that may be lost during dredging is not likely to be a source of major concern in many projects. *As the types and amount of soluble, or easily solubilized, contaminants increase in a sediment to be dredged, greater consideration should be given to the cost and environmental impact of mechanical dredging with watertight equipment relative to that of hydraulic dredging and water treatment at the disposal site.*

Disposal

There are four basic options for disposal of dredged material: open water, unconfined; confined aquatic; nearshore; and upland (Figures 2 and 3). Open water, unconfined disposal is the historic, traditional practice for navigation dredging in Puget Sound; the other three are confined options. Confined aquatic disposal involves the placement of cleaner sediments over more contaminated sediments (capping) to isolate the contamination. Dredged material also can be placed in nearshore or upland disposal sites within containing dikes. It is important to remember that there are trade-offs and environmental consequences to all four options. Figures 2 and 3 show the major pathways, or routes of effect, that sediment contamination can take. Summarized in Table 2, the first three pathways involve water losses. Effluent is water discharged from the site during actual dredging. Runoff is similar in that it is surface water discharged from a nearshore or upland site resulting from rainfall or pore water extruded during sediment consolidation after dredging is completed. Leachate is basically ground-water seepage, again after completion of dredging. In leachate, contaminants can move with the water (i.e., *convection*) or they can move through the water (i.e., *diffusion*). Plants and animals also can uptake and distribute contamination, as well as suffer adverse effects directly. Fine-grained particles are known to carry the greatest percentage of contamination and often control of suspended solids results in excellent control of contaminants. Contaminants can also

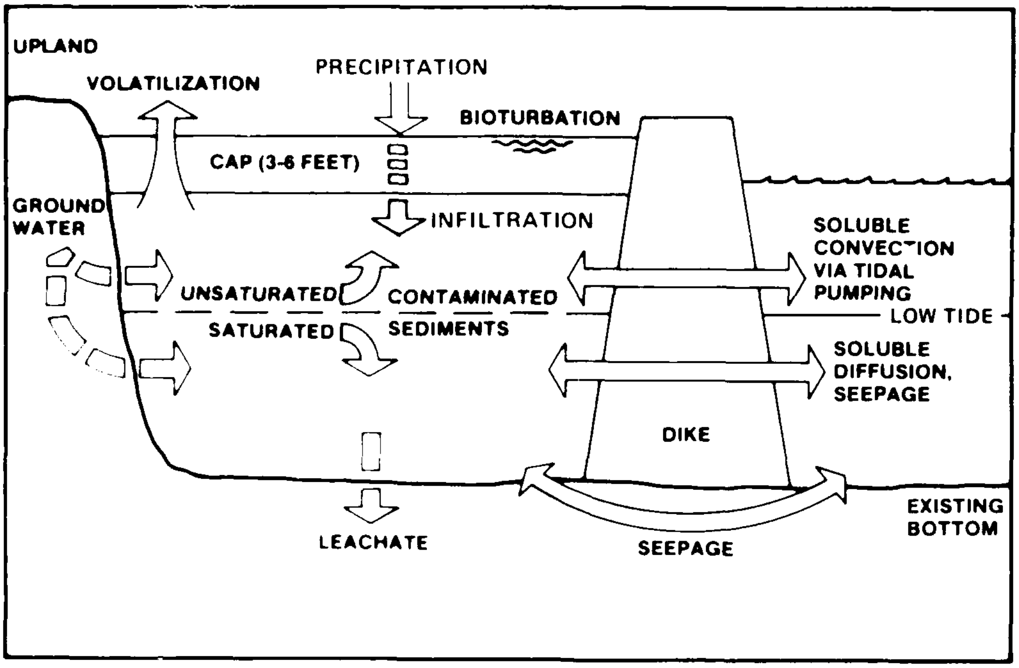


a. Open water, unconfined

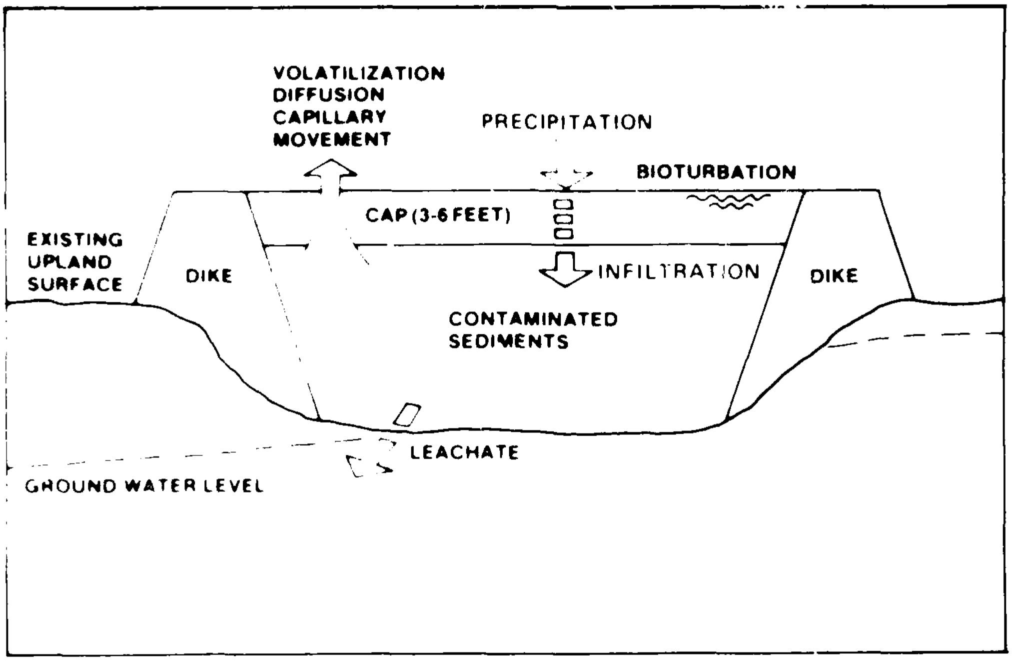


b. Aquatic confined

Figure 2. Open water, unconfined and aquatic confined disposal



a. Nearshore



b. Upland

Figure 3. Nearshore and upland disposal

TABLE 2. PATHWAYS OF CONTAMINANT MOBILITY

Water losses	
Effluent	
Runoff	
Leachate	- soluble diffusion
	- soluble convection
Uptake	- plants
	- animals
Particulates	- during dredging
	- effluent/runoff
Volatilization	

volatilize to air. These processes are of different relative importance in different disposal environments; and to confound matters further, different classes of contaminants will behave differently in different environments. Each assessment is, therefore, sediment- and environment-specific.

Contaminant mobility implies a quantifiable rate process and is a function of the chemical species, the solvent (water), the type of soil/sediment matrix, the physical state of the system, chemical concentration levels, and certain geometric factors. These factors are shown in Table 3.

Nearly all chemical contaminants are soluble to some degree; the degree of solubility depends on a variety of factors and can change as specific conditions are altered by the act of dredging, during transport of the dredged material, during its discharge into the disposal site, and following its disposal. The first three actions occur during a very short time frame and their influence on contaminant mobility is relatively instantaneous. Long-term condition changes and effects on contaminant phase, particularly with regard to solubility, are of greater complexity.

A percentage of the potentially soluble contaminant fraction will be in solution in the sediment interstitial water. The remainder will be bound to sediment particles. Thus, an equilibrium exists within the sediment phase, with the fraction of chemical contaminant potentially available for short-term release during dredging or disposal operations approximating the interstitial water concentrations and the loosely bound fraction in the sediments.

Once deposited at the disposal site, retention of soluble contaminants becomes difficult to assess. Contaminant fractions that were tightly bound to the sediment particles can, over time and due to altering conditions, become more prone to go into solution. This is often the case for heavy metals when placed in upland disposal sites. Under saturated conditions, many metals remain tightly bound to the sediments. In unsaturated conditions where oxidation can occur, these contaminants can become more soluble and may readily leach to surface runoff or ground water percolating through the material. Changes in pH have similar implications for other compounds.

TABLE 3. FACTORS RELEVANT TO LONG-TERM CONTAMINANT MOBILITY IN SOIL/SEDIMENT ENVIRONMENTS

- Chemical species
  - Molecular weight and chemical structure
  - Solubility in water and vapor pressure
  - Diffusivity in water and in pore gas
  - Partition coefficients and Henry's constant
- Solvents (pore water and pore gas)
  - Molecular weight of water and gas
  - Concentration and partial pressure of other species
- Soil/sediment matrix
  - Porosity (micro, macro)
  - Grain size (average and standard deviation)
  - Permeability (water and gas)
  - Organic fraction and ion exchange capacity
  - Water content (for soil)
  - Tortuosity
  - Level and depth of bioturbation
- Fluid properties
  - Temperature, pressure, and phase (G, L, S)
  - Water flow rate and direction
  - Gas flow rate and direction
- Total contaminant concentration level
  - Low: 50,000 ppm (wt)
  - High: 50,000 ppm (wt)
- Geometric factors
  - Length of diffusion and convection pathways
  - Cover layer depth
  - Disposal cell dimensions

In aerobic disposal environments, factors discussed above may have an even more pronounced effect on contaminant release. Brannon (1983) reported that sediments high in total sulfur content and low in $CaCO_3$ equivalent concentrations generally attained the lowest leachate pH's when incubated in an aerobic environment. Sulfide oxidation to sulfates with consequent formation of sulfuric acids is one mechanism postulated for this pH drop. In sediments from Seattle, Washington (total sulfide: 1,214 ppm), incubated under aerobic conditions, pH dropped from 8.2 to 6.6 in 1 month, to 5.5 in 3 months, and to 4.1 in 6 months of observation (Brannon 1983). Arsenic release from these sediments was associated with the pH drop. Aerobic disposal also affects organic contaminants: decay of TOC in upland disposal sites can change the mobility of these contaminants.

A comparison of the effectiveness of the three confined disposal methods is provided in Table 4. Soluble contaminants, or contaminants with the greatest potential to go into solution under certain conditions, are of highest

TABLE 4. COMPARISON OF DISPOSAL METHOD EFFECTIVENESS

Disposal Method	Relative Geochemical Effect on Contaminant Mobilization	Relative Magnitude of Contaminant Transport Mechanisms	Available Control/ Treatment Options	Relative Environmental Risks from Contaminant Release
Confined aquatic	Low	Diffusion: high Convection: medium Bioturbation: varies Erosion: Medium	Few	Low due to dilution (ecological risk)
Upland confined	High	Diffusion: low Convection: low Bioturbation: varies Erosion: low Volatilization: high	Many	Varies by contaminant (human health risk)
Nearshore confined	High in unsaturated zone; medium in saturated zone	Diffusion: high Convection: high Bioturbation: varies Erosion: low Volatilization: high	Some	Medium (human health and ecological risks)

concern because these are less readily contained. In general, open-water sites, especially those in deep water, have fewer transport mechanisms (e.g., air is absent) than upland sites and are geochemically more similar to in situ conditions at the dredging site. Nearshore sites have the most transport routes available and are located in a very active environment. Control of the releases and/or concern with the effects of the release must be considered. For example open-water disposal allows for very limited control of releases other than cap or liner thickness; the relatively stable physicochemical environment retards contaminant mobility and encourages a constant, gradual release to the overlying water body, especially once a cap has been saturated by migrating contaminants. The levels of contaminants released will be low (possibly unmeasureable) and will be diluted by the overlying water. The risk of significant damage in this environment is low and would not likely affect human health. Upland disposal, on the other hand, allows for the greatest control, through design considerations, monitoring capabilities, backup contaminant intercept systems, and treatment facilities. However, environmental risks may be higher than in open water because of proximity to ground water and potential human health concerns. The nearshore disposal option allows for some greater control of contaminants than in open water, but many fewer than are available in an upland situation. In addition, the risks to the environment and to human health often include both those of open-water and upland sites.

Site Control and Treatment Methods

The applicability and utility of various control and treatment alternatives for contaminated sediments in Commencement Bay are shown in Table 5.

TABLE 5. APPLICABILITY AND UTILITY USE OF VARIOUS CONTROL AND TREATMENT ALTERNATIVES FOR CONTAMINATED SEDIMENTS IN COMMENCEMENT BAY

	Disposal Method					
	Upland Disposal		Nearshore Disposal		Open-Water Disposal	
Utility	High	Low	High	Low	High	Low
Liners	Synthetic, soil			Soil	NA*	NA
Drains	Leachate drains		NA	NA	NA	NA
Capping	Synthetic, soil		Synthetic, soil		Sediment	
Sediment stabilization	Liming, palliatives	Sprinkling	Liming, palliatives	Sprinkling	NA	NA
Suspended solids removal	Sedimentation, clarification, filtration		Sedimentation, clarification, filtration		NA	NA
Removal of solubles	Precipitation, adsorption	Ozonation	Precipitation	Adsorption, ozonation	NA	NA
Dissolved solids removal		Distillation, RO, ED, IE**		Distillation, RO, ED, IE	NA	NA
Volatiles		Stripping		Stripping	NA	NA

(Continued)

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* NA = not applicable.

** RO: reverse osmosis, ED: electrodialysis, IE: ion exchange.

TABLE 5. (Concluded)

Utility	Disposal Method					
	Upland Disposal		Nearshore Disposal		Open-Water Disposal	
	High	Low	High	Low	High	Low
Biological control	Fencing, sediment cover	Biocides	Fencing, sediment cover	Biocides	NA	NA
Monitoring	Leachate, ground water	Volatiles	Berm, ground water	Volatiles, shellfish	Cap integrity, cap contaminants	Cap bioturbation
Remedial response	Leachate treatment	Groundwater treatment sediment removal		Slurry wall sediment removal	Dewatering berm	Add cap materials
Total number of available options	17	11	11	13	4	1

This table illustrates that the degree of site control and the number of available treatment options decreases from upland to nearshore to open-water disposal. This decreasing control is translated into reduced opportunities to design additional treatment measures that would prevent uncontrolled, sudden, or accelerated contaminant release into the environment and/or to avoid the extreme expense of sediment removal and relocation.

Dredging and Disposal Summary

For hydraulic dredging, the relative importance of losses at various times and from various phases during the sediment handling process is listed below in order of decreasing importance.

a. Short-term loss of sediment and water.

b. Long-term loss of water.

c. Short-term loss of volatiles.

d. Long-term loss of volatiles.

e. Long-term loss of sediment.

Normally, disposal will result in greater short-term loss of sediment and water than will dredging. Based on the above ranking of importance, short-term sediment and water loss during disposal will be the first consideration and the basis for selecting disposal method and treatment level. Concurrently, but on a secondary basis, the contribution of dredging to this loss should be evaluated. The next subsequent step should be selecting appropriate treatment, monitoring, and remedial response to address long-term loss of waterborne contaminants. Consideration of items c - e above would depend on sediment and site-specific conditions. For mechanical dredging, short-term loss of sediment and water and long-term loss of water from the disposal site (for upland or nearshore sites) may be equally important. The proportion of partially soluble contaminants in disposed sediment that is available for later leaching is increased relative to that in hydraulically dredged sediments. The amount of sediments and easily soluble contaminants lost at the extraction end is also increased relative to hydraulic dredging. However, the use of watertight buckets may reduce this loss substantially.

PARTITIONING

Assessing contaminant loss during dredging and disposal usually involves estimating the portion of the contaminants that might be found in the sediment interstitial water. While the preferred approach to this issue is to determine the portion empirically, through direct testing, it is also possible to theoretically calculate the soluble portion via application of chemical partitioning coefficients, as a way to flag potential problems and identify contaminants of concern for a given project.

Chemical phase partitioning information applied to select contaminants from Commencement Bay sediments has been used to illustrate one approach to a screening assessment of contaminated sediments. While the evaluation is useful as screening-level information, it must be emphasized that the values are highly variable and that present state of the art cannot provide reliable predictions of contaminant behavior or effects without sediment-specific tests. Equilibrium in chemical partitioning is often not present or maintained during dredging, such that kinetics must also be considered. Furthermore, while estimates of contaminant loss in the short term (during dredging, transport, and initial disposal) are possible, verified predictive techniques for long-term losses are not available. This further emphasizes the need to include long-term monitoring and remedial response capability as an integral design component for disposal of contaminated sediments.

Partitioning of Metals

Example Commencement Bay sediment chemistry is shown in Table 6. Metals in sediment can be partitioned to numerous forms of varying mobility, persistence, and bioavailability. Gambrell, Khalid, and Patrick 1975 list the following potential metal forms:

a. Soluble free cations.

b. Soluble organic or inorganic complexes.

c. Easily exchangeable cations.

d. Precipitates of metal hydroxides.

e. Precipitates with ferric oxyhydroxides.

f. Insoluble organic complexes.

g. Insoluble sulfides.

h. Residual metals.

Brannon et al. 1976 describe a selective extraction protocol that procedurally defines metal pools by their relative extractability. They suggest that short-term metal losses during dredging and initial disposal are likely related to the soluble (interstitial water) and easily solubilized (exchangeable phase) metal forms, the two metal phases most easily extracted in their procedure.

Dredging and disposal of sediments can influence the mobility of metals by changing basic factors that govern their phase partitioning such as pH and redox potential. The dredging process may influence these regulating processes in opposing directions. For example, most sediments found in the Commencement Bay waterways are low in oxygen content and consequently reduced. The dredging process often removes the sediments to oxidizing environments and/or introduces oxygenated water. The presence of sulfide under reduced conditions will precipitate divalent metal sulfides, resulting in very low solubility. Sulfides will oxidize under aerobic conditions, possibly releasing heavy metals. On the other hand, metals may be present in higher concentrations in sediment interstitial waters under reduced conditions due to reduction of iron and manganese hydrous oxides which tend to sorb or coprecipitate metals under oxidized conditions (Khalid et al. 1975). Which of these two factors will predominate is specific to a given sediment and dredging project.

TABLE 6. COMMENCEMENT BAY EXAMPLE SEDIMENT CHEMISTRY

Parameter	Value	Grain size
TOC*	8.83 percent	Sand: 66.7 percent
Solids:	58 percent	Silt: [illegible] percent
		Clay: [illegible].8 percent
Metals, ppm		
As	9,700	
Cu	11,400	
Zn	3,320	
Organics, ppb		
Phenol	310	
LMW PAHs**	20,000	
HMW PAHs†	31,000	
PCBs	26	
Hexachlorobutadiene	730	
Tetrachlorethylene	140	

* Total organic carbon.

** Low molecular weight polynuclear aromatic hydrocarbons.

† High molecular weight polynuclear aromatic hydrocarbons.

Given the number of variables that influence metal partitioning, estimates for the Commencement Bay sediment presented below are based on representative values derived empirically from other projects. Brannon et al. (1976) report values for soluble and exchangeable phase (easily removed from sediment) metals from various sediments collected around the United States. Representative values for use with the Commencement Bay sediment and the ranges of values seen in the Brannon et al. project report are shown in Table 7.

TABLE 7. REPRESENTATIVE AND RANGE OF VALUES FOR SOLUBLE AND EASILY SOLUBILIZED METALS*

	Soluble and Easily Solubilized, Percent	
Metal	Representative Value	Range of Values Reported
Cu	0.005	0.00004 - 0.0067
Zn	0.4	0.0001 - 2.4
As	0.3	0.0009 - 2.08

* Brannon et al. (1976).

For the example sediment, Table 8 shows that copper sediment concentrations of 11,400 ppm would translate into 0.57 ppm in the soluble plus exchangeable phase, assuming that a linear relationship exists between total metal concentrations and their respective concentrations in the soluble plus exchangeable phase. For zinc and arsenic, combining information in Tables 6 and 7 provides estimates of 13.28 ppm of zinc and 29.1 ppm of arsenic in the soluble and exchangeable phase.

TABLE 8. CASE STUDY PARTITIONING VALUES*

Parameter	C_S	C_{OC}	C_{IW}	C_{IS}
Metals, ppm				
Cu	11,400	NA	0.57	0.023
Zn	3,320	NA	13.28	0.170
As	9,700	NA	29.1	0.440
Organics, ppb				
Phenol	310	3,500	146	1
Tetrachloroethylene	140	1,590	7.9	--
Hexachlorobutadiene	730	8,270	4.1	--
PCB	26	290	0.001 - 0.006	0.030
LMW PAHs	20,000	226,500	45	10
HMW PAHs	31,000	351,100	0.4	10

* C_S = bulk sediment concentration.

C_{OC} = bulk concentration normalized to TOC content of sediments.

C_{IW} = estimated interstitial water concentration.

C_{IS} = interpretive standard or reference concentration.

Partitioning of Organic Contaminants

Blom et al. (1976) reported seeing no evidence that TOC in a sediment directly controlled release of metals during dredging, as no positive correlations between metals release and TOC or soluble organic carbon were observed. For many organic contaminants, however, TOC appears to play a central role in contaminant release and availability. Pavlou and Weston (1984) report that the variability in interstitial water concentrations of nonpolar, organic contaminants is a function of sediment TOC and not bulk sediment concentration. Consequently, a first-step estimation of interstitial water concentrations is possible by assuming thermodynamic equilibrium and using bulk sediment chemistry, TOC content, and information on the partitioning coefficient of each contaminant. Many of the necessary coefficients are available in the literature; a summary of organic carbon-water partitioning coefficients, K_{OC} for the example sediment chemicals is provided in Table 9. For the values shown in Table 8, bulk sediment concentrations were first normalized to TOC content, and then interstitial water concentrations were estimated using K_{OC} values from Table 9.

TABLE 9. PARTITIONING COEFFICIENTS FOR ORGANIC CONTAMINANTS

Contaminant	K_{oc}
Hexachlorobutadiene	2,000
PCB (2-6)	4,600-5,000
Phenol	24
LMW PAHs	
Acenaphthene	4,700
Naphthalene	890
Acenaphthylene	3,900
Anthracene	7,400
Phenanthrene	9,300
Fluorene	4,800
	$\bar{x}$ = 5,098
HMW PAHs	
Fluoranthene	45,000
Benzo(a)anthracene	37,000
Benzo(a)pyrene	300,000
Benzo(b)fluoranthene	500,000
Benzo(k)fluoranthene	840,000
Chrysene	77,000
Benzo(g,h,i)Perylene	1,800,000
Dibenzo(a,h)anthracene	160,000
Ideno(1,2,3-c,d)pyrene	4,100,000
Pyrene	33,000
	$\bar{x}$ = 789,200
Tetrachloroethylene	200

Bioavailability

McFarland (1984) reports that is is theoretically possible to predict the potential maximum level of bioaccumulation resulting in an organism from a given sediment. Proposed as a first-level screening device to identify important compounds and sediments worth further evaluation, the technique calculates thermodynamic bioaccumulation potential in lipid content of an organism based on bulk sediment contaminant concentration, TOC content, and partitioning coefficient equations. Using this technique for the PCB levels in the example sediment would show the following:

a. PCB bulk sediment concentration = 26 ppb.

b. TOC = 8.83 percent.

c. Assumed lipid content in animal muscle = 5 percent.

d. TBP (thermodynamic bioaccumulation potential in lipid) = (26 ppb/8.83 percent)/0.52 (equation from McFarland (1984)).

Screening Assessment

Data on soluble and exchangeable forms of metals, such as that summarized in Table 8, coupled with elutriate test results, can provide the basis for assessing potential contaminant losses in the water during dredging and initial disposal. Developed by use of K_{oc} values, the predicted water concentrations for organic contaminants can serve as a basis for assessing potential release. Note that indications of priority contamination ("contaminants of concern") as provided by bulk concentrations are sometimes reversed when partitioning is considered (e.g., copper versus arsenic, phenol versus low molecular weight PAHs). These predictions can be assessed by comparing them with available interpretive standards. These comparisons, shown in Table 8, again reorder the relative importance of various sediment contaminants. Predictions on potential muscle concentrations can also be compared with reference values, demonstrated effects data, and/or other decision criteria to determine if further testing with PCBs is warranted. The two-orders-of-magnitude difference between the estimated value (28 ppb) and interpretive value (2 ppm) for PCB concentrations provides a way to weight the intensity of further testing of these sediments.

The number of missing interpretive standards, and, in other cases, differing standard values for the same contaminant, highlight the difficulty in completing an assessment of dredged material.

COST

Theoretical (or empirical) estimates of soluble contamination and bioaccumulation potential serve to emphasize the need for decision criteria. Consideration of cost is a second emphasis for the same need. The Commencement Bay analysis found a very diverse array of technology that could be applied to dredged material disposal, including solids/water control, as well as treatment techniques and long-term management options. Table 10 notes that increased control and treatment has a proportional cost impact. Looking at detailed cost estimates (Tables 11 and 12), dredging can go from $1.50 per cubic yard to $9.40 per cubic yard with addition of a liner, to $15.10 per cubic yard with chemical clarification, and to $20.10 per cubic yard with carbon adsorption. A variety of advanced treatment systems can push costs even higher. As expected, the critical issue becomes the appropriate balance between environmental protection and cost.

DECISIONMAKING FRAMEWORK

To improve interpretation of theoretical or empirical contamination data on dredged material and to assist decisions associated with high cost of control technology, a decisionmaking framework was developed for Commencement Bay (Lee et al. 1985). The framework is a staged management strategy designed to assess, via a number of specific tests and interpretive guidance, the need for restrictions in dredging and/or disposal and to determine the appropriate level of those needed restrictions. It has five overall steps: (1) identify the potential contaminant effects at the disposal site, (2) characterize the

TABLE 10. COST OF DISPOSAL AND CONTROL OPTIONS FOR HANDLING CONTAMINATED SEDIMENTS*

Alternative	Cost**
Disposal	
Site preparation	
- upland/nearshore	\$500,000†
Weir construction	
- upland	\$25,000
- nearshore	\$35,000
Diking - imported materials	\$4./cu yd
- onsite materials	\$1./cu yd
Open-water vertical diffuser	
- construction	\$50,000-\$100,000
- operation	+\$1.-2./cu yd
Offsite material transport	
- truck	+\$.20/cu yd/mile
- barge	+\$.20-.25/cu yd/mile
Site control	
Open-water capping material	\$1.40/cu yd
Liners - soil (volume)	\$16.29-18.29/cu yd
- soil (area)	\$1.81-2.03/sq ft
- synthetic	\$.11-1.50/sq ft
Surface covers	\$1.27-24.20/sq yd
Underdrains	\$2,500/acre
Sediment stabilization	
- lime	\$10,000-14,000/acre
- dust palliatives	\$1,000-17,000/acre
- water sprinkling	\$2,000/acre
- chemical solidification	\$40/cu yd††

* Treatment costs not included because of their dependence on flow rates (see Tables 11 and 12).

** US dollars, January 1984.

† Average for potential sites identified in Commencement Bay. Includes diking and weir costs.

†† Cost based on proposed treatment of 37,000 cu yd of material in Commencement Bay.

dredged material by chemical and biological tests, (3) identify any specific contaminant mobility problems, (4) design problem specific solutions, and (5) prepare a site management plan, including any needed monitoring and remedial response.

Selection of dredging equipment and design of a disposal site for a given contaminated sediment first require specific tests to assess contaminant concentrations and types, to estimate contaminant release at various points during the handling process, and to predict contaminant effects under various disposal conditions. Test results are then compared with predetermined

TABLE 11. TREATMENT LEVEL COST COMPARISON FOR NEARSHORE 30-ACRE SITE*

Cost Component for Dredging 1,000,000 cu yd	Cost ($1,000) Construction	4-Month O&M	Unit Process 1,500	Cumulative Total 1,500
Level I				
Plain sedimentation	5,880	50	5,930	7,430
Chemical clarification	214	220	434	7,864
Filtration				
Option 1 - Pervious dike	75	5	80	7,944
Option 2 - Sandfill weir	86	20	106	7,970
Level II				(Assume Option 2 in Level I)
Precipitation	869	464	1,330	9,300
Level III				
Carbon adsorption - Option 1	5,000	495	5,500	14,800
Ozonation - Option 2	1,600	300	1,900	16,700**
Level IV				(Assume Option 2 in Level III)
Distillation	47,850	4,452	52,300	69,000
Electrodialysis	24,850	827	25,700	42,400
Reverse osmosis	28,850	598	29,450	46,150
Ion exchange	21,350	987	22,300	39,000

* Costs for site control and treatment at a nearshore site cannot be directly compared with costs for an upland site; treatment levels contain different site control options.

** Total cost includes Option 1, carbon adsorption, plus Option 2, Ozonation.

standards or reference values (i.e., decision criteria) to determine where predicted contaminant behavior will require some form of restriction or control. The purpose of the necessary restriction(s) is to reduce (or eliminate) the contaminant loss or effect to an acceptable level. Selecting an appropriate restriction method often requires additional sediment testing as part of project design. Thus, a given sediment to be dredged may undergo several different test series. For sediment "clean-up" projects, these are the following:

a. Testing to determine in-place hazard warranting cleanup (is it a "problem" sediment?).

b. Testing to determine if restrictions are required to dredge and dispose the sediment in a given disposal environment. (These tests are often tiered for cost efficiency.)

TABLE 12. TREATMENT LEVEL COST COMPARISON FOR NEARSHORE 80-ACRE SITE*

	Cost ($1,000)			
Cost Component for Dredging 1,000,000 cu yd	Construction	4-Month O&M	Unit Process 1,500	Cumulative Total 1,500
Level I				
Plain sedimentation	15,762	50	15,812	17,312
Chemical clarification	214	220	434	17,746
Filtration				
Option 1 - Pervious dike	75	5	80	17,826
Option 2 - Sandfill weir	86	20	106	17,852
Level II				(Assume Option 2 in Level I)
Precipitation	869	464	1,333	19,185
Level III				
Carbon adsorption - Option 1	5,000	495	5,495	24,680
Ozonation - Option 2	1,600	300	1,900	26,580**
Level IV				(Assume Option 2 in Level III)
Distillation	47,850	4,452	52,302	78,882
Electrodialysis	24,850	827	25,677	52,257
Reverse osmosis	28,850	598	29,448	56,028
Ion exchange	21,350	987	22,337	48,917

* Costs for site control and treatment at an upland site cannot be directly compared with costs for a nearshore site; treatment levels contain different site control options.

** Total cost includes Option 1, carbon adsorption, plus Option 2, Ozonation.

c. Testing to select and design the necessary restrictions.

Table 13 shows representative costs for conducting the testing outlined in the decisionmaking framework. Not all tests are needed for all projects; tests are specific to the disposal environment being evaluated.

APPLICATIONS IN PUGET SOUND EXAMPLES

Navy Homeport Facility at Everett, Wash.

The US Navy recently selected Everett, Wash., as its preferred location for a [illegible] to 15-fleet battle carrier group home ported on Puget Sound. Construction would involve dredging and disposal of 2.5 million cubic yards of marine sediment to deepen the harbor and excavation of 1.0 million cubic yards of marine-derived fill to reconfigure a mole pier. In June 1984, the Navy

TABLE 13. RELATIVE TIME AND COST ESTIMATES FOR CONDUCTING TEST PROTOCOLS

Test Protocol	Time months	Cost/ Test Run* dollars	Number of Test Samples Analyzed	Cost of Chemical Analysis** dollars	Total Cost dollars
Dissolved chemical analysis and mixing	2	100	6	6,000 - 9,000	6,100 - 9,100
Water column bioassay and mixing	3	1,000 - 5,000	16		1,000 - 5,000
Sediment comparison	2	100	8	8,000 - 12,000	8,100 - 12,100
Benthic bioassay and bioaccumulation †	5	1,000 - 5,000	16	16,000 - 24,000	17,000 - 29,000
Effluent quality	2	80	6	6,000 - 9,000	6,080 - 9,080
Runoff	6	4,000	15	15,000 - 22,500	19,000 - 26,500
Leachate ††	12	25,000	50	50,000 - 75,000	75,000 - 100,000
Plant uptake ‡	4	5,000	12	12,000 - 18,000	17,000 - 23,000
Animal uptake ‡‡	4	4,000	12	12,000 - 18,000	16,000 - 22,000

* One sediment sample.

** Estimated cost for PCBs, pesticides, 10 metals, and suspended solids per analyzed sample ranged from $1,000 to $1,500. Does not include cost of equipment or facilities or sample collection and transport to lab.

† Four species were used.

†† Leaching test and serial batch tests are currently under development. Routine test cost will be lower.

‡ Plant bioassay of 45-day exposure.

‡‡ Earthworm bioassay of 28-day exposure.

contracted with the Corps to provide them technical assistance for dredging and disposal of project sediments. Applying the Commencement Bay approach, the Corps first sampled and chemically characterized project sediments. This characterization effort identified two distinct layers of marine sediments that would be disturbed by the Navy's proposed dredging: one layer of organically rich, fine sediments (2 to 6 ft thick) overlying a layer of relatively cleaner "native" sediment. The organic layer was found to contain significantly elevated levels of chemical contaminants (metals and PAHs) and was determined to be unacceptable for unconfined, open-water disposal. In the Phase I report to the Navy, biological testing was recommended. Phase II studies and report concluded that the top 1 to 3 ft of the native material should be dredged and disposed with the overlying, contaminated fraction, but that the remainder of the material appeared acceptable for unconfined disposal. Phase III studies are currently under way at the WES applying the suite of tests previously described for the Commencement Bay decisionmaking framework; these studies are scheduled to be completed in mid-1986. The Navy is proceeding with state and Federal permit processing in order to meet a very tight construction schedule. Their present proposal is for confined aquatic disposal of the contaminated material (approximately 800,000 cu yd), on a 250-ft-deep shelf in adjacent Port Gardner, using the remainder of the project material for construction of the cap. Results of the suite of tests will be key to permit decisionmaking and final design of the Navy's dredging and disposal plans.

Puget Sound Dredged Disposal Analysis

The Commencement Bay work brought a number of issues and options to light; the Puget Sound Dredged Disposal Analysis (PSDDA) is a study to resolve the major issues associated with navigation dredging. A problem-solving component of an overall cleanup program for Puget Sound, the PSDDA study focuses primarily on unconfined, open-water disposal of dredged material. Sound-wide in its perspective, the 3-year study has three major objectives:

a. Locate and designate sites for open-water, unconfined disposal of dredged material.

b. Define evaluation procedures for dredged material. These are the chemical and biological tests and test interpretation needed to decide if dredged material is acceptable for open-water, unconfined disposal. Testing and test interpretation for material requiring confined disposal will also be covered.

c. Formulate site management plans for these open-water sites, likely involving a longer term and more carefully monitored approach to site management than used in the past.

At the heart of the PSDDA process will be several key decisions (shown in Table 14). These include whether to continue open-water, unconfined disposal shown as I; what level of chemicals should be placed and managed at these sites, shown as II; and where the sites should be located, shown as III. Initiated in April 1985, PSDDA is due for completion in April 1988, with a total study budget of $3.7 million.

TABLE 14. ALTERNATIVES FOR THE PUGET SOUND DREDGED DISPOSAL ANALYSIS

I. No unconfined, open-water disposal in Puget Sound
    a. No dredging
    b. All confined disposal

II. Unconfined, open-water disposal in Puget Sound
    a. No chemical effects in biological resources
    b. Minor chemical effects in biological resources
    c. Moderate chemical effects in biological resources
    d. Major chemical effects in biological resources

III. Alternative unconfined, open-water disposal site locations in Puget Sound

ISSUES

Dilution Versus Confinement

The Superfund effort in Commencement Bay has significantly heightened regional awareness of the contrasting management philosophies embodied in the Clean Water Act (CWA) and the Resource Conservation and Recovery Act (RCRA). In the past, concern about the environmental health of Puget Sound and its effects on human health concentrated principally on water quality considerations and standards. It was assumed that dilution would reduce pollution loading to safe levels and that these diluted wastes would be flushed from the Sound by normal water exchange with the Pacific Ocean. Passage of the CWA of 1972 eliminated direct sewerage and waste discharges without permits and required greater treatment of municipal wastes. Water quality criteria were established for a few, major compounds defining levels of pollutants that could not be exceeded. Research related to regulating safe levels of pollutant discharges concentrated on "mixing zones" that would ensure the proper dilution of wastes. Revision of the CWA in 1977 led to increased monitoring and more stringent requirements for discharges, but brought no changes to the dilution philosophy.

RCRA, passed in 1976, established a different philosophy toward chemical wastes and contaminants, based on controlling the contaminants from their creation to destruction (i.e., "cradle to grave"). Priority was given to the fate of contaminants, their long-term effects, liability of producers of these contaminants, and the objectives of "ultimate disposal" and maximum confinement. Originally targeted for upland control of chemical contaminants, RCRA makes no clear exception for dredged material that is contaminated. Researchers and regulators began to discover that dilution to maintain water quality standards did not resolve the problem of loading of chemical contaminants into marine waters. Often the contaminants became or remained bound to sediment particles where they were available for uptake by aquatic organisms and, through the food chain, potentially by humans. In addition, other research concluded that Puget Sound acted more like a giant bathtub whose water and bottom sediments sloshed back and forth rather than flushing into the open ocean. Pollutants discharged into Puget Sound tend to remain in Puget Sound and over the years chemical contaminants have accumulated in urban embayments and are apparently causing histopathological and visible pathological abnormalities in local biota.

Water Quality Versus Sediment Quality

Studies by the National Oceanic and Atmospheric Administration (e.g., Malins et al. 1984) were the first indicators that sediments, rather than water quality, constituted a problem potentially affecting the health of Puget Sound. The evidence strongly indicates that protection of water quality via application of water quality criteria to the water column does not appear to be the best approach. Dredging rarely violates Federal water quality criteria in the water column. Even the most contaminated sediments are unlikely to do so because the contaminants are predominantly bound to particles. So sediments are being contaminated even in the absence of water quality criteria violations.

With evidence of chemical contaminant enrichment and apparently related biological degradation at several deep water disposal sites in Puget Sound, questions and concerns have been expressed by agency personnel and the public over the contaminant levels in dredged material destined for disposal in Puget Sound. As a result, the RCRA philosophy is being heard more frequently during discussion of dredging projects: long-term confinement, mass loss, residence periods, and a "cradle-to-grave" solution as the means to address the many unknown effects and fates of chemicals in the environment.

Management goals for Puget Sound are being shifted from the water column to the sediments, and new decision criteria are being generated to reflect this shift in management. Increased vertical and horizontal stratification in sampling the dredging prism and complete chemical and biological testing of the sediments to be dredged have become normal requirements.

Regulatory Need for Simplicity

A fundamental problem associated with the selection of a management approach to contaminated sediments is the conflict between "technical adequacy" and "regulatory simplicity." Sediment contamination and its effects are infamously complex. However, an equally complex regulatory structure is often perceived as too complicated to implement. This latter view is reinforced when the time and cost of the technical assessment become apparent. This has led to numerous debates on the merits of a technically adequate assessment of contamination problems versus the use of a "best available technology" or overdesign approach.

The "Gray Zone"

Much of US law seems to be derived from a school of thought that believes that it is possible to identify specifically the level of contamination that is "safe" for a particular environment, such as marine waters. There are two reasons why such an approach will not work:

a. The definition of a "safe" level is often left to science; science is the first to admit that uncertainties abound in this field.

b. Safety is a relative term, requiring a social value judgment. Different cultures, different states, and different neighbors do not agree on what is an acceptable contaminant risk or effect and what is not.

The product of scientific uncertainty plus differing social values is the so-called "gray zone" in contaminant concentrations. This expansive (frequently orders of magnitude) zone requires decisions to assess trade-offs in effects and risks in different environments. It also is the source of the "regulatory flexibility" in agency decisions on individual projects.

Aquatic Disposal as a Viable Option

Aquatic disposal of dredged material is undergoing serious reevaluation at this time. Much of this reevaluation is aimed at addressing issues associated with sediment-bound contaminants, for example, establishing consistent testing and evaluation procedures. But another aspect is the reevaluation of aquatic disposal as the "least preferred," or "last resort," disposal option, embodied in current Federal law (e.g., Marine Protection, Research, and Sanctuaries Act of 1972 (Ocean Dumping Act)). In trying to minimize impacts in a single environment, such as the water, it is simplest to discourage its use by setting criteria as tightly as possible. But with the advent of RCRA, having a different philosophy and a "hammer provision" for ending landfilling, potential problems are readily apparent. The alternate perspective towards aquatic disposal is to consider it a viable disposal option. The advances of capping technology are key to this changed perspective. With many advantages in retarding contaminant mobility, and the presence of a reduced, buffered environment, the isolation of many contaminants can be very effective in confined aquatic sites. The Corps of Engineers (CE) has recently employed capped disposal in the Duwamish Waterway at Seattle, Wash., and the US Navy has proposed capping as part of their homeport proposal.

Single Environment Versus Multienvironment Assessments

Objective, technical selection of a dredged material disposal option is very much an exercise in risk management. Varying site resources, varying site geochemistry, varying contamination types and concentrations, etc., all imply that there are no "risk-free" options. The first conclusion of the CE congressionally mandated, $30 million research program on dredged material, conducted in the late 70's, was: "There is no single best disposal environment, no panacea." This is why the Corps actively supports EPAs efforts on multimedia, multienvironment assessments, which will eventually require several changes in Federal laws that are environment-specific. The approach is to assess each disposal option and its problems before selecting the best option.

New Liability

A direct repercussion of the intent of the Superfund law is a "new liability" that anyone having anything to do with contamination has to acknowledge. This intended "contagious" liability, required to fuel the Superfund cost recovery program, has set a clear precedent for contaminant transporters. A case in point for how this liability can affect projects is offered by the previously referenced Duwamish capping project which involved mechanical dredging of 1,500 cu yd of sediment contaminated primarily with metals and low-level PCBs. The 1,500 cu yd of contaminated sediments used for the capping project were initially to be placed in an upland area. The State of Washington approved the project with the condition that the Corps monitor the ground water, and if it became impacted, to "fix it." "Fixing" ground water is not a liability to be assumed lightly; remediation, if possible, always incurs high costs. With research data available for upland leaching from Duwamish sediments, it was in our view, an unacceptable liability. In our judgment, a more acceptable liability risk involved confining the contaminated material within the aquatic environment. The disposal barge was discharged over a depression (MLLW 60-70 ft) in the West Waterway of the Duwamish River, in Seattle, Wash. Cleaner sandy material from upstream, in several barge loads, was slowly released over the depression. The capping was hydroacoustically monitored after each cover load until the cap was in place. As a "technology demonstration," the project was closely monitored during construction; longer term monitoring involves use of a Vibracore. Samples are taken through the cap to check for cap thickness/integrity and possible leaching.

For the industrial waste managers, the best protection from liability is often said to be a continuous analytical track, an "accountability." This principle of dredged material accountability, combined with efforts toward objective consistency in regulations and guidelines, is at the heart of the PSDDA effort. Also, dredged material testing is now more than just for environmental protection; it is also a measure of possible liabilities with disposal.

Relocation Versus Reintroduction

This raises a challenge as to whether dredging and open water, unconfined disposal is introducing contamination into the Sound or simply relocating it within the Sound. Since much dredging is done for navigation purposes, with navigation monies, it can be viewed defensibly (to a certain extent) as a management opportunity to decide where and how to place again the contaminants. Where this becomes more challenging is when dredging is listed as a point source in a mass loading formula for the Sound. Are the sediment contaminants already "loaded" due to the industrial discharge, or does dredging reload them? The answer depends on what extent agencies and decisionmakers wish to take advantage of the navigation dredging opportunity to address existing "hot spots."

Single Technology Versus Problem Management

A related spin-off issue is the conflict between contaminant handling by specified, perhaps "best available," technology and handling by problem identification and problem-specific management. In the CWA, different wastes (for example, municipal sewage versus dredged material) received different handling philosophies. But RCRA, with its statutory design standards, has involved dredged material with several questions. Should detailed tests be conducted on dredged material to identify what problems it might cause, and then solutions to those problems be designed, or should we minimize testing and go with a single-approach disposal site design?

Two Cost References

Because of the many new issues and concerns described above, managers now have two reference points to evaluate the cost of dredged material disposal:

- The traditional cost of unconfined, open-water disposal ($[illegible] to $[illegible] per cubic yard)

- The RCRA best available technology design cost for permanent landfill ($40 to $60 per cubic yard)

In addressing the continuum of contaminant concentrations in dredged material, the splitters and lumpers will produce two very different cost curves depending on how they apply technology to address contaminant effects. Figure 4 illustrates examples of these two cost curves. Given the two reference points in cost for dredged material disposal (the traditional cost of open water, unconfined disposal, and the high cost of RCRA "ultimate" disposal), the direction headed in Puget Sound involves increasing up-front testing cost, to identify problems and design problem-specific solutions, as a way to minimize construction costs.

Long-Term Planning

One consequence of these increased costs, of the ongoing debate of best approach to managing contaminated sediments, and of the overall imperative of improved accountability, is the need for long-term planning. Dredged material "management," a comprehensive look at quantities and characteristics of the material and a detailed inventory of the disposal site "resource," is essential. Following the path blazed by industrial waste managers, with their multimillion dollar regional incinerators, multiuser disposal sites for dredged material are an obvious next step. This approach, for both confined and unconfined disposal, is the best way to focus resources, afford proper design and monitoring, and maximize accountability in the dredging program.

Source Control Versus Sediment Removal

Assessing progress in cleanup first requires us to define what we mean by "cleanup." Common sense dictates that the place to deal with aquatic contamination is at the introduction point, the source. This is the main thrust of the "cleanup" programs under way in several Puget Sound urban bays, identification and elimination of contaminant sources to the aquatic environment. This is a reasonable sentence, as revised, and necessary approach. But there are additional things to consider:

a. Many of the chemicals we find in marine sediments (e.g., DDT and other pesticides and PCBs) are no longer produced or allowed in present industrial discharges. In Puget Sound, residual, or historic, contamination is being found even in recently deposited sediments.

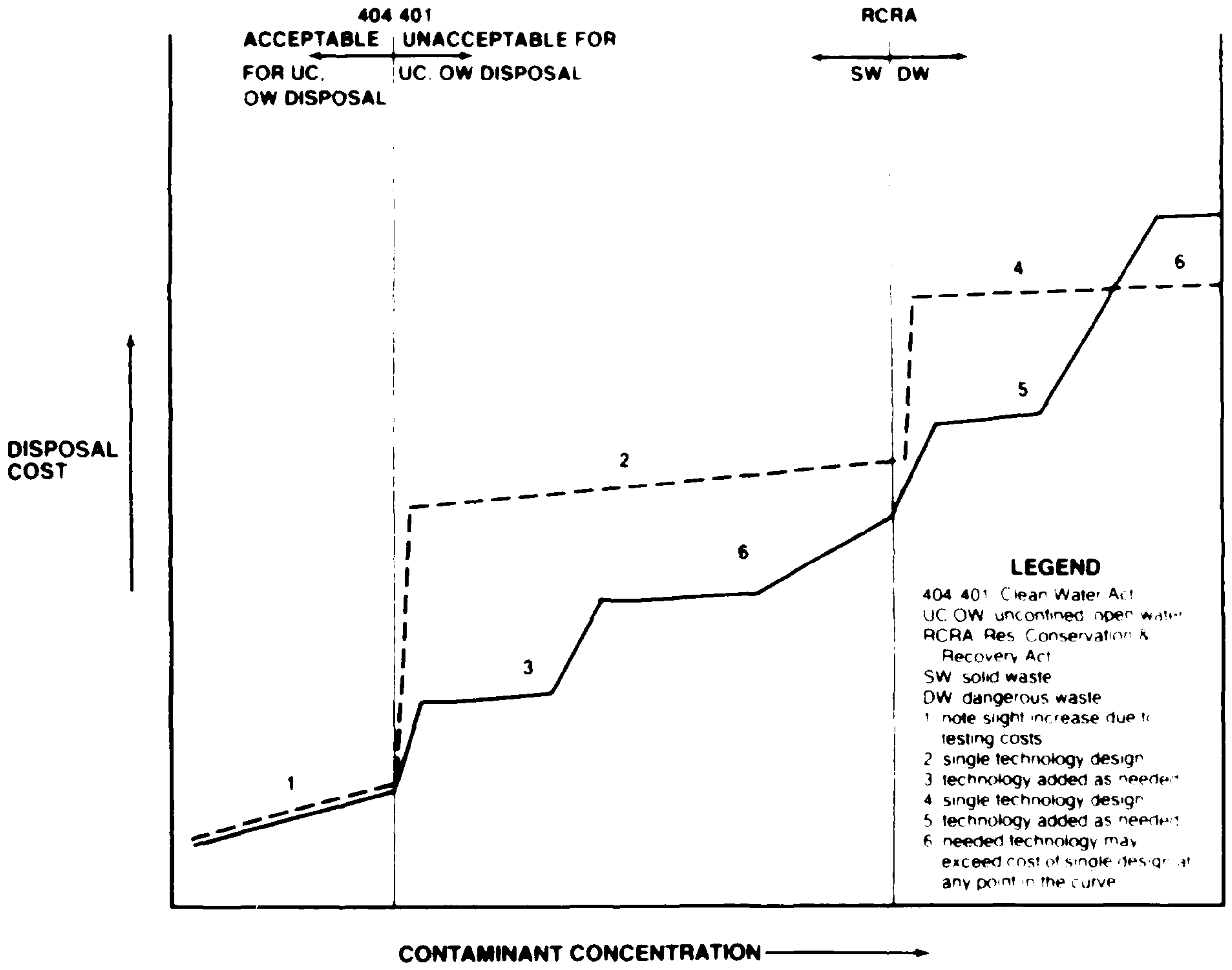

Figure 4. Cost implications of management approaches

b. Current sediment contamination in urban areas is likely the result of a chronic, long-term discharge. If contaminated sediments are removed, it may be many years before contaminated sediments are reestablished given current industrial contaminant discharges allowances. Combined with historic contamination sources, this has significant implication to the relative cost effectiveness (per given unit of reduction of risk to the environment and human health) of sediment removal versus source control. Both become of comparable importance.

c. Most treatment experts agree that in the near term (decades), effluent control technology will still require residual discharge of contamination. These much lower concentrations will take proportionately longer to significantly recontaminate sediments near effluent discharge points.

d. Given treatment pathway-length limitations and the prospect of long-term residual releases, it appears logical, though perhaps radical, to extend the concept of "treatment processes" to include the sediments outside the effluent pipe as "further treatment." These sediments could be monitored and periodically removed, resulting in a more cost-effective method than some prohibitive advanced treatments, and perhaps also be more risk acceptable. Given the residuals, both better source treatment and sediment control should be considered as complementary in developing source control permits. At least it would provide some discharger responsibility beyond the pipe.

e. On the other side of this issue, natural sedimentation in Puget Sound, along with biodegradation of some contaminants, provide a conceptual, though unknown, assimilative capacity for contaminant discharges. The location of, and possible transport routes to, these "natural graves," are also unknown. But it suggests that source reduction may perhaps work as a long-term solution.

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EFFECTS OF SEDIMENT REMOVAL IN TOKYO BAY

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INTRODUCTION

Although the removal of organic sediment is conceptionally understood as an effective countermeasure against environmental pollution, few data exist concerning the extended removal effects of sediment. To study this problem, the Second District Port Construction Bureau of the Ministry of Transport conducted several surveys on organic sediment of Tokyo Bay to observe dredging effects in 1982-1984. This paper reports the findings of these surveys.

TEST LOCATIONS AND TEST APPARATUS

The test locations given in Figure 1 (A, B, and C) have three different removal depths.

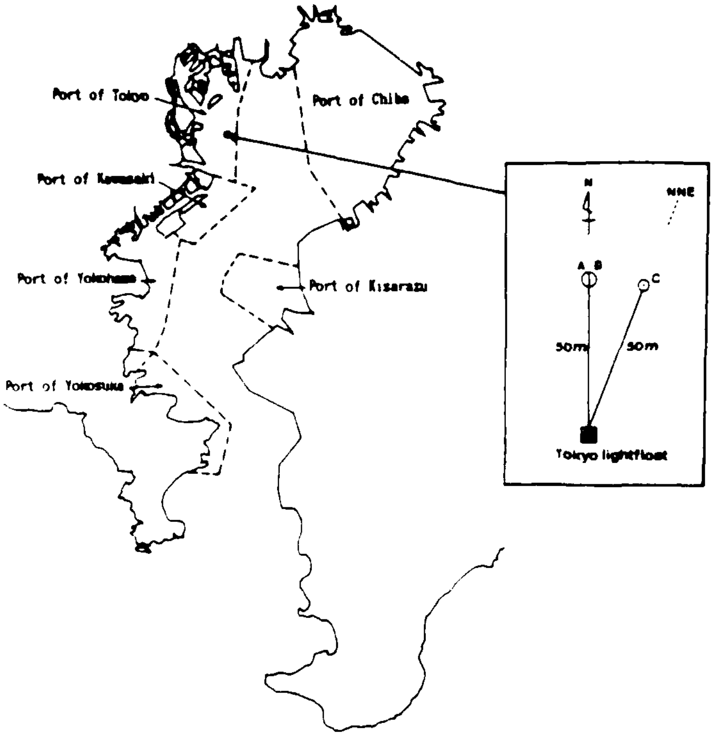
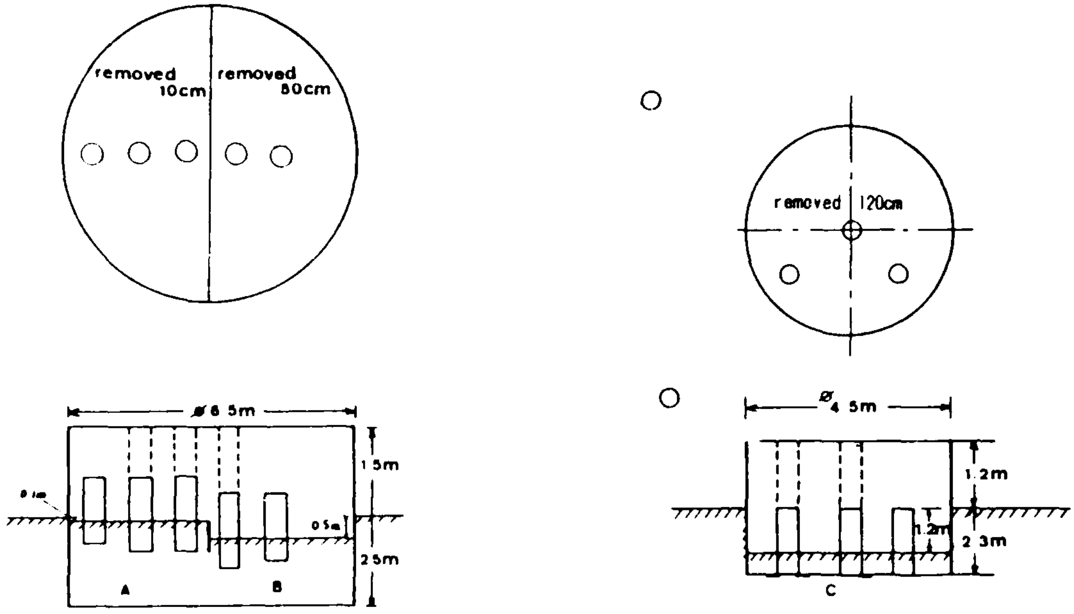


Figure 1. Test locations

The first test was conducted in 1982 at C, where a sediment depth of 120 cm was removed inside a steel circular frame of 4.5 m Ø , as shown in Figure 2. Three release test pipes, one of which has a cover, were inserted inside the frame and two pipes, one of which has a cover, were outside the frame. When not in use, the release test pipes without caps were extended to the surface of the frame by additional pipes.



Test Points A and B

Test Point C

Figure 2. Test points A, B, and C

The second test was performed in 1983 at A and B, where a sediment depth of 10 cm was removed from half of a steel circular frame of 6.5 m Ø (area A) and 50 cm from the other half (area B). Five release test pipes were inside the frame.

The test pipes without caps were lengthened when not in use in the same way as those in the 1982 test.

RELEASE TEST PIPES/IN SITU

Each release test pipe (both with and without caps) had a diameter of 500 mm and a length of 1,500 mm (Figure 3). Each pipe was inserted into the sediment to a depth of 500 mm. Test procedures are illustrated in Figure 4.

In Case I, in which some amount of floating sediment flows into the pipes, the durability of sediment removal is observed. In Case II, in which no floating sediment flows into the pipes, the dredging effects against time are measured. In Case III, in which floating sediment has no contact with the sediment, the release rates from floating sediment are measured.

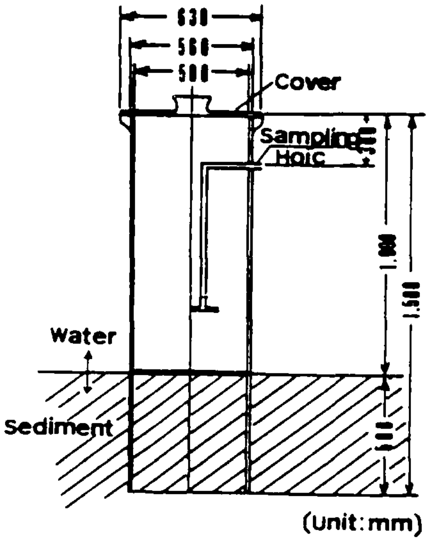


Figure 3. Release test pipe

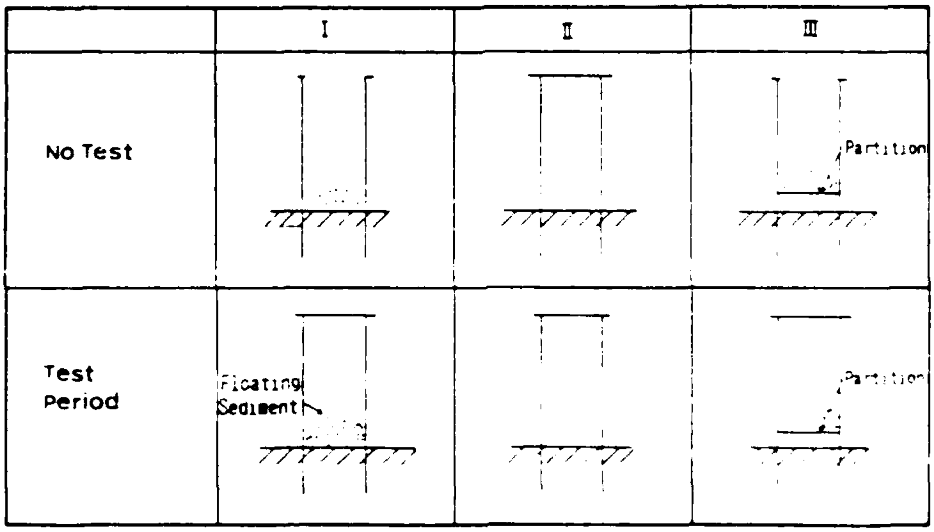


Figure 4. Test procedures

CHANGE OF POLLUTANT CONCENTRATION IN SEDIMENT

As implied by the vertical distribution of pollutants in sediment (Figure 5), it is natural that the pollutant concentration in sediment decrease after sediment removal. Figure 6 shows two examples. The removal of 50 cm of sediment was carried out in September [illegible]. Measurements began in September 1983 and were conducted seven times during a year. The pollutant concentrations in the upper layers of sediment are increasing gradually with time. The differences in pollution concentrations outside and inside the test frame are diminishing with time. This may be due to the precipitates which settled on the sediment during the observation period.

Figure [illegible] indicates the reduction of pollutant concentrations after sediment removal. The reduction of sulfide and T-N is large, but for T-P is small.

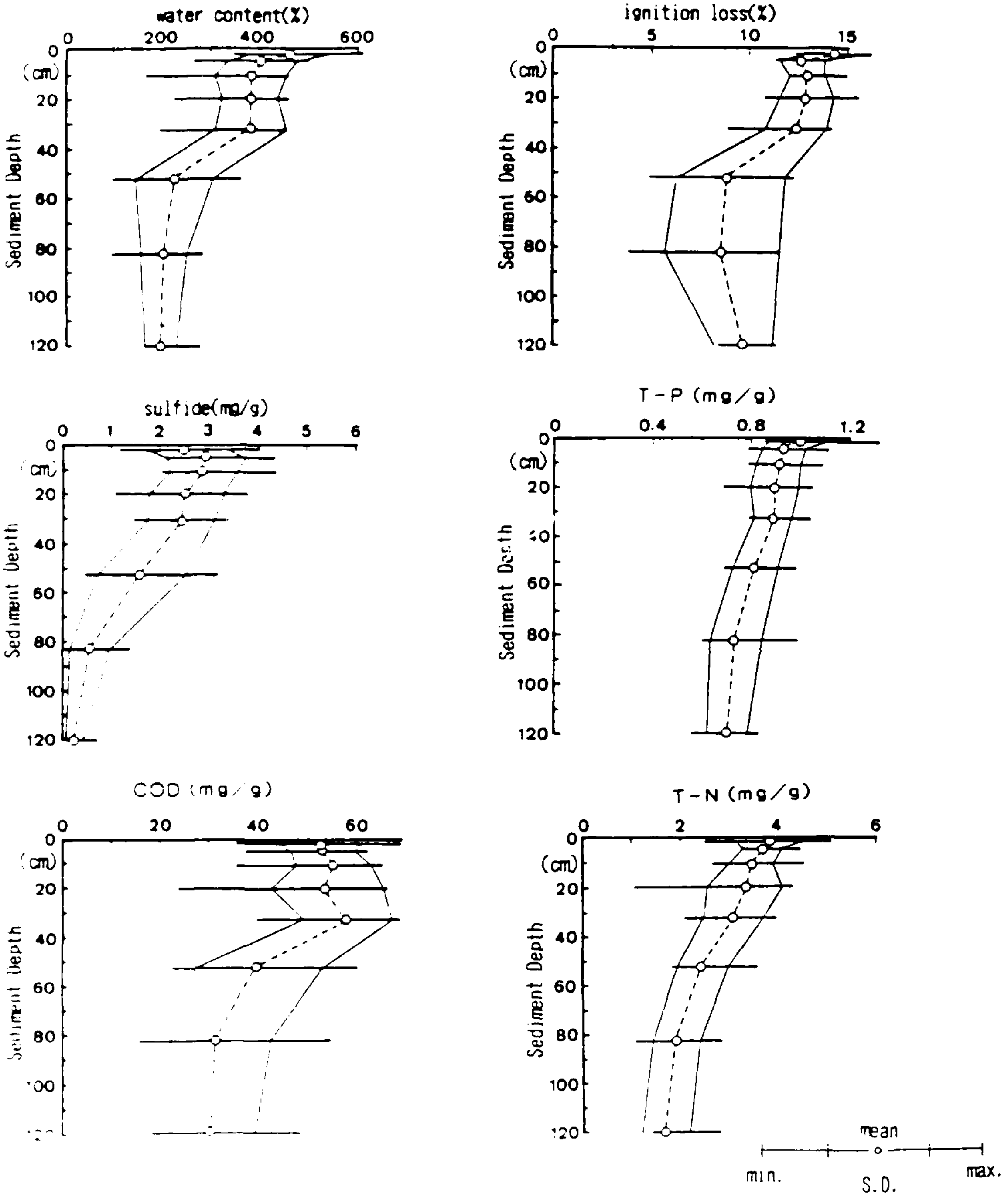


Figure 5. Changes in pollutant concentrations versus sediment depth

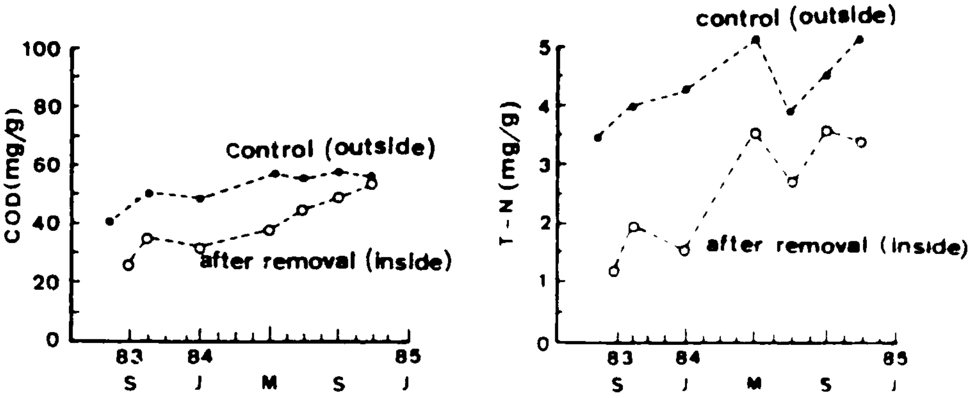
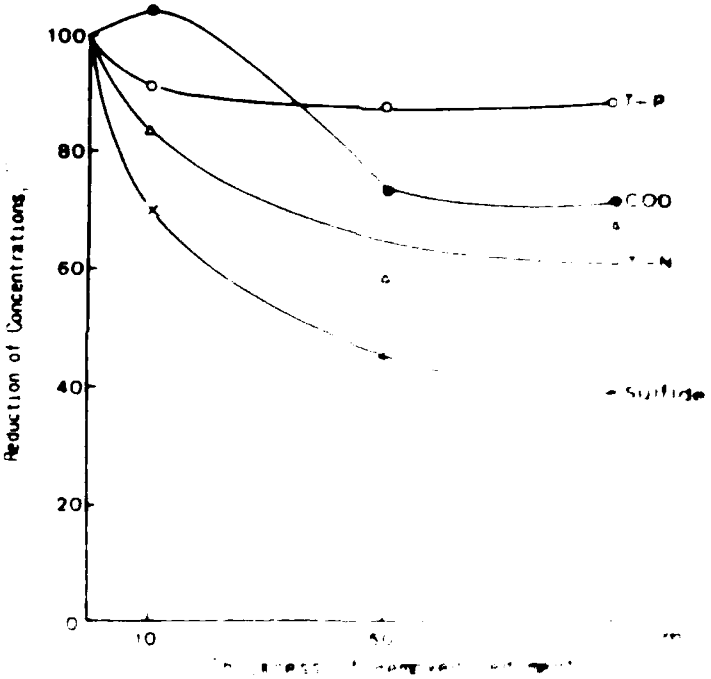


Figure 6. Change in pollutant concentration with time
(inside and outside the test frame)



In the [illegible] were [illegible] outside [illegible] was most [illegible] data [illegible]

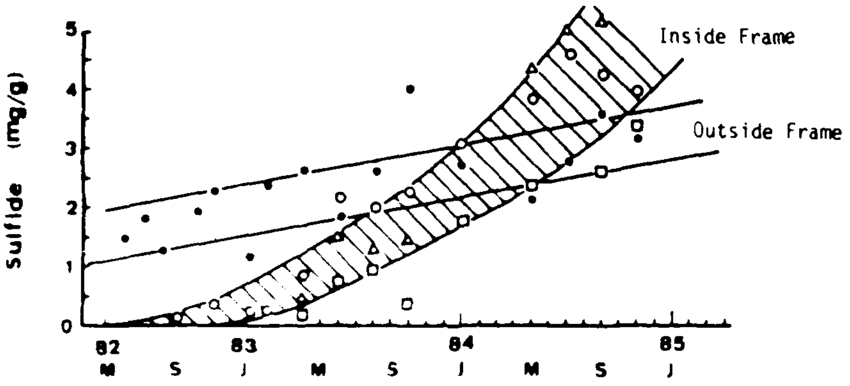
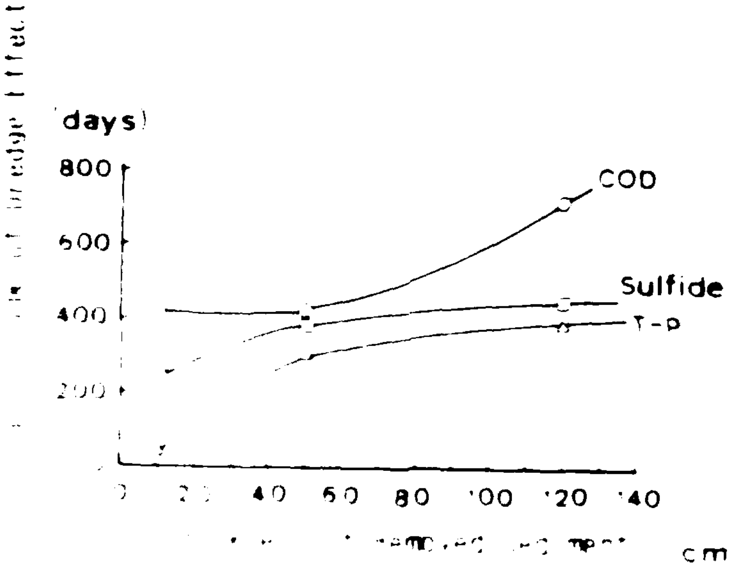


Figure 8. Change in sulfide concentration with time



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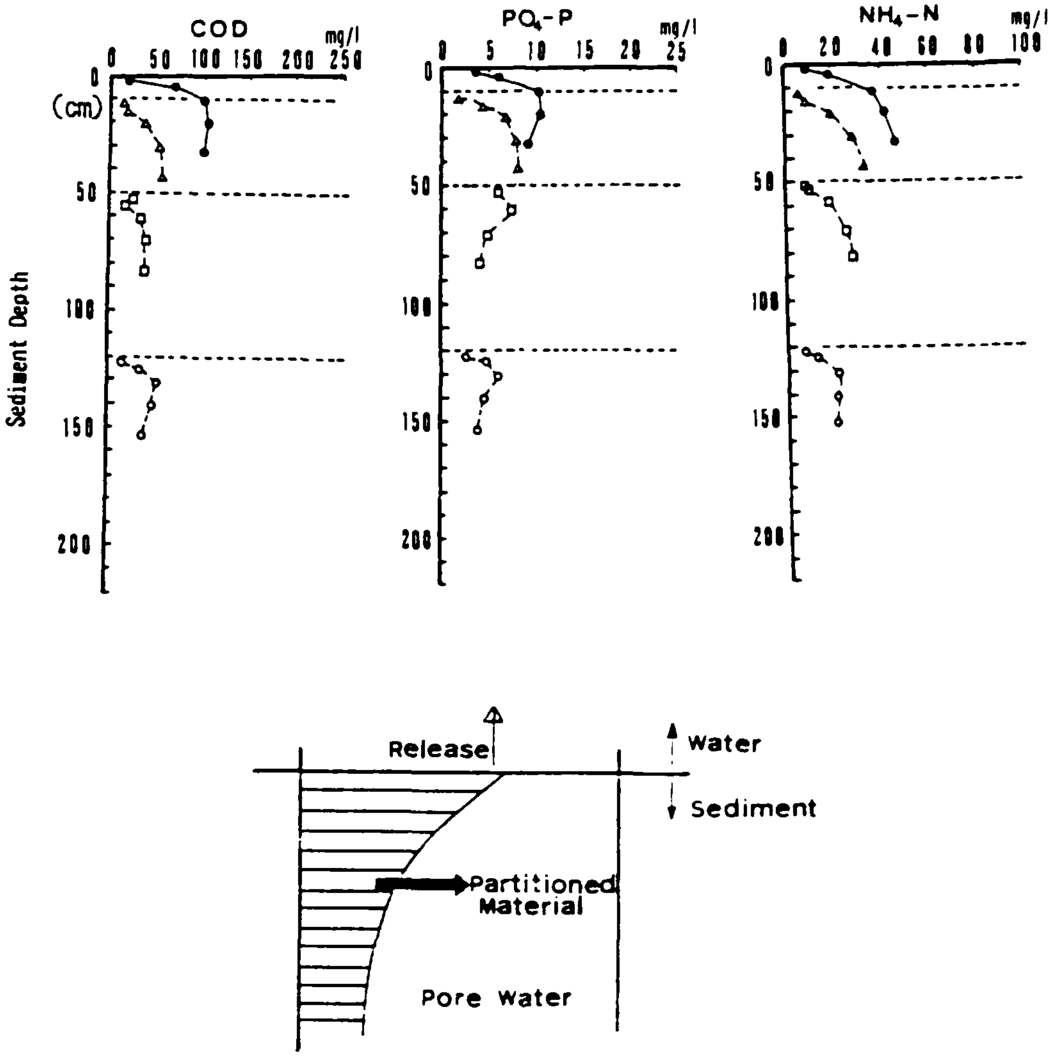

Figure 10. Change in pollutant concentrations in pore water

CHANGE IN RELEASE RATES

Figure 11 shows the release rates in test pipes with and without caps outside the test frame. The release rates in test pipes without caps are much larger than the ones in pipes with caps. This may be caused by foreign material such as newly settled solids or floating sediment which moved into the pipes.

Therefore, the measured data should be analyzed considering such effect of foreign material.

Figure 12 shows the reduction of release rates in test pipes with caps according to the removed depth. In this case the effects of sediment removal [illegible] the effects of foreign material. The effects of sediment removal [illegible] [illegible] thicker sediment removal.

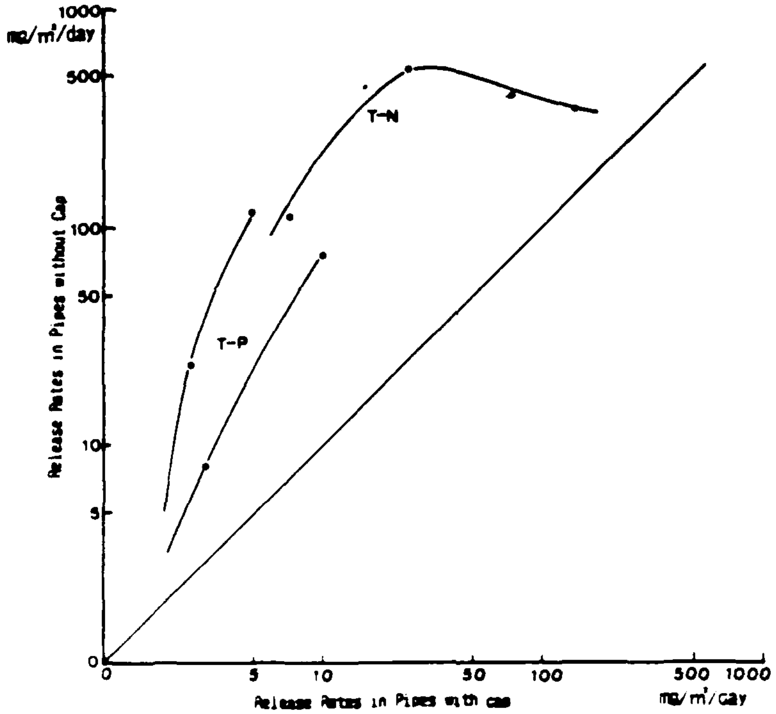

Figure 11. Release rates in pipes with and without caps outside test frame

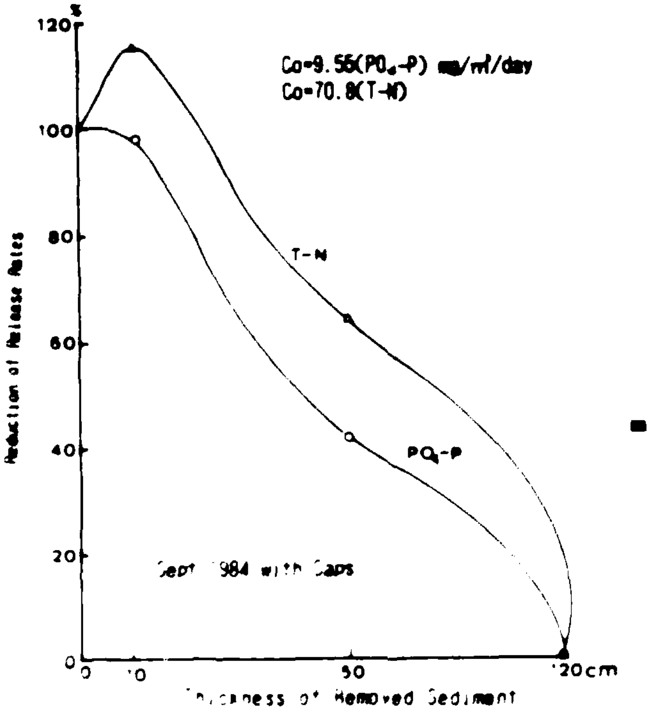

Figure [illegible]. Reduction of release rates by sediment removal

Figure 13 shows the reduction of release rates in test pipes without caps. In test pipes without caps the release rates decrease to a smaller extent than the pipes with caps. As mentioned before, this may be caused by foreign material in the test pipes.

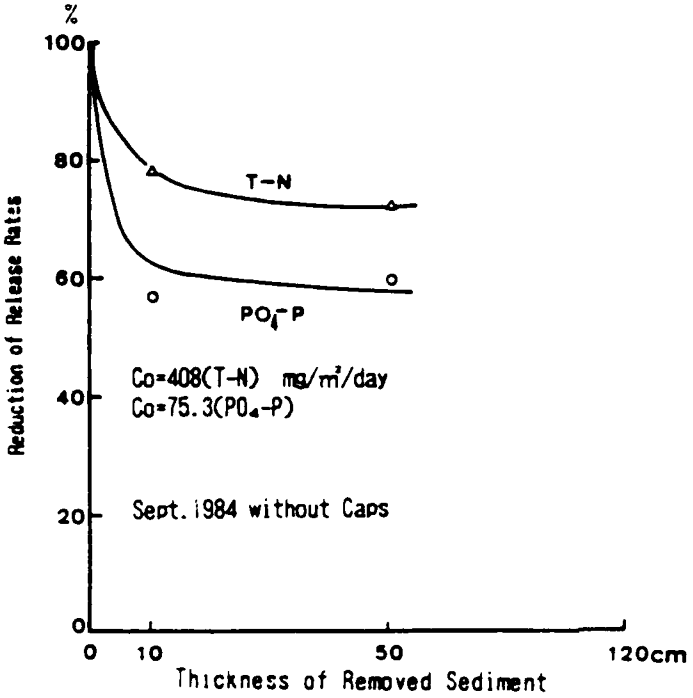

Figure 13. Reduction of release rates in pipes without caps

The measurements at test points A and B were conducted 12 months after removal, and at C, 24 months after removal. Therefore, at the start of the tests much larger quantities of foreign material had flowed into the test pipes of C than those of A and B.

As Figure 14 shows, the release rates of PO_4-P at C is about four times larger than the original. It is no wonder that the foreign material which flowed into the pipes during 24 months manifested a big release. The release rates of test point C are believed to result from the foreign material which flowed into the pipes before the tests. This means that the foreign material is rich in pollutants.

The levels of PCB were used to determine the amount of foreign material which had flowed into the steel frame. PCB is relatively stable in an environment and is scarce in the lower layer of natural sediment (Figure 15). Therefore, little PCB was detected at test point C immediately after dredging. After 3 months, however, some PCB was observed in the 0- to 1-cm layer and, at present, is observed in the 10- to 11-cm layer. This suggests that foreign material does flow into the test point C frame.

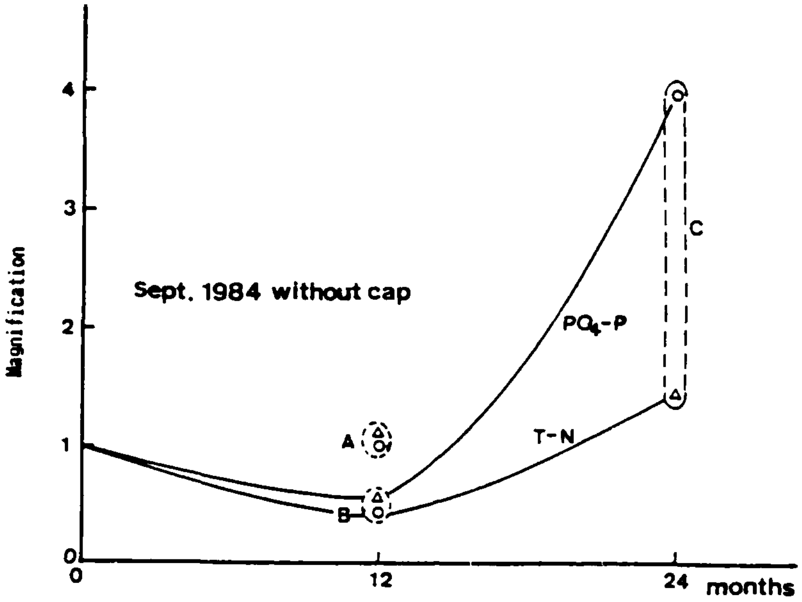

Figure 14. Differences in release rates between A, B, and C

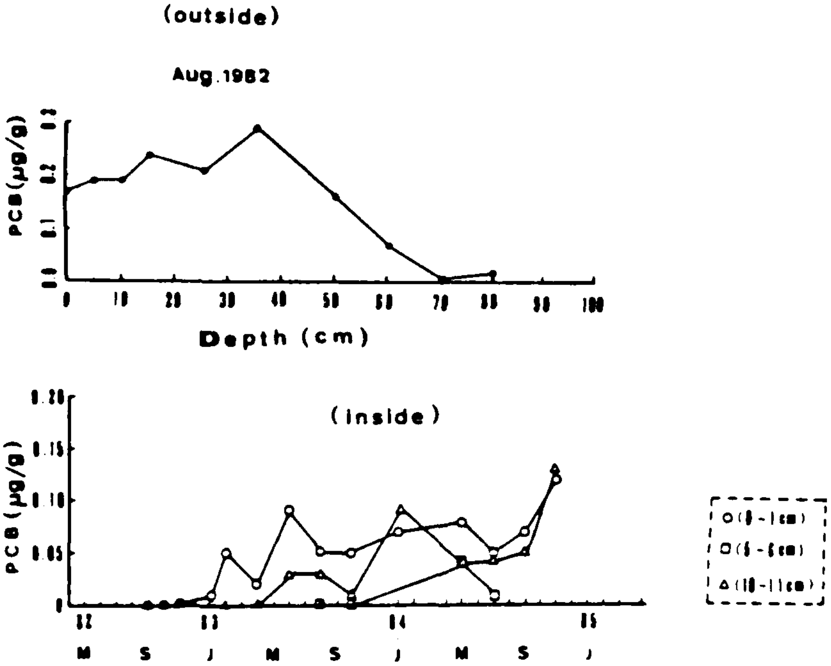

Figure 15. Change in PCB concentration in sediment test (point C)

EFFECTS OF SEDIMENT REMOVAL

Nutrient concentrations in sediment increase gradually with time after sediment removal under the influence of foreign material such as newly precipitating matter or suspended sediment. This gives the impression that no restorative effects can be expected from sediment removal. However, it should be studied from the viewpoint of release rates. Figure 16 shows the change in release rates with time. As Figure 16 (Case II) indicates, the release rates increase with time as the foreign material accumulates, and result in an early increase of release rates, thereby cancelling the removal effects.

Figure 16 (Cases I and III) indicates the release rates excluding the released volume from foreign material. Although some variations are found, release rates generally decrease.

It is believed that the decrease in release rates in the early stages may result from diffusion into the underlying sediment because the underlying sediment is clean due to removal of the polluted material. Diffusion into the underlying sediment continues until both the concentrations of release material from foreign materials and in sediment pore water become equal. After that the release rates increase again.

Figure 17 shows the relationship between dredging depth and release rate of COD, PO_4-P, and T-N released from the bed for 1 year after the dredging. Both cases (with and without cap) show that the release rates decrease with increasing thickness of the sediment removal. This feature is much clearer in the case with the cap.

CONCLUSIONS

These tests were conducted to study the restorative effects of sediment removal. At sea there are many conditions such as tides, newly precipitating material, and suspended sediment. These conditions cause complicated sea conditions that require much time and plenty of data to obtain a decisive opinion.

Our tests faced this problem. In spite of the unsatisfactory conditions such as short test times and small test spaces, the test results gave us some valuable data.

In summary, we succeeded in determining the effects of sediment removal on reducing release rates. An even more significant finding of this study is that foreign materials accumulating on conventional sediment have a great influence.

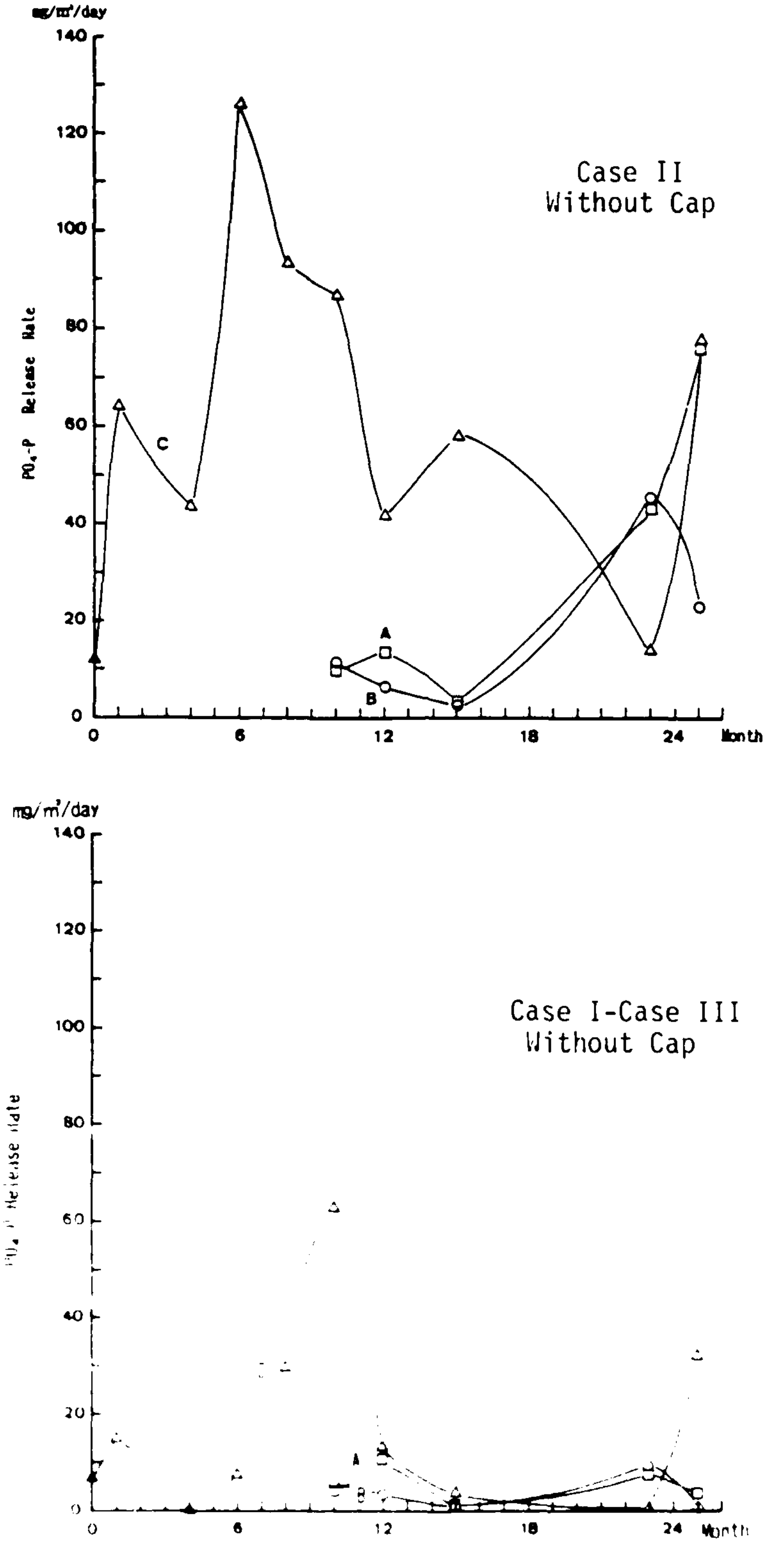

Figure [illegible]. Change in release rates versus time

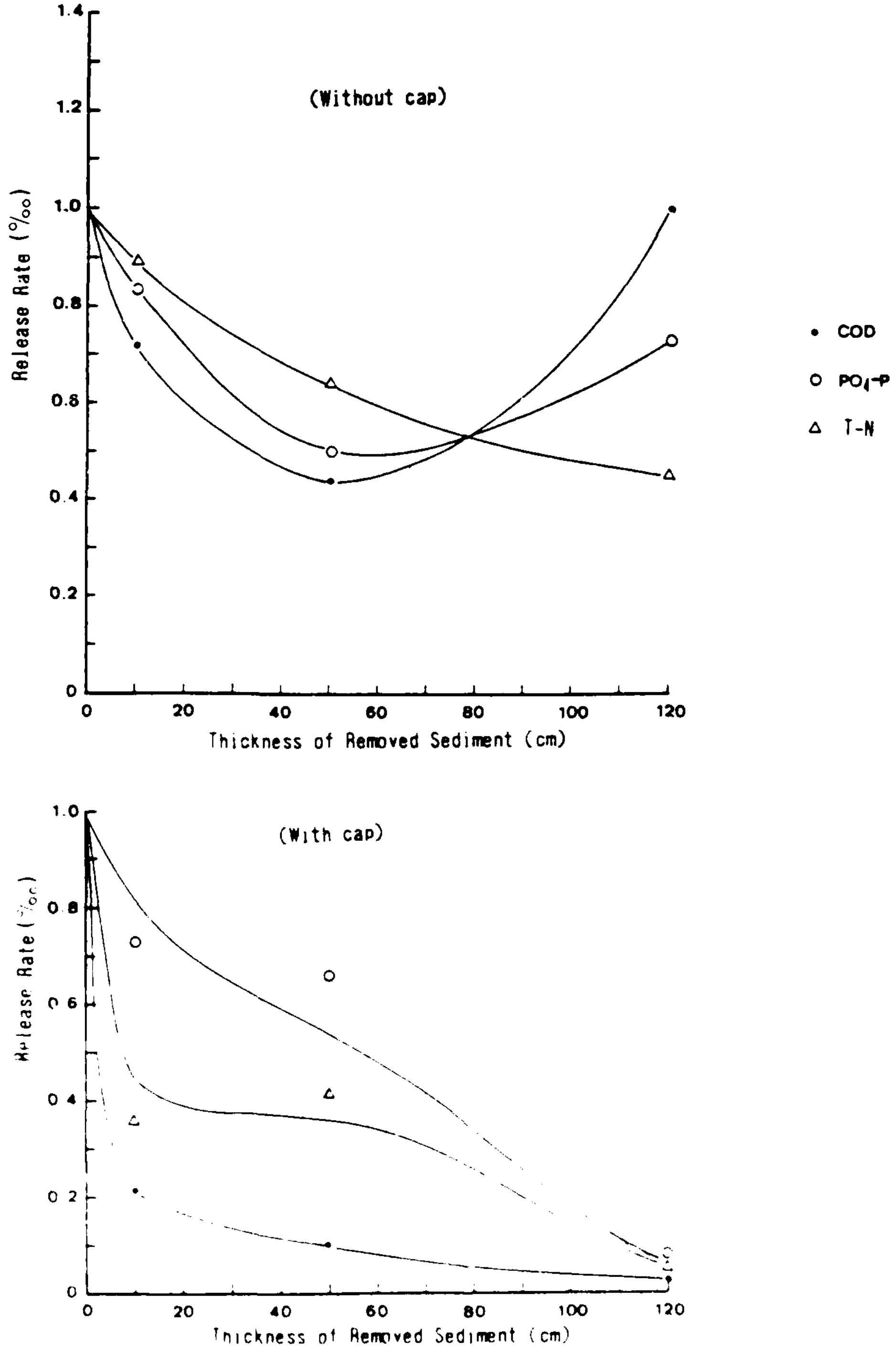

Figure 17. Reduction of the annual amount of released COD, PO_4-P, and T-N due to sediment removal (release rates are normalized by the values before dredging)

DEVELOPMENT OF LEACHATE TESTING PROTOCOLS FOR DISPOSAL OF CONTAMINATED SEDIMENTS

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ABSTRACT

A testing protocol is described herein for evaluating desorption characteristics of contaminants from dredged material. The procedure relies on comparing the effluent trace predicted by the one-dimensional advection-dispersion equation containing an equilibrium source term with that from an operating laboratory column. System specific parameters (v, ρ, θ, D_p, K_d) required to solve the transport equation are evaluated separately. Such comparisons provide insight into the mechanisms responsible for contaminant desorption.

The procedure was applied to a highly contaminated sediment from Indiana Harbor, Indiana. Preliminary leaching data for two heavy metals, arsenic and cadmium, can be interpreted as film-controlled desorption rather than equilibrium partitioning.

INTRODUCTION

Bottom sediments contaminated with organic matter, heavy metals, polychlorinated biphenyls, and polyaromatic hydrocarbons are present in most urban waterways. When contaminated dredged material is placed in a confined disposal facility (CDF), the potential exists to generate leachates that may have adverse impacts on ground waters. When the potential for adverse environmental impacts exists, disposal of contaminated material must be planned to limit these impacts by restricting contaminant mobility. To design facilities and systems necessary to satisfy site-specific requirements for environmental protection, information is needed on potential contaminant mobility. Lacking specific quantitative information on contaminant mobility, project engineers are forced to adopt contaminant containment strategies that are possibly more conservative than necessary, resulting in increased costs without a commensurate increase in environmental protection. If leachate quality and generation rates can be predicted, the potential impacts of using a CDF for disposal of contaminated dredged material can be determined, allowing the most cost-effective site design to be utilized.

At present, there is no routinely applied laboratory testing protocol capable of predicting, or even approximating, leachate quality from confined dredged material disposal sites. Testing procedures to predict leachate quality are therefore needed to fully evaluate the confined disposal alternative for contaminated dredged material. In order to predict leachate quality with any degree of confidence, information concerning the fundamental processes controlling contaminant transfer from the sediment solids to the aqueous phase is needed.

SCOPE

This paper describes research that is being conducted by the US Army Engineer Waterways Experiment Station to develop testing procedures for evaluating desorption mechanisms that control leaching in contaminated sediments. The theoretical basis for understanding leaching of dredged material is briefly described, and selected findings from an ongoing study are presented.

TECHNICAL APPROACH

Hill, Myers, and Brannon (1985) have presented a theoretical basis for understanding the leaching of dredged material. Several models based on mass transport theory were considered for describing contaminant transfer from the dredged material solids to the aqueous phase. Mathematical descriptions of equilibrium, film diffusion, intraparticle diffusion, reversible reaction, irreversible reaction, and solubility limited, kinetic models were developed. The one recommended by Hill, Myers, and Brannon (1985) for application to dredged material uses an equilibrium distribution (partitioning) coefficient to relate aqueous phase concentration to sediment phase concentration.

The approach described below uses numerical as well as experimental results to test the assumption that leaching is governed by equilibrium-controlled desorption. Sequential batch leach tests are used to estimate a distribution coefficient for each contaminant of interest. The distribution coefficient is considered an intrinsic parameter that describes contaminant release from the sediment. This parameter is then used in a one-dimensional mass transfer equation containing an equilibrium source term to predict effluent quality from a continuously leached column. Interpretation of batch and column data is based on a comparison of the predicted curve to the observed data. Hence, this is an integrated approach that combines batch data, column data, and mass transport theory. A schematic of the integrated approach is shown in Figure 1.

The one-dimensional mass transport equation describing contaminant transport in confined dredged material is

$$D_p\left(\partial^2 C_i/\partial Z^2\right) - V\left(\partial C_i/\partial Z\right) - \left(\rho/\theta\right)K_d\left(\partial C_i/\partial t\right) = \partial C_i/\partial t \qquad (1)$$

where

D_p = bulk dispersion coefficient for contaminant i, L^2/t

C_i = aqueous phase contaminant concentration, M_c/L^3

Z = direction

V = average pore water velocity, L/t

ρ = sediment bulk density, Ms/L^3

θ = volumetric water content of sediment, L^3/L^3

K_d = slope of desorption isotherm, L^3/M

t = time

Details of the development of this equation have been described by Hill, Myers, and Brannon (1985). If the laboratory leaching column is short, it may be possible to assume that the liquid within the pores remain completely mixed, thus allowing Equation 1 to be simplified and written as the ordinary differential equation, shown below:

$$dc_i/dt = -(Q/V_v)\ C - \left(\rho/\theta\right)K_d(dc_i/dt) \tag{2}$$

where V_v = volume of voids, L^3. Equation (2) has the solution

$$\ln\left(C/C_o\right) = -V/\theta L\left(\frac{1}{1 + \frac{\rho}{\theta}\,K_d}\right) t \tag{3}$$

where

C_o = contaminant concentration in the pore water at $t = 0$, Mc/L^3

L = column length, L

The bulk density ρ, void volume V_v, and volumetric water content θ are obtained using standard soil test procedures. The dispersion coefficient D_p is obtained using continuous flow column tests as described by Levenspiel (1972). The pore water velocity and column length L are obtained from column operating records. The initial contaminant concentration C_o is obtained by chemical analysis. The desorption coefficient K_d is computed independently of the column using sequential batch testing. After these data have been obtained, Equations 1-3 can be solved, and the resulting trace of contaminant concentrations versus pore volume can be compared with measured effluent concentrations from an operating column. The use of such a protocol implies the following:

a. The experimental column, termed a permeameter and shown in [illegible], is designed and operated such that the flow is essentially [illegible] dimensional.

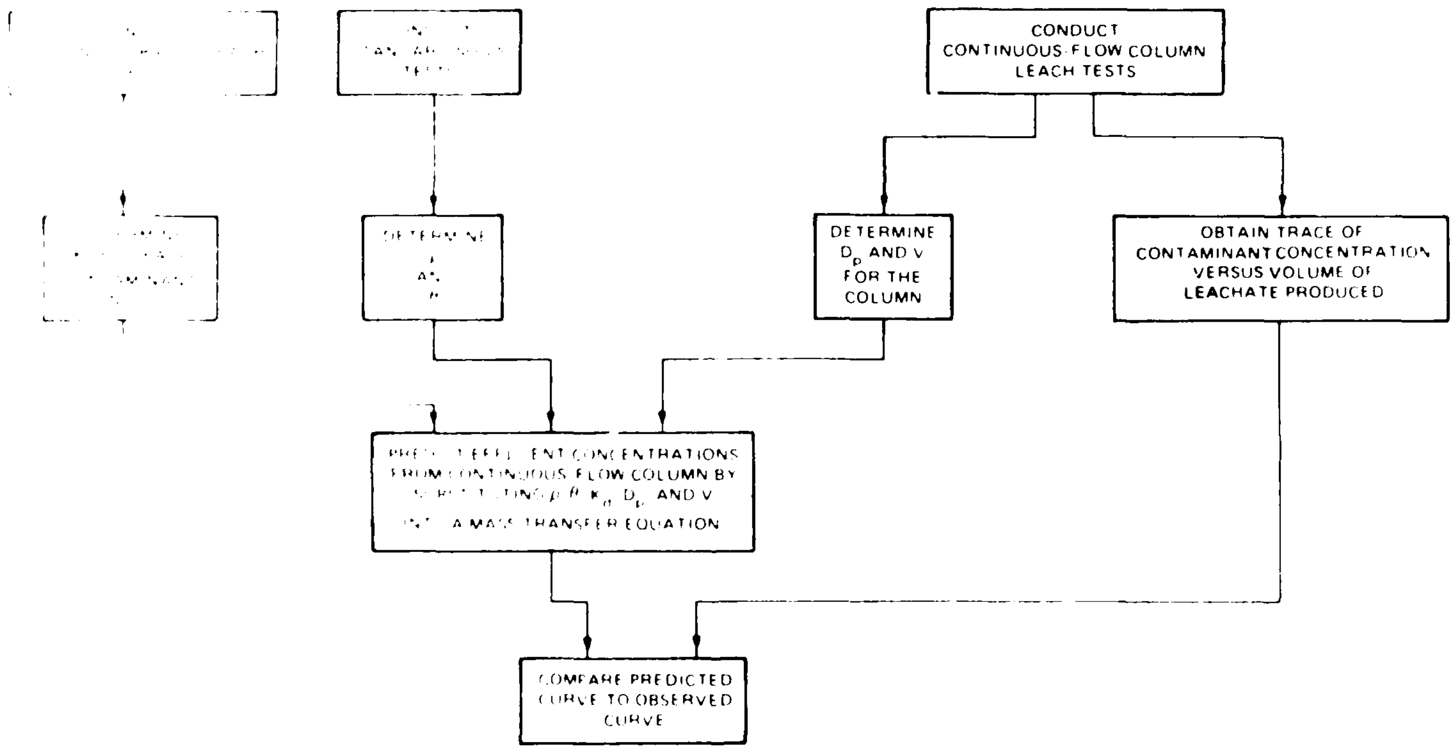

Figure 1. Schematic of integrated approach

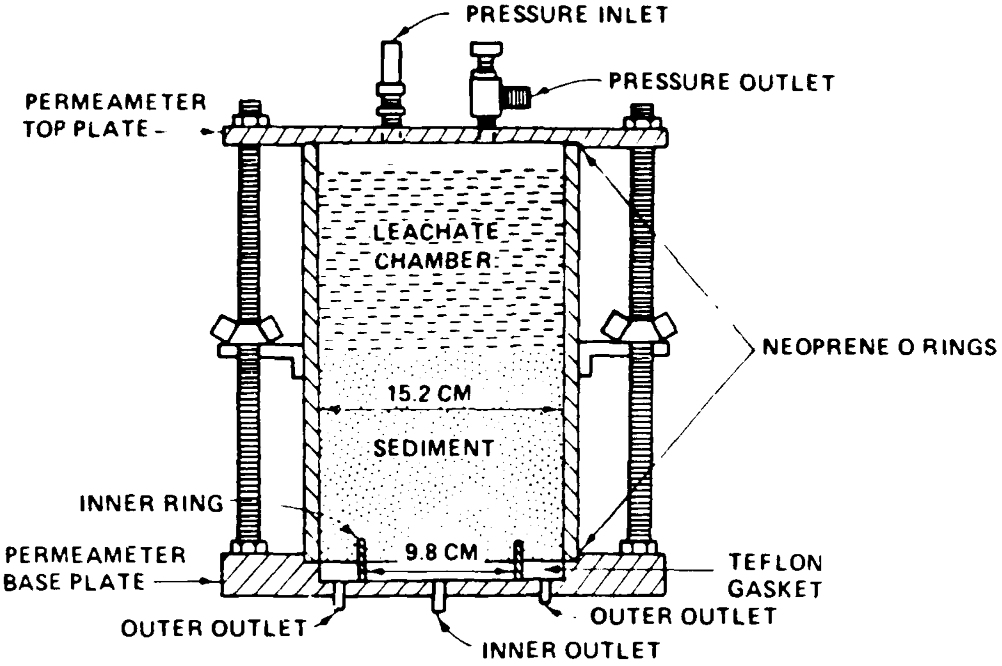

Figure 2. Divided-flow permeameter used for continuous flow column testing

b. Column test conditions used to obtain the dispersion coefficient D_p are identical to those in the laboratory column.

c. The physical-chemical mechanisms governing desorption are the same in batch tests as in continuous flow systems, thus allowing the use of a K_d determined from batch studies in Equations 1-3.

d. The degree of dredged material saturation and consolidation in the column is homogeneous and does not vary during the test or is known and can be accounted for.

A SPECIFIC APPLICATION

The integrated approach is presently being used (October 1985) to assess the assumption of equilibrium-controlled desorption of contaminants from a sediment obtained from Indiana Harbor, Indiana. In some reaches of the Indiana Harbor Canal, the sediments are heavily contaminated with oil and grease, organics, and heavy metals.

Current testing involves using sediment maintained under anaerobic conditions. Contaminants of interest are the heavy metals As, Cd, Cr, Pb, and Zn as well as selected organics. Prior to making comparisons, a value of K_d is determined for each contaminant (Figure 1). The K_d is the slope of a desorption isotherm constructed from a table of equilibrium values for solid phase contaminant concentration q and aqueous phase contaminant concentration c. Various solid and corresponding aqueous phase concentrations are obtained by sequentially leaching the sediment in batch tests as outlined in Table 1. A typical desorption isotherm is shown in Figure 3.

TABLE 1. TEST SEQUENCE FOR [illegible]
TESTING OF ANAEROBIC [illegible]
FOR METALS AND ORGANIC [illegible]

Step 1	Load sediment into appropriate centrifuge [illegible] polycarbonate for metals and [illegible] contaminants. Add sufficient water [illegible] water to sediment ratio of [illegible]. Sufficient [illegible] tubes must be loaded to obtain enough leachate [illegible] for use in leaching fresh sediment.
Step 2	Shake horizontally at 160 cycles per minute [illegible]
Step 3	Centrifuge for 30 min at 6,500 [illegible] organics and [illegible] metals. Prior to filtering, centrifuged leachate [illegible] through acid-washed glass wool for metals and [illegible] glass wool for organics. Samples for organic [illegible] repetition on step 3 using clean stainless steel centrifuge tubes to remove oil.
Step 4	Acidify leachate for organic analysis with [illegible] and leachate [illegible] metals analysis with ultrex nitric acid. Store leachate for organic analysis in acetone-rinsed glass bottles and leachate for metals analysis in plastic bottles.

Note: The anaerobic integrity of the sample was maintained during sample addition to centrifuge tubes, shaking, centrifugation, and filtration

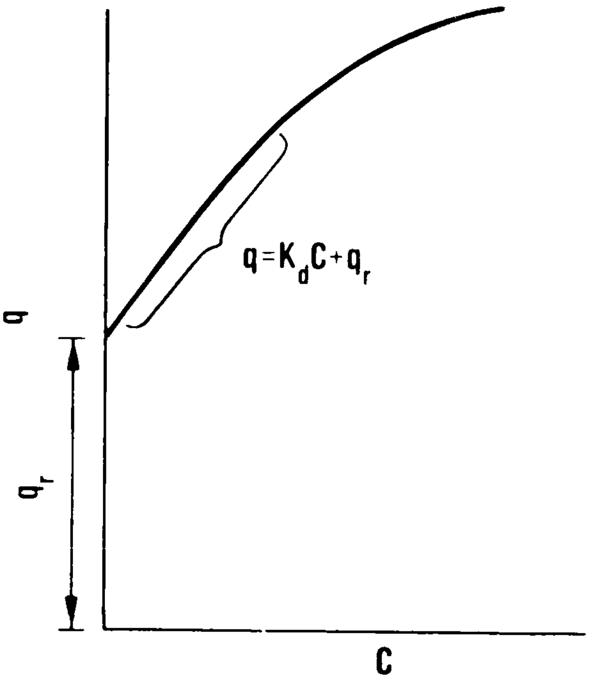

Figure 3. Typical desorption isotherm

However, obtaining reliable data with which to construct desorption isotherms for Indiana Harbor sediment was hampered by the oil and grease in the sediment. The vigorous shaking used in the batch tests separated much of the oil from the sediment solids, resulting in a water-oil and grease emulsion. It was necessary to separate the oil and grease from the leachate prior to contaminant analysis in order to obtain K_d values indicative only of sediment water interactions. As shown in Table 1, for metals this required centrifugation followed by passage through glass wool in order to avoid clogging of the 0.45-μm membrane filter used in phase separation.

Initial metal concentrations in the pore water were determined by centrifuging the sediment at its natural moisture content and then filtering the supernatant through a 0.45-μm membrane filter. The bulk sediment concentrations, initial pore water concentrations, and equilibrium distribution coefficients for arsenic and cadmium are presented in Table 2.

TABLE 2. BULK SEDIMENT AND INITIAL PORE WATER ARSENIC AND CADMIUM CONCENTRATIONS AND EQUILIBRIUM DISTRIBUTION COEFFICIENTS FOR INDIANA HARBOR SEDIMENT

Metal	Bulk Sediment mg/kg dry wt.	C_o mg/ℓ	K_d ml/g
Arsenic	29.5	0.0137	7.47
Cadmium	20.0	0.0047	2.87

Application of the integrated approach is illustrated in Figure 4. The points shown in Figure 4 are measured contaminant concentrations from three column effluents. The curves in Figure 4 result from plotting Equation 3. In one instance desorption is assumed to be an equilibrium-controlled process characterized by the desorption coefficient, K_d, for that particular contaminant. The other curve assumes no desorption ($K_d = 0$). For both arsenic and cadmium, experimentally determined concentrations are lower than predicted by Equation 3 using the appropriate value of K_d. In fact, the points appear to lie along the curves which assume no desorption ($K_d = 0$). It appears that once the original interstitial liquid is pushed from the column, desorption of arsenic and cadmium effectively stops. This could be due to an oil and grease coating on the sediment particles that inhibits contaminant transfer from the sediment to the aqueous phase.

The contaminant concentration pore volume trace shown in Figure 4 does not account for consolidation (pore volume decrease) that occurs during the column tests. After the column leach tests are terminated, the sediment profile will be examined for consolidation, and the effects of consolidation can then be taken into account. If consolidation has been significant, the observed points shown in Figure 4 will be shifted to the right. In addition,

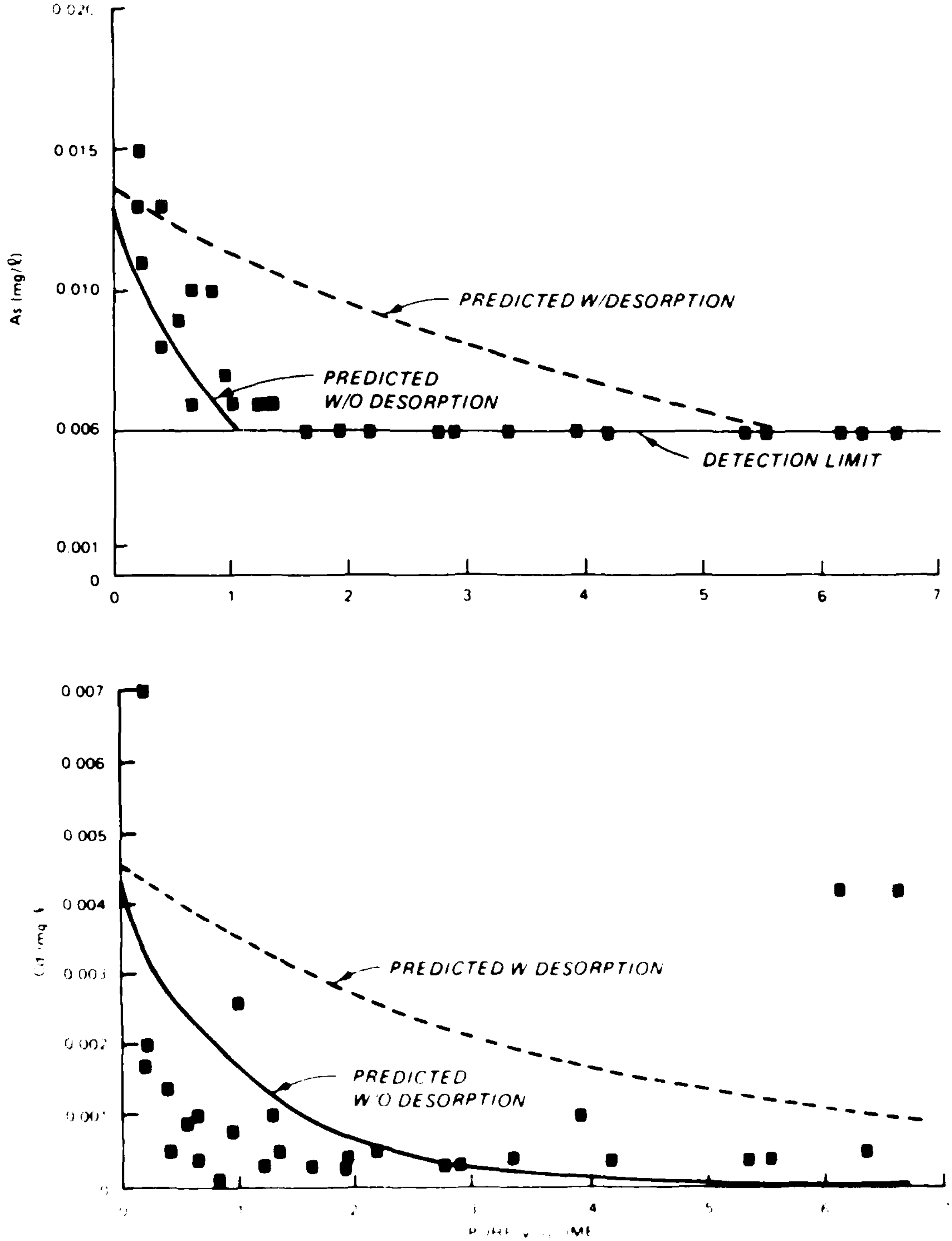

Figure 4. Comparison of arsenic and cadmium concentrations in column effluent with predicted values

tests are in progress to determine the dispersion in the column. The availability of this information will aid in the ongoing interpretation of column desorption data.

SUMMARY

An integrated approach to leachate testing involving batch and column leach tests and mass transport equations was described. The integrated approach appears to be a useful tool for characterizing the process that governs contaminant leaching from sediment solids. Preliminary results suggest that leaching of arsenic and cadmium from Indiana Harbor sediment is governed by an oil and grease film rather than equilibrium partitioning between the sediment and aqueous phase.

ACKNOWLEDGEMENTS

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METHOD FOR ESTIMATION OF METHYLMERCURY DISSOLVED INTO WATER FROM DREDGED MATERIAL BY USING THE MARINE COPEPOD *ACARTIA CLAUSI*

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ABSTRACT

Floatable marine copepods (*Acartia clausi*) were reared for 24 hr in seawater containing sediment. The sediment contained 0.002 to 0.003 ppm of methylmercury and 32.47 to 59.01 ppm of total mercury. Before the start of the rearing experiments, the sediments were irradiated with sunlight for 6 hr a day for 23 days. The population density of the copepods was 10 to 41 individuals per litre. The copepods were collected and methylmercury and total mercury in copepods were measured. Then dissolved methylmercury concentration in the seawater was estimated by the use of the correlation between the bioconcentration factor of methylmercury and population density of copepods, and by the use of methylmercury concentration in the copepods. The concentrations of methylmercury and total mercury in the copepods increased by 0.0045 to 0.17 ppm and 0.7 to 1.706 ppm, respectively. The estimated concentrations of dissolved methylmercury were 0.000012 to 0.00026 ppb. These concentrations correspond to 1/100 to 1/6 of methylmercury contents in the sediments used in the experiments. From the experiments, it was concluded that short-term sunlight irradiation and short-term exposure of aerobic condition did not form methylmercury from inorganic mercury in the sediments, but dissolved methylmercury and inorganic mercury into the seawater from the sediments.

INTRODUCTION

In dredging it is very common for the dredged material to contain toxic substances. When inorganic mercury exists in the dredged material, it is feared that inorganic mercury changes into methylmercury in the sediments by the irradiation of sunlight and aerobic condition (Fujiki and Tajima 1974, 1976). If methylmercury is formed in the dredged material, methylmercury will go into the supernatant of the water and then the river or seawater will be polluted by the methylmercury. Therefore, monitoring of the supernatant water from dredged sediment ponds is indispensable. However, the bioconcentration factor of methylmercury is predominant. Granted that the methylmercury accumulation from water into fish exists and is detectable, the methylmercury concentration in the water is so slight that it is undetectable.

Hirota et al. (1982) reported that the negative correlation existed between the bioconcentration of methylmercury and the density of copepods. The methylmercury accumulation into fish is a slow process, but the methylmercury accumulation into copepods is rapid and the methylmercury concentration in copepods reaches a plateau in 24 hr. Therefore, copepods are useful indicators for methylmercury pollution. To develop a method of estimation for dissolved methylmercury into water from the sediment containing mercury, the following experiments were conducted.

EXPERIMENTAL METHODS

Experiment 1

Floatable marine copepods (*Acartia clausi*) obtained from the fish pond at the Oyano Branch of the Kumamoto Prefectural Fisheries Experimental Station in May 1982 were used in this experiment. Filtered seawater was placed in a 500-ℓ fish-rearing tank. Bottom sediment containing mercury obtained from Minamata Bay was added to the seawater in the fish-rearing tank so that the concentration of suspended substance was maintained at about 500 ppm. Irradiation with sunlight and aeration with an air pump to the seawater in the fish-rearing tank were continued for 6 hr. Then the fish-rearing tank was removed to an indoor laboratory, and the copepods at a density of 15 to 41 individuals per litre were added to the seawater. The copepods were reared in the seawater aerated with an air pump. The water temperature was 25.6 to 26.9° C during the experiment. After 24 hr, the copepods were collected with a plankton net (mesh size was 328 μm), picked out by using a needle, rinsed briefly with distilled water, and dried in an oven at 60° C until they reached a fixed weight. The dry materials were stored in a desiccator. Then methylmercury and total mercury in the copepods were measured.

Experiment 2

Floatable marine copepods (*Acartia clausi*) obtained from the fish pond at the Oyano Branch of the Kumamoto Prefectural Fisheries Experimental Station in June 1982 were used in this experiment. Bottom sediment containing mercury obtained from Minamata Bay was placed in a 30-ℓ fish-rearing tank, and filtered seawater was added to the fish-rearing tank so that the seawater came up to the surface of the sediment. Then the fish-rearing tank was set outdoors and the sediment irradiated with sunlight for 23 days. The weather during this irradiation with sunlight was as follows: 10 days fine, 9 days fine/cloudy, and 4 days cloudy/rainy. After 23 days, the sediments irradiated with sunlight were added to filtered seawater in a 500-ℓ fish-rearing tank so the concentration of suspended substances was maintained at about 500 ppm. The fish-rearing tank was removed to an indoor laboratory, and the copepods at a density of 10 to 13 individuals per litre were added to the seawater. The copepods were reared in the seawater aerated with an air pump. The water temperature was 24.6 to 27.0° C during the experiment. After 24 hr, the copepods were collected with a plankton net (mesh size was 328 μm), rinsed briefly with distilled water, and dried in an oven at [illegible] until [illegible] weight. The dry materials were stored in [illegible] and total mercury in the copepods were measured.

Analytical Method of Methylmercury

Analysis of methylmercury in the copepods and the sediment was conducted as follows: copepods were soaked in [illegible] % hydrochloric acid and allowed to stand at room temperature for [illegible] days, after which the solution was filtered, and the methylmercury contained in the filtrate extracted into benzene. The benzene layer containing methylmercury was transferred to another separating funnel, and methylmercury in the layer was extracted into 0.[illegible] percent glutathion solution. The glutathion solution was transferred to an ashing flask to which concentrated sulfuric acid and 4 percent potassium permanganate solution were added; the flask was then heated. After ashing, 10 percent hydroxylamine hydrochloride solution was added to reduce the potassium permanganate and manganese dioxide. Mercury in the solution was measured by flameless atomic adsorption spectrophotometry.

Analytical Method of Total Mercury

Analysis of total mercury in the copepods and the sediments was conducted as follows: copepods were placed in an ashing flask, concentrated nitric acid was added, and the flask was heated until the solution became clear. Concentrated sulfuric acid and 4 percent potassium permanganate solution were then added to the flask, which was heated again. After ashing, 10 percent hydroxylamine hydrochloride solution was added to reduce the potassium permanganate and manganese dioxide. Mercury in the solution was measured by flameless atomic absorption spectrophotometry.

RESULTS AND DISCUSSION

The experimental results on mercury uptake in *Acartia clausi* are presented in Tables 1 and 2. In these tables, the mercury contents of the copepods are shown on a dry weight basis, and the mercury contents of the sediments are shown on a wet weight basis. The initial concentrations of mercury in the sediments used in these experiments were as follows: in HH-1 and HH-2, methylmercury and total mercury were 0.002 ppm and 56.45 ppm, respectively; in HH-3 and HH-4, methylmercury and total mercury were 0.003 ppm and 40.14 ppm, respectively; in HH-5, methylmercury and total mercury were 0.003 ppm and 59.01 ppm, respectively; in HH-6, methylmercury and total mercury were 0.003 ppm and 32.47 ppm, respectively.

The initial concentrations of methylmercury and total mercury in the copepods were 0.069 ppm and 0.081 ppm (dry weight basis), respectively. After the rearing experiment, the concentrations of methylmercury and total mercury in the copepods were as follows: in HH-1, methylmercury and total mercury were 0.028 ppm and 0.808 ppm, respectively; in HH-2, methylmercury and total mercury were 0.033 ppm and 0.145 ppm, respectively; in HH-3, methylmercury and total mercury were 0.099 ppm and 1.526 ppm, respectively; in HH-4, methylmercury and total mercury were 0.134 ppm and 0.643 ppm, respectively; in HH-5, methylmercury and total mercury were 0.125 ppm and 1.431 ppm, respectively; in HH-6, methylmercury and total mercury were 0.353 ppm and 2.152 ppm, respectively.

TABLE [illegible]. MERCURY UPTAKE IN COPEPODS, EXPERIMENT 1

Number	Mercury Concentration in Sediments, ppm, Wet Weight Basis		Mercury Concentration in Copepods, ppm, Dry Weight Basis		Population Density of Copepods Individuals/ℓ
	Total Mercury	Methyl-mercury	Total Mercury	Methyl-mercury	
HH-[illegible]	[illegible]	0.002	0.808	0.028	41
HH-[illegible]	[illegible]	0.002	0.145	0.033	37
HH-3	[illegible]	0.003	1.526	0.099	15
HH-[illegible]	40.14	0.003	0.643	0.134	19
Control	--	--	0.081	0.069	--

TABLE 2. MERCURY UPTAKE IN COPEPODS, EXPERIMENT 2

Number	Mercury Concentration in Sediments, ppm, Wet Weight Basis		Mercury Concentration in Copepods, ppm, Dry Weight Basis		Population Density of Copepods Individuals/ℓ
	Total Mercury	Methyl-mercury	Total Mercury	Methyl-mercury	
HH-5	59.01	0.003	1.431	0.125	10
HH-6	32.47	0.003	2.152	0.353	13
Control	--	--	0.081	0.069	--

In experiment 1, the concentration of methylmercury accumulated in the copepods for 24 hr was 0.074 ppm (average). This concentration was higher than initial concentration (0.069 ppm), and it increased by 0.0045 ppm. The concentration of total mercury accumulated in the copepods for 24 hr was 0.781 ppm (average). This concentration was higher than initial concentration (0.081 ppm), and it increased by 0.700 ppm. In experiment 2 the concentration of methylmercury accumulated in the copepods for 24 hr was 0.239 ppm (average). The concentration was higher than initial concentration (0.069 ppm), and it increased by 0.170 ppm. The concentration of total mercury accumulated in the copepods for 24 hr was 1.787 ppm (average). This concentration was higher than initial concentration (0.081 ppm), and it increased by 1.706 ppm.

In experiment 1, on the assumption that mercury in the sediments (the suspended substance is maintained at about 500 ppm) dissolves completely into the seawater in the 500-ℓ fish-rearing tank, increased concentrations of methylmercury and total mercury in the seawater were 0.0013 ppb (average) and 24.15 ppb (average), respectively. In experiment 2 on the assumption that mercury in the sediments (as 500 ppm of suspended substance) dissolves completely into the seawater in the 500-ℓ fish-rearing tank, increased concentrations of methylmercury and total mercury in the seawater were 0.0015 ppb (average) and [illegible] ppb (average), respectively. In experiment 1, the bioconcentration factors of methylmercury and total mercury were 3,6[illegible] times and 29 times, respectively. In experiment 2, the bioconcentration factors of methylmercury and total mercury were 11[illegible],333 times and [illegible] times, respectively.

On the intake of methylmercury from seawater into copepods, Hirota et al. (1982) reported that a good negative correlation (r = -0.931) existed between the bioconcentration of methylmercury and the density of copepods (Figure 1). By using the correlation in Figure 1 (log Y = 6.4530 - 0.6014 • log X) the bioconcentration factors for a population density of 28 individuals per litre (experiment 1 group) and for a density of 11.5 individuals per litre (experiment 2 group) were estimated. The bioconcentration factor for the experiment 1 group was about 382,000 times, and the bioconcentration factor for the experiment 2 group was about 653,000 times. By using the bioconcentration factor obtained from the estimation and the concentration of methylmercury taken into the copepods, it is possible to estimate the concentration of dissolved methylmercury in the seawater. Obtained results were as follows. In the experiment 1 group, the concentration of dissolved methylmercury in the seawater was 0.000012 ppb. This concentration corresponds with 1/100 of methylmercury contents in the sediment. In the experiment 2 group, the concentration of dissolved methylmercury in the seawater was 0.00026 ppb. This concentration corresponds with 1/6 of methylmercury contents in the sediment.

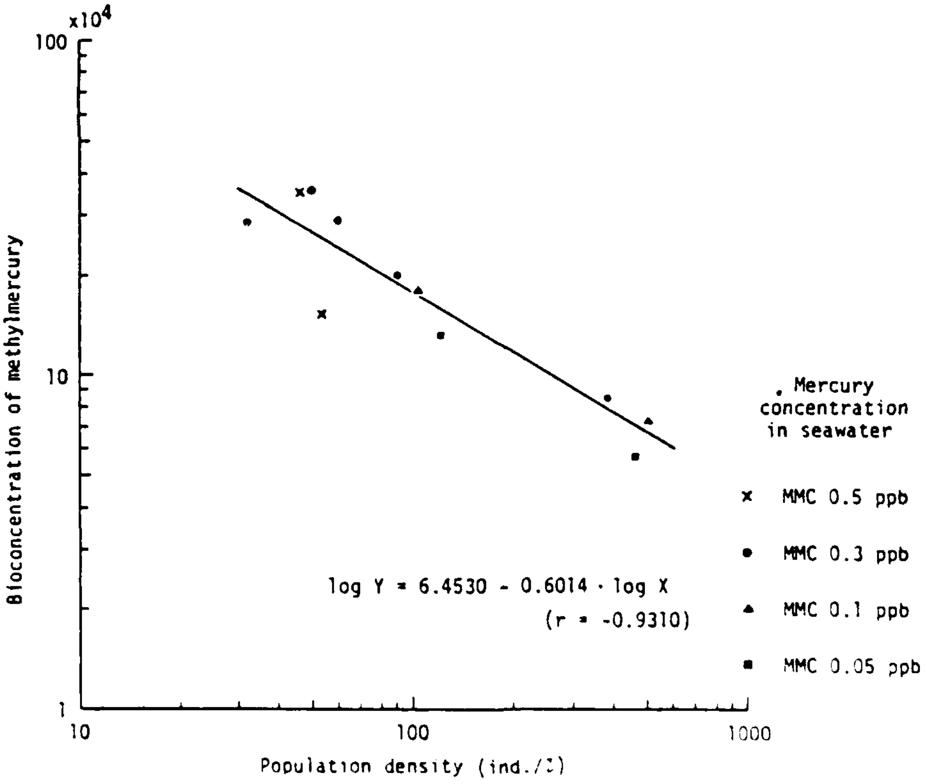

Figure 1. Correlation between bioconcentration of methylmercury (MMC) and density of *Acartia clausi*

In experiment 2 it was expected that methylmercury formation from inorganic mercury in the sediments existed, and that the concentration of methylmercury in the copepods increased by 0.17 ppm (dry weight basis). However, the estimation method explained in this paper suggested that the contents of increased methylmercury in the copepods was brought about by the initial contents of methylmercury in the sediments. It was considered that methylmercury and inorganic mercury in the sediments dissolved into the seawater with sunlight irradiation.

CONCLUSIONS

Marine copepods were reared in seawater containing sediment with methylmercury and inorganic mercury on the particles. Then dissolved methylmercury concentration in the seawater was estimated by using the correlation between the bioconcentration factor of methylmercury and the population density of copepods and by using methylmercury concentration in the copepods. From the experiments, it was estimated that short-term sunlight irradiation and short-term exposure of aerobic condition did not form methylmercury from inorganic mercury in the sediments, but dissolved methylmercury and inorganic mercury into the seawater from the sediments.

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WATER QUALITY MANIPULATIONS IN A EUTROPHIC IMPOUNDMENT

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ABSTRACT

Poor water quality in Eau Galle reservoir, located in west-central Wisconsin, has been associated historically with an overabundance of nuisance planktonic algae (phytoplankton) during the summer months. In an attempt to improve water quality, a variety of experiments was conducted over a 2-year period in large (10 m diameter) enclosed water columns. Specific objectives of these experiments were to reduce phytoplankton standing crop and to promote favorable changes in species composition, i.e., away from nuisance algae (cyanophytes and dinoflagellates) toward more desirable algae (diatoms and chlorophytes). Experimental treatments, implemented singly and in combination, included destratification by mixing, addition of soluble silica, sediment sealing with sand, and precipitation of phosphorus with block aluminum sulfate. Mixing, alone or in combination with silica addition, extended the presence of vernal diatom populations into the summer in one investigation. In contrast, addition of silica to the water column without mixing had no effect on diatom production. In general, mixing stimulated phytoplankton production by increasing phosphorus availability. However, phosphorus inactivation with block aluminum sulfate suspended in the water was sufficient to overcome this effect. Individual effects of phosphorus precipitation and sediment sealing were similar; both decreased phytoplankton standing crop in association with decreased total phosphorus concentrations. Since most of the phosphorus contributed to the phytoplankton in Eau Galle reservoir derives from the sediment, complexation of sediment phosphorus is recommended to improve water quality.

INTRODUCTION

Algae are an essential component of aquatic ecosystems. These organisms are responsible for the synthesis of organic materials, providing much of both the organic structure and energy upon which other components of aquatic systems are dependent. Yet, when the production of algae [illegible] becomes asynchronous with the capacity of the ecosystem to [illegible] them, excessive algal populations may develop.

Dense algal populations occur in many temperate reservoirs [illegible] with documented regularity. The excessive growth of algae is frequently associated with an advanced state of eutrophication and [illegible] of limnological conditions favoring high algal population densities (Wetzel 1983). In addition to obvious reductions in the aesthetic [illegible] of reservoirs supporting excessive algal biomass, the decomposition of algae can result in reductions in dissolved oxygen and the formation of potentially toxic organic residues (Henning and Kohl 1981). Algae-related water quality problems are a source of considerable public concern related to the use of reservoir resources.

Many lakes and most reservoirs receive nutrients from nonpoint and other sources that are not readily amenable to control. In such systems it is not feasible to regulate algal standing crop by modifying nutrient loadings. Thus in-lake approaches to algal management, perhaps involving changes in species composition (i.e., from nuisance species to more desirable species), need to be developed. Toward this end we have conducted enclosure experiments in attempts to manipulate both the standing crop and species composition of the planktonic algae (phytoplankton) of Eau Galle reservoir, a classically eutrophic system located in west-central Wisconsin. Results of these experiments are reported here for initial consideration in formulating a management plan specific to algal problems in Eau Galle.

SITE DESCRIPTION

Eau Galle reservoir is a small (0.6 km^2), shallow (3-m mean depth), moderately alkaline (2-4 meq l^{-1}) dimictic impoundment. It has one major and two minor tributaries, which deliver nutrients from cattle lots and associated agricultural lands, and a single outflow, which receives water from both surface and bottom withdrawal points. Major water exchange and external nutrient loading occur during spring thaw, when as much as half of the reservoir volume is replaced within a period of several days. Throughout the remainder of the year, hydraulic exchange is relatively minor. During the summer, bottom withdrawal of water in combination with wind-driven mixing processes promote hypolimnetic warming (≥15° C), which accelerates mineralization rates and enhances nutrient cycling. Historically, algal blooms in Eau Galle reservoir have been severe.

The seasonal succession of phytoplankton in Eau Galle is fundamentally similar to that described for many temperate eutrophic lakes (Wetzel 1983). Eau Galle is dominated by diatoms during spring and fall periods of destratification and codominated by cyanophytes (blue-green algae) and dinoflagellates during summer stratification. Three species (*Stephanodiscus hantzschii* Grun., *Aphanizomenon flos-aquae* (L.) Ralfs, and *Ceratium hirundinella* (O.F.M.) Schrank) typically contribute 50 percent or more of total annual phytoplankton biomass. Depressed epilimnetic silica concentrations occur during spring and fall periods of diatom growth. The development of nitrogen-fixing cyanophyte populations during the summer reflects reduced inorganic nitrogen availability. The combination of changes in thermal conditions, water column [illegible] diatoms at a rate exceeding [illegible] destratification mixing was [illegible] water column stability during summer [illegible] mentation of diatoms and other [illegible] flagellates and those cyanophytes [illegible] vertical migration. Mixing was provided [illegible] electric motors (Quintero and Garton [illegible] intended to encourage diatoms principally, and [illegible] algae) at the expense of cyanophytes and [illegible]

TABLE 1. EXPERIMENTAL [illegible]

Column Number	[illegible]
1	[illegible]
2	[illegible]
3	[illegible]
4	[illegible]

* Silica was applied continuously as [illegible] 20 g SiO_2 per cubic metre per month.

The principal treatments during [illegible] silica sand and phosphorus precipitat[illegible] was implemented alone and [illegible] 1983 experiments. Principal [illegible] standing crop by diminish[illegible] with principal treatment [illegible] reduced levels [illegible]

TABLE 2. EXPERIMENTAL TREATMENTS IN 1984

[illegible]	Treatment
[illegible]	Sediment Seal* + Mixed
[illegible]	Alum** + Mixed
[illegible]	Sediment Seal
[illegible]	Alum

* [illegible] was "sealed" with 5 cm of washed silica sand. Total mass added [illegible] per column.

** [illegible] (55 percent) was added in block form. Blocks were contin[illegible] to keep pace with dissolution rate.

[illegible] two integrated samples (0-3 m) were taken from each enclosed water [illegible] and the open water. Separate aliquots of each sample were analyzed for [illegible], chlorophyll a content corrected for phaeopigments (American Public Health Association (APHA) 1980), total phosphorus, total soluble phosphorus, total nitrogen, total soluble nitrogen (Raveh and Avnimelch 1979, for N and [illegible] and dissolved silica (automated molybdosilicate method, APHA 1980). Aliquots for algal counts and cell volume determinations were fixed in Lugol's iodine; subsamples of 1.0 to 50 ml were transferred to sedimentation chambers and examined with a Wild M-40 inverted microscope following the technique of Lund, Kipling, and LeCren (1958). Cell volume determinations were made in accordance with techniques described by Barko et al. (1984).

RESULTS

Phytoplankton Response in 1983

The 1983 investigation had three phases. During the first phase (May 9 to June 13), columns #1 and #2 were mixed continuously, and silica was added to columns #1 and #3 (Table 1). Silica enrichment continued through the second phase (June 13 to July 19), but mixing was intermittent due to storm-related power failures on three occasions: June 13, June 28-29, and July 3-6. During the final period (July 19 to August 16), a secondary treatment (addition of liquid alum) was implemented in column #1, silica enrichment continued in columns #1 and #3 until July 22, and mixing in columns #1 and #2 until August [illegible].

[illegible] phytoplankton biomass and community composition varied among study [illegible] (Figure [illegible]). Diatoms and cryptophytes were most abundant during the [illegible] investigation. Diatom biomass during the first period was [illegible] in the mixed columns than in either the unmixed columns or the [illegible]ely, cryptophyte biomass was greatest in the unmixed

[illegible] phase of investigation, total phytoplankton biomass [illegible] in the open water and in all columns except #2. [illegible] this period was due almost entirely to the develop[illegible] dinoflagellate blooms as populations of diatoms and

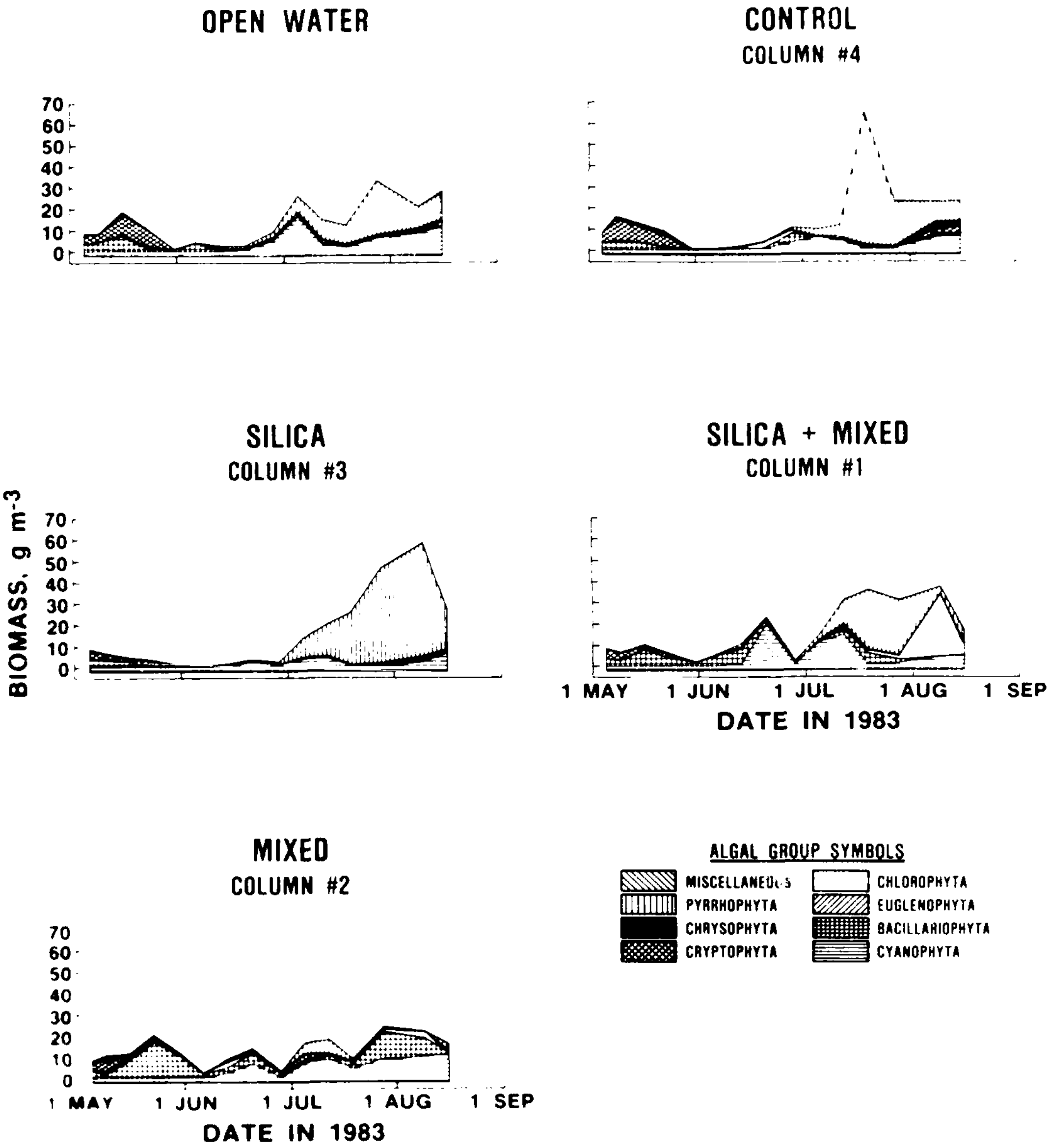


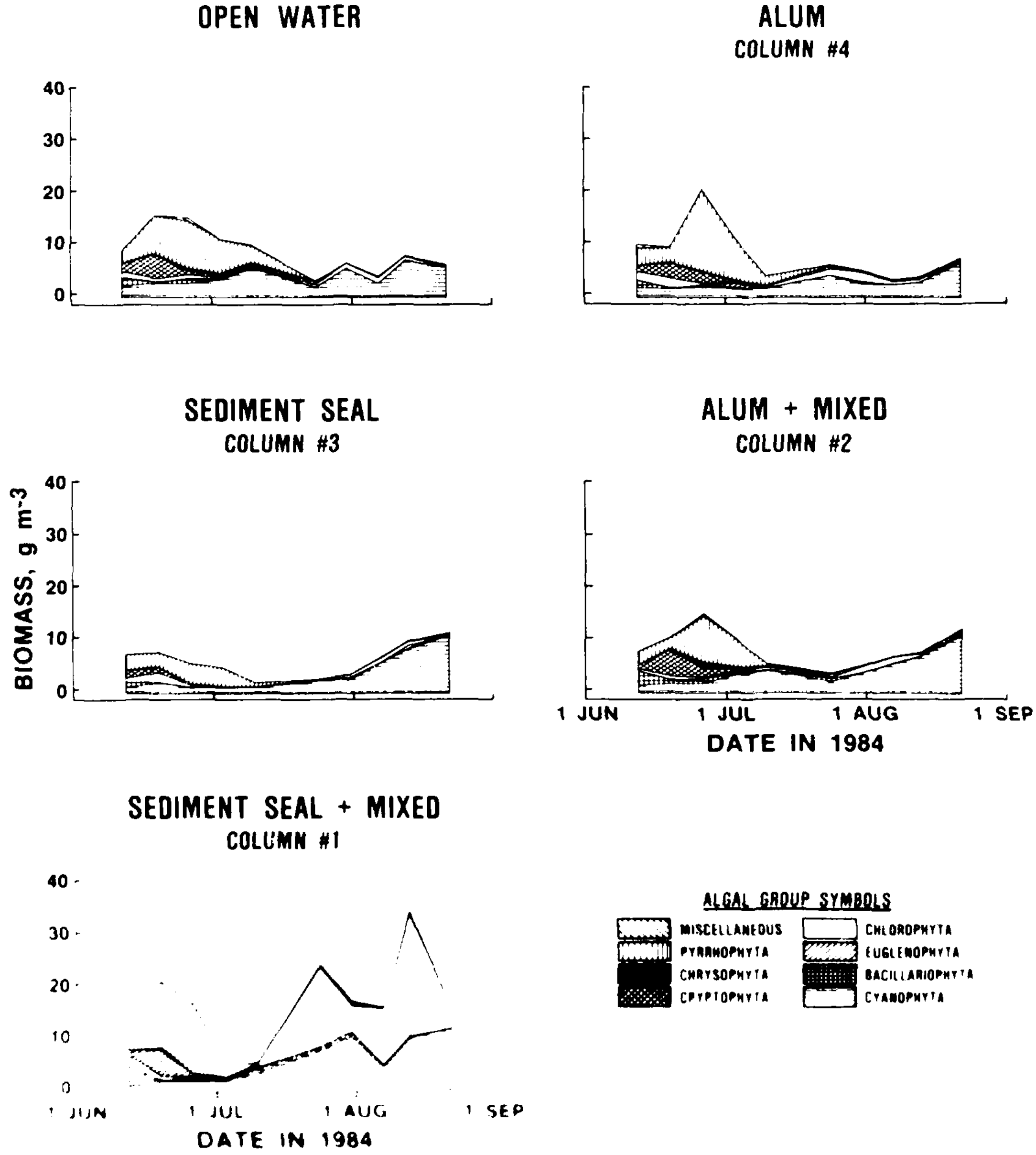
Figure 1. Contributions of various algal groups (taxonomic divisions) to total phytoplankton biomass in the open water and in experimentally enclosed water columns during the summer of 1983

[illegible] significantly, however, the proportion of diatom biomass [illegible] remained greater in the mixed than in the un[illegible] water during this period.

[illegible]

Phytoplankton Response in 1984

The 1984 investigation was continuous, with no interruptions in mixing. Phytoplankton biomass in 1984, with the exception of that in column #1, was about half the biomass in 1983 (contrast Figures 1 and 2). Greater biomass in column #1 than elsewhere was due solely to the development of a massive chlorophyte population (principally the filamentous species, *Mougeotia*) during mid through late summer (Figure 2).



Dinoflagellates peaked in the open water and in all columns except #3 during June of 1984, approximately 1 month earlier than in the previous year. These peaks corresponded with the waning of relatively minor diatom and chrysophyte populations. During mid-July cyanophytes assumed dominance over dinoflagellates. Biomass in all columns except #1 at that time was exceptionally low, but comparable to that in the open water. Throughout the remainder of the study, cyanophytes continued to dominate all columns and the open water. Dominance in column #1, however, was shared by chlorophytes.

Physical and Chemical Effects on Phytoplankton

Additions of silica to the water column in 1983 significantly increased silica concentrations, up to threefold greater than in the open water, but this treatment alone had no effect on diatom biomass. Diatom production was stimulated by addition of silica in combination with mixing (column #1), but to a lesser extent than mixing alone. Dissolved SiO_2 concentrations in 1984 increased progressively from <3 mg ℓ^{-1} to 9 mg ℓ^{-1} throughout the study, did not vary among treatment columns or the open water, and were unrelated to diatom abundance.

Total nitrogen concentrations were essentially unaffected by the various treatments in either investigation. In contrast, total phosphorus was higher and ratios of total nitrogen to total phosphorus were lower in mixed columns and the open water than in unmixed columns, particularly in 1984 (Figure 3). Part of this effect was due to reductions in phosphorus associated with sediment sealing (1984, column #3) and addition of alum (1984, column #4). However, the same trend in 1983 with no experimental reductions in phosphorus indicates that mixing did elevate total phosphorus levels. In both investigations there were strong associations between high total phosphorus concentrations, low nitrogen-to-phosphorus ratios, and high cyanophyte biomass. Conversely, low total phosphorus and high ratios of total nitrogen to total phosphorus were associated with high dinoflagellate biomass.

Concentrations of total phosphorus were significantly correlated positively with total phytoplankton biomass ($r > 0.55$, $p < 0.01$) and negatively with Secchi depth ($r < -0.73$, $p < 0.01$) during both study years. Secchi depth decreased in these investigations proportionately with increasing phytoplankton biomass, and was further depressed periodically in mixed columns due to turbidity caused by sediment resuspension. Mixing increased the rate of phosphorus flux from sediment interstitial water into the overlying water (Table 3). Notably, however, sediment sealing reduced the rate of phosphorus flux to less than half that determined under unsealed conditions. All experimental water columns remained aerobic from surface to sediment surface; thus, differences in phosphorus flux were probably not determined by differences in redox potential.

DISCUSSION

Certainly silica concentration was not the sole factor controlling diatom population dynamics in the enclosures. Diatom populations in the silica-enriched columns in 1983 were as sparse in the spring as those in the control columns, and [illegible] were [illegible] absent from these unmixed columns [illegible] the

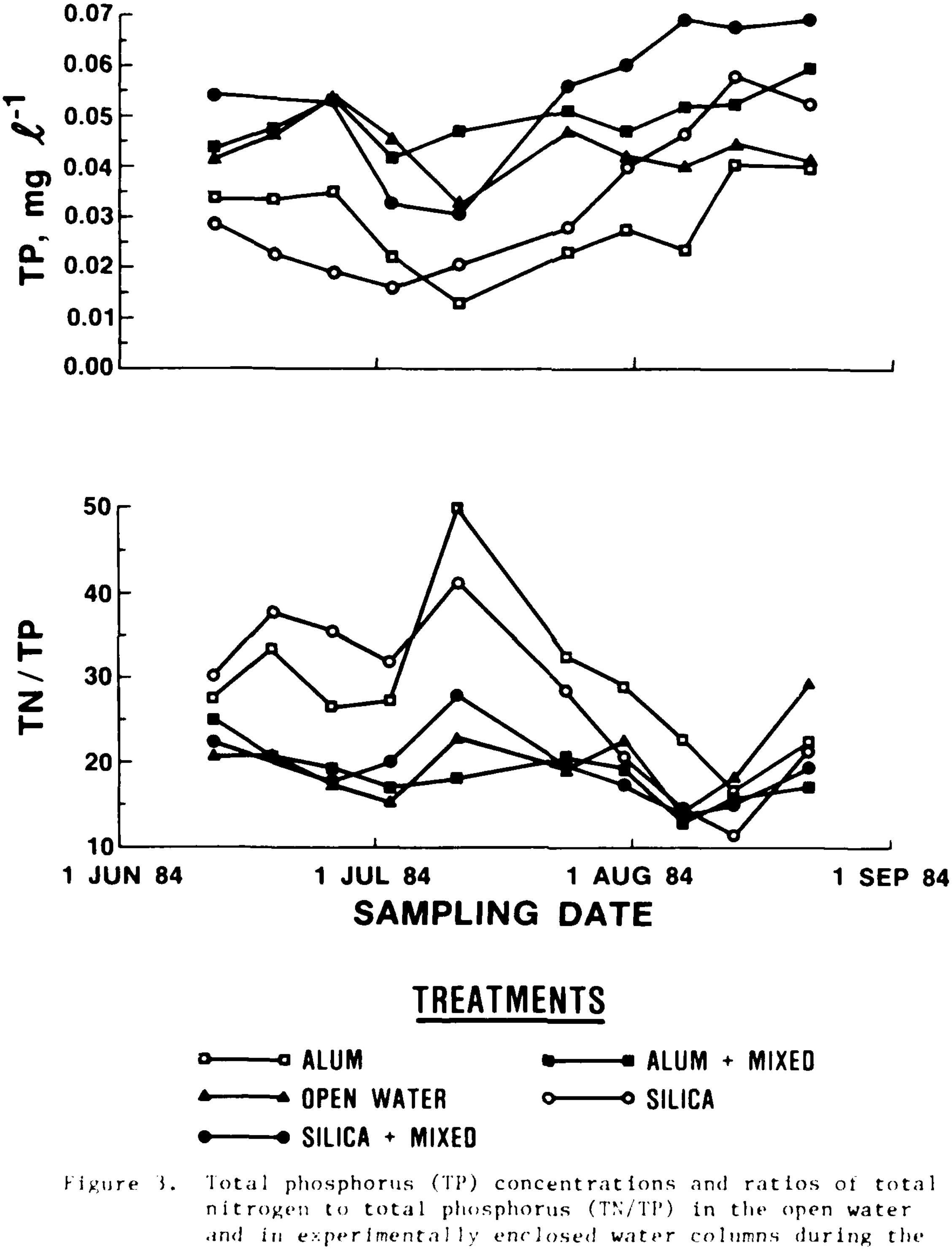


Figure 3. Total phosphorus (TP) concentrations and ratios of total nitrogen to total phosphorus (TN/TP) in the open water and in experimentally enclosed water columns during the summer of 1984

TABLE 3. FLUX RATES (AT 15° C) OF ORTHOPHOSPHATE FROM SEDIMENTS WITHIN SELECTED EXPERIMENTAL COLUMNS DURING THE 1984 INVESTIGATION

Treatment Condition	Flux Rate, $mg\,m^{-2}\,day^{-1}$*
Sediment Sealed	14.4
Sediment Sealed + Mixed	20.6
Sediment Unsealed	32.9

* Calculated using a modification of Fick's equation from vertical profiles of orthophosphate concentration measured in sediments.

open water as well as in the unmixed columns, diatoms began declining in mid-May as water column stability (i.e., resistance to mixing) increased. Diatoms, because they are relatively dense and lack the ability to maintain position within the water column, are particularly disadvantaged in hydraulically stable environments (Hutchinson 1967). Destratification by mixing in 1983 facilitated the extension of spring diatom populations into the summer, but this effect was not enhanced by silica additions. Thus diatom population dynamics in Eau Galle appear to be more responsive to hydraulic conditions than to the availability of silica.

Unlike conditions in 1983, experiments in 1984 were initiated a full month following the collapse of vernal diatom populations (Barko, unpublished), and in contrast to the previous year's results, diatom production in 1984 was not encouraged by mixing. Apparently, mixing in Eau Galle is only effective in maintaining vernal diatom populations when initiated prior to diatom collapse (see also Nicholls, Kennedy, and Hammett 1980).

We were partially successful in maintaining preferred algae (diatoms) by mixing, but not at the expense of nuisance algae (cyanophytes and dinoflagellates). In general the growth of cyanophytes was stimulated by mixing, while that of dinoflagellates was either depressed somewhat or unaffected under the same conditions. The growth of chlorophytes, principally filamentous *Mougeotia*, which is not a desirable species, was also stimulated under some circumstances by mixing--in combination with alum treatment in 1983 and sediment sealing in 1984. Chlorophytes were essentially absent from columns in which mixing was implemented alone.

Trimbee and Harris (1984) suggested that the most notable effect of periodic natural mixing episodes on phytoplankton was a shift in dominance from nonnitrogen-fixing to nitrogen-fixing species. In our investigations nitrogen-fixing cyanophytes dominated at times under both mixed and unmixed conditions, but were generally more abundant in mixed columns. The most prominent cyanophyte taxa in mixed columns were *Anabaena* and *Aphanizomenon* species, both of which readily fix nitrogen. Nitrogen fixation has been viewed as a response to stoichiometric imbalance between nitrogen and phosphorus availability (Schindler 1977). Accordingly, relatively low ratios of total nitrogen to total phosphorus (TN/TP), as in our mixed columns, tend to favor nitrogen-fixing cyanophytes (Smith 1983).

Dinoflagellates, principally *Ceratium*, were dominant in the absence of mixing during mid through late summer in 1983. However, in 1984 mixing had no obvious effect on dinoflagellates, which achieved dominance in all columns and in the open water during the early summer. An interesting observation applicable to both years of investigation is that dinoflagellates generally fared best where cyanophytes fared poorest. On the whole, 1983 was dominated by dinoflagellates while in 1984 cyanophytes dominated. There may be an antagonistic interaction that accounts for the apparent inverse relationship between dinoflagellate and cyanophyte population development (Vance 1965; Dottne-Lindgren and Ekbohm 1975; and Nicholls, Kennedy, and Hammett 1980), but causal mechanisms have not been satisfactorily explained.

In general, the production of phytoplankton biomass was stimulated by mixing, and this effect was most pronounced in 1983. We attribute this response to increased phosphorus availability caused by sediment resuspension and enhanced phosphorus diffusion from sediments under mixed conditions. Notably, phosphorus inactivation with block aluminum sulfate under these conditions was sufficient to overcome stimulated biomass production. Although these results are strictly applicable only to relatively shallow regions (less than about 3 m) of Eau Galle, they do underscore the important influence of sediments on phosphorus availability and resultant phytoplankton production.

The sediment is the major source of phosphorus in Eau Galle during the summer months, when inflows account for no more than a few percent of the total phosphorus budget (James, Kennedy, and Gaugush 1985). During this period phosphorus in concentrations >2,000 $\mu g\ \ell^{-1}$ in the anoxic hypolimnion is made available to the phytoplankton by periodic weather-related mixing events (Gaugush 1984). Mixing in Eau Galle is facilitated by relatively low water column stability due to the effects of bottom withdrawal of water on the thermal regime. Similar effects of weather-induced mixing on phosphorus dynamics have been reported for Shagawa Lake, Minnesota, another relatively shallow basin located within the same climatic setting as Eau Galle (Stauffer and Armstrong 1984).

Sediment sealing with sand in the absence of mixing was fairly effective in reducing total phosphorus release from sediments under ambient aerobic conditions, but would probably have had a lesser effect under anaerobic conditions due to enhanced phosphorus availability at low redox potentials (Wetzel 1983, p 261). Mixing largely negated the effect of sediment sealing on phosphorus release, indicating that the advantage gained by increasing the diffusion distance between sediment surface and overlying water was minor. Addition of alum in the 1984 investigation decreased total phosphorus concentrations in an unmixed water column, and thus was effective in precipitating a portion of the phosphorus released from underlying sediment.

Improved water quality in Eau Galle will require a reduction in phosphorus release from sediments during the summer months. We propose to do so by administering alum directly onto the sediment surface within the profundal region (i.e., >3-m depth) according to procedures described in Kennedy and Cooke (1982). The advantage to using alum as opposed to an inert sediment seal is that the former selectively complexes phosphorus in chemical forms that are unaffected by changes in redox potential. Prerequisite to the successful application of this approach is the knowledge that internal loading of phosphorus from sediments dominates the midsummer phosphorus budget. Alum treatment of sediments in natural lakes to control phytoplankton production has been generally quite successful (Cooke and Kennedy 1981), but has not been applied to a reservoir system.

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THE EFFECTS OF SEDIMENT ON THE SURVIVAL OF MYSIDS EXPOSED TO ORGANOTINS

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ABSTRACT

A 10-day flow-through toxicity test was conducted by exposing mysids (*Acanthomysis sculpta*) to organotin antifouling (AF) leachates. A paired design with and without sediment was used to determine if sediment affects the survival of mysids exposed to organotin AF leachates. Since the Navy is considering a fleetwide conversion from copper-based AF coatings, it is studying the potential environmental impact of organotins and their effect on the dredging permit process.

Eighteen-day-old mysids were exposed to tributyltin (TBT) concentrations averaging 0.60 ppb for 10 days in the presence and absence of sediment. TBT concentrations in tanks with and without sediment ranged from 0.41 to 0.80 ppb and 0.54 to 0.78 ppb, respectively. Sediment did not accumulate detectable quantities of TBT during 31 days of exposure.

Sediment and toxicant both affect the survival of mysids. Sediment became a statistically significant factor in the survival of mysids by day 7 and continued to be a factor until the end of the experiment. Mysids exposed to TBT exhibited significantly higher survival in tanks with sediment than in tanks without sediment. A significant decline in survival in the TBT tanks without sediment was observed 2 days before such a decline in the TBT tanks with sediment. The TBT toxicant became a significant deleterious factor by day 8. After 10 days control survival was 100 percent with sediment and 95 percent without sediment; TBT treatment survival was 46 percent with sediment and 30 percent without sediment.

INTRODUCTION

AF coatings for ship hulls contain toxic compounds such as copper and organotin which leach out of the coating matrix to inhibit attachment of marine organisms. These toxicants enter the water column and a significant portion is then adsorbed by the sediment. Since the Navy is considering a conversion from copper to organotin-based AF coatings, it is studying the potential environmental impact that might be attributed to fleetwide implementation of organotin-based AF coatings. In a related program the Navy is also studying the potential effects of organotins on dredging and disposal, and the dredging permit process.

Sediment dredging is a major and continuing Navy requirement which is currently approaching 15 million cu yd/year. With a projected introduction of deep draft vessels and a potential 600-ship Navy by the year 2000, this burden will only increase. Anticipating this potential problem, the Navy has developed a program to expedite Navy dredging by optimizing the dredging permit process. As part of this program, a Memorandum of Agreement (MOA) with the Army Corps of Engineers (CE) has been proposed. As part of this program a bioassay was conducted on organotin-contaminated sediment from San Diego Bay (Salazar and Salazar 1984).

Although some commercial fleets and private owners currently use organotin-based AF coatings, the behavior of organotin compounds in the marine environment is not clearly understood. To date there are no regulatory criteria for the disposal of organotin-contaminated sediments. Since there are potential environmental problems associated with organotin AF coatings, the Navy has sponsored a research program over the last 5 years to study the fate and effects of organotins in the marine environment. Much of this time was spent developing techniques for accurate measurement of organotins in seawater, sediment, and tissues. Concurrently a number of programs were developed to measure the toxicity of organotins.

The principal toxicant in organotin coatings is TBT either in unbound mixtures or in controlled release polymers. The TBT is released from the coating through hydrolysis to marine waters. It is generally accepted that the TBT cation is the toxic component and the anion is not a factor in its toxicity. TBT degradation products are less toxic than the parent compound. Our work has shown that TBTs are nearly an order of magnitude more toxic than dibutyltins and two orders of magnitude more toxic than monobutyltins (Salazar and Salazar 1985). Smith (1979) has reported similar results. Smith also reports that the most important use of TBT oxide is in marine AF paints. For these reasons we have conducted almost all toxicity tests with the TBT form. Only recently have the most meaningful results been produced due to refinements in both the biological assessments and analytical techniques.

Our early work, as well as the work of others, has shown that sensitive marine species exhibit significant mortality at TBT concentrations below 1 ppb. During the past year we have measured TBT concentrations in seawater as high as 0.93 ppb in an enclosed yacht basin in San Diego Bay (Valkirs et al. 1985a). This concentration exceeds the 0.60 ppb TBT reported to cause effects on sensitive marine organisms like copepods and mysids in laboratory tests (U'Ren 1983; and Valkirs, Davidson, and Seligman 1985). To assess future dredging problems, we conducted a standard dredged material bioassay with organotin-contaminated sediment from an area in San Diego Bay where we had previously measured the highest concentration of TBT in sediment. The results from that study indicate organotins bound to and released from sediments were not as toxic as anticipated. We were surprised to find no statistically significant toxicity in tanks with high TBT concentrations in the sediment. Elutriate concentrations in that test approached those producing significant mortalities in other studies.

The relationship between sediment and TBT toxicity to mysids was examined in more detail during the present study. The purpose of this work was to determine if the survival of mysids exposed to organotin AF leachates is affected by the presence of sediment.

METHODS AND MATERIALS

Experimental Design

Four experimental conditions were used to test the effects of sediment and organotin leachate on mysid survival. They were: clean seawater only (CSO), clean seawater with sediment (CSS), TBT leachate seawater only (TBT-SO), and TBT leachate seawater and sediment (TBT-SS). The first two conditions, CSO and CSS, were designed as controls. Mysid survival data for treatments were compared with controls with a two-way analysis of variance (ANOVA) (Figure 1). The variables compared were toxicant ("control" versus "organotin exposures") and substrate ("seawater" versus "sediment"). The ANOVA also permitted testing for significant interaction of TBT and sediment on mysid survival. The conditions tested can be summarized by the following hypotheses:

$H_{o}1$: Organotins have no effect on mysid survival.

$H_{o}2$: Sediment has no effect on mysid survival.

$H_{o}3$: Organotins and sediment do not interact to affect mysid survival.

To test the hypothesis that sediment increases the survival of mysids exposed to organotin, the experimental design consisted of monitoring the survival of mysids exposed to TBT-contaminated seawater with and without sediment present. Five replicate tanks were prepared for each test condition with 20 mysids per replicate. A total of 20 test tanks was used, ten with sediment and 10 without. The test was run for 10 days to duplicate conditions recommended in the Implementation Manual (Environmental Protection Agency (EPA)/CE 1977) for the solid-phase portion of the standard dredged material bioassay and in the bioassay conducted with organotin-contaminated sediment (Salazar and Salazar 1984).

		TOXICANT	
		NO TBT	TBT
SUBSTRATE	NO SEDIMENT	CLEAN SEAWATER ONLY (CSO)	TBT SEAWATER ONLY (TBT-SO)
	SEDIMENT	CLEAN SEAWATER & SEDIMENT (CSS)	TBT SEAWATER & SEDIMENT (TBT-SS)

Figure 1. Experimental design

Test Animal

The mysid [illegible] was selected as the test animal because of proven sensitivity and reliability demonstrated in over [illegible] bioassays conducted by the Naval Ocean Systems Center (NOSC) to qualify dredged material from Navy projects in San Diego Bay for ocean disposal. Mysids are appropriate sensitive marine organisms for toxicity testing, with death as a measurement. They are sensitive to a variety of toxicants (Smith [illegible]), reliable in terms of experimental repeatability (EPA/CE [illegible]), and representative by nature of a close association with the bottom and migrations throughout the water column (Mauchline 198[illegible]).

[illegible] is a hypoplanktonic crustacean that generally schools close to the bottom sediment during the day for protection and migrates to the water surface at night to feed on smaller zooplankton (Mauchline 1980). When associated with [illegible] kelp beds, [illegible] remains within the kelp canopy and does not undertake significant vertical migrations. They are afforded protection by the kelp and have a supply of food from surface zooplankton. Mysids are on the approved species list for use in dredged material bioassays (EPA/CE 1977). Further, they are considered important enough to be required for both the liquid particulate phase and the solid phase portions of the standard bioassay.

Mysids were collected on 1 March 1985 just outside the mouth of San Diego Bay within a 500-m radius of the laboratory seawater intake. They were collected in the surface kelp canopy with buckets and returned to the laboratory holding system. These mysids were maintained in flow-through rearing tanks. Released juveniles were collected and maintained separately so that all test animals would be in the same age class. Juveniles used in this test were separated from adult tanks on 4 April 1985 and held for 18 days before the test began. All mysids were fed maintenance levels of brine shrimp (25-30 nauplii/mysid) twice per day.

Test Sediment

Sediment was collected with a stainless steel Van Veen grab from a site on the west end of North Island, San Diego Bay (Figure 2). The North Island site was selected for sediment collection because grain-size composition and contaminant levels have been well documented. The sediment has been demonstrated to be reasonably "uncontaminated" during previous bioassays. A physical/chemical description of the sediment used in this test is given in Table 1. It is classified as very fine sand (Wentworth 1922).

Sediment from the grab was stored in 60-ℓ ice chests lined with plastic bags. Ice chests were used for ease in handling sediment and to maintain the temperature of sediment between collection and storage. The sediment was stored for 72 hr at 4° C. Approximately 50 ℓ of sediment was collected. The sediment was sieved through a 1.0-mm stainless steel screen to remove endemic organisms and achieve homogeneity. After sieving, 1 ℓ of sediment was distributed to each of the 10 sediment tanks (five controls and five treatments). This 1 ℓ of sediment produced a 20-mm layer on the bottom of each "substrate" test tank. Seawater was allowed to flow through all 20 test tanks (10 with sediment and 10 without sediment) for 21 days before starting the test. Seawater to half of these tanks contained the TBT leachates.

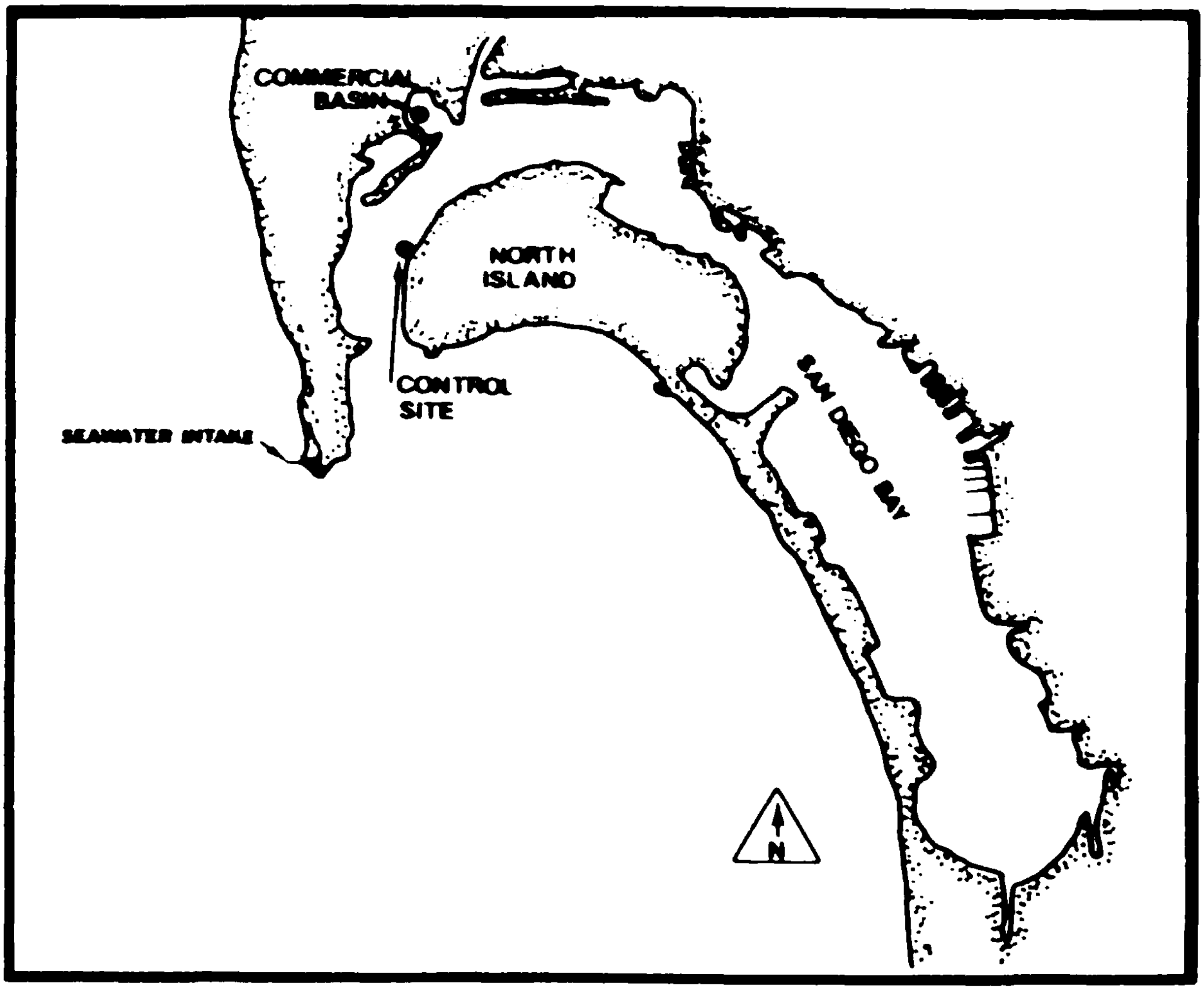

Figure 2. Site map

Seawater

Nearshore seawater available at the NOSC Marine Life Sciences Laboratory was used for this toxicity test. The seawater intake for this system was approximately 250 m from shore (Figure 2). This natural seawater was passed through large sand filters prior to laboratory use.

In the laboratory physical and chemical parameters of incoming seawater were measured during the test with the Seawater Quality Monitoring System (SQMS) developed by NOSC. This system consists of several commercially available sensors arranged in an in-line manifold system. Output is recorded with a data acquisition system linked to a microcomputer. Parameters monitored daily included pH, temperature, and salinity. Dissolved oxygen was measured with a hand-held meter. Seawater temperature was measured at 14.1 to 15.3° C, salinity at 32 to 33 ppt, dissolved oxygen at 6.5 to 7.9 ppm, and pH at 7.8 to 8.0. The light regime consisted of a 14L:10D cycle as recommended in the

[illegible] PHYSICAL/CHEMICAL DESCRIPTION OF "CONTROL SEDIMENT" [illegible]SED IN THE MYSID/ORGANOTIN LEACHATE EXPERIMENT

Sediment Composition Information

[illegible]	Percent Composition			
	Sand	Silt	Clay	Classification
[illegible]	80.1	11.7	8.2	Very fine sand

Contaminant Information, ppm

[illegible]	Chromium	Copper	Mercury
[illegible]	15.0	18.0	0.13

Silver	PHF	PCB	Pesticides
[illegible]	0.100	0.006	0.001

Organotins, ppb

Tributyltin:	8.0
Dibutyltin:	3.0
Monobutyltin:	5.0

Implementation Manual (EPA/CE 1977). Irradiance was measured with a photometer and ranged between 100 and 200 μw/cm^2 for the *Acanthomysis* [illegible] test tanks.

Test Conditions

Mysids were held in 4-ℓ polycarbonate tanks. Our research on [illegible] metals and organometallic contaminants has shown that polycarbonate [illegible] quality glass are similar in their nonabsorptive properties (Dooley [illegible] 1983). Ease in drilling for flow-through fittings, durability, [illegible] polycarbonate tanks preferable for toxicity tests. The flow rate [illegible] tank was approximately 170 ml/min. This corresponds to rough[illegible] changes per hour (assuming complete exchange).

Mysids were fed brine shrimp nauplii (25 to [illegible] day and aerated at a rate of 3 to 7 ml/min. Dead an[illegible] and removed. With 20 animals per 4-ℓ tank, live [illegible] count. We counted dead animals because the white [illegible] on the dark sediment. Usually mysid bodies [illegible] 1 to 3 days after death. By counting fresh [illegible] live individuals could be made. By counting [illegible] live animals on day 10, we were able to [illegible] counting method. We could account for [illegible] experiment, 98.8 percent.

AD-A186 119 MANAGEMENT OF BOTTOM SEDIMENTS CONTAINING TOXIC 3/4
SUBSTANCES: PROCEEDINGS O..(U) ARMY ENGINEER WATERWAYS
EXPERIMENT STATION VICKSBURG MS ENVIR.. T R PATIN
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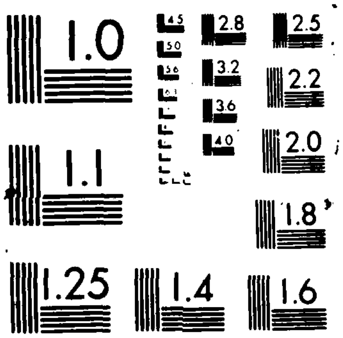
1.0
4.5
5.0
5.6
2.8
2.5
3.2
2.2
3.6
4.0
2.0
1.1
1.8
1.25
1.4
1.6

Toxicant

The TBT toxicant was introduced as a leachate from AF paint on Plexiglas panels. Panels (15 by 15 cm) were coated with SPC-956 (International Paint, Inc.). This paint contains 9.4 percent TBT methacrylate, 0.5 percent bis(TBT) oxide, and 44.7 percent cuprous oxide as the toxicants. Two coats were applied by dipping the panels into the paint can, allowing them to dry overnight, and dipping them again. These panels were installed in the polycarbonate troughs of a flow-through seawater delivery system previously developed (Meador, U'Ren, and Salazar 1984). In this system (Figure 3) seawater is forced to flow over a series of panels at a rate of 6 ℓ/min before distribution to test tanks. The concentration of TBT in seawater delivered to test tanks is determined by the flow rate of incoming seawater and the number of panels in the mixing trough.

During an 8-week equilibration and stabilization period without sediment in test tanks, chemical measurements were made to determine the TBT concentration in the leachate. By altering flow rates and removing panels, the approximate desired concentration of 0.60 ppb TBT was achieved. Previous measurements of leachates from this coating system have demonstrated that the copper concentration is about twice the TBT concentration. However, due to differences in effective toxicity, the effects of copper are relatively insignificant. This concentration was selected after review of the most current toxicity data. In a previous flow-through experiment without sediment, 0.61 ppb TBT produced 50 percent mortality in 4 days with 1- to 3-day-old *Acanthomysis*

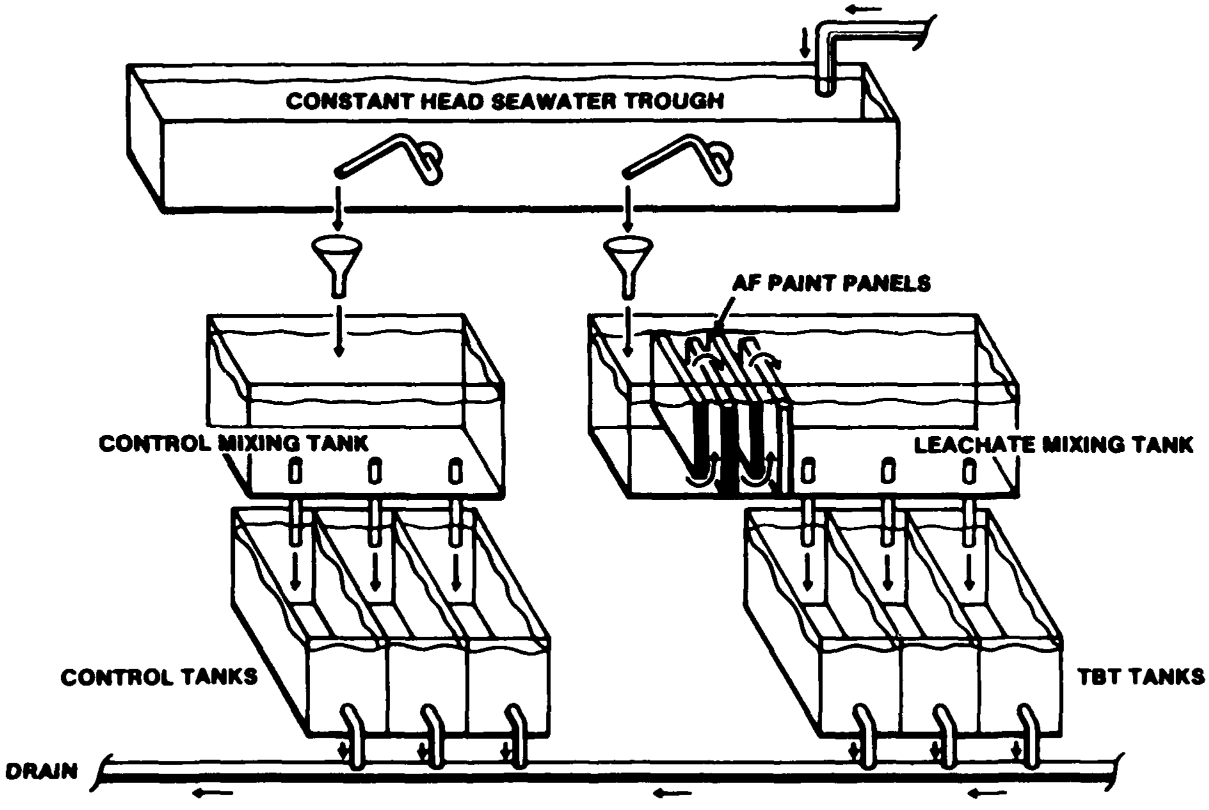

Figure 3. Flow-through bioassay system

sculpta juveniles (Valkirs, Davidson, and Seligman 1985). TBT concentrations were measured by hydride derivatization atomic absorption spectrophotometry (HD-AAS) (Valkirs et al. 1985b).

Organotin Measurements

The concentration of TBT in seawater and sediment was monitored before and during the assessment. Four sets of tanks were prepared as chemical blanks. By comparing TBT measurements in tanks with and without mysids, we could assess the effect of mysids on TBT chemistry. Each set consisted of two tanks, one tank with sediment and one without (total of eight tanks). No mysids were added to these chemical blanks. Three sets were used to monitor organotin concentrations in the seawater during the test. The fourth set was used as a chemical blank for the controls to monitor potential TBT contamination. Seawater from the TBT chemical blank tanks was measured prior to starting the test to confirm the concentration of TBT in the leachate solution. A 50-ml aliquot of seawater from each of the three chemical blank tanks was pooled for one composite sample. In addition to samples from the chemical blanks, seawater to be used for chemical analysis was collected from the outflow ports of treatment tanks to avoid stressing the test animals.

Composite seawater samples of 100 ml each were collected from the following sets of TBT treatment tanks on a daily basis: mysids with sediment, mysids without sediment, chemical blanks with sediment, and chemical blanks without sediment. Polycarbonate bottles were used for the samples. After collection, samples were frozen until chemical analysis. The HD-AAS was used to determine the TBT concentration in both sediment and seawater samples. All samples were not analyzed due to constraints in analytical support.

RESULTS

The sediment/organotin toxicity test with mysids was conducted between March-May 1985. Daily mysid survival (from daily counts of dead animals) is presented in Table 2 by replicate, with total percent survival by test condition. Percent survival versus time for each of the four test conditions is presented in Figure 4. Inspection of the data shows there was no significant difference in survival among any of the treatments during the first 6 days of the test. The two-way ANOVA confirmed no statistically significant effect in either substrate or toxicant treatments. Sediment did not become a statistically significant factor until day 7 (Table 3). The effect was positive in that mysid survival was greater with sediment than without sediment. Mysid survival in TBT-SO tanks began a significant decline 2 days before a significant decline in TBT-SS tanks. By day 8 there were two statistically significant effects on mysid survival, sediment and TBT. The sediment (substrate) effect remained positive, but the TBT (toxicant) effect was deleterious. Control survival in the CSO and CSS tanks remained constant at 95 and 100 percent, respectively, from day 7 to day 10, the end of the experiment. Survival data from days 8, 9, and 10 showed significant effects due to sediment and TBT. The sediment (substrate) effect was positive while the TBT (toxicant) effect was deleterious. No interaction effect was shown.

In the TBT-SO tanks mysid survival dropped rapidly from 95 percent on day 6 to 90 percent on day 7, 83 percent on day 8, 64 percent on day 9, and 30 percent on day 10. Survival of mysids in TBT-SS tanks remained high at

TABLE 2. SURVIVAL OF MYSIDS EXPOSED TO ORGANOTIN-CONTAMINATED SEAWATER WITH AND WITHOUT SEDIMENT

Tank	Rep	Time, days										
		0	1	2	3	4	5	6	7	8	9	10
CSO	1	20	20	20	20	20	20	20	20	20	20	20
	2	20	20	20	20	20	20	20	20	20	20	20
	3	20	20	20	19	19	19	19	19	19	19	19
	4	20	20	18	18	18	17	17	17	17	17	17
	5	20	20	19	19	19	19	19	19	19	19	19
		100%	100%	97%	96%	96%	95%	95%	95%	95%	95%	95%
CSS	1	20	20	20	20	20	20	20	20	20	20	20
	2	20	20	20	20	20	20	20	20	20	20	20
	3	20	20	20	20	20	20	20	20	20	20	20
	4	20	20	20	20	20	20	20	20	20	20	20
	5	20	20	20	20	20	20	20	20	20	20	20
		100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
TBT-SO	1	20	20	20	20	20	20	20	19	18	11	6
	2	20	20	20	20	20	20	19	19	19	17	10
	3	20	20	20	20	20	20	20	18	16	10	2
	4	20	20	19	19	19	19	19	18	15	12	3
	5	20	20	19	18	18	18	17	16	15	14	9
		100%	100%	98%	97%	97%	97%	97%	90%	83%	64%	30%
TBT-SS	1	20	20	20	19	19	19	19	19	19	17	11
	2	20	20	20	20	20	20	20	20	20	16	8
	3	20	20	20	20	20	20	20	20	19	17	7
	4	20	20	19	19	19	19	19	19	19	16	11
	5	20	20	20	20	20	20	20	20	19	17	9
		100%	100%	99%	98%	98%	98%	98%	98%	96%	83%	46%

96 percent until day 8 and then dropped rapidly to 83 percent on day 9 and 46 percent on day 10. The difference in mysid survival between toxicant tanks was statistically significant on days 7, 8, and 9. Survival of mysids exposed to TBT with sediment was greater than survival of mysids exposed to TBT without sediment. The difference on day 10 was not statistically significant. In addition to no significant interaction effect on survival between sediment and organotin, there was no statistically significant difference in survival between controls (CSO and CSS) when compared with a Mann-Whitney U Test (Tables 3 and 4).

The concentration of TBT toxicant in the mixing trough 3 days before the test began was measured at 0.60 ppb TBT. During the test TBT concentrations

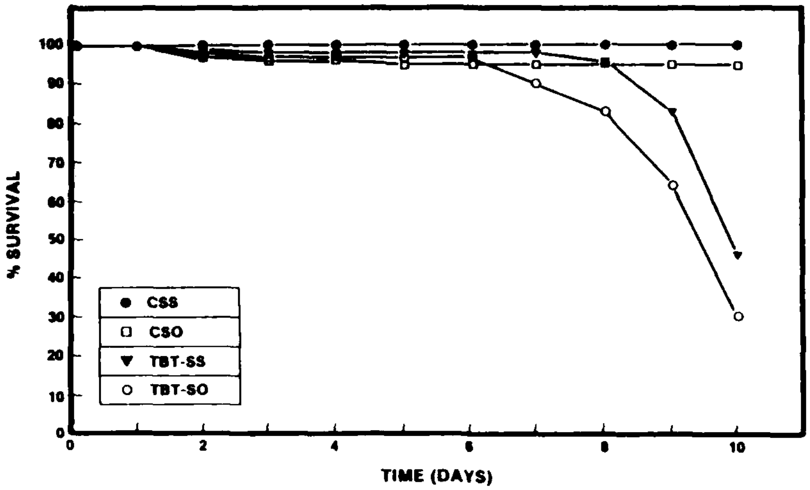

Figure 4. Mysid survival

TABLE 3. RESULTS OF TWO-WAY ANOVA PERFORMED ON THE MYSID SURVIVAL DATA

		Probability Level			
Day	"F" Value	Prob > F	Toxicant	Substrate	Interaction
7	4.5858	0.01683*	0.10410	0.00557*	0.47088
8	8.5866	0.00126*	0.00558*	0.00240*	0.12916
9	21.5859	0.00001*	0.00000*	0.00308*	0.05918
10	57.0658	0.00000*	0.00000*	0.03779*	0.25287

* A < 0.05 indicates a significant effect.

in the TBT-SO tanks with mysids varied from 0.41 ppb to 0.80 ppb TBT with a mean of 0.60 ppb TBT. TBT concentrations in the TBT-SS tanks varied from 0.54 ppb to 0.78 ppb with a mean of 0.60 ppb TBT. The means were the same for both TBT treatment conditions. The TBT-SO chemical blanks varied from 0.37 ppb to 1.11 ppb TBT and had a mean of 0.78 ppb TBT. The TBT-SS chemical blanks varied from 0.47 to 0.80 ppb TBT and had a mean of 0.61 (Table 5).

The clean sediment used in this test had TBT levels below the limits of detection. Further, sediment did not accumulate detectable quantities of organotins during 31 days of exposure to seawater containing an average TBT concentration of 0.60 ppb. There were no measurable increases in tri-, di-, or monobutyltin in test sediments since organotins were below the limit of detection before and after exposure to TBT leachates (Table 5).

TABLE 4. RESULTS OF MANN-WHITNEY U TEST*

Day	U-Critical	U-Calculated
	TBT-SO Versus TBT-SS Data	
7	21.0	23.0*
8	21.0	23.0*
9	21.0	36.5*
10	21.0	19.5
	CSO Versus CSS Data	
10	21.0	20.0

* If U-Calculated > U-Critical, there is a statistically significant difference at the 95 percent confidence level when a = 0.05 .

TABLE 5. ORGANOTIN CHEMISTRY SUMMARY TABLE
(CONCENTRATIONS IN PPB TBT)*

Composite Samples	Seawater Chemistry							
	T_o-3	T_o	Day 2	Day 3	Day 7	Day 9	Day 10	Mean
Chemical blanks								
TBT-SS		0.47			0.57		0.80	0.61
TBT-SO	0.6	0.37			0.87		1.11	0.78
Mysid tanks								
TBT-SS		0.58	0.55	0.57	0.54	0.78		0.60
TBT-SO		0.41	0.46	0.53	0.79	0.80		0.60

TBT-SS Samples	Sediment Chemistry**	
	T_o-6	Day 10
Tributyltin	<8.0	<8.0
Dibutyltin	<3.0	<3.0
Monobutyltin	<5.0	<5.0

* At T_o the control sediment had been exposed to a nominal concentration of approximately 0.6 ppb TBT for approximately 21 days. By day 10 the sediment had been exposed to TBT for over 31 days.

** All butyltin concentrations were below detection limits.

DISCUSSION

Based on the results of the two-way ANOVA there was no significant effect attributed to a 0.60-ppb TBT exposure after 6 days. By day 7 there was a significant effect attributed to sediment and the effect was positive. The reason for this statistically significant effect is higher survival in both sediment treatments (CSS = 100 percent, TBT-SS = 98 percent) than in the seawater treatments (CSO = 95 percent, TBT-SO = 90 percent). This may appear to be anomalous, but we have routinely seen higher survival of mysids in dredged material bioassays in the presence of sediment than in the absence of sediment. This positive effect attributed to sediment continued from day 7 to day 10, the end of the experiment.

By day 8 there was also a significant effect attributed to the TBT toxicant. This effect was deleterious. Mysid survival dropped from 90% on day 7 to 83 percent on day 8. This deleterious effect attributed to the TBT toxicant continued from day 8 to day 10. Mysid survival dropped precipitously in both TBT-SO and TBT-SS treatments near the end of the experiment. These data suggest that juvenile mysids are capable of withstanding low-level exposures to TBT for periods up to 6 days. The actual toxic insult was present during the entire 6 days, but the result of that insult was not expressed as death until day 7. This trend has been noted in previous work performed with TBT and mussels (*Mytilus edulis*) (Smith 1979). One possible explanation is a cumulative effect over time or a deterioration of the animal's defense mechanisms. A 6-day experiment with the same design and TBT concentrations followed by a recovery period without TBT would clarify the 6-day effects of 0.60 ppb TBT in *Acanthomysis sculpta*. Pickwell and Steinert (1984) have described a quantifiable sublethal stress measurement for mussels exposed to copper. A similar system would prove useful with mysids or other sensitive test animals. Until those techniques are available, death is the most reasonable measurement.

Daily inspection of the control and treatment tanks revealed that mysids exposed to TBT leachates were generally less active than control animals. Treatment mysids did not respond as quickly to stimuli such as changes in light intensity or the presence of moving objects in test tanks, i.e., pipettes to remove dead animals. In many instances the mysids from treatment tanks were not actively swimming but rather "resting" or debilitating on the bottom. Similar responses with copepods have been described by U'Ren (1983). Throughout the test the control animals were very active and demonstrated behavior patterns similar to freshly collected mysids held under standard laboratory conditions.

During the experiment mysid survival was higher in the presence of sediment. However, at the end of the experiment when mysids remained on the bottom for extended periods, the presence of sediment may have become a deleterious factor. We believe the animals were smothered by sediment if they remained on the bottom in a weakened condition. The ANOVA could not extract this effect on day 10 due to the magnitude of the positive effect in both the CSS tanks (100% survival) and the TBT-SS tanks. The Mann-Whitney U Test showed that mysid survival in TBT-SS tanks was significantly greater than in TBT-SO tanks on days 7, 8, and 9. There was no statistically significant difference in mysid survival between those tanks on day 10. There was no statistically significant difference in mysid survival between CSO and CSS tanks at any time. Chemically, the positive effects of sediment in the TBT-SS tanks were still present, but they were probably tempered by the physical effects of smothering. This was not the case in the TBT-SO tanks where mysids could remain on the bottom without being covered with sediment.

The question of environmental significance in every toxicity test is an important one. Since we have shown that mysids exposed to organotins exhibit higher survival in the presence of sediment than the absence of sediment, it can be assumed that in other toxicity tests with mysids, similar results would occur if each experiment had a paired design such as this one. The next question is which experimental design best represents conditions in the marine environment. We suggest that toxicity tests with sediment are more realistic than toxicity tests without. Sediment may reduce toxicity and bioavailability through physical and chemical processes related to sediment particles and naturally occurring organics. There may also be some chemical in the sediment that increases mysid survival. Our experiment may underestimate toxicity since the volume of sediment was a significant portion of the total volume in the tank. However, standard toxicity tests in sterile glass aquaria with few organics, particles, or sediment probably overestimate toxicity. This raises doubts about much of the TBT toxicity data that exist in the literature. The expected toxicity of organotin-contaminated sediment from a previous experiment (Salazar and Salazar 1984) was not realized because it was probably affected by some of these physical/chemical processes.

Waldock and Thain (1983) have shown that sediment affected oyster (*Crassostrea gigas*) growth rates. Compared with the control without sediment, oyster growth rates were higher in the tanks with sediment. When exposed to TBT concentrations between 0.15 ppb and 1.6 ppb, growth rates were higher in tanks with sediment and TBT than in tanks with TBT only. The only mortalities occurred in tanks with TBT.

Another problem in assessing potential environmental impact is interpreting the relationship between toxicity data reported from the laboratory and field measurements from contaminated areas. The problem with the TBT toxicity literature has just been discussed. The problem with field measurements is that they are usually discrete samples taken at one point in time and space and may be quite variable. Zirino, Lieberman, and Clavell (1978) have shown that the concentration of certain heavy metals can vary by a factor of three or more with tidal cycle. They have explained that within San Diego Bay the incoming tide transports a large volume of clean, uncontaminated ocean water into the bay. The outgoing tide carries a large volume of water that has been contaminated from sources within the bay. Recent measurements by our laboratory at one particular location within San Diego Bay have shown that variations in TBT concentration approach an order of magnitude during the tidal cycle. These measurements raise even more questions regarding field measurements previously reported for San Diego Bay (Valkirs et al. 1985a) and their significance as related to toxicity values previously reported. Further, the variability in TBT concentration in our test system precludes a precise LC-50 determination, which enhances the problem. Attempting to extrapolate environmentally significant numbers for regulatory purposes by using these data is not reasonable.

The bioavailability of contaminants and their actual toxicity in seawater are highly variable. Although a minimum concentration of certain trace metals is important for phytoplankton growth (Guillard and Ryther 1962), the form or chemical state of those metals determines whether or not they are usable by organisms (Johnston 1964). Studies on some species of phytoplankton show that growth inhibition and copper uptake are related to cupric ion activity and not to total copper concentration (Sunda and Guillard 1976). Along with others, we have shown that toxicity is also related to the state of the cupric ion tested (Smith 1979). In experiments with mysids and brine shrimp nauplii, TBT was far more toxic than its degradation products, dibutyltin and monobutyltin (Seligman 1984). The addition of sediment to test tanks reduces the toxicity of copper to *Neanthes arenaceodentata* (Pesch and Morgan 1978) while the type of sediment influences copper toxicity and copper bioaccumulation (Pesch 1979).

It has been shown that organic chelators are often the most crucial factor in bioavailability of nutrients for phytoplankton growth in seawater (Johnston 1964, Barber and Ryther 1969). It has also been suggested that, except for copper and iron, complexing agents have a relatively minor effect on initial complexation in seawater and that organic compounds adsorbed on the surface of sediment particles control the adsorption of metals (Morel and Morel-Laurens 1981). Although the actual chemical mechanism is unclear, some organics found in natural sediments can reduce toxicity and bioavailability. The results of our previous experiment (Salazar and Salazar 1984) suggest that something in the TBT-contaminated sediment had the capacity to reduce toxicity and bioavailability of TBT in that sediment.

The present study suggests that something in the sediment may have reduced the toxicity and bioavailability of the TBT leachates. Another alternative may be that the sediment itself adsorbed a significant amount of TBT. However, TBT adsorption was not measurable. Measurements of TBT from sediment in TBT-SS tanks was below 8.0 ppb before, during, and after the experiment, and not significantly different from the CSS tanks.

In addition to naturally occurring organic chelators in sediments that reduce toxicity and bioavailability, animals also have the ability to sequester and detoxify contaminants. There are several possible methods of detoxification. Accumulated organotins may be modified by the animals to become biologically unavailable, or they may combine with lipids to reduce toxicity. Organotins may bind to metallothioneins (Roesijadi 1981) or be sequestered within cellular organelles (George et al. 1978). In each case, there is a reduced opportunity for toxic expression.

Thus, when estimating the potential toxicity of organotin AF leachates from Navy hulls to marine organisms, more effort should be directed toward duplicating natural conditions. This will prevent overestimating toxicity values and reduce the significance of questionable values currently appearing in the literature. For our purposes, however, the survival of mysids exposed to TBT leachates was higher in the presence of sediment than the absence of sediment. Questions could be raised about the variability of TBT concentrations during the 10 days of exposure. The flow-through leachate system described here is not perfect, but the alternative is less desirable. TBT solutions are very unstable, time-consuming to prepare, and present a storage problem for experiments of any length. To clearly define an LC-50 for *Acanthomysis sculpta*, a flow-through leachate system as used here is not the most appropriate technique. As in LC-50 tests we conducted with *Metamysidopsis elongata* (Smith 1979) and those conducted with *A. tonsa* (U'Ren 1983), a more appropriate technique would be a static renewal test using leachates. In the present study using 18-day-old juveniles, the 10-day LC-50 for *Acanthomysis sculpta* exposed to TBT leachates in the presence of sediment was approximately 0.60 ppb TBT. Only one concentration was used in this experiment, and thus it is impossible to predict precisely the comparable 10-day LC-50 for mysids exposed to TBT leachates. A 30 percent survival at a nominal 0.60 ppb TBT was observed after 10 days. The concentration necessary to produce 50 percent survival under these conditions is higher than for mysids exposed to leachates and sediment simultaneously.

An additional problem is that new contaminants like organotins are not well characterized. As they are studied further, new insight should be gained into their toxicity, means of improving the required ecological evaluation of dredged material and standard toxicity tests. The Navy research programs to expedite dredging and study the fate and effects of organotins in the marine environment will help guide the assessment of the potential for environmental impact. However, as the use of organotins increases, so will the need for additional work to understand fully the chemistry and bioavailability of organotins and to predict their effect on the marine environment.

CONCLUSIONS

The results of this toxicity test demonstrate that sediment affects the survival of mysids exposed to organotins. Juvenile *Acanthomysis sculpta* exhibited higher survival in the presence of sediment than in the absence of sediment. The variability in TBT concentrations during the course of the experiment precludes an accurate LC-50 prediction and raises doubts about LC-50 values reported by others using similar systems. Further, the absence of sediment in some of these other experiments probably indicates an overestimation of organotin toxicity. The environmental significance of all available data with regard to fleetwide implementation of organotin AF coatings is not clear. Additional work is required to understand fully the chemistry and bioavailability of organotins and to predict their effect on the marine environment. The Navy and regulatory agencies like CE/EPA should work closely together to establish realistic criteria for fleetwide implementation and dredging. The approach in testing organotin toxicity should be reevaluated in terms of environmental significance using protocols that address TBT variability in test tanks, the significance of field measurements, and the effects of sediment on survival.

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FOOD CHAIN STUDIES AT TIMES BEACH CONFINED DREDGED MATERIAL DISPOSAL SITE
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ABSTRACT

The 46-acre Times Beach Confined Disposal Facility (CDF) Buffalo, New York, was half filled with polluted dredged material between 1972 and 1976. Filling was stopped at the request of environmental groups because of the large numbers of migratory waterfowl and birds which occupied the area. In the intervening years, open water, wetland, and wooded upland zones have evolved with associated wildlife and vegetation. An objective of the biological and chemical studies in progress at Times Beach is to determine the extent to which contaminants in the dredged material are bioaccumulating in plants and animal food chains in the three environments of the Times Beach CDF. A further long-term objective is to determine if bioaccumulation or biomagnification produces deleterious effects on food chains including chronic toxicity, suppressed reproduction, carcinogenicity, and disruption of community structure.

Up the the present time, wetland and aquatic plant communities, earthworms, fish, insects, and toads have been sampled from the disposal area and analyzed for heavy metals, polychlorinated biphenyls (PCBs), and polyaromatic hydrocarbons (PAHs). Results of these samplings and analyses are reported in this paper. The data showed less accumulation of metals in plant seeds than in leaves or rhizomes. Cattail rhizomes were high accumulators of heavy metals, especially cadmium and arsenic. Aquatic plants accumulated greater concentrations of metals than wetland plants. Very low concentrations of organic contaminants were found in plant samples.

Accumulation of metals in earthworms at Times Beach was much higher than accumulation in earthworms from a natural reference area. Cadmium levels in earthworms were elevated. In general, metal levels in insects and toads from dredged material and a

contiguous background location were comparable. Cadmium appeared to be higher in insects from the dredged material.

Concentrations of heavy metals in fish indicated that mercury was more elevated in fish from the Times Beach site compared with a reference site at the mouth of the Buffalo River. PCBs and PAHs were also more elevated in fish from the Times Beach CDF.

Mammal, fish, and bird populations have been inventoried at the Times Beach CDF. Future study will be directed at sampling and analyzing key higher food chain species at the Times Beach for contaminant bioaccumulation and effects.

INTRODUCTION

The Times Beach disposal area is a 46-acre confined dredged material disposal area near the confluence of Lake Erie and the Buffalo River. The site received approximately 550,000 cu yd of polluted dredged material from the Buffalo River and Buffalo Harbor from 1972 to 1976. While being filled, the area became intensively used by migratory waterfowl, songbirds, and other wildlife. Over 200 species of birds and ducks have been observed at the Times Beach site (Andrle 1985). At the request of local environmental groups, filling was stopped in 1976 because of the large numbers and varieties of birds using the site. The completely enclosed area now contains about 50 percent shallow open water (1 to 7 ft deep), 25 percent marsh, and 25 percent scrub trees. Figure 1 is an aerial photograph of the site.

The marsh wetland is vegetated primarily with cattails, smaller areas of phragmites, rice cutgrass, purple loosestrife, and minor inclusions of sedges and rushes. The more elevated upland portion of the disposal area contains primarily cottonwood trees generally in the 2- to 10-in.-diam range. Understory consists primarily of dogwood, goldenrod, willow, and impatiens.

A 1985 animal survey (Andrle 1985) showed muskrat to be common in the marsh area, while cottontail rabbits frequented the wooded area. This survey inventoried 26 species of resident and breeding species of birds at the site.

Although the area is observed to foster lush vegetative growth and abundant wildlife, there is concern that contaminants from the sediment may accumulate and bioconcentrate in plant and animal food chains. Consequently, a long-term program has been initiated to study bioaccumulation by aquatic and terrestrial plants and animals, and possible effects on organisms including growth, reproduction, vitality, and carcinogenicity. This paper describes initial work on chemical characterization of sediment and bioaccumulation by plants, earthworms, fish, insects, and toads.

Figure 1. Aerial photograph of Times Beach disposal area

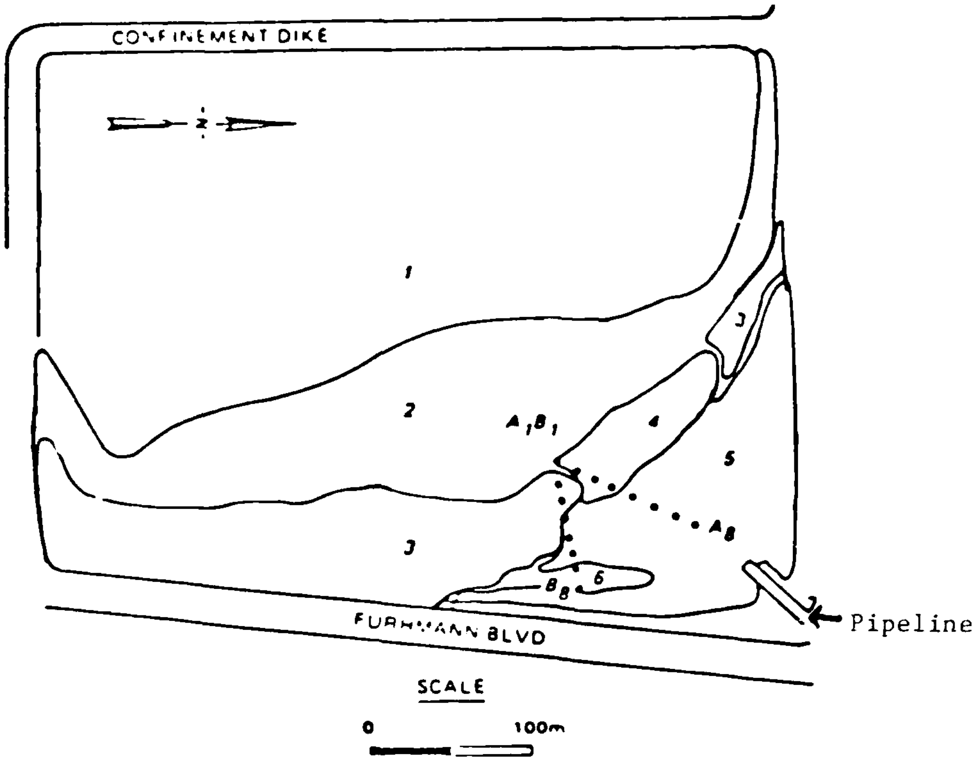
Although much work has been done on bioaccumulation by the above organisms in controlled experiments in the laboratory and at the Times Beach site (Marquenie, Simmers, and Kay 1985), this paper will be concerned with bioaccumulation by natural plant and animal communities.

STUDY METHODOLOGY

Vegetation Associations

Vegetation specialists examined the Times Beach disposal site and identified and mapped five major vegetation associations as shown in Figure 2. Area 1 is open water varying from 1 to 7 ft in depth with a mean depth of 3.5 to 4.0 ft. This area exhibits prolific algal growth and an abundance of attached and floating aquatic plants including potamogeton (pondweed) and elodea.

Area 2 is characterized almost entirely by broad and narrow leaf cattail (typha), while area 3 has mixed cattail and phragmites. Area 4 has many sedges, and the area 5 woodland is dominated by cottonwood (populus). Area 6 is dominated by phragmites. Rice cutgrass and purple loosestrife are also abundant in area 4.



1 - Open Water
2 - Cattail (Typha)
3 - Mixed Phragmites and Cattail
4 - Sedges
5 - Woodland (Poplar)
6 - Phragmites

Figure 2. Location of Times Beach vegetation zones and sampling transects

Transects (A and B in Figure 2) were laid out for sampling of vegetation, soils, and earthworms from vegetation areas 2 through 5. In addition, sediments, plants, and fish were sampled from the open water of area No. 1. Insects and toads were sampled from pitfall traps located in the wooded area as shown in Figure 3 (area 5 of Figure 2). Part of area 5 is nondredged soils and was sampled for background levels of contaminants in insects.

The approximate location of the pipe which discharged hydraulically dredged sediments from the Buffalo River and Harbor into the Times Beach is shown in Figure 2. This method of discharge resulted in deposition of coarser material (sand and gravel) and higher elevated topography in the vicinity of the discharge pipe. Silt and clay were carried further out into the disposal area at lower elevations.

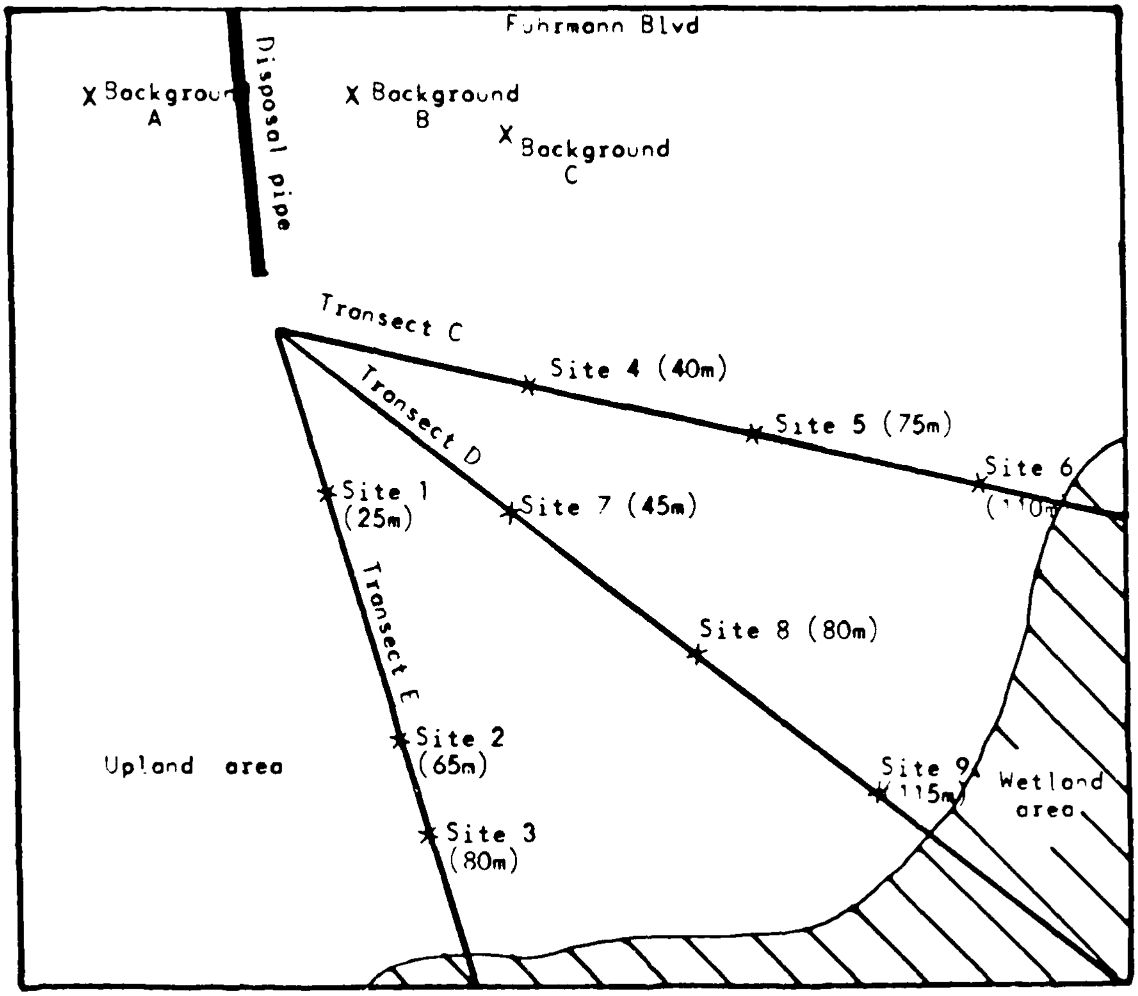


Figure 3. Position of pitfall traps placed in the upland area of the Times Beach confined disposal site

Reference Sites

It was considered useful to compare bioaccumulation in some organisms present at Times Beach with uptake by these organisms from other areas in the vicinity of Buffalo. Reference samples of fish were obtained from the Buffalo River at the point where it flows into Lake Erie, just north of the disposal site. Reference samples of earthworms were obtained from Grand Island located in the Niagara River about 12 miles north of Buffalo. The soils of this area were undisturbed but probably had received atmospheric fallout of pollutants from the Niagara Falls/Buffalo industrial complex.

RESULTS AND DISCUSSION

Soils

As discussed previously, sandier sediments were deposited on the surface nearer to the pipeline and finer silts and clays carried out further.

Figure 4 illustrates sediment profiles advancing from the pipeline in the woodland through the wetland and into the open water. In the intervening 9 years since disposal ceased, there has been considerable drying and oxidation in the sands of the wooded area, whereas the silts of the wetland and open water bottom remain saturated and chemically reduced. These physical and chemical differences are believed to be of considerable significance to the chemical transformation and bioavailability of inorganic and organic pollutants present in the sediment. Sampling transects were arranged to examine the physical, chemical, and biological differences proceeding from the discharge pipe point of entry to the open water of the disposal area.

The brown sandy surface sediment, especially in the woodland, has a "natural" soil appearance and odor with no visual or olfactory suggestion of contaminants. This layer dries out in summer months and is oxidized with the appearance of reddish iron-oxide colors. The underlying black to grey silty sediment has a semiliquid consistency and a strong oily odor. This chemically reduced material appears to be practically unchanged from the time of deposition, and the depth to this material appears to coincide with the permanent water table.

In June 1981, 16 surface samples of sediment were analyzed from the wetland and woodland. Metal and organic pollutants found are given in Table 1. Additional sediment analyses are given with plant uptake data. Major organic pollutants include PAHs and aniline compounds. The major source of PAHs is believed to be coking facilities from steel plants in the Buffalo Harbor and Buffalo River. Aniline compounds originate from a dye plant on the Buffalo River.

Vegetation

Four different plant species from the wetland area were identified as important food sources for local birds and ducks. Samples of these species were collected along with soil around their roots. Species collected included two rushes (*Carex stipata* and *Scirpus atrovirens*), cattail (*Typha latifolia*), and *Phragmites austrailis*. In addition, whole plant samples of pondweed (*Potamogeton*) and elodea were collected from the open water. Seeds, leaves, and, in the case of cattail, rhizomes were analyzed individually. Results of the plant analyses are given in Table 2.

The data show that, for all species analyzed, accumulation of metals in seeds is less than in leaves or rhizome. Seeds and leaves of the wetland plants (cattail, phragmites, rush) do not appear to accumulate toxic heavy metals to any extent. Aquatic plants (elodea, potamogeton) accumulated heavy metals to a much greater extent than the wetland plants.

Cattail rhizomes are seen to be high accumulators of heavy metals and may biomagnify cadmium and arsenic. Since rhizomes constitute an important food for muskrats, which are common at the site, future work will be directed at obtaining and analyzing muskrats for transfer of metals through the food chain. Roots of the wetland plants have also been collected for analysis since the rhizome data indicate that roots of other plants may have higher metal levels than leaves or seeds.

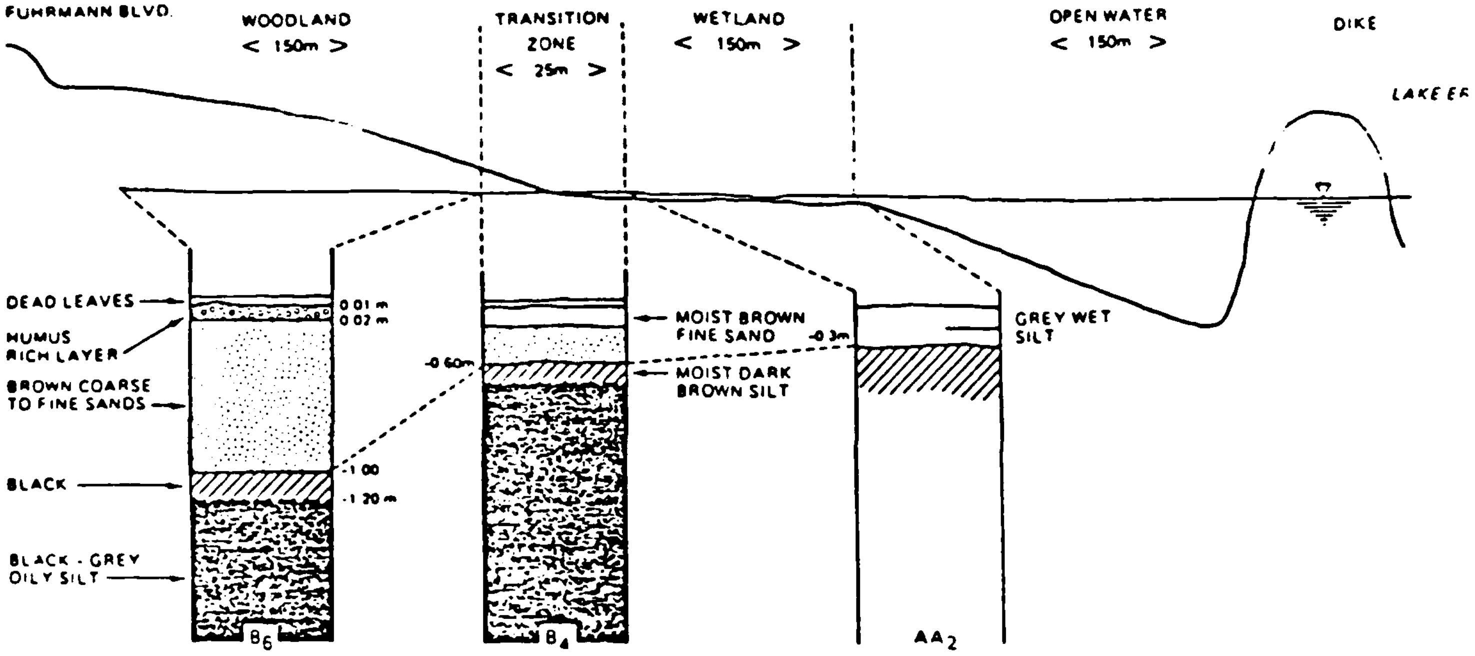


Figure 4. A soils transect through Times Beach confined disposal site (Marquenie, Simmers, and Kay 1985)

TABLE 1. METAL AND ORGANIC ANALYSES OF TIMES BEACH SURFACE SEDIMENTS (μg/g DRY WEIGHT)*

Pollutant	Range	Mean
Metal		
Zinc	1031-1845	1283
Cadmium	10.9-13.3	11.9
Copper	238-269	251
Arsenic	2.6-58.9	22.7
Mercury	2.9-9.4	4.8
Nickel	49-63	55
Chromium	302-393	332
Lead	156-1037	497
Organic		
Bis(2-ethylhexyl) phthalate	1.5-5.5	3.0
PCB(Aroclor 1242)	0.42-1.0	0.75
PCB(Aroclor 1254)	0.62-2.5	1.5
Aniline	1.7-2.8	2.3
1-amino-napthalene	1.7-4.1	2.7
N-benzyl-N-ethyl-aniline	2.4-7.0	4.5
4-(dimethyl-amino) benzo phenone	<0.1	<0.1
4,4-methylene bis (N, N-dimethyl-aniline)	0.54-1.4	0.93
N,N,N',N'-tetramethyl benzidine	<0.1	<0.1
p,p'-benzylidene bis (N, N-dimethyl-aniline)	2.6-4.7	3.3
benzo-(a)-pyrene	20-96	3.9
Chlordane	<0.2	<0.2
Toxaphene	<0.2	<0.2
Dimethyl phthalate	<0.2	<0.2
Diethyl phthalate	<0.2	<0.2
Dibutyl phthalate	<0.2	<0.2
Benzyl butyl phthalate	<0.2	<0.2
Lindane	<0.1	<0.1
Heptaclor	<0.1	<0.1
Aldrin	<0.1	<0.1
p,p'-DDE	<0.1	<0.1
Dieldrin	<0.1	<0.1
Endrin	<0.1	<0.1
p,p'-DDD	<0.1	<0.1
p,p'-DDT	<0.1	<0.1
Methoxyclor	<0.1	<0.1
Mirex	<0.1	<0.1
Phenol	<1.7	<1.7

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* 16 samples.

TABLE 2. METALS CONCENTRATION, VEGETATIVE ANALYSES FOR TIMES BEACH

	Metal, μg/g dry weight									
	As		Cd		Cr		Cu		Hg	
Sample	S-1	S-2	S-1	S-2	S-1	S-2	S-1	S-2	S-1	S-2
					C. stipata (Rush)					
Seed	0.17	0.18	0.22	0.12	0.57	0.16	7.5	8.8	0.029	0.031
Leaf	0.22	0.27	0.97	0.24	0.92	0.81	11	6	0.028	0.036
Sediment	74	48	4	4	177	129	228	232	4.71	3.83
					T. latifolia (Cattail)					
Seed	0.07	0.05	ND	0.48	0.03	ND	16	13	0.003	0.009
Leaf	0.15	0.14	2.76	1.44	0.26	0.21	4.9	3.7	0.021	0.023
Rhizome	59	39	6.85	5.88	29	70	49	96	0.294	1.19
Sediment	117	39	8.89	2.27	268	106	350	149	9.16	2.41
					P. australis (Phragmites)					
Leaf	0.15	0.39	0.06	0.11	0.4	2.3	15	15	0.024	0.052
Sediment	81	62	3.3	4.8	206	287	249	215	4.27	5.88
					S. atrovirens (Rush)					
Seed	0.05	0.11	0.03	0.07	0.12	0.13	11	10	0.02	0.02
Leaf	0.17	0.17	0.41	0.13	0.5	0.9	8.5	8.9	0.05	0.04
Sediment	28	26	1.03	0.76	73	100	94	77	1.21	2.18
					Aquatic Plants					
		As		Cd		Cr		Cu		Hg
Potamogeton		12		1.7		11		11		0.19
Elodea		6		2.3		6.3		26		0.13

EARTHWORMS

Native worms were collected from the wooded area of the Times Beach site and from the reference area on Grand Island. Very few earthworms were found in the wetland portion of Times Beach, probably because of the high water table and periodic inundation. The only worm species collected at the Times Beach site was *Lumbricus rubellus*, a species which confines itself to the top layer, rich in humus. Two composite samples consisting of 85 and 57 worms each were collected and analyzed. Two composite samples each of three worm species (*Lumbricus rubellus*, *Octalasium lacteum*, and *Allobophora chlorotica*) were collected and analyzed from the reference site. Concentrations of heavy metals, PCBs, hexachlorobenzene (HCB), and PAHs for the worm samples from both sites are given in Tables 3-5. PCB and PAH components are identified in Table 6.

The data show significantly greater accumulation of heavy metals by *L. rubellus* at the Times Beach site compared with the reference site. Biomagnification of cadmium and perhaps arsenic is indicated. It should be noted that metal concentrations in worms from the reference site are also elevated when compared with values cited in the literature.

Table 4 shows much greater accumulation of PCBs by worms from the Times Beach woodland despite somewhat low sediment levels. Similarly, PAHs were much higher in worms from Times Beach. PAHs are elevated in Times Beach sediments.

Insects and Toads

Insects and toads were collected from pitfall traps at the 12 locations shown previously in Figure 3. All samples were collected from the wooded portion of the site. Earthworms which fell into the traps were also analyzed. The purpose of this sampling was to identify dominant groups of soil-dwelling invertebrates; to determine levels of the heavy metals Zn, Cu, Cd, Ni, Pb, and

TABLE 3. METAL CONCENTRATIONS (μg/g ASH-FREE DRY WEIGHT) IN NATIVE WORMS AT TIMES BEACH AND A REFERENCE AREA

Area	Species	Cd	Cu	Hg	As
Times Beach	*L. rubellus*	113.0	59.7	1.33	30.8
		84.4	58.3	1.95	52.9
Reference	*L. rubellus*	17.6	20.2	0.469	8.84
		22.4	30.2	0.549	11.8
	A. chlorotica	24.3	11.4	1.76	10.8
		22.5	10.4	2.00	10.4
	O. lacteum	36.5	12.5	1.77	6.47
		50.7	14.8	2.34	9.59

TABLE 4. PCB CONCENTRATIONS (μg/g ASH-FREE DRY WEIGHT) IN NATIVE WORMS AT TIMES BEACH (TB) AND A REFERENCE AREA (REF)

		PCB Component									
Area	Species	28	52	49	70	101	87	153	138	180	HCB
TB	*L. rubellus*	0.126	0.460	0.220	0.330	0.320	0.160	0.130	0.130	0.050	0.260
		0.207	0.900	0.460	0.590	0.560	0.280	0.200	0.210	0.160	0.370
Ref	*L. rubellus*	all values below detection limits =				0.040					0.013
	A. chlorotica	all values below detection limits =				0.040					0.012
	O. lacteum	all values below detection limits =				0.008 0.040					0.008

TABLE 5. PAH CONCENTRATIONS (μg/g ASH-FREE DRY WEIGHT) IN NATIVE WORMS AT TIMES BEACH (TB) AND A REFERENCE AREA (REF). APPROXIMATE DETECTION LIMITS ARE INDICATED (d)

Area	Species	PAH Component										
		1	2	3	4	5	6	7	8	9	10	
TB	*L. rubellus*	0.36	0.13	0.62	0.37	0.013	0.94	0.13	0.53	0.48	0.32	
		0.20	0.059	0.24	0.17	0.014	0.56	0.038	0.22	0.23	0.23	
Ref.	*L. rubellus*	0.11	0.0048	0.051	d	d	d	d	0.012	0.020	d	
		0.17	0.0090	0.086	d	d	d	d	0.022	0.040	d	
	A. chlorotica	0.20	0.0064	0.090	d	0.071	0.045	d	0.012	d	d	
		0.13	0.0042	0.066	d		0.023	d	0.0070	0.014	d	
	O. lacteum	0.088	0.0040	0.039	d	0.040	0.013	d	0.0074	d	d	
		0.11	0.0051	0.057	d		0.020	d	0.0072	0.027	d	
		11	12	13	14	15	16	17	18	19	20	21
TB	*L. rubellus*	d	0.19	0.45	0.27	0.59	0.16	0.15	0.76	0.48	d	0.14
		d	0.13	0.27	0.14	0.31	0.086	0.13	0.64	0.30	d	0.086
Ref.	*L. rubellus*	d	0.0048	0.024	0.010	0.011	d	d	0.10	d	d	d
		d	0.0027	0.034	0.015	0.018	d	d	0.14	d	d	d
	A. chlorotica	d	d	0.0083	0.0051	d	d	d	0.17	d	d	d
		d	d	0.0085	0.0049	0.0028	d	d	0.18	d	d	d
	O. lacteum	d	d	0.0074	0.0047	d	d	d	0.14	d	d	d
		d	0.0015	0.019	0.0080	0.0065	d	d	0.13	d	d	d
d		0.15	0.0015			0.003	0.015	0.015		0.015	0.004	0.003

TABLE 6. NOMENCLATURE OF PCB AND PAH COMPONENTS UNDER STUDY WITH REPORTING CODE NUMBERS

PCB Components

28	2,4,4',	-	trichlorobiphenyl
52	2,5,2',5',	-	tetrachlorobiphenyl
49	2,4,2',5',	-	tetrachlorobiphenyl
70	2,5,3',4',	-	tetrachlorobiphenyl
101	2,4,5,2',5',	-	pentachlorobiphenyl
87	2,3,4,2',5',	-	pentachlorobiphenyl
153	2,4,5,2',4',5',	-	hexachlorobiphenyl
138	2,3,4,2',4',5',	-	hexachlorobiphenyl
180	2,3,4,5,2',4',5',	-	heptachlorobiphenyl

PAH Components

1	phenanthrene
2	anthracene
3	fluoranthene
4	pyrene
5	3,6-dimethylphenanthrene
6	triphenylene
7	benzo(b)fluorene
8	benzo(a)anthracene
9	chrysene
10	benzo(e)pyrene
11	benzo(j)fluoranthene
12	perylene
13	benzo(b)fluoranthene
14	benzo(k)fluoranthene
15	benzo(a)pyrene
16	dibenzo(a,j)anthracene
17	dibenzo(a,i)pyrene
18	benzo(g,h,i)perylene
19	indeno(1,2,3-c,d)pyrene
20	3-methylcholanthrene
21	anthanthrene

Cr in each taxa; and to assess the potential value of these organisms for use as indicators of heavy metal mobility in the ecosystem. A complete inventory of the numbers of individuals trapped in each order and family is contained in Strafford (1985). Although sufficient carnivorous and detrivorous insects for analysis were collected in this study, herbivorous invertebrate numbers were insufficient for analysis. Future studies will include larger numbers of herbivores.

Tables 7 and 8 contain data on metal levels in soil invertebrates, and Table 9 contains data on metal levels in toads. Table 10 compares levels of metals in insects from Times Beach to contaminated and uncontaminated sites from the literature. In general, metals levels in insects and toads from

TABLE 7. MEAN METAL CONCENTRATIONS IN SOIL INVERTEBRATES, TIMES BEACH WOODLAND DREDGED MATERIAL (μg/kg DRY WEIGHT)

Invertebrate	Zn	Cu	Ni	Cd	Cr	Pb
Carnivore						
Beetles (Coleoptera)	104	15	<1	2	4	<5
Spiders (Araneida)	376	193	<17	85	40	<30
Centipedes (Chilopoda)	246	54	<8	11	13	<12
Detrivore						
Millipedes (Diplopoda)	208	632	2	3	7	8
Wood Lice (Isopoda)	177	141	3	39	13	14
Earthworm (Oligochaeta)	833	94	12	60	41	62

the wooded dredged material area and the wooded background soil are comparable. However, cadmium appears to be higher in insects from the dredged material. Table 10 indicates that metal levels in invertebrates from the Times Beach dredged material are higher than uncontaminated sites from the literature, and comparable with contaminated sites.

From the results available, it appears that spiders reflect the greatest accumulation of the elements Zn, Cu, and Cd at the carnivore trophic level. Uptake of Zn, Cu, and Cd is evident from metals and analyses of wood lice. However, both Zn and Cu appear to have been accumulated to a greater extent by millipedes. Strafford (1985) examined the stomach contents of *Bufo americanas* and suggested that these toads had been feeding on many of the same species as were collected in the pitfall traps. Comparison of metal levels in the invertebrates collected at Times Beach (Tables 7 and 8) with metal levels measured in the toads (Table 9) indicates little or no bioaccumulation of trace elements between trophic levels. This may imply little mobility of the heavy metals from invertebrate prey into vertebrate predators, but may also be an indication that *B. americanas* would be a poor indicator species for use in studies of metal mobility.

Discussion of the mobility of heavy metals between trophic levels in the upland zone of the Times Beach ecosystem is incomplete due to the lack of data on heavy metal concentrations in herbivorous invertebrates. Specimens representative of the herbivore trophic level could be acquired using a longer sampling period. From the results available at present and summarized in

TABLE 8. MEAN METAL CONCENTRATIONS IN SOIL INVERTEBRATES, TIMES BEACH WOODLAND BACKGROUND (μg/g DRY WEIGHT)

Invertebrates	Zn	Cu	Ni	Cd	Cr	Pb
Carnivore						
Beetles (Coleoptera)	139	16	0.65	0.95	2.3	<2
Spiders (Araneida)	339	97	<6	14.6	23	<17
Centipedes (Chilopoda)	314	93	2.08	4.16	11.3	<6
Detrivore						
Millipedes (Diplopoda)	189	642	1.75	1.89	7.4	12
Wood Lice (Isopoda)	200	173	2.76	18	14.5	10
Earthworm (Oligochaeta)	942	144	10.9	39	22	27

TABLE 9. MEAN METAL CONCENTRATIONS IN TOADS, TIMES BEACH (μg/g DRY WEIGHT)

Sample	Zn	Cu	Ni	Cd	Cr	Pb
Whole specimen						
Dredged material	132	53	2	5	5	6
Background	132	56	1.1	6	6	7
Liver						
Dredged material	130	136	<4	8	13	<15
Background	107	272	<5	5	18	<11
Kidney						
Dredged material	194	48	<11	12	39	<40
Background	183	89	<10	7	49	<32
Bone						
Dredged material	179	7	<1	3	15	<5
Background	149	13	<2.3	1.9	14	<8

TABLE 10. COMPARISON OF METAL LEVELS FOR INVERTEBRATE TROPHIC LEVELS AT TIMES BEACH WITH LITERATURE VALUES (μg/g DRY WEIGHT)

Site	Zn	Pb	Cd	Cu
Uncontaminated sites (literature)*				
Carnivores	60	55	2 - 6	23
Detrivores	--	--	2	19
Contaminated sites (literature)*				
Carnivores	270	130	11 - 33	568
Detrivores	--	--	13	343
Times Beach (dredged material)				
Carnivores	242	<5 - <30	33	87
Detrivores	193	11	21	387

* Strafford (1985).

Table 7, there is some indication of accumulation of Cd and Zn, but not Cu or Pb between the detrivore and carnivore level of the ecosystem. Further studies involving some replication which would enable statistical analysis to be carried out are necessary to confirm any trends suggested by the results given in Table 7.

Fish

Fish species were collected from the open water of the Times Beach confined disposal site and from a reference area in the Buffalo River. Each species was pooled into one sample. In order to make a risk assessment for local human consumption, fish fillets (muscle) were prepared. Liver tissues were prepared in order to evaluate the presence of contaminants on a physiological basis and to compare Times Beach with the reference site for contaminants that do not accumulate in muscle tissue (e.g., cadmium).

Concentrations of heavy metals in fishes from Times Beach and a reference site (mouth of the Buffalo River) are given in Table 11. These results indicate that only Hg in fish is more elevated at Times Beach and approaches but does not exceed human food safety levels (Marquenie, Simmers, and Kay (1985).

PCB and HCB concentrations in fishes are shown in Table 12. Higher concentrations were found in fishes collected at Times Beach than in fishes of the aquatic reference site. The highest levels were found in muscle tissues of the carp. The total concentration of the nine isomers that were analyzed exceeded 1 μg/g (fresh weight basis). The concentrations in the livers were much higher for all species. The highest total concentration exceeded 8 μg/g for yellow perch.

Table 13 provides the concentrations of PAH components in fishes from Times Beach and the aquatic reference area. Clearly, only the most soluble and most volatile PAH components are found in the tissues. Known mutagens

TABLE 11. METAL CONCENTRATIONS (μg/g ASH-FREE DRY WEIGHT) IN FISHES AT TIMES BEACH AND THE ADJACENT MOUTH OF THE BUFFALO RIVER

Area	Species	Organ	Cd	Cu	Hg	As
Times Beach	Yellow perch	Muscle	<0.013	2.44	1.16	0.214
		Liver	0.042	1.32	0.102	0.936
	Pumpkinseed	Muscle	<0.019	3.30	0.717	0.579
		Liver	0.316	8.00	0.355	1.89
	Rock bass	Muscle	<0.04	1.07	2.80	0.541
		Liver	1.29	12.8	1.25	1.83
	Carp	Muscle	<0.025	2.73	0.767	0.801
		Liver	0.125	8.37	0.252	0.751
Buffalo River	Yellow perch	Muscle	<0.020	1.92	0.428	0.161
		Liver	0.280	3.96	0.053	0.575
	Pumpkinseed	Muscle	<0.028	1.99	0.730	0.534
		Liver	1.26	10.9	0.363	1.82

were absent or only occasionally detected in very low concentrations. The concentrations of the components that could be quantified were about five to ten times as high in fishes from Times Beach as in those from the reference site.

Mammal, fish, and bird populations have been inventoried at the Times Beach confined disposal facility. Future study will be directed at sampling and analyzing key higher food chain species at Times Beach for contaminant bioaccumulation and effects.

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TABLE 12. PCB CONCENTRATIONS (μg/kg ASH-FREE DRY WEIGHT) IN FISHES AT TIMES BEACH (TB) AND THE ADJACENT MOUTH OF THE BUFFALO RIVER (BR)*

Area	Species	Organ	PCB Components									
			28	52	49	70	101	87	153	138	180	HCB
TB	YP	m	100	180	130	160	130	59	68	56	22	38
		l	1,800	3,400	2,500	2,900	2,300	840	750	580	330	580
	PS	m	150	290	210	290	210	110	120	100	50	46
		l	420	750	590	780	600	360	370	330	190	100
	RB	m	160	370	280	400	310	150	190	210	83	35
		l	200	5,200	4,000	5,800	5,200	2,500	3,300	3,700	1,500	360
	C	m	630	1,100	830	1,400	570	280	260	310	110	190
		l	2,100	3,200	2,500	4,100	1,800	920	800	870	220	600
BR	YP	m	26	38	26	27	41	23	63	50	33	4.4
		l	530	900	650	600	920	87	900	580	230	110
	PS	m	18	26	25	28	33	26	47	34	28	6.5
		l	42	70	50	64	90	25	130	93	73	15

* YP = yellow perch, PS = pumkinseed, RB = rock bass, C = carp, m = muscle, l = liver.

TABLE 13. PAH CONCENTRATIONS (μg/g ASH-FREE DRY WEIGHT) IN FISHES AT TIMES BEACH (TB) AND THE ADJACENT MOUTH OF THE BUFFALO RIVER (BR). APPROXIMATE DETECTION LIMITS ARE INDICATED (d)*

			PAH Component									
Area	Species	Organ	1	2	3	4	5	6	7	8	9	10
TB	YP	Muscle	0.071	0.023	0.024	d	0.0051	0.056	d	d	d	d
		Liver	0.77	0.26	0.25	0.19	0.042	0.58	d	0.015	d	d
	PS	Muscle	0.11	0.044	0.065	0.025	d	0.16	d	d	d	d
		Liver	0.53	0.16	0.33	0.11	0.017	0.95	d	d	d	d
	RB	Muscle	0.12	0.018	0.031	d	d	d	d	d	d	d
		Liver	0.45	0.17	0.20	0.08	d	d	d	d	d	d
	C	Muscle	0.39	0.095	0.095	0.16	d	d	d	d	d	d
		Liver	0.63	0.20	0.16	0.20	d	d	d	d	d	d
BR	YP	Muscle	0.036	0.0025	0.0066	d	d	d	d	d	d	d
		Liver	0.11	0.022	0.045	0.069	0.013	0.026	d	d	d	d
	PS	Muscle	0.031	0.0022	0.0012	d	d	d		d	d	d
		Liver	0.059	0.0070	0.037	0.051	0.0070	d	d	d	d	d
d		Muscle				0.02	0.003	0.01	0.008	0.007	0.01	0.03
d		Liver						0.02	0.02	0.01	0.03	0.08

(Continued)

* YP = yellow perch, PS = pumkinseed, RB = rock bass, C = carp.

TABLE 13. (Concluded)

			PAH Component										
Area	Species	Organ	11	12	13	14	15	16	17	18	19	20	21
TB	YP	Muscle	d	d	d	0.0041	d	d	d	0.15	d	d	d
		Liver	d	0.070	d	0.011	d	d	d	0.55	d	d	d
	PS	Muscle	d	d	d	0.0027	d	d	d	0.13	d	d	d
		Liver	d	d	0.0095	0.014	d	d	d	0.63	0.13	d	d
	RB	Muscle	d	d	d	0.0055	d	d	d	d	d	d	d
		Liver	d	d	d	0.0067	d	d	d	d	d	d	d
	C	Muscle	d	0.012	0.036	d	d	d	d	d	d	d	d
		Liver	d	d	d	0.0059	d	d	d	d	d	d	d
BR	YP	Muscle	d	d	d	0.0030	d	d	d	0.14	d	d	d
		Liver	d	0.0075	d	0.0015	d	d	d	0.047	0.0084	d	d
	PS	Muscle	d	d	d	0.0033	d	d	d	0.15	d	d	d
		Liver	d	d	d	0.0048	d	d	d	0.22	d	d	d
d		Muscle	0.1	0.0015	0.003		0.0025	0.01	0.01		0.01	0.0035	0.003
d		Liver	0.35	0.0035	0.0009		0.007	0.040	0.03		0.0015	0.0095	0.008

THE INDIANA HARBOR EXPERIENCE

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ABSTRACT

Indiana Harbor and Canal is part of a highly industrialized watershed in northwestern Indiana. The bottom sediments within the navigation channel are highly polluted, with two reaches of the canal having polychlorinated biphenyls (PCBs) contamination in toxic levels (>50 ppm). The Corps has been unable to maintain the channel for over 10 years because of the lack of a suitable disposal site. The Chicago District is addressing the disposal of maintenance dredgings from Indiana Harbor and Canal along two lines. For the bulk of polluted dredgings, the District is preparing plans for a confined disposal facility to be constructed in Lake Michigan at East Chicago, Indiana. For those sediments contaminated with elevated levels of PCBs, the Chicago District has contracted with the Waterways Experiment Station (WES) to conduct studies on the disposal alternatives for these materials. In addition, WES is assisting the District in evaluating innovative dredging and disposal technologies and conducting a study on the impacts of the no-action alternative.

INTRODUCTION

The Indiana Harbor and Canal is located in East Chicago, Indiana, on the southern shore of Lake Michigan (Figure 1). Indiana Harbor is the fourth busiest port on the Great Lakes, having a deep draft navigation channel maintained by the Corps of Engineers (CE). The Grand Calumet River, tributary to Lake Michigan via the Indiana Harbor and Canal, has a relatively small watershed which is entirely urban/industrial. The major industries along the canal are steel, petrochemical, lead, and gypsum. The Grand Calumet River/Indiana Harbor Canal (GCR/IHC) has a long history of water quality problems and has been identified by the International Joint Commission of the Great Lakes (IJC) as a major area of concern.

The Chicago District, CE, maintained the navigation channel at Indiana Harbor and Canal by periodic dredging until 1972. Prior to 1970, dredgings were disposed to the open waters of Lake Michigan. After 1970, Federal environmental regulations prohibited the unconfined disposal of polluted

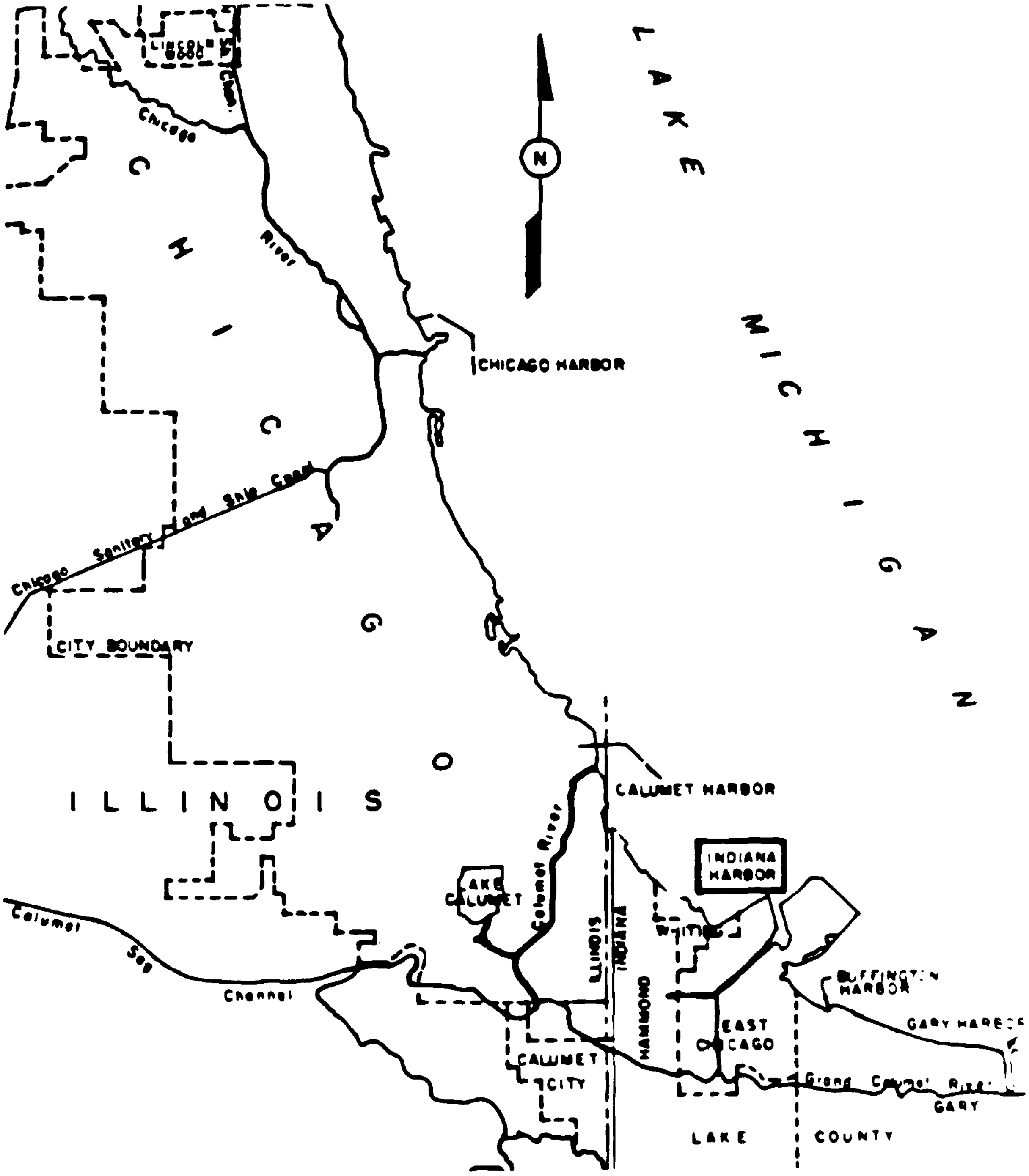

Figure 1. Indiana Harbor, Indiana, vicinity map

dredgings. In 1971-1972 maintenance dredgings were disposed to an enclosed lakefill on Inland Steel. Since 1972 the Corps has been unable to maintain the navigation channel because no acceptable disposal site was available. Despite several attempts, the Corps could not locate a site or local sponsor for almost 10 years. In 1983 the Corps completed a Site Selection Study for potential dredged material disposal sites. The Lake County Board of Supervisors offered to act as sponsor for the construction of a confined disposal facility (CDF) in Lake Michigan at East Chicago. The Corps has proceeded with the preparation of a Draft Environmental Impact Statement to be released in November 1985.

The dredging and disposal of polluted sediments from Indiana Harbor and Canal have received an unusually high amount of notoriety, including considerable media attention. A number of local citizens and environmental groups have expressed their opposition to the proposed project, fearing the dredging and disposal might endanger the quality of Lake Michigan. In November 1983, a few months after the release of the Site Selection Study, a meeting between the Corps, US Environmental Protection Agency (USEPA), State of Indiana officials, and representatives of citizens groups was held at the IJC biennial conference in Indianapolis, Indiana. The citizens groups described the GCR/IHC as a valuable resource forgotten by the State and USEPA and expressed doubt that the Corps would use the most up-to-date technologies in its dredging and disposal plans. At this meeting both the USEPA and Corps of Engineers, North Central Division, pledged to take actions to address the concerns raised. The USEPA identified its intent to prepare a Master Plan for the GCR/IHC, coordinating the regulatory and remedial actions necessary to improve water quality. The Corps stated it would explore innovative technologies for dredging and disposal of contaminated sediments.

USEPA MASTER PLAN

In 1984-85 the USEPA, Region V, prepared a "Master Plan for Improving Water Quality in the Grand Calumet River/Indiana Harbor Canal." The Master Plan report included a discussion of existing environmental problems and pollutant sources, a presentation of existing water quality control programs, and recommendations for improving water quality and aquatic habitat conditions.

The summary of existing environmental problems in the GCR/IHC stated that there were "high concentrations of conventional, nonconventional and toxic pollutants in the sediments and overlying water column. Although improved point source controls have resulted in significant improvements in ambient water quality conditions in recent years, the contaminated sediments continue to represent a major in situ reservoir of accumulated pollutants." The Master Plan report indicated that "the continued discharge of toxic and nonconventional pollutants (including contaminants leaching from sediments) are now the major limitation to the biological recovery of the GCR/IHC system."

The USEPA Master Plan formulated the following recommendations:

a. Continue the existing emphasis on pollutant controls.

b. Clarify the role of toxic pollutants in the river system.

c. Develop any additional toxic pollutant control programs that are necessary for restoration of the GCR/IHC.

The recommendations are to be implemented through a cooperative, interagency effort which is directed by USEPA. The Corps' participation is centered around the Environmental Impact Statement on Indiana Harbor maintenance dredging and its research and development study. The Corps' "assessment of alternatives to dredging and spoils disposal from the Harbor and Ship Canal will contribute to public and agency perceptions of the feasibility of remedial actions proposed for upriver sediment contaminations."

INDIANA HARBOR STUDIES

In 1984 the Chicago District contracted with the WES in Vicksburg, Mississippi, to conduct an R&D study. The purposes of the study were threefold. The first purpose was to fulfill the pledge made at the November 1983 IJC meeting to investigate innovative dredging and disposal technologies. This part of the R&D study was conducted last month in the form of an equipment demonstration at Calumet Harbor, Illinois. The second purpose of the R&D study by WES (which represents the major effort) was an evaluation of disposal alternatives for PCB-contaminated sediments from Indiana Harbor Canal. Thirdly, the Chicago District asked WES to investigate what has become known as the "no-action" alternative.

Equipment Demonstration

The purpose of this study was to demonstrate two pieces of innovative dredging/disposal equipment: a "Matchbox" hydraulic suction head and a submerged diffuser. The "Matchbox" is a Dutch-designed dredge head which had been used effectively in Rotterdam Harbor for the dredging of highly contaminated sediments with minimal resuspension. The submerged diffuser is a Corps-designed "flange" to be placed at the discharge end of a hydraulic dredge pipeline for controlled placement of dredgings directly on the ocean/lake floor. The diffuser reduces exit velocities, reduces resuspension, and can be used to place and cap contaminated sediments.

The equipment demonstration took place in October 1985 at Calumet Harbor in Chicago, Illinois (Figure 1). The Corps' hydraulic dredge "Dubuque" from the St. Paul District was used to demonstrate the equipment. The performance of the "Matchbox" was compared with a standard cutterhead. The submerged diffuser was demonstrated within the Chicago Area Confined Disposal Facility (CDF) at Calumet Harbor. The results of the demonstration and water quality monitoring should be available in January 1986.

Disposal Alternatives for PCB-Contaminated Sediments

The bottom sediments of the Indiana Harbor and Canal are heavily polluted with metals and organic contaminants such as PCBs and polynuclear aromatic hydrocarbons (PAHs). The levels of contamination present preclude any disposal of dredgings to the open water. Most of the sediments contain levels of contamination suitable for confined disposal in an upland or in-lake CDF. Two reaches of the canal contain levels of PCBs exceeding 50 ppm. These sediments are subject to Federal regulation under the Toxic Substances Control Act (TSCA), and the disposal alternatives are less clear. WES was asked by the Chicago District to evaluate the feasibility of three specific disposal alternatives for these PCB-contaminated sediments and to analyze the control measures required under each alternative. These disposal alternatives are: (1) upland CDF, (2) in-lake CDF, and (3) confined aquatic disposal (CAD).

In October 1984, 20 barrels of sediment from Indiana Harbor Canal and Lake Michigan were collected and transported to WES. A battery of chemical, engineering, and biological tests were performed with these sediment samples, including:

a. Physical/engineering properties.

b. Settling and consolidation tests.

c. Solidification/stabilization tests.

d. Bulk chemical analysis.

e. Modified elutriate tests.

f. Column capping tests--chemical/biological.

g. Plant uptake.

h. Animal uptake.

i. Leachate tests.

j. Runoff tests.

The sediment collected from Indiana Harbor was a highly plastic clay, having a high concentration of oil and grease and 12.5 percent total organic carbon content. Metals were present in elevated levels, as expected given the industrialized area. PAHs were present in high concentrations, including napthalene and benzo(a)pyrene. The mixed sediment sample also contained 33.4 ppm of PCB (Arclor 1248).

The tests performed by WES were used to evaluate the conditions expected during the three disposal alternatives. Modified elutriate tests were used to assess the quality of effluent from a CDF. Column capping tests were used to determine the thickness of cap required to isolate the contaminated sediments from the overlying water column. For these tests, aquatic organisms (fish, clams, and crayfish) were placed in the column over capped and uncapped sediments. Plant (sedge) and animal (worm) uptake tests were used to determine the potential for bioaccumulation in an upland CDF.

In addition to the above testing procedures, WES scientists have been developing new methods to simulate the processes of leaching and runoff in confined dredged material disposal. This effort was necessary because there was either no standardized method available or the existing method was too costly or time-consuming. Mr. Tommy Myers of WES will be giving a presentation at this meeting on the development of leachate testing protocol.

The preliminary results of the WES analysis of disposal alternatives for PCB-contaminated sediments from Indiana Harbor Canal are being reviewed by the Chicago District and North Central Division at this time. It appears that, with certain design and operational control measures, all three disposal alternatives would be feasible from an engineering and environmental basis. The specific control measures are beyond the scope of this paper. The final report should be available in January 1986. Following review of this report by the USEPA and State of Indiana, the Chicago District intends to discuss the selection of a disposal alternative with these agencies.

No-Action Alternative

The performance of maintenance dredging has long been viewed as a necessary evil by the public and environmental groups--a messy operation having no redeeming value except to navigation users. This preconception has been challenged only recently. If the USEPA can use dredging as a remedial action for a cleanup under Superfund, how is maintenance dredging that different?

When the Corps prepares an Environmental Impact Statement for a planned maintenance dredging and disposal, no action is one of the alternatives evaluated. Typically this alternative is examined only for economic impacts. Yet, if the proposed action is to remove polluted sediments from a river or harbor and place them in a confined environment, are there not some positive environmental impacts as well? Navigation channels are often located at the mouth of a river and act as a settling basin trapping sediments which would otherwise be transported out to the receiving water body. The maintenance of this sediment trap should be especially important if the sediments are heavily polluted and the quality of the receiving water is highly valued.

The Chicago District asked WES to evaluate the no-action alternative at Indiana Harbor. Since it was recognized that this was a large undertaking, WES agreed to do a "first-cut" investigation during 1985. This study was to examine the existing information on the GCR/IHC system, literature on the interaction between bottom sediment-water column, and sediment transport information.

The study would attempt to prioritize the mechanisms by which the polluted bottom sediments impact the water quality of the river and adjacent lake and to identify laboratory or field investigations necessary to quantify these impacts. Preliminary findings indicate that sediment-bound contaminants represent the most important mechanism for water quality impacts and that field studies of the suspended sediment sources and transport are of highest priority. The conclusions and recommendations of the no-action alternative study will be available in January 1986.

SUMMARY

The experiences of the Chicago District at Indiana Harbor have shown the current perceptions of the public and regulatory agencies toward the environment, dredging, and the removal of in-place toxic pollutants. The public is becoming increasingly impatient with regulatory agencies tasked with restoring and maintaining environmental quality. Organizations such as the IJC have recognized that environmental problems are not solved by study alone. At their recent biennial conference in Kingston, Ontario, the Water Quality Board of the IJC expressed this in the theme of their presentation "Let's Get On With It."

At Indiana Harbor, the Corps and USEPA have opportunity to make significant improvements to the existing environmental conditions. The USEPA has formulated a Master Plan to coordinate regulatory and remedial measures. The Corps has proposed a plan for dredging most of the navigation channel and has completed studies on innovative dredging technologies, disposal alternatives for PCB-contaminated sediments, and the impacts of the no-action alternative. With this information in hand, the Corps should be able to remove and confine a portion of the in-place pollutants from Indiana Harbor under its navigation authority. The USEPA will be able to define the environmental benefits from the removal of in-place pollutants beyond the navigation channel and evaluate the feasibility for a remedial cleanup action using dredging/disposal alternatives developed by the Corps.

THE MOVEMENT OF SEDIMENT UNDER A PROGRESSIVE WAVE

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ABSTRACT

Bottom sediments in the shallow sea and in-bay have very high water and organic matter content. These sediments are easily resuspended and settled, and flow easily under tidal currents or waves. These phenomena have caused many environmental problems, such as water pollution and decreased fairway depth in harbors and bays. In order to solve these problems, an hydraulic experiment on the movement of sediment under a progressive wave was carried out under differing conditions of water content.

It was found that the length of the sediment ripple on the interface between the water layer and sediment layer, the wave height of sediment oscillation, and the attenuation rate of progressive wave height reached maximum values when the water content was about around 247 to 250 percent.

Further, these characteristics could be evaluated by a nondimensional number $\phi(=\sqrt{\sigma/2 \cdot \nu \text{ sediment}} \cdot d)$ which was obtained from the thickness of both laminar boundary layer and sediment.

INTRODUCTION

A great volume of soft sediment has settled in the shallow sea and in-bay due to transportation of organic matter and sands from rivers and deposition of suspended substances. The following are characteristics of this kind of sediment:

a. High composition of very fine soil particles.

b. High natural water content.

c. Easily resuspends and flows.

d. High content of organic matter.

e. Very small bearing capacity.

This kind of sediment is easily resuspended and settled, and flows easily by tidal currents or waves. Therefore, these phenomena have caused constant water pollution and decreasing fairway depths in harbors and bays. The movement of the bottom sediment is very complicated, and studies of these phenomena are few (Ohtubo 1983, Sumitomo, Ishibashi, and Kuriyama 1975, Nagai, Yamamoto, and Ludwig 1983). In particular, the study identifying the difference of physical properties of sediment is not almost carried out.

To solve the problems caused by these phenomena, a method to predict the movement of sediments under the currents or waves must be obtained.

This paper will focus attention on the movement of sediments with differing properties, such as water content, under the progressive wave in order to clarify the mechanism of movement in the shallow sea and in-bay.

EXPERIMENTAL APPARATUS AND PROCEDURE

The wave tank used in this test was 21 m long, 1.5 m deep, 1.5 m wide, and was equipped with a piston-type wave maker, as shown in Figure 1.

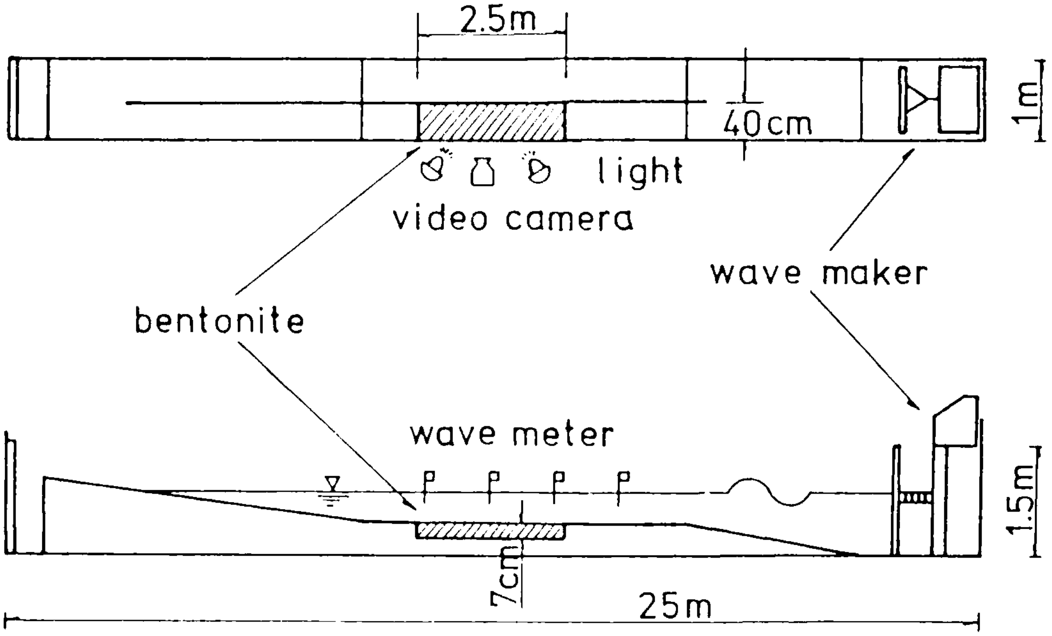

Figure 1. Experimental apparatus

A uniform bottom slope having a gradient of 1/20 was set up at the down-wave side of the tank as a wave absorber, and bentonite (as shown in Table 1) was put in the trench, which was 2.5 m long and 7 cm deep at the center of the tank.

Experiments were carried out under 91 conditions. Water content and wave height were changed from 200 to 270 percent and from 0.3 to 8.3 cm, respectively. Wave period, water depth, and sediment thickness were fixed at 1.05 sec, 20 cm, and 7 cm, respectively.

TABLE 1. CHARACTERISTICS OF BENTONITE

Specific gravity	2.60
Median diameter	34 μm
Texture	Sand 4%
	Silt 42%
	Clay 54%
Liquid limit	160.2%
Plastic limit	24.7%

The following were measured or observed:

a. Attenuation rate of wave height over sediment.

b. Development process of sediment ripple.

c. Resuspension of sediment.

d. Wave height of sediment oscillation.

e. Phase lag between wave motion and sediment oscillation.

Viscosity of sediment was measured by a rotary viscometer (cylinder type) in order to determine flow characteristics. The influence of consolidation was neglected because the test was completed within a few hours.

RESULTS

Physical Properties of Sediment

Physical properties of sediment should be separated into (a) that independent of water content such as chemical components, ingredient texture, liquid and plastic limits; and (b) that dependent on water content such as density and viscosity. In this experiment, apparent density and flow characteristics were tested.

Shearing rate and shear stress are used as the physical indexes of shearing flow, and yield value is used for plastic flow. Figure 2 shows flow curves of bentonite at each water content. In this figure, D is shearing rate, S is shear stress, and the gradient of curve shows viscosity. According to Figure 2, the viscosity of bentonite becomes high with decreasing of water content, and the flow curve shows that bentonite is a non-Newtonian fluid.

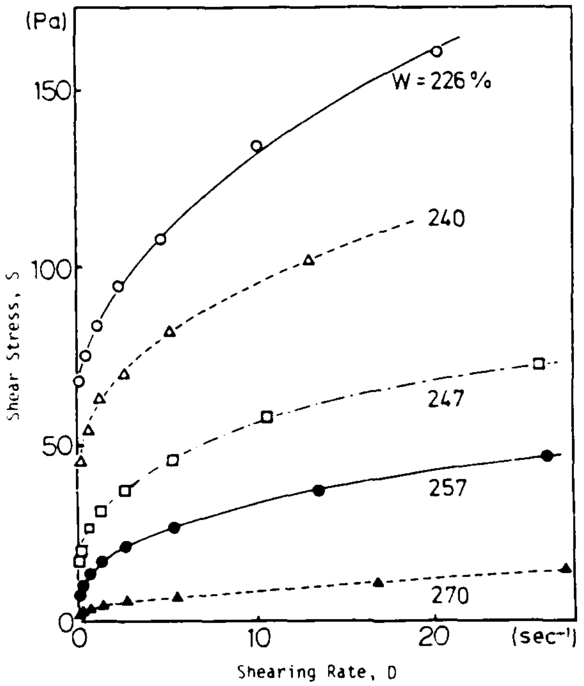

Figure 2. Flow curve of bentonite

The viscosity η_o for the low shearing rates (0-10 sec^{-1}) was calculated using this curve. The kinematic viscosity $\nu_{sediment}$ that was calculated using η_o and the apparent density ρ_o are shown in Figure 3. This figure shows that η_o and W (water content) are proportional at the range of the water content in this experiment. However, the viscosity was too high to measure by the viscometer in the case of bentonite with W of 200 and 211 percent. Therefore, the values of η_o were determined by extrapolation.

Movement of Sediment

The sediment behaved as described below with increasing wave height:

a. Appearance of sediment ripple on the surface of sediment. The sediment ripple appeared on the sediment surface when the progressive wave height was fairly small.

b. Growth of sediment ripple. As the progressive wave height increased, the length and height of the sediment ripple became large. Moreover, the profile of the sediment ripple was asymmetric, leaning in the opposite direction of the progressive wave.

c. Resuspension of sediment from the crest of sediment ripple. When the progressive wave height became larger, resuspension of sediment from the crest of the sediment ripple was observed.

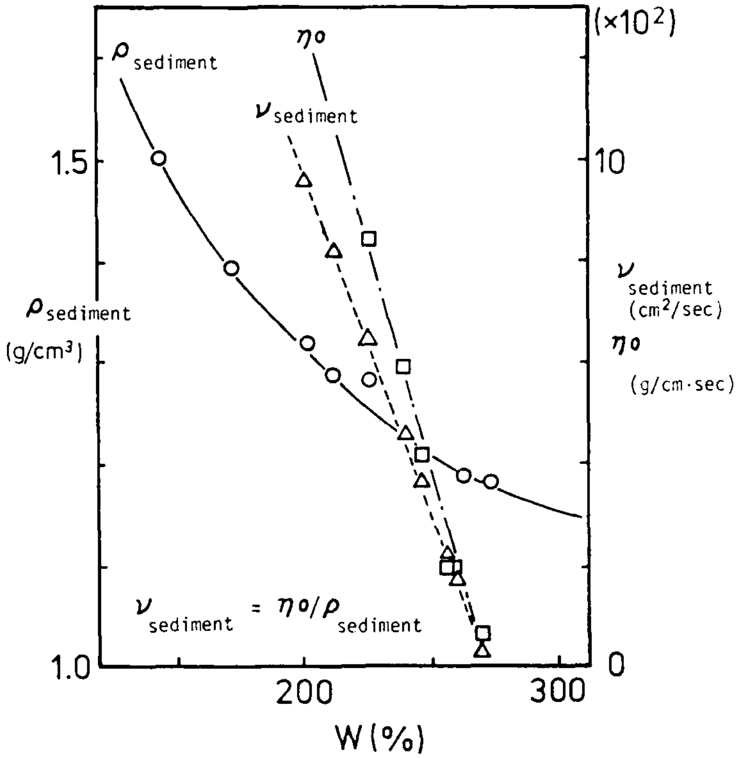

Figure 3. Viscosity of bentonite

d. Oscillation of sediment. The phenomena mentioned in a and c were observed on the sediment surface. However, when the progressive wave height exceeded a certain height, the whole sediment began to oscillate.

e. Wave motions and destruction of sediment. The oscillation of sediment mentioned in d became large gradually, resulting in the wave motions or destruction shown in Figure 4.

The appearance time and the scale of these phenomena vary with differing sediment physical properties. Therefore, in order to study further the movement of sediment, the influence of water content on the growth of the sediment ripple and the oscillation of the sediment was examined. Figure 5 shows the relation between the wave length λ at the center of the trench and the water content W . In this figure, the progressive wave height H1 was measured at the fixed bed on the incident side. The length of the sediment ripple increased as the progressive wave height increased. Figure 5 shows that the lengths of the sediment ripple are maximum at W = 247-250 percent for each wave height.

Similarly, Figure 6 shows the relationship between the wave heights (Hm1 and Hm2) of sediment oscillation and water content, in which Hm1 and Hm2 were measured at a position 1 and 2 m from the end of the trench on the incident side. The wave height of sediment oscillation increased with increasing wave height, and maximum values were reached at W = 247-250 percent in all cases.

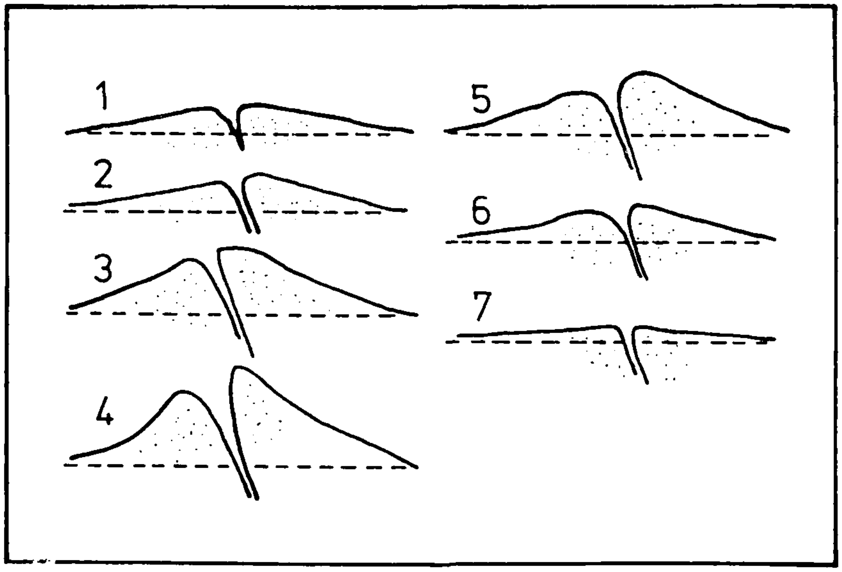

Figure 4. Process of destruction of sediment

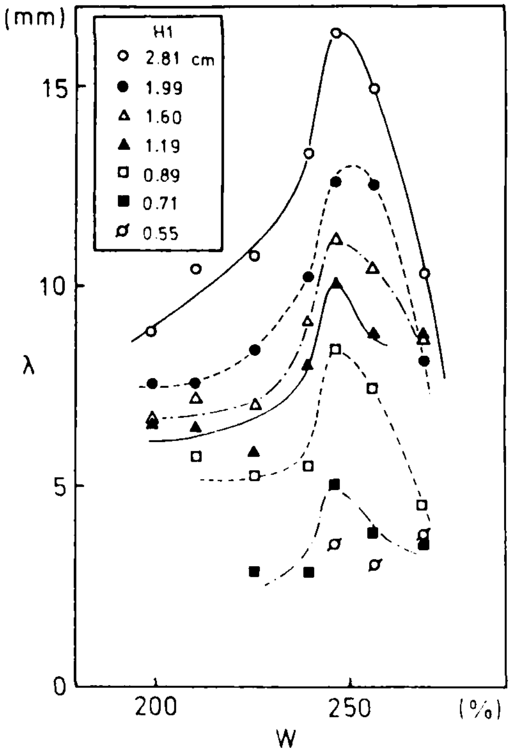

Figure 5. Length of sediment ripple at given wave heights and water contents

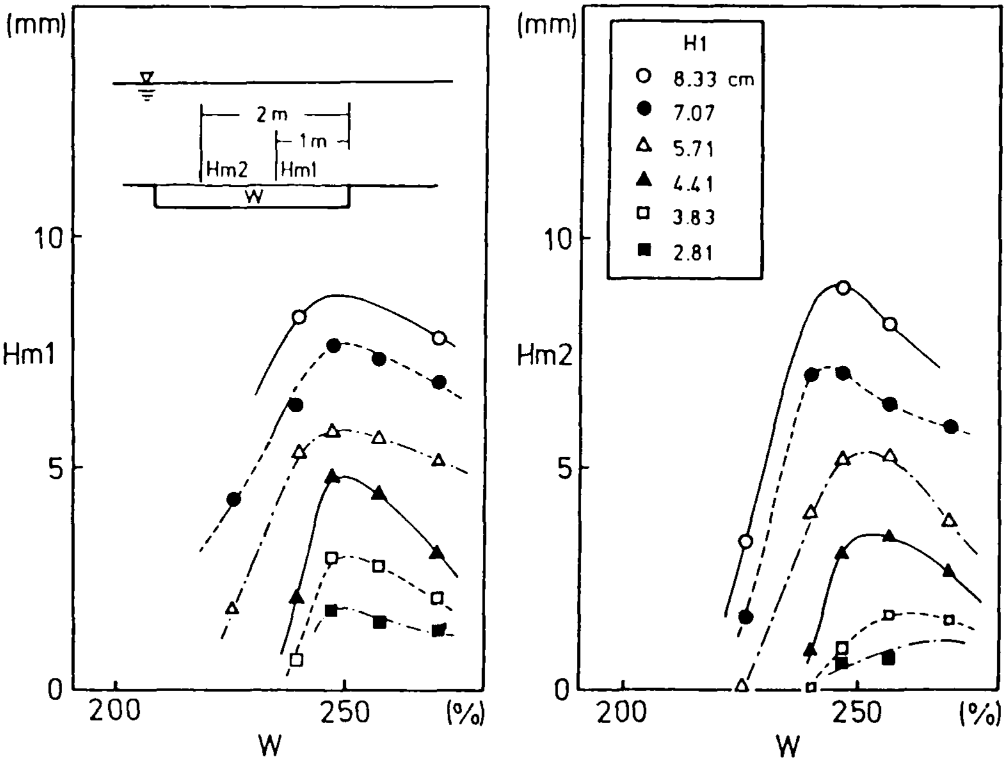

Figure 6. Relationship of wave height of sediment oscillation and water content

Interaction of Wave and Sediment

The movement of sediment was discussed above. As the sediment movement increased, the sediment and wave began to interfere with each other. In this experiment, the wave attenuation, which progressed over the sediment, was observed as reported by Nagai, Yamamoto, and Ludwig (1983).

The change of wave height as it progresses over the sediment is shown in Figure 7, in which the abscissa is a measuring position of wave height. The attenuation of wave height became considerable as the progressive wave height became large. Further, it seems to be maximum at $W = 247$-250 percent in all cases.

The attenuation rate Dw of wave height when the wave progresses by unit length is shown in Figure 8, which indicates that the maximum value of Dw exists at $W = 247$-250 percent.

The phas lag δ between the sediment movement and the progressive wave (the wave height is 8.3 cm) is shown in Figure 9, where δ is positive when the phase of sediment movement proceeds with one of wave. According to Figure 9, the phase lag δ is zero at $W = 247$ percent. Moreover, this value of W is the same water content value as when the length of sediment ripple, the wave height of sediment oscillation, and the attenuation rate of progressive wave height are maximum.

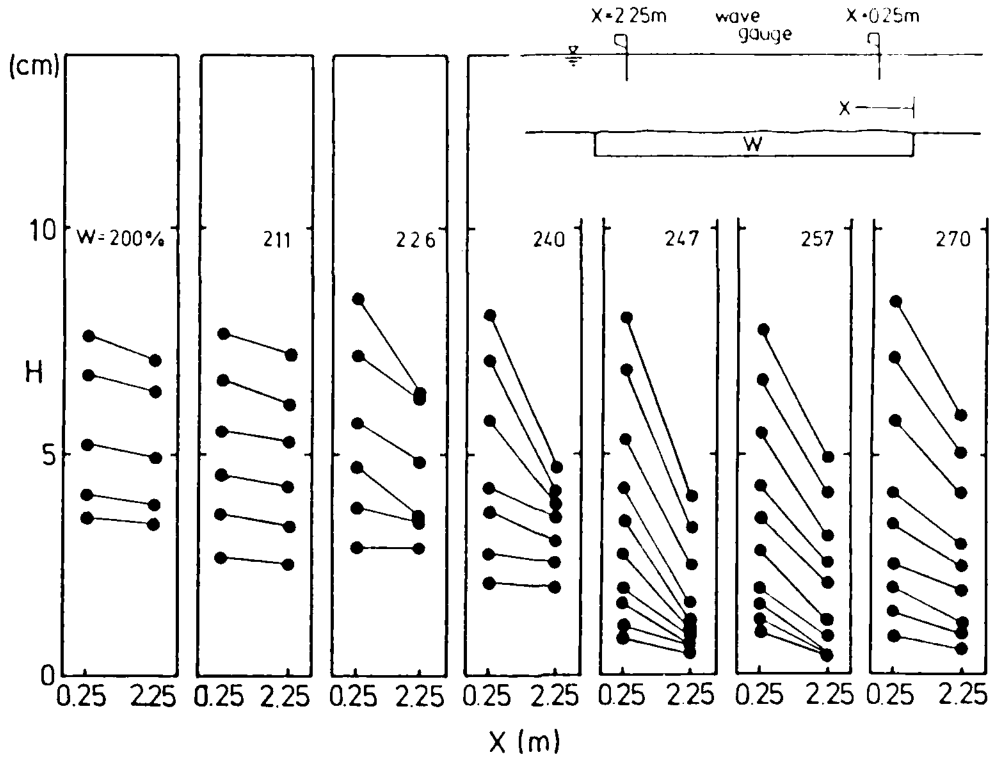

Figure 7. Attenuation of wave height progressing over sediment

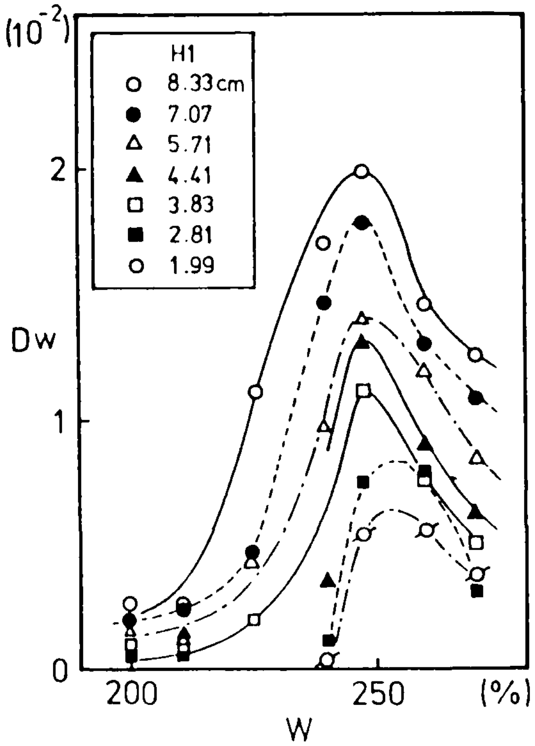

Figure 8. Relation of attenuation rate and water content

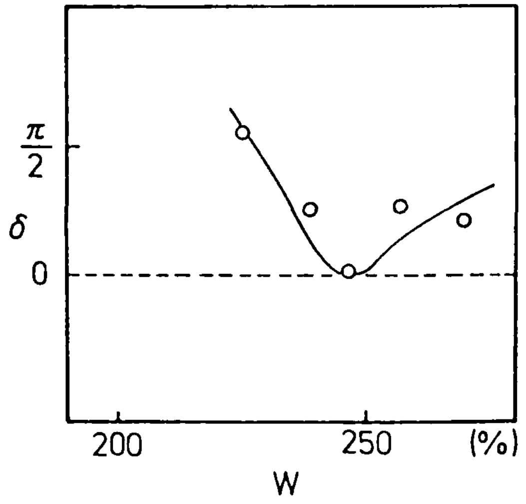

Figure 9. Phase lag between sediment oscillation and progressive wave

DISCUSSION

According to the results presented earlier, the attenuation of the wave progressing over the sediment and the movement of sediment are concerned with the phase lag between wave and sediment oscillation. Therefore, the two layered model shown in Figure 10 was examined with reference to Dalrymple • Liu's (1978) method. In this figure, the upper fluid is water which has the density ρ and the kinematic viscosity ν , and the lower fluid is sediment which has ρ sediment and ν sediment (= η_o/ρ sediment).

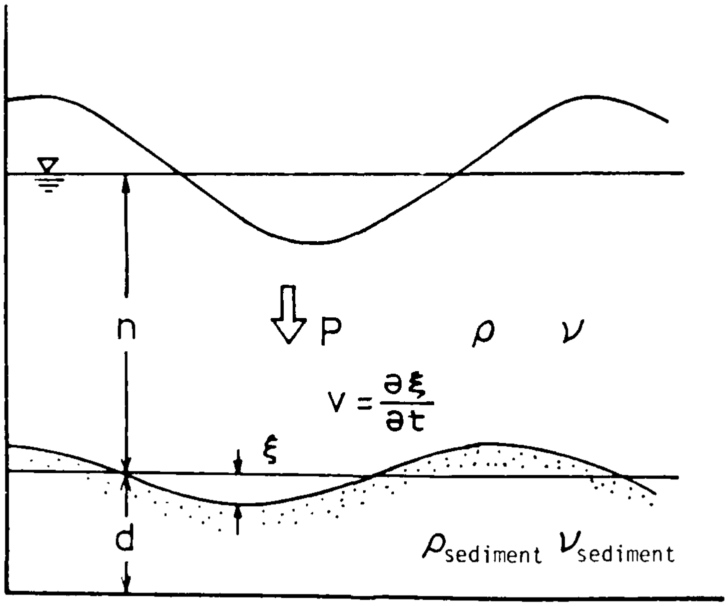

Figure 10. Two-layered model

When the wave progresses over the sediment, the pressure of the wave acts vertically on the sediment, and the velocity V due to the vertical displacement ξ occurs on the face of the sediment oscillation ($V = d\xi/dt$). At this time, the upper fluid does the work $P \cdot V$ against the lower fluid. When the wave and the sediment oscillation have the same phase, the work $P \cdot V$ is the greatest. Further, the attenuation of wave height also increases because the energy transmission from the wave to the sediment becomes great.

The following will be pointed out as physical properties which control the movement of sediment:

a. Sediment thickness : d

b. Kinematic viscosity of sediment : ν sediment

c. Angular frequency of wave : $\sigma = 2\pi/T$

The nondimensional number ϕ is given by the thickness of laminar boundary layer and the thickness of sediment as follows:

$$\phi = \sqrt{\sigma/2 \cdot \nu \text{ sediment}} \cdot d$$

The results, which are taken from Figures 5, 6, 8, and 9 by using ϕ, are shown in Figures 11-14. Accordingly, these figures, values of the length of sediment ripple, wave height of sediment oscillation, and attenuation of wave height, are arranged by the use of ϕ, and these values are maximum at $\phi = 0.6$.

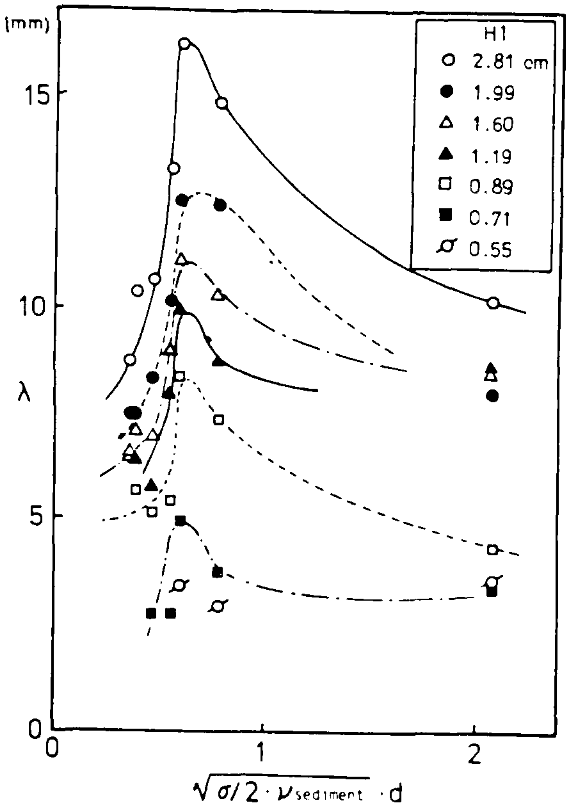

Figure 11. Relationship between length of sediment ripple and ϕ

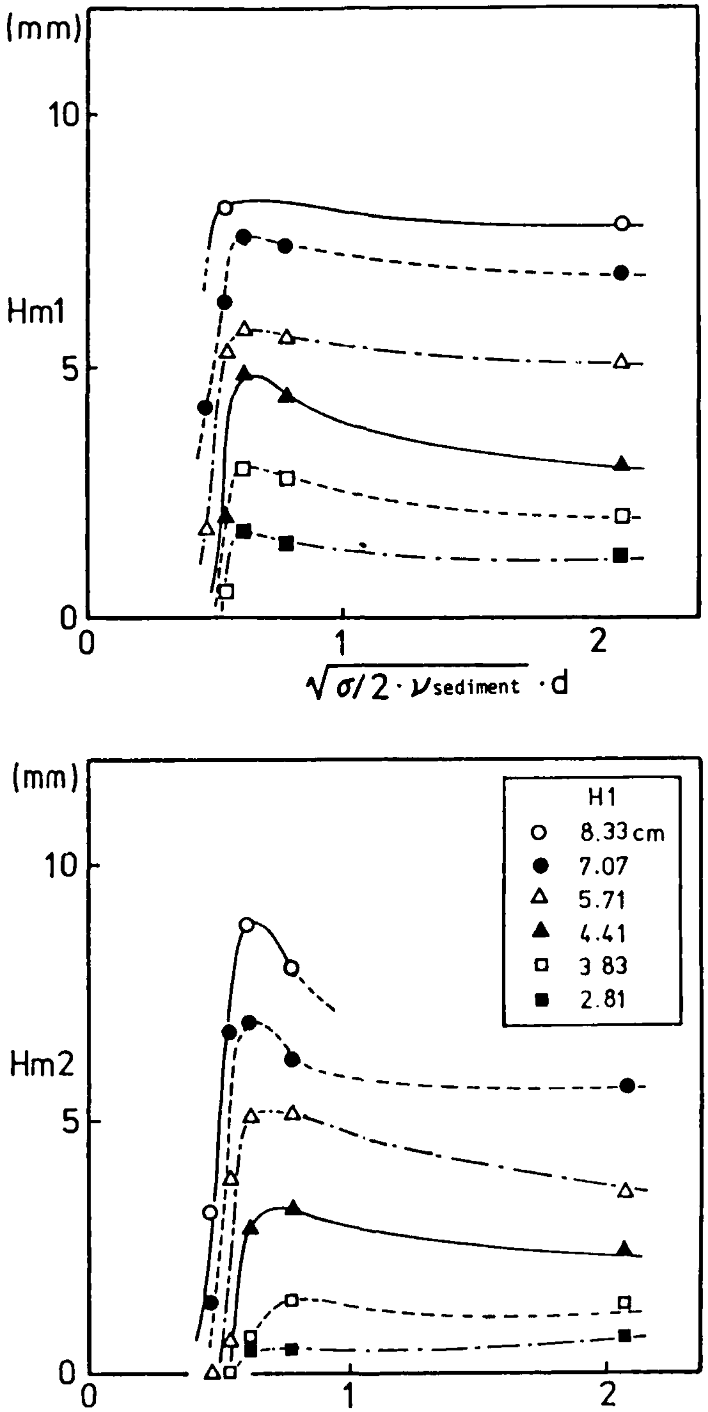

Figure 12. Relationship between wave height of sediment oscillation and ϕ

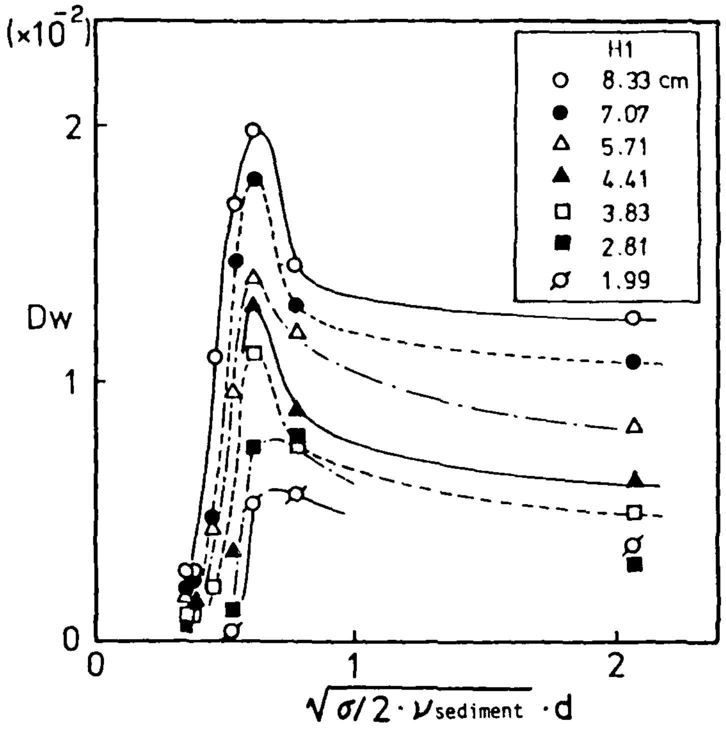

Figure 13. Relationship between attenuation rate of wave height and ϕ

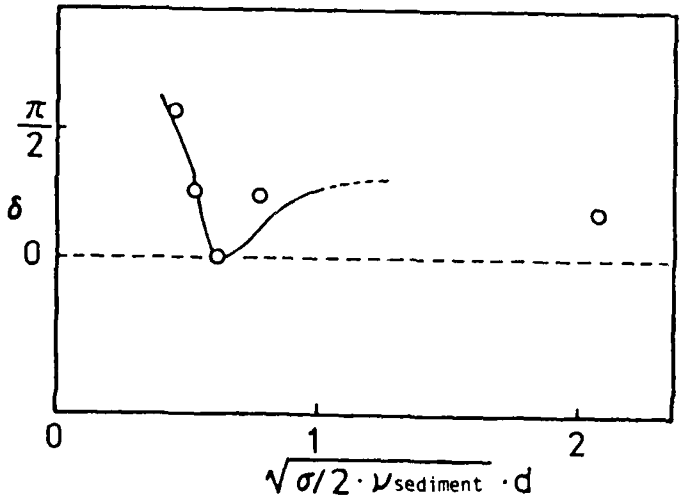

Figure 14. Relationship between phase lag and ϕ

An experimental study where the wave period and the sediment thickness were fixed was conducted to study the viscosity of sediment. More studies are needed to clarify the sediment movement. For this purpose, a series of experiments under conditions with differing wave periods and sediment thicknesses is planned.

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THE LONDON DUMPING CONVENTION AND ITS ROLE IN REGULATING DREDGED MATERIAL: AN UPDATE

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ABSTRACT

The Convention on the Prevention of Marine Pollution by Dumping of Waste and Other Matter--known as the London Dumping Convention or LDC--was formed in 1972 and came into force in the United States in 1975. The policy, regulatory, and technical aspects of the Corps of Engineers' ocean dumping program for dredged material are directly affected by the United States being signatory to the LDC. The contracting nations have met annually since LDC-1 met in September 1976. Since that time, an extensive program of work was adopted related to the implementation of the Convention concerning all aspects of dumping at sea. Requirements of the LDC are discussed in relation to the United States' domestic ocean dumping regulations. Emphasis is placed on application of the technical Annexes to the Convention. An update of the LDC annual meetings includes LDC-8 and LDC-9--which met in February 1984 and September 1985, respectively. Future strategy at the Convention emphasized the need for a multidisposal media approach at waste management that places ocean disposal on an equal basis with land based alternatives in the early planning phase of a project.

INTRODUCTION

Navigable waterways of the United States have played a vital role in the Nation's economic growth through the years. The Corps of Engineers (CE), in fulfilling its mission to maintain, improve, and extend these waterways, is responsible for the dredging and disposal of large volumes of sediment each year. Dredging is a process by which sediments are removed from the bottom of streams, rivers, lakes, and coastal waters; transported via ship, barge, or pipeline; and discharged to land or water. Annual quantities of dredged material average about 290 million cubic metres in maintenance dredging operations and about 78 million cubic metres in new work dredging operations with the total annual cost exceeding $150 million (Boyd et al. 1972). Ocean disposal accounts for about 15 to 20 percent of the Corps' annual disposal operations.

Sediment contamination has generated concern that dredging and disposal may adversely affect water quality and aquatic organisms. Consequently, most of the concern has centered on aquatic disposal. In recent years, the CE has disposed of approximately half of the material at open-water sites. Because many of the waterways are located in industrial and urban areas, sediments are often contaminated with wastes from these sources.

The lead responsibility for development of ecological criteria and guidelines regulating the transport and disposal of dredged and fill material was legislatively assigned to the Environmental Protection Agency (EPA) to share in consultation and conjunction, respectively, with the CE. Moreover, the enactment of Public Laws 92-532 (the Marine Protection, Research, and Sanctuaries Act (MPRSA) of 1972) and 92-500 (the Federal Water Pollution Control Act Amendments of 1972) requires the CE to participate actively in developing guidelines and criteria for regulating dredged and fill material discharge. The focal point for the developmental research on these procedures was the CE Dredged Material Research Program (DMRP) (Boyd et al. 1972, Brannon 1978).

Recent emphasis on deepwater port development for coal export and petroleum import and transfer will generate large quantities of dredged material in specific locations. As an example, planned deepening of Norfolk and Mobile harbors will generate about 35 and 141 million cubic metres, respectively. Ocean disposal may be the only viable alternative for this generally clean material. The ocean as an acceptable location for disposal of this material must be thoroughly investigated.

LEGAL REQUIREMENTS

The London Dumping Convention

The United States is signatory to an international treaty that is implemented through the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Dumping Convention (LDC)). The signatory countries to this convention shall then take all practicable steps to prevent the pollution of the sea that is liable to create hazards to human health, to harm living resources and marine life, to damage amenities, or to interfere with other legitimate uses of the sea (Article I of the LDC).

The LDC was negotiated in November 1972. After ratification by 15 member nations, the LDC came into force in the United States on 30 August 1975. The contracting or member nations have met annually since September 1976.

The construction of the LDC is quite simple and straightforward. The articles which comprise the legal framework of the Convention develop the formal regulatory foundation. The Annexes (I-prohibited materials, II-materials requiring special care, and III-provisions for developing regulatory criteria) develop the technical framework to implement the articles, and the interim guidelines for interpreting the Annexes or Articles comprise the day-to-day working level application of the LDC. The simplicity ends here. Many legal and technical terms and constraints were ill defined, or even undefined by the founders of the LDC, leaving the signatory countries a very difficult job in implementation through their domestic procedures. The LDC meets annually and conducts business on a consensus basis after debating the issues at hand. A vote is rarely taken and is regarded as choice of last resort. Issues (questions, legal and/or technical positions, rule changes, modifications or proposals, etc.) can only be introduced to the annual consultative meetings by member nations and thusly debated and acted upon by said member nations. Technical and/or legal issues can also be referred to intersessional groups for further study and/or resolution. The LDC is an active and dynamic treaty that tries to incorporate the state of the art in its deliberations while remaining responsive to many opposing views regarding ocean disposal.

The policy, regulatory, and technical aspects of both the Corps' and EPA's ocean dumping programs for dredged material are directly affected by the United States being signatory to the LDC. Domestic criteria must, at a minimum, be equivalent to and contain all of the basic constraints set forth in the international regulations.

A Scientific Group was formed by the LDC to meet intersessionally to make technical and scientific recommendations to the Consultative Body concerning the ecological aspects of regulating dumping at sea. The group met in September 1977 and has since met annually approximately 6 months prior to each annual LDC meeting.

A Task Team 2000 has been organized to identify long-term goals of the LDC. The team's specific objective is to project the role of the Convention in the Year 2000.

An ad hoc Legal Experts Group on Dumping also meets intersessionally on an as-needed basis to discuss and/or resolve legal interpretation of the LDC. Critical issues under discussion by this group are incineration at sea and seabed emplacement of high level radioactive wastes.

The Deputy Director of Civil Works of the Office, Chief of Engineers, attends the annual meetings of the LDC as the Corps' policy representative. Dr. Robert M. Engler of the Waterways Experiment Station (WES) is the Corps' technical representative at both LDC and Scientific Group meetings. Corps participation is subsequently through membership on the US delegation (headed by State Dept.) to the LDC.

Marine Protection Research and Sanctuaries Act

The constraints of the LDC affecting dredged material are promulgated through Section 103 of the MPRSA specifying that all proposed operations involving the transportation for dumping of dredged material into ocean waters must be evaluated to determine the potential environmental impact of such activities (Federal Register 1973, 1977). Implementation of this evaluation program is aided by use of an EPA/CE Implementation Manual (EPA/CE Technical Committee 1977). The ocean dumping criteria also require that the published (Federal Register 1977) interim ocean disposal sites be designated as final ocean disposal sites by January 1980 or as amended. This designation can only follow a comprehensive ecological investigation of the site and preparation of a site designation (Environmental Impact Statement (EIS)) by the EPA (Federal Register 1977, EPA 1977).

An ocean dumping evaluation must consider materials prohibited from disposal by international treaty (LDC) (EPA 1977) (Public Law 92-254); the environmental impact; the general compatibility of the material with the disposal site; the need for ocean dumping with a thorough review of alternatives; impacts on aesthetics, recreation, and economics; and impacts on other uses of the oceans. Evaluations in CE regulation 33 CFR Parts 320 through 329 must also be applied.

Final regulations and criteria controlling ocean disposal of dredged sediments were published by the EPA on 15 October 1973 (CE 1976) in the Federal Register. The procedures (criteria) for assessing the suitability of dredged sediments for ocean disposal consisted primarily on the Elutriate Test (Saucier et al. 1978) in place of total sediment analysis. This procedure was used to address short-term water quality impacts but not the longer term benthic impacts. Bioassays were recommended only in general terms.

The MPRSA further required that the criteria for ocean disposal be updated at least every 3 years. The first updated criteria, which are currently in effect, were published in the 11 January 1977 Federal Register (1977). These criteria account for provisions of the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (Convention) and reflect recent legal challenges by the National Wildlife Federation as to the adequacy of the 1973 criteria (Federal Register 1973, 1977). The Convention bans the ocean dumping of materials containing other than traces of certain compounds. Compounds on the prohibited list are considered to be present in trace quantities when the dumping of dredged sediments containing these contaminants will not cause significant undesirable effects (1977).

EVALUATIVE REQUIREMENTS

The potential effects of the ocean disposal of dredged material on marine organisms and human uses of the ocean may range from unmeasurable to important. These effects may differ at each disposal site and must be evaluated on a case-by-case basis. The Federal Register (1977) provides criteria for such an evaluation, with an emphasis placed on direct assessment of biological impacts, and the appropriate technical procedures are found in Parts 227 and 228 (Federal Register 1977). These procedures and their relationship to each other are illustrated diagrammatically and completely described in the EPA/CE Manual (EPA/CE Technical Committee 1977).

PROHIBITED MATERIALS

The first evaluation involves the presence of certain prohibited substances (prohibited pursuant to the LDC and domestic law) that may not be ocean dumped under any circumstances. If materials such as high radioactive wastes or chemical or biological warfare agents are present, the permit application must be denied without further consideration. Dredged material, however, is highly unlikely to contain these substances and must usually receive the full technical evaluation required by the criteria.

EXCLUSIONS FROM TESTING

There are cases where dredged material is not considered chemically contaminated and would, therefore, cause negligible pollutional impact when discharged at an appropriate disposal site. Thus, material that meets the requirements of paragraph 227.13(b) of the Federal Register (1977) may be excluded from the technical evaluations required by Section 227.13(b) and need be evaluated only in terms of its compatibility with the disposal site and the considerations of Subparts C, D, and E, and the appropriate sections of Part 228. Dredged material that does not meet the exclusions must receive full testing for its potential for environmental impact. The evaluative procedures emphasize biological effects, rather than simple chemical presence of possible contaminants. Dredged material is separated for evaluation into three phases, as defined in paragraph 227.32(b) (1) of the Federal Register (1977). All three phases must be evaluated.

LIQUID PHASE

The liquid phase of dredged material may be analyzed chemically and the results evaluated by comparison to water quality criteria for all contaminants after allowance for initial mixing. The period of initial mixing, discussed in the EPA/CE Manual (EPA/CE Technical Committee 1977), must be allowed before comparing the predicted concentrations with water quality criteria. If the water quality criteria approach is not taken, the liquid phase must be evaluated by bioassays. The direct bioassay approach is to be used when the liquid phase may contain major constituents not included in the water quality criteria or when there is reason to be concerned about possible synergistic effects of certain contaminants. In these cases liquid phase bioassay can aid in evaluating the importance and the total net impact of dissolved chemical constituents released from the sediment during disposal operations.

SUSPENDED PARTICULATE PHASE

The suspended particulate phase of dredged material must be evaluated for potential environmental impact only by use of bioassays. The bioassays are used to evaluate directly the potential for biological impacts due to both the physical presence of suspended particles and to any biologically active contaminants associated with the particulates and/or the dissolved fraction. These bioassays must also be conducted in light of initial mixing.

SOLID PHASE

It is generally felt that if a dredged material is going to have an environmental impact, the greatest potential for impact lies in the solid phase. This is because it is not mixed and dispersed as rapidly or to such an extent as the liquid and suspended-particulate phases, and bottom-dwelling animals live and feed in and on the deposited solid phase for extended periods. Therefore, unless there is reason to do otherwise, the major evaluative efforts should be placed on the solid phase. Bioassays are required for evaluation of the potential impact of the solid phase. Solid-phase bioassay must also be interpreted in light of initial mixing and must be conducted with appropriate sensitive marine organisms consisting of at least three species of one filter-feeding, one deposit-feeding, and one burrowing species.

BIOACCUMULATION

All biological evaluations of the suspended particulate and solid phase are required by law (Federal Register 1977) to include an assessment of the potential for contaminants from dredged material to be bioaccumulated in the tissues of marine organisms. This is intended to assess the potential for the long-term accumulation of toxins in the food web to levels that might be harmful to the ultimate consumer, often man, without killing the intermediate organisms. Since concern about bioaccmulation is focused on the possibility of gradual uptake over long exposure times, primary attention is usually given to the solid phase that is deposited on the bottom. Bioaccumulation from the suspended-particulate phase is considered to be of secondary concern due to the short exposure time resulting from rapid dispersion of the suspended particulates by mixing. Because of the long-term nature of the concerns, bioaccumulation from the solid phase is best evaluated at present in the field. This can be done only when a historical precedent exists for the proposed operation; that is to say, past projects of similar pollutional characteristics were disposed at the site under assessment. Under these conditions a field assessment provides the most useful information because the animals have been exposed to the sediment under natural conditions for periods greater than are now generally practical in the laboratory.

TRACE CONTAMINANTS

As described in the EPA/CE Manual (EPA/CE Technical Committee 1977), the presence or absence of trace contaminants must be determined for all three phases of the material. Section 227.6 of the Federal Register (1977) is perhaps the key section of the criteria since dredged material may not be ocean dumped if it contains any of the listed substances in greater than trace amounts. This is not defined in terms of numerical chemical limits whose environmental meaning is uncertain, but rather "...EPA came to the conclusion that the basis for regulation (of trace contaminants) should be the probable impact of these constituents on the biota and that the measurement technique used should be bioassays on the waste itself" (EPA 1977). Section 227.6(b) expresses in regulatory language the idea that trace concentrations should be defined as those too low to cause an environmental effect (Federal Register 1977).

PERSPECTIVE

It should be made very clear at the outset that the policy of the United States acting through the Marine Protection Research and Sanctuaries Act of 1972 (Ocean Dumping Act) with application of the LDC constraints does not strictly prohibit the ocean dumping of dredged material. Rather, it regulates the dumping to prevent or strictly limit dumping which would adversely affect human health, welfare, or amenities of the marine environment, ecological systems, or economic potentialities (Federal Register 1973). The LDC prohibits the dumping of some materials as other than trace contaminants; however, dredged material contaminated with these prohibited materials may be ocean disposed using capping or other special care techniques that will rapidly render the contaminant material harmless.

These critical domestic and international objectives are implemented through criteria promulgated by EPA which incorporates state-of-the-art biochemical evaluative guidance and testing procedures. The legislation (Federal Register 1973, 1977) then requires that a determination be made such that the dumping will not unreasonably degrade or endanger human health, welfare, or amenities or the marine environment ecological system or economic potentialities.

Additional evaluative guidance is set forth in EPA's regulations to implement the Ocean Dumping Act and the international constraints. These are known as the Ocean Dumping Criteria and present the testing and evaluative procedures and specific constraints that must be applied to a proposed dumping operation. With regard to dredged material, it also presents important exclusions from the mandatory and comprehensive testing requirements. It should be noted that the LDC has also promulgated broad exclusions from the mandatory testing and specific prohibitions for dredged material and sewage sludge proposed for ocean disposal. For those materials that cannot be excluded, a comprehensive laboratory and field evaluative scheme is applied to the specific project. The general constraints outlined in the Criteria (Federal Register 1977) are such that various tests are conducted, so that when the materials are dumped, reasonable assurance is provided that no significant undesirable effects will occur due either to toxicity or bioaccumulation. As previously mentioned, there are numerous other constraints which must also be met, but these are the most rigorous.

Obviously, none of this language, which is taken directly from the legislation, the treaty, and the regulations, strictly prohibits ocean dumping with the exception of the trace contaminants provisions of the LDC. The legislation allows significant latitude and discretion in determining when effects are adverse and will result in unreasonable degradation or endangerment or result in undesirable effects.

As a consequence, ocean disposal should then be considered a viable alternative of these regulations and evaluated on an equal basis to inland based alternatives.

What have we learned about ocean disposal of dredged material since 1972 or passage of the MPRSA? Studies of existing ocean disposal sites (Engler 1981; Herner and Company 1978; Lee, Engler, and Mahloch 1976; Saucier, Calhoun, Engler, Patin, and Smith 1978), both domestically and worldwide, have not revealed the presence or formation of an ecological desert at any site investigated. The physical effects, however, are obvious. The dumped materials form somewhat stable mounds. The site recolonizes rapidly with marine organisms and many sites become favored fishing grounds; there has been no documented effect found outside the legal boundaries of the disposal site (Engler 1981; Herner and Company 1978; Lee, Engler, and Mahloch 1976; Saucier, Calhoun, Engler, Patin, and Smith 1978). Sites used as often as the Mud Dump Site in the New York Bight obviously remain in a state of physical stress as would any routinely used part of our ecosystem, be it a superhighway, a navigation channel, or an intensively farmed agricultural area. Moreover, there are adequate and appropriate guidelines (Federal Register 1977) and ecological testing procedures (EPA/CE Technical Committee 1977) to indicate the small percent of the highly contaminated and toxic dredged material. This mat may be unacceptable for ocean disposal and must be carefully managed and contained in a land disposal site, or it may be ocean disposed with appropriate special care techniques.

LDC UPDATE

The LDC applies to dredged material as well as all other materials considered for ocean disposal. However, guidance adopted by the past Consultative Meetings to the LDC has recognized that all the factors used to evaluate the ocean disposal of some material may not be applicable to dredged material and may in fact be excluded from the rigorous evaluation. First, the "Interim Guidelines of the Implementation of Paragraphs 8 and 9 of Annex I (prohibited materials) to the LDC" provide that, with respect to dredged material, the test procedures to be used for determining the potential for toxic effects, persistence, inhibition of life processes, and bioaccumulation. This information is used to determine whether Annex I materials are "trace contaminants" or "rapidly rendered harmless" (and thus exempt from the prohibitions of Annex I). This information may not be needed if chemical characterization of the material and knowledge of the receiving area allows an assessment of the environmental impact. In addition, Appendix 1 to the Interim Guidelines, "Test Procedures for the Interpretation of 'Trace Contaminants' and 'Harmlessness'" exempts dredged material, in the absence of appreciable pollution sources, from the chemical and biological testing requirements of the Annex I prohibitions.

The LDC, meeting in February 1983, was asked by its Scientific Group (which makes technical and scientific recommendations to the LDC concerning the ecological aspects of regulating dumping at sea) to consider whether or not dredged material contaminated with prohibited substances could be capped with clean material under the provisions of the LDC. It was generally agreed that capping techniques are considered to be technically and scientifically feasible in low-energy environments and should continue as field research and that the projects should be monitored until sufficient information is obtained as to the acceptability of capping for a broader selection of ocean environments. (*Note: Only 5 to 10 percent of dredged material considered for ocean disposal may fall into the LDC category of being unacceptable for ocean disposal.*) The Corps' research will continue to assess the technical applicability of innovative techniques such as capping and subaqueous borrow pits for disposing of contaminated dredged material. The engineering aspects of capping contaminated dredged material and the efficiency of capping in isolating the contaminated material from the water column and bottom-dwelling organisms are published and are available from the Waterways Experiment Station.

The Eighth Consultative Meeting to the LDC (February 1984) adopted these guidelines for the interpretation of Annex III to the Convention; these guidelines recognize that not all of the environmental factors enumerated in Annex III need be applied to the ocean disposal of "some uncontaminated dredged material." In addition and in regard to the evaluation of alternatives to ocean dumping, it was accepted by the 8th LDC that a general weighing and balancing of alternatives be conducted by a comparative assessment of human health risks, environmental risks, hazards, economics, and future uses of disposal areas for both sea disposal and the alternatives. If this comparative assessment shows the land alternative to be more practical, sea disposal should not be permitted. This is a major change in the philosophy that ocean disposal is a "choice of last resort." A report of the "Task Team 2000" included the US position that the ocean disposal of dredged material meeting appropriate standards is an acceptable alternative and may be evaluated with all other alternatives and be considered throughout the planning and decision process. This shift in past direction of US ocean disposal policy signals a need for a comprehensive total waste management strategy that considers environmental protection of land and its vital water and food resources as well as that of the ocean. The LDC-VIII meeting also rejected a proposal to expand the list of prohibited materials to include lead (Pb); their decision was based on a strong US position documented by a thorough evaluation of the ecological effects of lead in dredged material, sediments, and industrial wastes. The US evaluation demonstrated that a total prohibition of lead disposal was unnecessary and that lead-contaminated wastes could be adequately handled through other provisions of the LDC. A very divisive and still unresolved issue discussed at the seventh and eighth LDC is the possible prohibition of all materials containing low level radioactive materials and wastes. This could be construed to include some dredged material. It is the current US position that dredged material is nonradioactive for the purposes of regulation. The radioactive material issue will be resolved at LDC-IX in September 1985.

The ninth meeting of the LDC (LDC-IX) in conjunction with the eighth meeting of the Scientific Group (SG-VIII) made decisions on a number of issues significant to dredging and dredged material disposal. Criteria for allocating toxic substances to the Annexes or even removing existing substances from the Annexes were adopted unanimously by LDC-IX. These criteria now form a sound scientific and technical basis for designating which substances need stringent, moderate, or minimal regulation with regard to ocean disposal. It was determined at LDC-VIII, IX, and SG-VIII that the Guidelines for Implementing Annex III were inappropriate for dredged material and in many cases tended to unnecessarily overregulate dredged material. Additionally, several technical/regulatory terms remain undefined that could if adequately defined lead to a less complex and fractured regulatory scheme. Consequently, the SG-VIII recommended and the LDC-IX approved an intersessional meeting of LDC experts in dredged material disposal to develop specific Guidelines for Implementing Annexes I, II, and III for dredged material. This group will also attempt to define terms such as "rapidly rendered harmless" and "special care."

The LDC-IX also approved, by roll call vote, a resolution calling for the suspension of ocean disposal of low level radioactive waste (LLRW). The resolution is one of principle and is not binding on the Contracting Parties to the LDC. It is important to note that the vote was based on almost purely political reasons rather than on a scientific and technical basis as is called for in the articles of the Convention. A proposal to amend the Annexes to ban ocean disposal of LLRW was tabled for some future action. LDC-IX action on seabed emplacement of high level radioactive waste was delayed until LDC-X. Intersessional groups meeting prior to LDC-X will discuss disposal of persistent plastics, ropes, netting, etc., with a view of proper regulation and incineration at sea.

The Intersessional group of LDC dredging experts met in London 28-31 October 1985. The experts group agreed that specific guidelines for applying all technical aspects of the LDC were necessary. Consequently, the group developed and recommended for adoption by the LDC Scientific Group and subsequently to the Consultative Body "Guidelines for the Application of the Annexes to the Disposal of Dredged Material." These are "stand alone" guidelines that incorporate previous LDC Annex guidance, modified as appropriate for dredged material, and present a new section entitled "Disposal Management Techniques." All previous LDC Annex guidance will have dredged material references delected. The "new" Guidelines have exemptions and exclusions appropriate to dredged material and discuss the acceptability of sea disposal in relation to techniques to mitigate effects of highly contaminated material. The "new" Guidelines present a complete and holistic approach at managing the disposal of dredged material in conformity with Annexes I, II, and III of the LDC. Moreover, the "new" Guidelines recognize various disposal management techniques that may be used to incorporate the "rapidly rendered harmless" and "special care" terms into a disposal management strategy. These disposal management techniques are considered to be any actions or processes by which conformity with the LDC and the Annexes is achieved.

Future meetings of the LDC include the Scientific Group Meeting in April 1986 and the main consultative body in September 1986. Issues affecting dredged material at the SG-IX include (1) review of Annexes with regard to cadmium and lead, (2) sea disposal of dredged material, (3) clarification of undefined terms in Allocation Criteria, (4) field verification of predictive laboratory tests, and (5) review of comparative assessments between sea disposal and land-based alternatives. LDC-IX will address the findings of SG-IX and seabed emplacement of high level radioactive waste issues. Numerous other nondredging issues will be addressed. Because of the significant importance of the LDC to navigation and ocean disposal of dredged material, the Corps will continue to maintain an active participation and leadership role in all domestic and international aspects of the LDC with regard to policy and technical issues. Corps participation includes numerous interagency scientific and technical working groups and committees as well as State Department directed ocean policy groups. Continued Corps participation at these levels will ensure that current research on dredged material and the Corps' vast experience in managing dredged material disposal are integral to the ultimate direction of the LDC.

FUTURE STRATEGY

In closing, there is a critical need to develop a more integrated approach internationally and, more importantly, domestically to uses of the ocean in general, and to waste disposal in particular. Demands on the ocean from multiple sources (defense, fisheries, marine mining, waste disposal, etc.) have increased significantly in recent years with a corresponding increase in conflicts between specific user groups. At present, regulation of ocean uses is typically along generic lines, in some cases to the exclusion of other potential users of the ocean environment.

In regards to dredged material disposal, the present domestic approach to this disposal is medium-by-medium regulation. This approach results in the protection of a particular medium for its use in waste disposal without regard for the impact of these regulations on other disposal media. Available research clearly indicates that no disposal alternative or media is without some form of environmental constraint, nor does any one disposal alternative or media reflect the best management practice or offer the soundest environmental protection. It would seem far more appropriate to develop criteria and guidelines that regulate the waste and not the medium to ensure that all disposal alternatives are fully investigated and weighed during the planning process and treated on an equal basis. It is imperative that a final decision be made based on all available facts. The LDC is moving in this direction, although somewhat laboriously, and the US must continue to maintain a leadership role in this area.

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ANALYTICAL SURVEY OF ACCUMULATIONS OF SEDIMENTS IN CANALS OF TOKYO PORT

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ABSTRACT

The canal beds around the Tokyo port are covered with thick oozy sediments including floating sediments. These sediments often cause not only traffic problems but also environmental problems. In order to remove the sediments, a dredging project has been conducted. In order to perform the project effectively, observations and analyses were carried out to identify sediment characteristics, sediment sources, sediment supply, and sedimentation rates.

The thickness of oozy sediments ranged from 60 to 80 cm, and the color of the sediments was black due to the highly unaerobic conditions. The characteristic values of bottom sediments, the water content, the ignition loss and chemical oxygen demand, were about 200 percent, 10 percent, and 40 mg/g, respectively. These values indicate that the sediments contain a lot of organic substances.

To estimate the rate of sedimentations, samples were collected in two ways, short-term (1 day) collections and long-term (3 years) collections. The former method gave a considerably high rate of about 20 cm/year, and the latter gave a lower rate of 1.5 cm/year. However, direct observations of the sedimentation rate provided a range of 1.6 to 11.4 cm/year.

The major sources of the sediment are rivers (48 percent of the total load to the canals), phytoplankton productions (37 percent), and sewerage plants (15 percent).

INTRODUCTION

As shown in Figure 1, the Tokyo port is located at the bottom of Tokyo Bay and is the harbor at the estuary of the River Sumidagawa. It has played an important role as an import and export base for goods to support industrial activities and civil life for the past 300 years.

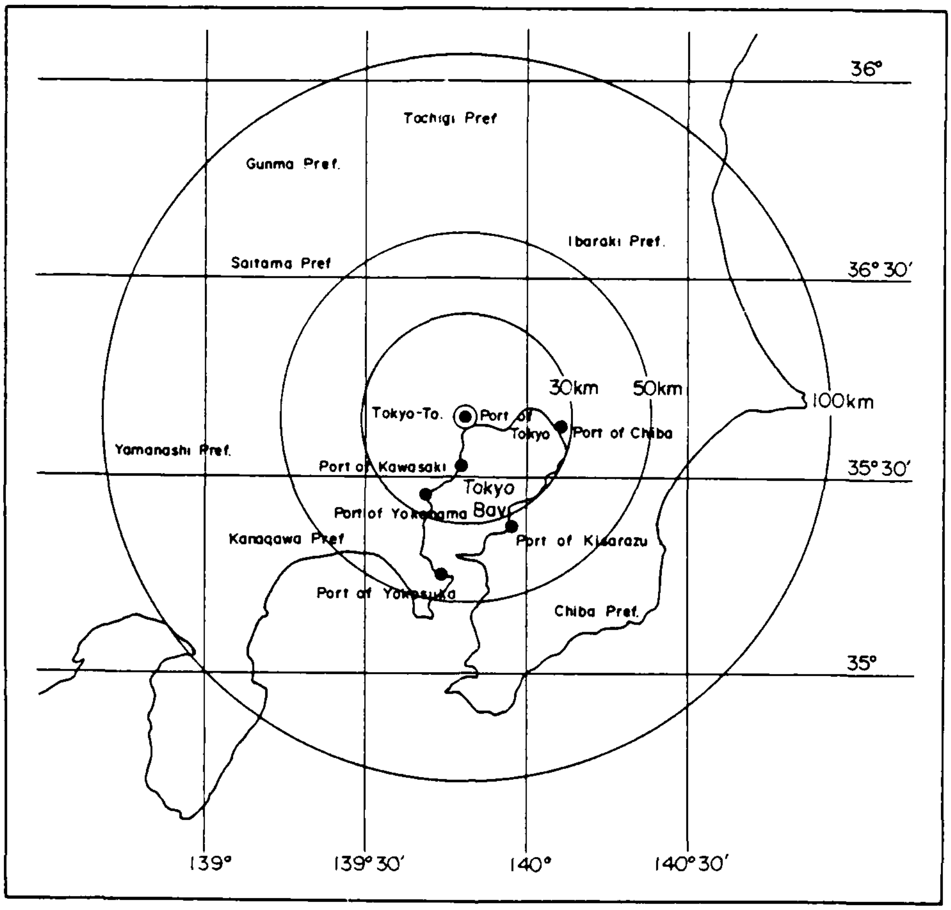


Figure 1. Location of Tokyo Bay

Today, Tokyo port is a gateway for the metropolitan area of Tokyo, as well as for the eastern region of Japan. In addition, it has occupied a position as an international trading harbour which represents Japan.

Canals in the Tokyo port are located at the boundary between the existing town zone and the harbor. Most of these canals are artificial and were constructed in the process of "reclamation" accompanying the enhancement of harbour facilities. The total length of the canals is about 55 km, the width ranges from 30 to 200 m, the depth from 2.0 to 4.0 m, and the total area about 600 ha. Warehouses and shops which need canal facilities are built among the canals, and these places use lighters. Small vessels such as fishing boats, passenger boats, and governmental boats are also moored here. In addition, a monorailroad, highways, sewerage treatment plants, seawalls, marine parks, business offices, and mansions are built; the canals are utilized for many different purposes.

The chemical oxygen demand (COD) value, which is used to reveal the condition of the aqueous environment around the Tokyo port (the reference value is below 8 ppm), had been lowered until 1980 through encouragement of various environmental integrity actions, and thus water quality had been improved. In 1981, however, the COD value began to increase. Since that time, particularly in summer, the value often exceeds the reference value for water environment.

A major cause of the increased COD value is organic sediments on the canal beds. Such sediments cause deterioration of the water quality, producing offensive odors, making people uncomfortable, and significantly lowering their living environment. To eliminate these problems, since 1972 the Tokyo Metropolitan Government has been engaged in gradually removing the sediments from the canal over a total length of about 40 km and a total area of about 370 ha. This has resulted in considerable improvement of the environment including prevention of offensive odors. However, because of the placement of the canals and the existing sediment/water conditions, new sediment buildup is occurring.

This paper introduces services performed in the Tokyo port for integrity of the aqueous environment, as well as summarizes results of the site survey performed since 1979 on properties of sediments, sedimentation rates, sources of the sediments, and sediment supply to the canals in order to encourage sediment removal.

OVERVIEW OF SERVICES

Our bureau is currently engaged in the following services in berths, sea passages, and canals to maintain integrity of the aqueous environment in the Tokyo port.

a. Sea surface monitoring. The port is always monitored by five patrol boats to prevent illegal disposal of soil, dust, and waste oil and to inform the agency of such illegal disposal.

b. Collection of waste oil from vessels. To prevent oil pollution of the port, waste oil is collected from the incoming vessels by oil collecting boats.

c. Prevention of oil diffusion. Seven oil treatment boats equipped with oil fences are available.

d. Sea surface cleaning. Seven cleaning boats collect and treat floating dust in the port.

e. Treatment of sunken wood and bark. The circulation of logs handled on the sea may produce sunken wood and bark. To prevent these from being sedimented on the sea bottom and from adversely affecting the water quality, the bark is treated through self-propulsive incineration boats and an on-land recycle center, by contractors.

f. Treatment of sunken and scrapped vessels. Sunken and scrapped vessels left in the port are obstacles to vehicle passage and adversely affect the water quality, and are therefore taken to a disposal area for incineration, etc.

g. Removal of sediments at canals. Services are being performed to remove sediments in the port, particularly in the canals:

(1) Plan: pollution prevention project in the Tokyo metropolitan area (determined in December 1972 and revised for the third time in March 1983).

(2) Performed in: 1972 to 1986.

(3) Amount of sediment planned to be removed: about 3.1 million cubic metres.

(4) Amount of sediments actually removed: about 2.6 million m (1972 to 1984).

(5) Sediment removal standard: six-point estimation method (tentative standard). Ignition loss, COD, and sulfides are measured. If any or the total of these values exceeds a predetermined range, the sediments are to be removed.

(6) Method: Dredging: Pump type dredging boat, sealed type grab dredging boat; transportation: Sealed type (box type) soil transportation boat, bottom opening type soil transportation boat; disposal to treatment plant: Pump type sediment raising method, secondary sediment feeding method.

DISCUSSION

Characteristics of Bottom Sediments

Stratification of Sediments

Sediments were sampled at station B-1 through B-5 as shown in Figure 2. Over the entire canal, the thickness of the sampled sediments was about 60 to 80 cm, while the thickness of sediments sampled at the Koto district was 40 to 70 cm. The thickness was lower here, indicating obvious layer divisions not observed at the other stations.

Irrespective of the sampling point, the color of sediments was black; this indicates that sediments in the shallow sea are significantly organically polluted. The sediments are black due to a combination of Fe^2 with hydrogen sulfides formed by decreased oxidation-reduction potential.

Except for Shibaura (St. B-2), gray-green to dark-gray layers 3 cm thick were observed at the surface section. The layers seem to have been produced in winter because water masses were mixed and thus bottom water became aerobic.

Characteristics of Sediments

Table 1 shows the results of analyzing the sediments discussed above. A distinct odor indicated the presence of hydrogen sulfides at all stations except Harumi (St. B-3). Water content ratio was not less than about 200 percent at all stations except Koto (St. B-4). Wet density was not greater than about 1.3 g/cm^3, ignition loss, COD, and total organic carbon (TOC) as indices of organic pollution were over 10 percent, 40 mg/g, and 20 mg/g, respectively,

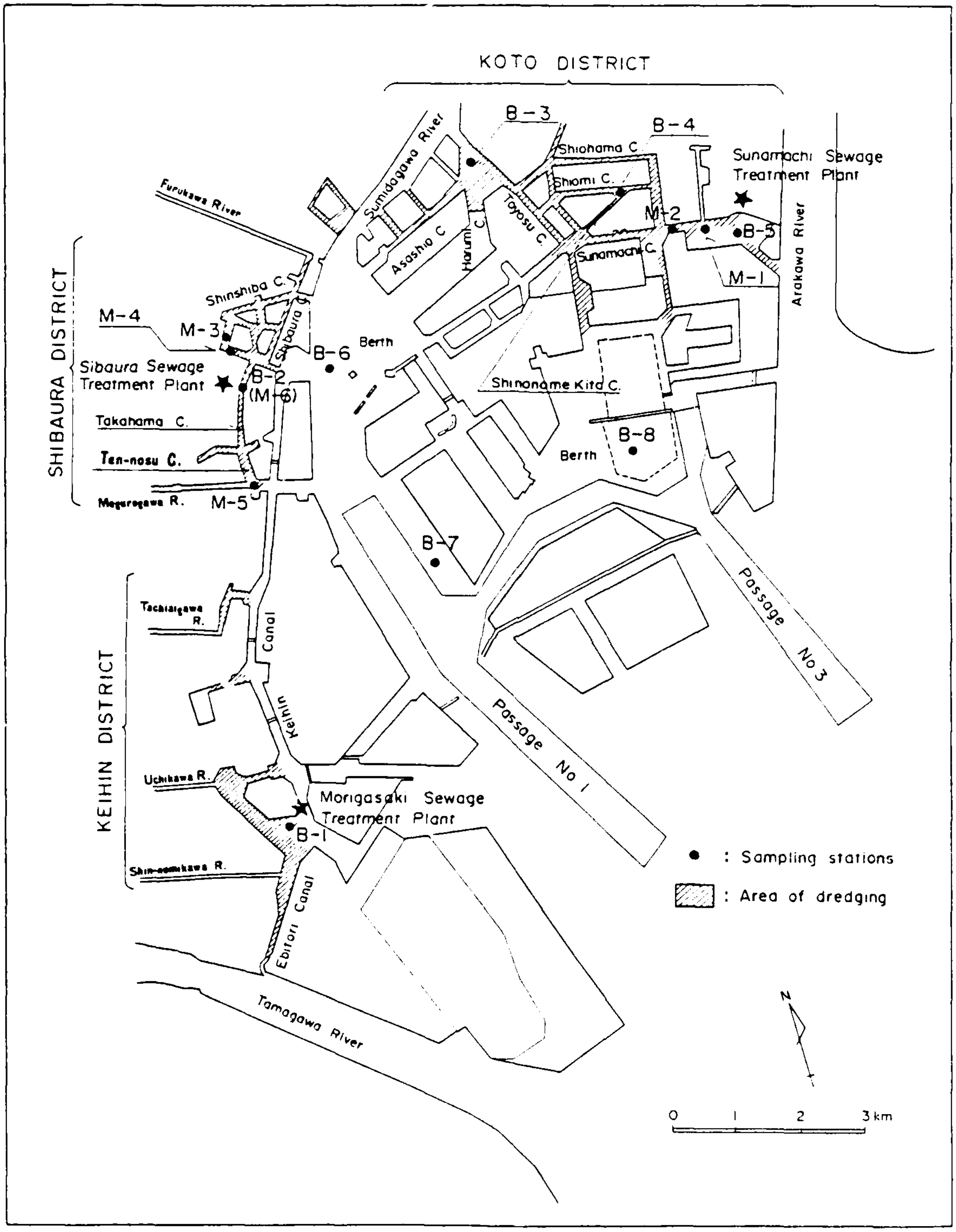


Figure 2. Station locations

TABLE 1. VERTICAL CHANGES OF SEDIMENT CHARACTERISTICS

Items	Sediment Classification	Station				
		B-1	B-2	B-3	B-4	B-5
Odor	A*	None	None	None	Hydrogen sulfide	Hydrogen sulfide
	B**	Hydrogen sulfide	Hydrogen sulfide	None	None	None
	C†	None	Hydrogen sulfide	None	None	None
Water content, percent	A	184	720	215	283	232
	B	251	625	178	108	199
	C	140	502	154	80.8	253
Wet density, g/cm^3	A	1.27	1.06	1.23	1.17	1.22
	B	1.20	1.07	1.27	1.43	1.24
	C	1.35	1.08	1.31	1.56	1.19
Ignition loss, percent	A	11.2	30.8	13.0	13.6	11.1
	B	14.1	31.7	13.0	7.2	12.1
	C	9.7	30.4	12.1	5.9	15.5
COD, mg/g dry	A	38.8	125	50.0	43.7	49.1
	B	54.5	114	51.6	16.6	55.7
	C	32.7	119	40.4	14.0	71.0
TOC, mg/g dry	A	26.3	104	30.2	21.8	21.8
	B	35.7	113	30.3	15.3	24.7
	C	18.6	101	26.5	12.2	37.7
Sulfide, mg/g dry	A	8.57	9.90	3.18	4.80	2.85
	B	2.17	8.30	6.41	0.57	3.93
	C	3.55	11.1	3.78	0.50	8.23

Sampling period: December 1979.

* Very high water content sediment, partially oxidized layer, found in top 3-cm layer.

** High water content and reduced sediment.

† Low water content sediment.

each indicating high values and revealing the progression of organic pollution in the sediments in the canal. Sulfates showed a high-value of about 3 mg/g (dry sediments) except for Koto; the black sediments may be due to sulfides.

The sediment analyses showed that the water content ratio of the sediments sampled in Shibaura (St. B-2) was high (500 to 700 percent), the index of organic matter was high, and the sediment was polluted organically. The canal of Shibaura is located inland and is narrow; the organic matter produced here or discharged from the sewerage treatment plants will probably settle in the canal.

Characteristics of Sediments and Sedimentation Rate

Qualitative and quantitative measurements have been made both daily and annually of the canal bed sediments.

Short-Term Measurement Using Polyethylene Bottles

One-liter polyethylene bottles as shown in Figure 3 were installed 1 m above the canal bed to sample material sedimented for 1 day. Table 2 shows the results of this sampling. The amounts of material were high in Keihin (St. B-1) and Sunamachi (St. B-5), and low in Koto (St. B-4) and Sunamachi (St. M-2). A comparison of these values was made with values observed in Funka Bay from summer to fall (Handa and Tagami 1981), in Tokyo Bay during a red tide (Sato 1977), and in Kizaki Lake (Koyama 1980). This comparison showed that the same amount, or more material settled during the red tide in Tokyo Bay as in the canals of Tokyo Bay, depending on the measuring point.

Table 3 shows the sedimentation rate calculated from the amount of material sampled using the polyethylene bottles.

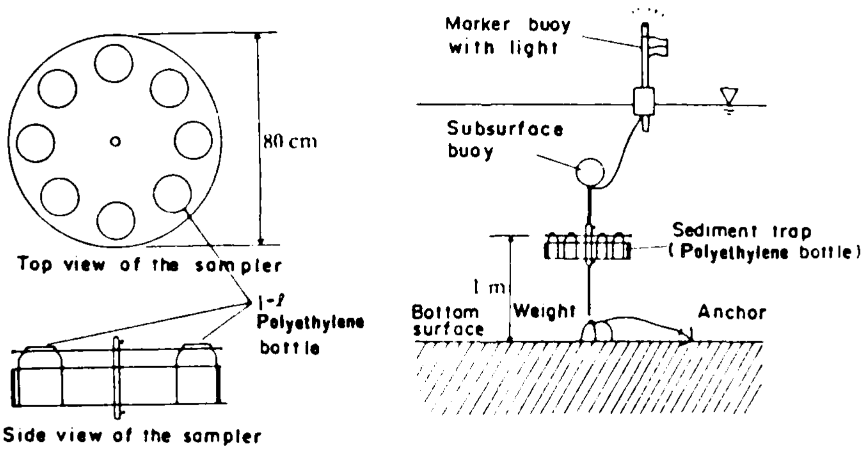


Figure 3. Sediment trap and mooring method

TABLE 2. AMOUNT OF MATERIAL ($g/m^2/day$) COLLECTED BY THE POLYETHYLENE BOTTLE

Period of Sampling	Station						
	B-1	B-2	B-3	B-4	B-5	M-1	M-2
December 1979	241	124	(454)*	33.8	286	136	60.4
January 1980	326	160	138	34.6	252	117	63.4

* The value in parentheses seems to be influenced by ship propellers.

TABLE 3. BOTTOM SEDIMENTATION RATE (cm/year) ESTIMATED FROM THE AMOUNT OF MATERIAL COLLECTED

Period of Sampling	Station						
	B-1	B-2	B-3	B-4	B-5	M-1	M-2
December 1979	19.7	35.0	42.5	4.0	28.4	13.5	6.0
January 1980	26.6	45.2	12.9	4.4	25.0	11.6	6.3

* The rate of sedimentation is calculated by the following equation:

$$h = s \times \frac{1}{\rho} \times \frac{100 + w}{100} \times 365$$

where

h = Bottom sedimentation rate, cm/year

s = Collected amount, $g/cm^2/day$

ρ = Wet density, g/cm^3

w = Water content, percent

The value of ρ and w were shown in Table 1 (very high water content sediment).

The calculated rates of sedimentation are considerably higher than both the results of the previous survey and the direct measurement in this survey.

Long-Term Measurement Using Canvas Buckets

As shown in Figure 4, 12 buckets were placed immediately above the canal beds to collect material. These buckets were retrieved one by one at 3-month intervals from 1979 to 1981.

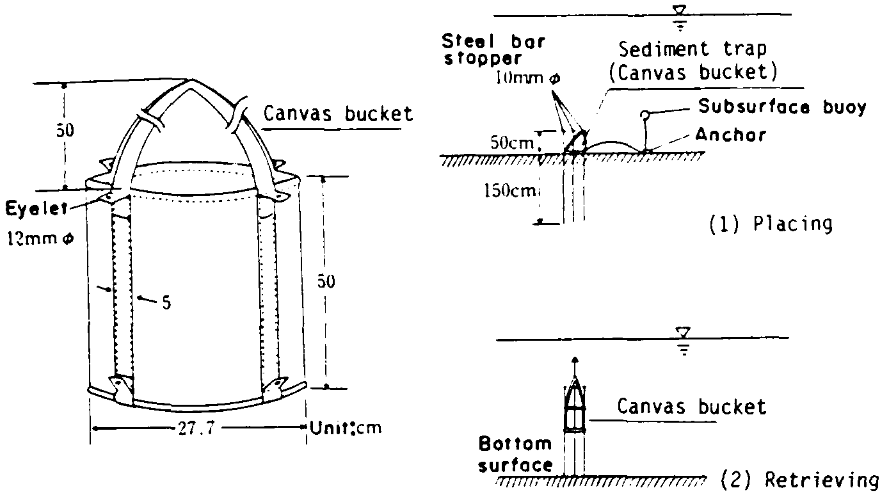


Figure 4. Sediment trap and placement method

Table 4 shows the sedimentation rates estimated from the water content ratios, wet densities, and amounts of material measured at stations in the Koto District.

TABLE 4. CHARACTERISTICS OF SEDIMENTS COLLECTED BY THE LONG-TERM TRAPPING EXPERIMENT AND SEDIMENTATION RATES DERIVED

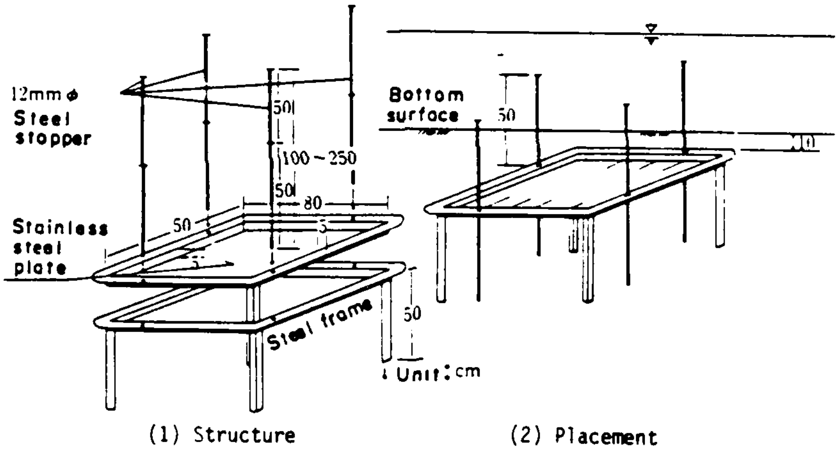
	Station (Koto District)		
Item	B-5	M-1	M-2
Water content, percent	296	159	237
Wet density, g/cm^3	1.17	1.25	1.19
Collected amount, $g/m^2/day$	15.5	18.7	15.5
Sedimentation, rate, cm/year	1.9	1.4	1.6

Trapping period: December 1979-December 1981.

The values of ignition loss, COD, and TOC as indices of organic matter tend to decrease as the sampling period becomes longer. Characteristics of sedimentations collected in an earlier period were similar to those of suspensions in the water. On the other hand, settlings collected later were similar to bottom sediments.

Direct Measurement of Sedimentation Rate

In general, the rate of sedimentation is measured using radioactive nuclides including ^{14}C and ^{210}Pb as the tracers (Matsumoto 1975). Lately, organic chlorides such as PCB stable in the atmosphere have also been used as tracers (Aizawa 1975). However, methods using tracers can be applied only to seabeds with very slow bottom flow and are not adapted to the water areas, like the canals of interest, with violent agitations caused by dredging and floods. In our case, stainless steel reference plates were placed about 10 cm below the canal bed, as shown in Figure 5, and the rate of sedimentation was determined on the basis of the plates from aging of the thickness of sedimentations.



(1) Structure

(2) Placement

Figure 5. The measuring method of sediment thickness using a stainless steel plate

Since the canal waters were highly turbid and dark, it was difficult to directly measure the thickness of sedimentation on the plate. Divers sampled sedimentation at three positions on the reference plates using three acrylic transparent tubes (having an inside diameter of 3 to 5 cm), and their thickness was measured on a boat. Figure 6 shows the changes with time of the thickness of the sedimentation, wherein the maximum value of the thickness is regarded as the actual thickness of sedimentation. The sedimentation rates determined from the 3-year-long experiment were 1.6 to 11.4 cm a year. The rate of sedimentation, which is affected by the local characteristics of each canal, is considerably different depending on the measuring point.

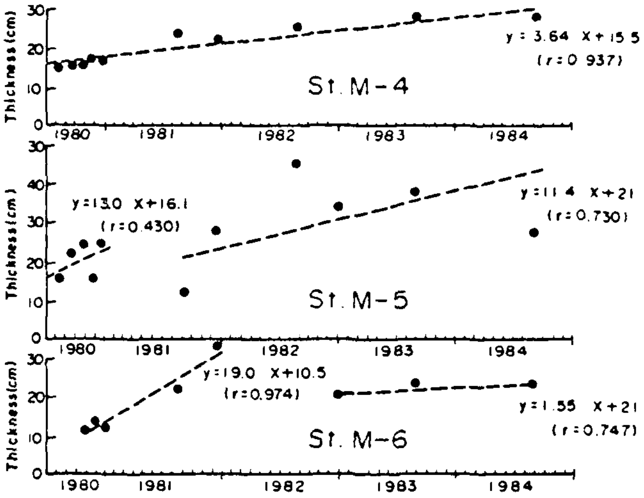


Figure 6. Changes with time of sediment thickness on the plate of stainless steel (Shibaura District)

Comparison of Methods to Determine Sedimentation Rates

Table 5 shows the values determined from the three measuring methods: stainless steel plate, canvas bucket, and polyethylene bottle.

TABLE 5. COMPARISON OF BOTTOM SEDIMENTATION RATES (cm/year) OBTAINED FROM THE THREE METHODS*

	Station									
Method	B-1	M-4	M-5	M-6 (B-2)	B-3	B-4	B-5	M-1	M-2	Remarks
Stainless steel plate	--	3.6	11.4	1.6	--	--	3.8*	--	--	Observed
Canvas bucket	--	--	--	--	--	--	1.9	1.4	1.6	Calculated
Polyethylene bottle	23.2	--	--	40.1	27.7	4.2	26.7	12.6	6.2	Calculated

* Observation period: August 1980-August 1981.

From Table 5 it can be seen that:

a. The rate of sedimentation measured at the same measuring point is significantly different depending on the measuring method. The rate of sedimentation measured using the stainless steel plate is 1.6 to 11.4 cm a year.

b. The rate of sedimentation calculated from the value measured using the polyethylene bottles is 4.2 to 40.1 cm a year, which is about three to four times higher than the results of the direct measurement. The former seems to ignore decomposition of organic matters and resulting consolidation of the sediments.

c. The rate of sedimentation calculated from the value measured using the canvas buckets is 1.4 to 1.9 cm a year. This value, as it has been determined from data obtained over an observation period of 2 years, is near the value corresponding to the actual rate of sedimentation.

Sedimentation Rate Determined from Previous Water Depth Measurement

For comparison, the sedimentation rate was determined by water depth measurement performed in 1979 and 1982. According to this depth measurement, the rate of sedimentation around Showa Jima, where a Morigasaki sewerage treatment plant is located, was 13 to 17 cm a year.

In the water area round the Shibaura District, which is almost enclosed and which receives disposal water from the Shibaura sewerage treatment plant, the rate of sedimentation was 4 to 10 cm a year. Before 1978, a high rate of sedimentation of 32 cm a year had been estimated at a Shinshiba canal.

At Toyosu, Harumi, and Asashio canals located at the estuary of the River Sumidagawa, the rate of sedimentation was 16 to 23 cm a year. In Koto and Sunamachi Districts, the rate of sedimentation at Shiohama, Shiomi, and Shinonome-Kita canals was 0.7 to 14 cm a year.

This comparison of sedimentation rate indicates few differences between our value and the value based on water depth measurement.

Sources of Sediments and Their Supply

The thickness of sediments on the seabed is increased by suspended solids (SS) (hereinafter referred to as settlings) existing into the water and produced within the water. There are three major sources of sediments: discharged from sewerage plants, river runoff, and phytoplankton production in the water. These three factors are discussed below along with the sources of material by water area.

Sewerage Treatment Plant

In the canals of the Tokyo port, there are three sewerage treatment plants: Morigasaki, Shibaura, and Sunamachi. The amount of material discharged from the sewerage treatment plants has decreased from 4,700 tons a year in 1980. However, BOD loads to the sewerage treatment plants and the amount of treatment water have not changed.

The sewerage plants treat both the public sewerage and the drainage basin sewerage. This means that when a large amount of rainwater flows into the plants, the untreated sewerage is discharged directly to the canal.

The amount of material produced due to overflow from the sewerage treatment plant into the canal must be identified. According to preliminary studies of the untreated loads and studies done in the United States, the SS loads are about 15 times as high as the treatment water (Hoshikuma 1982). Hence, the amount of settlings originating from the sewerage plants and discharged into the canal has been estimated to be about 80,000 tons a year.

River Runoff

Six rivers, Sumidagawa, Furukawa, Megurogawa, Tachiaigawa, Uchikawa, and Shin'nomikawa, flow into the canal sections of the Tokyo port. In contrast to a large river, the small river with a short length and small basin is characterized by sudden increases in discharge and wide variations in discharge. The discharge during rainfalls is much greater than the discharge during sunny weather; in particular, the discharge during a heavy rainfall is greater by several orders of magnitude. This also applies to the rivers which flow into the canals in the Tokyo port; while the amount of material which flows from rivers into the canals during a heavy rainfall seems to occupy a great proportion of the overall material, no data have been obtained which indicate a correlation between discharge during a heavy rainfall and change in water quality.

A report about characteristics of SS-loaded discharge during a heavy rainfall measured at Yabata River (which is located at a crowded place in Toshima Ward) reveals that the SS load is proportional to the square of the discharge (Yamaguchi 1977). The amount of material carried by the rivers into the canals has been estimated to be about 250,000 tons per year on the basis of the above report, 85 percent of the amount resulting from heavy rainfalls.

The amount of sediments on the river beds flowing into the canal was estimated to be about 600,000 cu m by our survey; some part of these sediments flow into the canal during heavy rainfalls.

Phytoplankton Production

Eutrophication of the water leads to a large phytoplankton production and seems to result in shallowing the canals. Red tides (abnormally high production of phytoplankton) occur every year in Tokyo Bay, which includes the Tokyo port. To what extent the plankton productions contribute to the shallowing needs to be investigated. To estimate potential settlings due to plankton production, observations were made in 1981 at the eight stations shown in Figure 2.

Measurement of daily phytoplankton production using an in situ bottle method has shown relatively high values in summer in some canals and passages and very low values in winter at every measuring point relative to summer.

From the results of surveying the amount of phytoplankton production, the amount of production per unit area was calculated, which was then multiplied by the water area of the Tokyo port to determine the amount of produced organic carbons over the entire area. The amount of phytoplankton production has been converted from the amount of produced organic carbons by regarding their chemical composition as following the Redfield's model.

Thus, the amount of produced organic carbons in the Tokyo port originating in phytoplankton has been estimated to be about 50,000 tons a year, and the SS amount becoming settlings on the bed at about 190,000 tons.

Then, for the canal, unsteady phytoplankton production during a red tide, etc., must be examined. The amount of net production during red tides in Tokyo Bay has been estimated to be 12 to 57 g $O_2/m^2/day$. Thus, by applying the above-mentioned calculation, the amount of material settling due to phytoplankton in 1979 (during which a red tide occurred frequently) has been estimated at about 270,000 tons, which is about twice as many as that in 1977 (during which fewer red tides occurred).

Sources of Settlings by Water Area

Figure 7 shows the relationships between TOC and chlorophyll contained in the sediments.

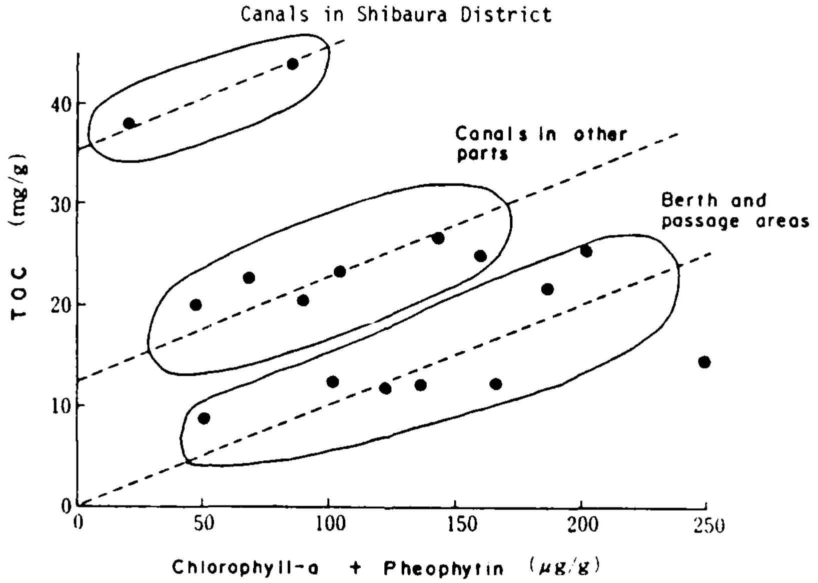


Figure 7. Relationship between TOC and photosynthetic pigments in the bottom sediments and their changes with region

The relationship, by water area, between chlorophyll and TOC in the sediments has shown that TOC increases at a certain rate with the increase of chlorophyll, although the ordinate intersections are different by water area. This indicates that the loads of allochthonous organic matters are different depending on the measuring point and organic carbons originating from phytoplanktons loaded on the sediments. For the canal, the rate of contribution of the loads is different by water area. The paragraphs below discuss the sources of settled material by water area estimated from positions and water quality.

Around Showa Jima in the Keihin District, where the Morigasaki sewerage treatment plant is located, are the rivers Ebitorigawa, Shin'nomikawa, and Uchikawa, and Ebitori canal. While an attempt to investigate the sources of sediments in the water area was made, the contributions of the rivers, the sewerage treatment plant, and phytoplankton production could not be identified, though these three factors must contribute to sedimentations in a complex manner.

The water area in the Shibaura District, which is surrounded by Takahama, Shibaura, and Shin-shiba canals, is highly closed. There is a sewerage treatment plant in the district which affects sedimentations significantly, phytoplankton production contributing very little to the sediments. The water area around Ten'nosu canal is affected considerably by Megurogawa River.

The water area around Tsukishima and Harumi, which consists of Harumi and Asashio canals, is affected significantly by the River Sumidagawa, but phytoplankton production cannot be ignored.

Shiomi, Shiohama, and Shinonome-Kita canals in the Koto District are affected significantly by phytoplankton production. Near the Sunamachi canal, belonging to Sunamachi District, is Sunamachi sewerage treatment plant; phytoplankton production cannot be ignored, but estimating the material settled is difficult since the water flow is high.

As described above, the source of the sediments differs depending on the causes and places, and thus cannot be discussed together.

Estimation of the Amount of Settlings Applied to Tokyo Port

The SSs that come into the Tokyo port (inside a central breakwater) including canals, berths, and passages originate from sewerage treatment plants, incoming rivers, and phytoplankton production, and their amount varies significantly between steady and unsteady (heavy rainfall, red tide, etc.) conditions. A rough estimation of the thickness of sediments from the amount of settlings supplied to the whole Tokyo port leads to the following results:

a. 520,000 tons of SS are sedimented a year in the whole Tokyo port.

b. 48 percent of these originate from rivers; they are carried into the port particularly during heavy rainfalls.

c. 37 percent of the SS originate from phytoplankton production; this ratio tends to increase in years during which red tides occur frequently.

d. 15 percent of the SS originate from sewerage treatment plants, which affect the near canals significantly.

The yearly SS have been converted to the corresponding volume, which has then been divided by the water area of the whole Tokyo port to estimate the mean rate of sedimentation. The result was about 6 cm a year. This value closely approximates the values measured by the stainless steel plate method (1.6 to 11.4 cm, mean value of 6.5 cm a year), and thus seems to be appropriate. The mean value (about 6 cm a year) is based on an assumption that every loaded SS has been sedimented on the bed of the whole Tokyo port. As the loaded sources contribute differently to settlings depending on the water area, applying the mean value directly to the object of design as the rate of sedimentation for the water area is accompanied by a number of problems yet to be solved.

CONCLUSIONS

The degradation of the environment in closed waters such as canals and inland bays is caused by loads from lands and by red tides occurring in the waters. It has been pointed out that sediments on the beds of the rivers and the sea significantly affect this degradation. The effects of sediments on the water area include offensive odors, rolled up dirt, consumption of dissolved oxygen, and release of COD components and nutrients.

Based on this background, our bureau is engaged in sediment removal to restore the aqueous environment of the canals. Since sediments have been observed to resettle after removal, our bureau performed this survey to understand the characteristics of sedimentation in the canal and to encourage sediment removal.

This survey, performed for the first time in Japan, has led to an understanding of the characteristics of the bottom sediments and of the rate of sedimentation, and to the identification of the sources of the sediments and the amounts. However, this survey revealed that the sources of sediments are different for different canals; conditions such as heavy rainfalls and red tides produce significant effects; and it is difficult to quantitatively identify settled materials and sediments. Investigations will continue in order to solve these problems.

While this survey has aimed at determining the amount of sedimentation, the effects of sediments on water quality must be identified quantitatively before the water environment of the canal of the Tokyo port can be restored. A survey for this purpose began in 1983 and will end in 1986.

Through understanding sediments both qualitatively and quantitatively, our bureau aims at revising the current tentative removal criteria and including the revised criteria in the next pollution prevention plan (prepared in 1987) to establish a commercial entity for such plan implementation.

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TECHNIQUES FOR MONITORING STABILITY OF OFFSHORE SUBMERGED DISPOSAL MOUNDS

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The physical fate of dredged material released in the marine environment often is of vital concern. If currents spread the material beyond the disposal site, problems could arise, such as shoaling of adjacent channels or smothering of biological resources. In other instances spreading may be desirable, for example, to increase site capacity, nourish adjacent beaches, or simply return the material to the natural system from which it was captured.

In addition to site-specific questions regarding sediment migration, it is crucial to obtain good field data for verification and calibration of numerical models so that rates and directions of spreading can be predicted for various conditions. Such predictive ability would lead to design criteria to minimize problems and maximize beneficial uses of dredged material.

Several standard geophysical techniques are presently available to provide needed basic field data. A few new tools offer promising potential to augment today's techniques.

Hydrographic surveys can document changes in the shape of any substantial sediment accumulation. Subbottom seismic reflections can reveal subsidence of the foundation beneath the accumulated materials. Side-scan sonar surveys record the surface characteristics across wide swaths of the bottom, and can be used to identify and map edges of disposal deposits which are too thin to be recognized on sounding or seismic profiles. Sediment samples allow direct examination that may include sedimentary, chemical, or microscopic analysis to confirm subtle distinctions between disposal and native bottom materials. Sediment cores provide stratified sequences that record layer-by-layer accumulation and compaction within the deposit. Cores can also indicate where currents or organisms are reworking or mixing the different sediment types.

In addition to these surface-operated techniques, the observations of technically trained SCUBA divers greatly increase our understanding of bottom characteristics and our confidence in the interpretation of more expansive, remote techniques. Repeated measurements by divers of the bottom elevation will indicate accretion and erosion independent of, and on a different scale from, that captured on sounding records.

The foregoing techniques use readily available equipment. New technology offering potential for monitoring disposal deposits includes (1) swath sounding systems that, from a single pass, reveal a three-dimensional picture of the bottom; (2) seabed drifters that move with the bottom currents and indicate where moving sediments may accumulate; and (3) sediment traps that can be used in conjunction with current meters to quantify sediment movement. If the disposal depth or other conditions preclude direct observation by divers, remotely controlled television or photography can still provide a view of the bottom. A newly developed camera system photographs even the sediment-water interface in cross section using a lens prism that penetrates the upper few centimetres of sediment. All these techniques depend, of course, on good positioning.

Incorporating the discussed elements in an efficient monitoring plan depends on familiarity with the capability and requirements of each element. Because of the usual demands and expense of working at sea and the challenge of distinguishing native from disposed material, as many of these overlapping techniques as possible should be incorporated into any survey plans. Mobilization and boat time are major cost items. Thus, multiple element surveys often provide not only the best insurance of success, but also the most cost-effective approach for tracking disposed material.

USE OF ANTI-TURBIDITY CURTAINS AT A SAND COMPACTION PILING AREA IN YOKOHAMA HARBOUR

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Japan Bottom Sediments Management Association

ABSTRACT

Sea work, such as dredging and sand compaction piling, causes turbidity in the surrounding sea area. To prevent diffusion of turbidity, anti-turbidity curtains are generally used in the working area. In such cases it is important to know how the curtains affect suspended solids in seawater. In 1984 anti-turbidity curtains were used at a sand compaction work area in Yokohama Harbor, and the behavior of suspended solids around the curtains was observed. This paper reports the data measured at that time.

CURTAIN APPARATUS

There are two kinds of anti-turbidity curtains, continuous and jointed. The continuous type requires a large space for assembly on land and much work to place it at sea (Figures 1 and 2).

Compared with the continuous type, the jointed type requires a minimum of space for assembling and erecting (Figure 3).

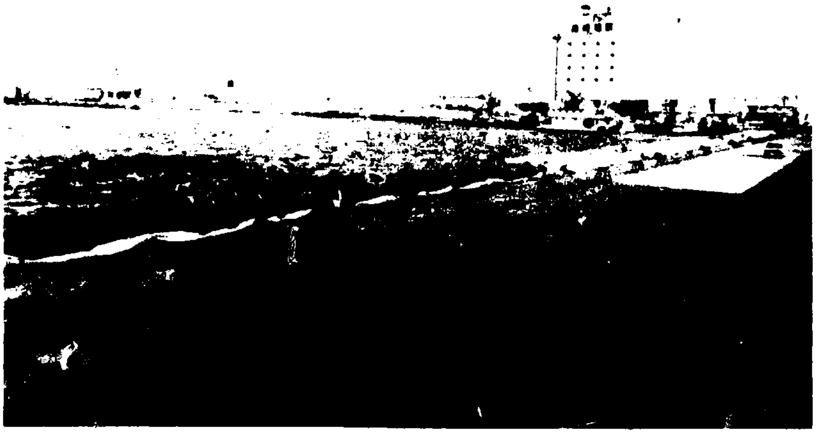


Figure 1. Assembling onshore

Figure 2. Transporting from shore

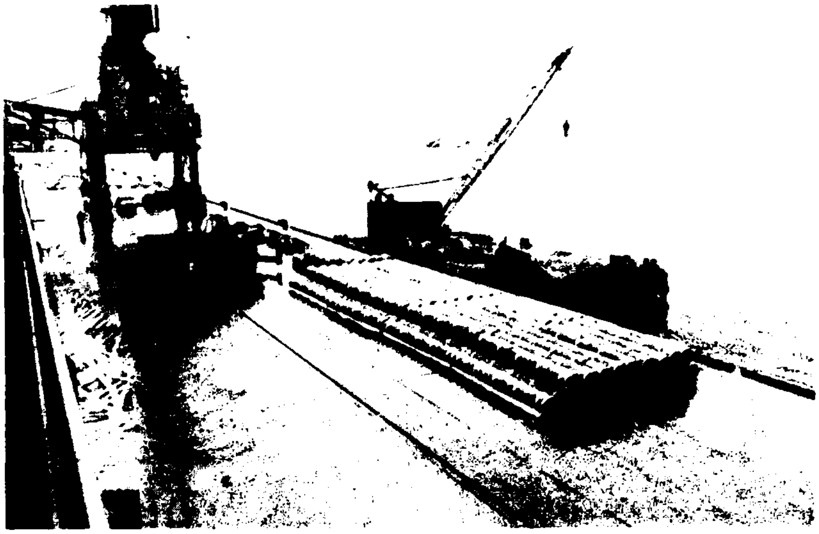


Figure 3. Assembling of floats

The jointed type can also be placed at sea by connecting the elements. The jointed anti-turbidity curtains in our country are called by the commercial name "Patoresia." The apparatus used in Yokohama Harbour is indicated in Figures 4 and 5.

BACKGROUND OF WORK SITE

The City of Yokohama's reclamation project, "Harbour Future 21," began in February 1984 (Figures 6 and 7).

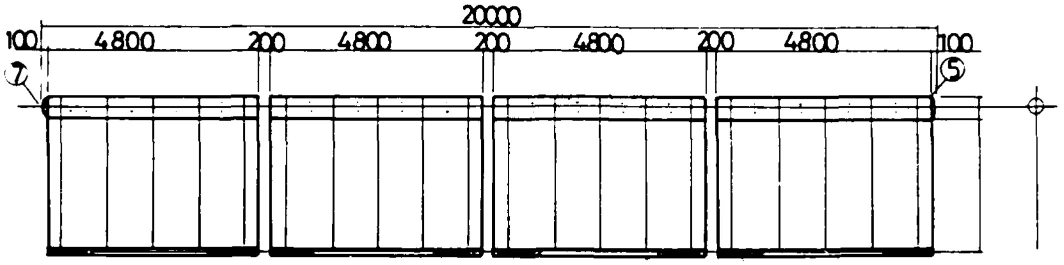

Patoresia P-2

Figure 4. Appearance of curtains

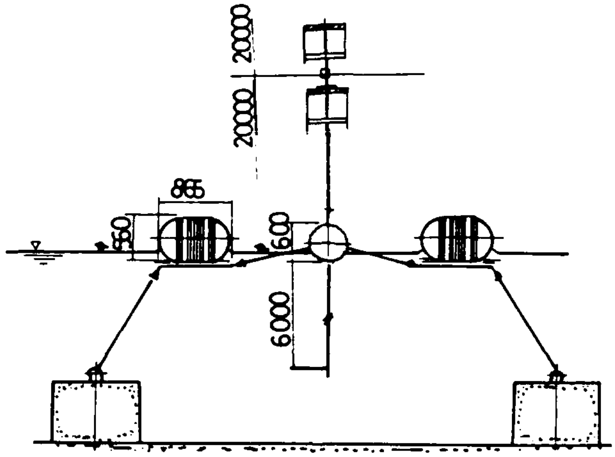


Figure 5. Aggregate of curtain apparatus

The land to be reclaimed has an area of 76 ha and after completion will be utilized for port facilities, international cultural exchange facilities, park and green spaces, etc. Figure 8 shows an anticipated view of the new port area when completed. The sand compaction piling is used to stabilize the bottom sediment at the reclamation area.

MEASURING SITES

The suspended solids (SS) concentrations near the anti-turbidity curtains were measured at two different locations on February 3 and May 21, 1984. The measuring sites are shown in Figures 9 and 10.

Figure 11 shows the work area surrounded by curtains.

Flow velocities and directions as well as SS concentrations were measured. From these it was found that a reflection wave occurred (Figures 12 and 13).

It is natural that there is no pulsating wave at the lower layer underneath the curtains and outside them. The tide velocities were about 10 cm/sec. The Froude number of the pulsating wave was $F = 0.0056$.

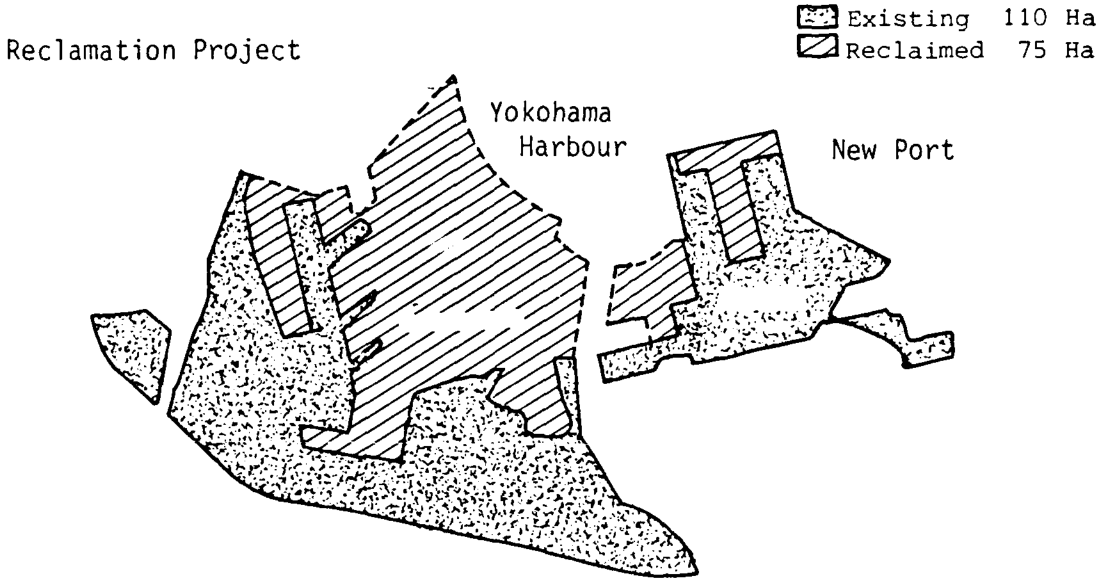


Figure 6. Reclamation project "Harbour Future 21"

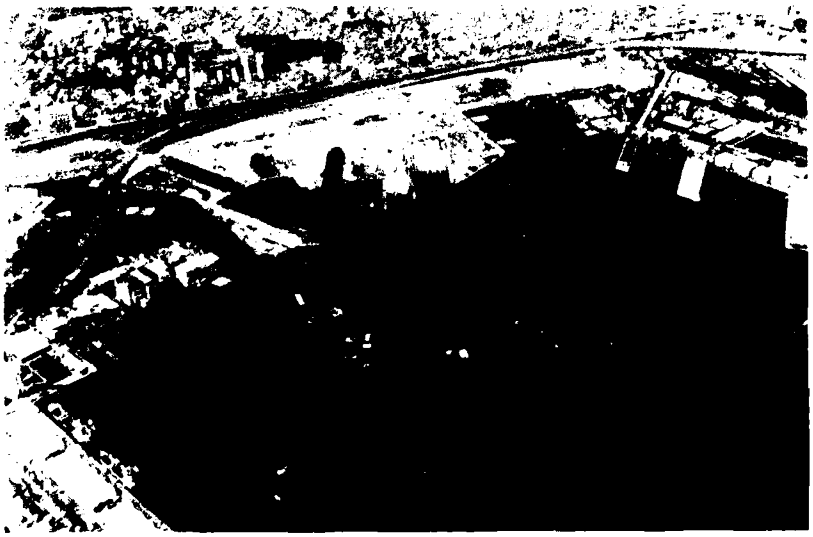


Figure 7. Work site

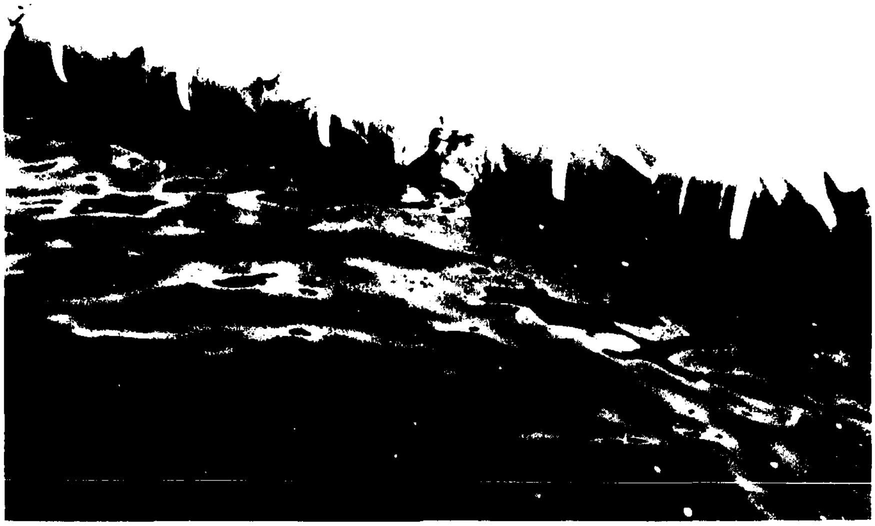


Figure 8. Anticipated view of completion

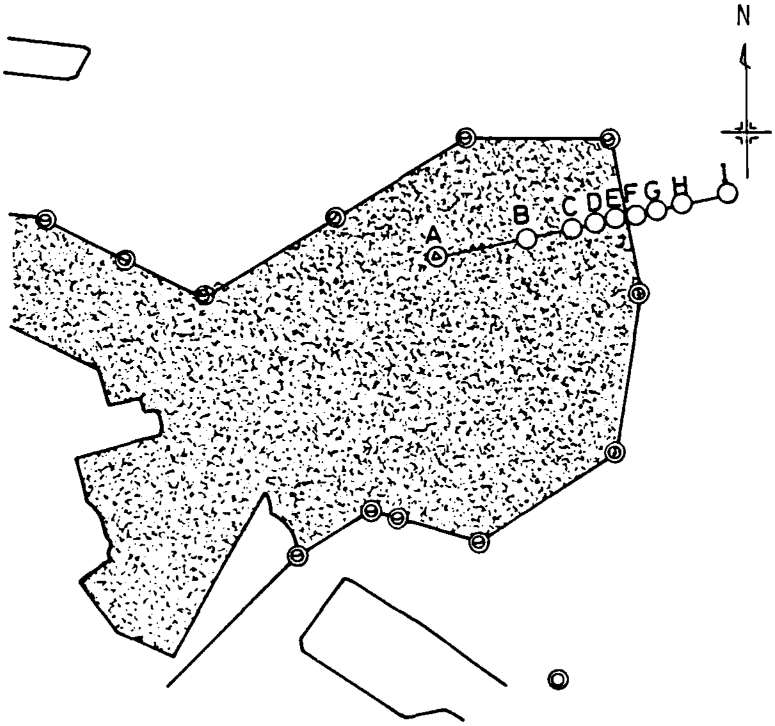


Figure 9. Measuring site of February 3

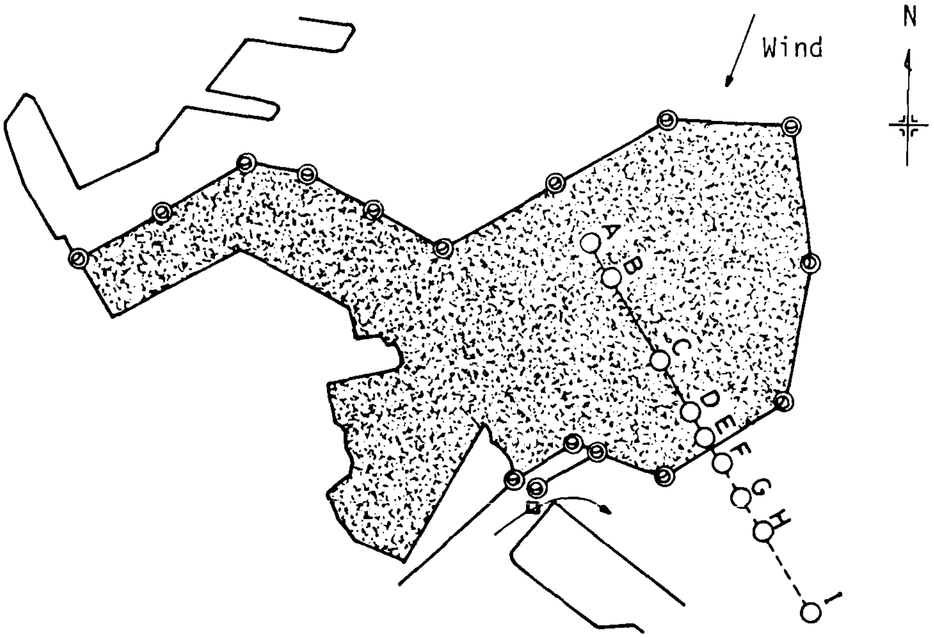


Figure 10. Measuring site of May 21

Figure 11. Work area surrounded by curtains

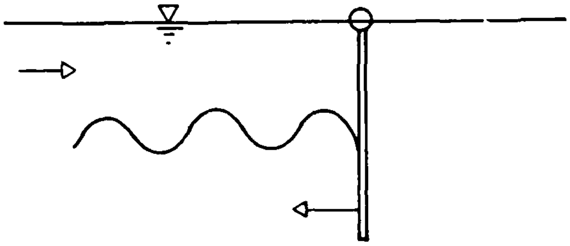


Figure 12. Reflection curve

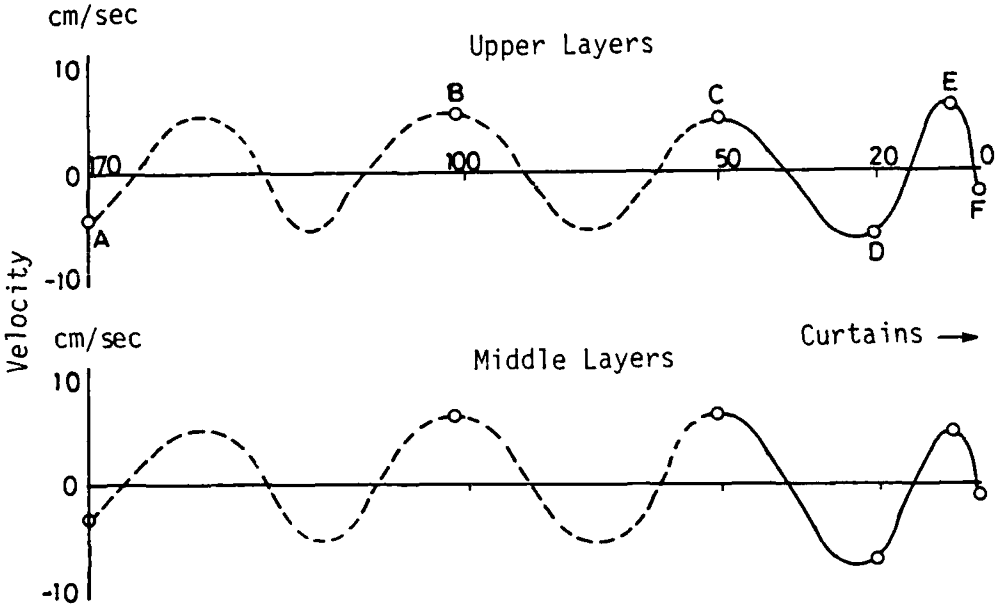


Figure 13. Pulsating wave reflected by curtains

RESUSPENSION COLUMN

At the sand compaction point some amount of sediment particles are resuspended upwards by the turbulence (Figure 14). In this case movement of

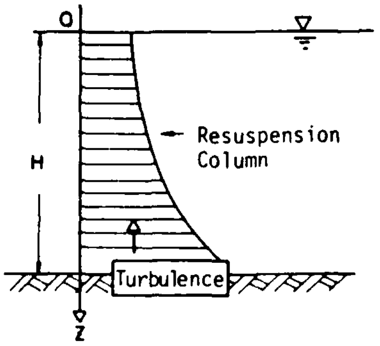


Figure 14. Turbulence resuspension

the resuspended particles is represented as follows (tide velocity neglected):

$$Kz \frac{\partial^2 c}{\partial z^2} - Ws \frac{\partial c}{\partial z} = 0 \qquad (1)$$

where

Kz = diffusion coefficient in the vertical direction

c = concentration of resuspended solids

Ws = mean settling velocity of resuspended solids

The solution of Equation 1 is obtained as follows:

$$C = A \frac{Kz}{Ws} e^{\frac{Ws}{Kz} Z} + B \qquad (2)$$

One of the boundary conditions is $Z = 0$; $C = Co$.

$$B = Co - A \frac{Kz}{Ws}$$

Then we have

$$C = Co + A \frac{Kz}{Ws} \left(e^{\frac{Ws}{Kz} Z} - 1 \right)$$

The other condition on the sea surface is

$$Z = 0; \quad -WsC + Kz \frac{\partial c}{\partial z} = 0$$

Then

$$A = \frac{Ws}{Kz} Co$$

Finally, we have

$$C = Co\ e^{\frac{Ws}{Kz} Z} \tag{3}$$

Equation 3 indicates the resuspension column has a vertical distribution with an exponential form.

Figure 15 shows two vertical distributions of the resuspension column obtained on February 3 and May 21, 1984. The SS concentrations at the turbulence points were 900 mg/ℓ and 1,100 mg/ℓ, and the ones at the sea surface (H = 0.5 m), 26 mg/ℓ and 140 mg/ℓ, respectively. Both vertical curves clearly have exponential form.

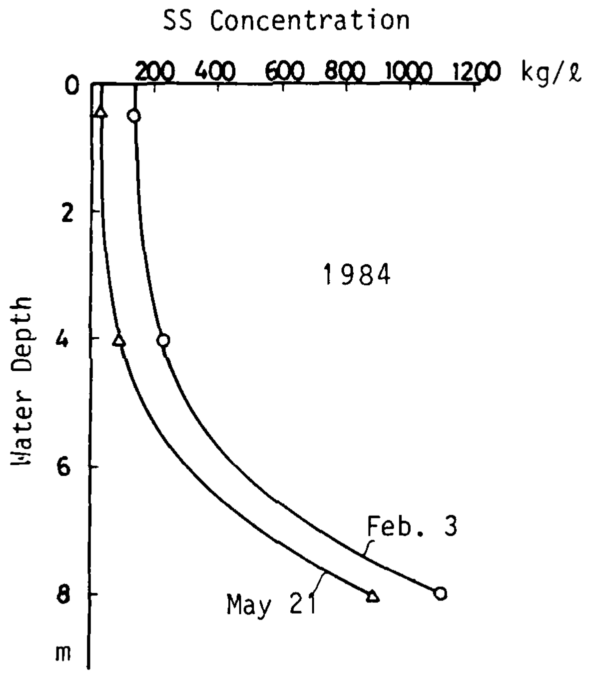


Figure 15. Resuspension column, May 21 and February 3

When we plot the data of May 21 on half-exponentially graduated paper, we have a curve consisting of two straight lines as Figure 16 shows. For the upper and lower halves we have

$$\frac{Ws_1}{Kz} = 0.1418\ 1/m$$

$$\frac{Ws_2}{Kz} = 0.3818\ 1/m$$

Then assuming $Kz = 2\ \text{cm}^2/\text{sec}$, we have

$$Ws_1 = 2 \times 0.1418 \times 10^{-2} = 0.002836\ \text{cm/sec}$$

$$Ws_2 = 2 \times 0.3918 \times 10^{-2} = 0.007824\ \text{cm/sec}$$

Using Stoke's equation

$$d^2 = \frac{18\mu Ws}{g(\rho s - \rho w)}$$

We can obtain the mean diameters of the resuspended particles:

$$d_1^2 = \frac{18 \times 0.01 \times 0.002836}{980\ (2.6 - 1.03)} = 3.256 \times 10^{-7}\ \text{cm}^2$$

$$d_1 = 0.57 \times 10^{-3}\ \text{cm} = 5.7\ \mu$$

$$d_2^2 = \frac{18 \times 0.01 \times 0.007824}{980\ (2.6 - 1.03)} = 8.982 \times 10^{-7}\ \text{cm}^2$$

$$d_2 = 9.48 \times 10^{-3}\ \text{cm} \doteqdot 9.5\ \mu$$

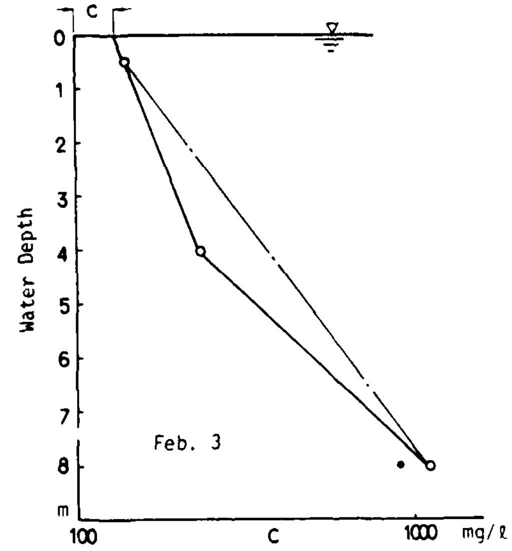


Figure 16. Resuspension column, February 3

From the data of May 21, we get 2 particle diameters $d_1 = 6.54\ \mu$ and $d_2 = 8.02\ \mu$ by the same way. From these data, it is found that the resuspension column consists of two particle groups having a mean diameter of $5.7 - 6.5\ \mu$ at the upper layers and $8 - 9.5\ \mu$ at the lower layers.

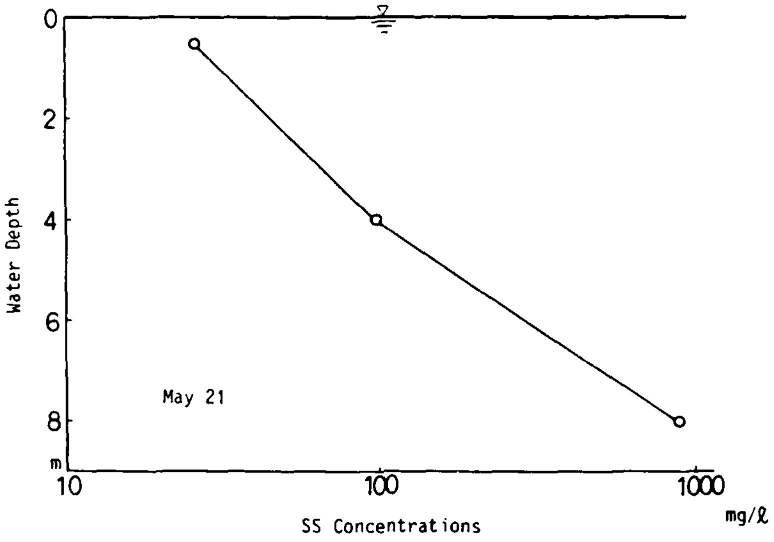
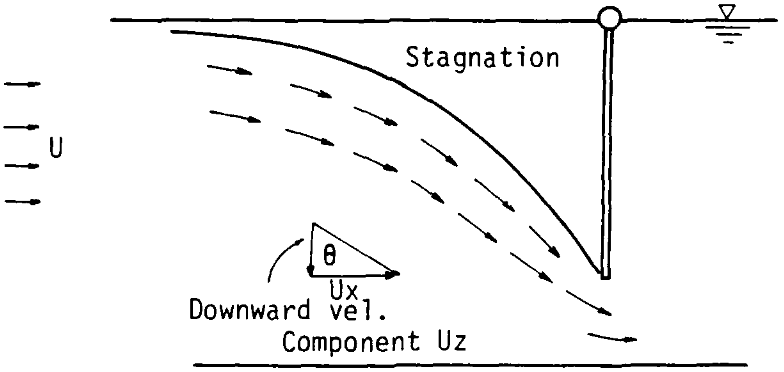


Figure 17. Resuspension column, May 21

SEDIMENTATION PROMOTED BY CURTAINS



Inside the curtains, where the horizontal tide flow collides with the curtains, a stagnation flow is generated. Suffering this potential flow, the seawater flows downwards and underneath the curtains. Meanwhile, the suspended solids contained in the seawater settle rapidly. The change in SS concentrations influenced by the curtains is represented as follows:

$$\frac{\partial c}{\partial t} + Ux \frac{\partial c}{\partial x} + (Uz + Ws) \frac{\partial c}{\partial z} = Kx \frac{\partial^2 c}{\partial x^2} + Kz \frac{\partial^2 c}{\partial z^2} \qquad (4)$$

where

c = concentration of suspended solids

Ux = horizontal flow velocity component of tide

Uz = vertical flow velocity component of tide

Ws = mean settling velocity of SS particles

Kx = diffusion coefficient in the horizontal direction

Kz = diffusion coefficient in the vertical direction

The turbidity soil particle with a 10-μ diameter has the following settling velocity:

$$Ws = \frac{\rho s - \rho w}{18\ \mu}\ gd^2$$

$$= \frac{2.65 - 1.03}{18 \times 0.01} \times 980 \times (0.001)^2$$

$$= 0.009\ \text{cm/sec}$$

With a tide of 10 cm/sec, the downward flow velocity component can be taken as follows, when the inclined flow inside the curtains has an average of 5°.

$$Uz = 10 \sin 5^\circ = 10 \times 0.087 = 0.87\ \text{cm/sec}$$

Then we have

$$Uz + Ws = 0.87 + 0.009 = 0.879\ \text{cm/sec}$$

Therefore we obtain the following magnification

$$E = \frac{\text{vertical vel. component with curtains}}{\text{vertical vel. component without curtains}}$$

$$= \frac{Uz + Ws}{Ws} = \frac{0.879}{0.009} \doteq 98$$

This means that the curtains increase the settling velocity of the turbidity soil particles by about 100 times. In other words, the function of the curtains can be regarded as increasing the particle size of the turbidity soil particles by $\sqrt{100} = 10$ times, compared with the soil particles without curtains. In such a way the curtains play an excellent role in promoting the sedimentation of suspended solids. Figure 18 shows the change of SS concentrations inside the curtains.

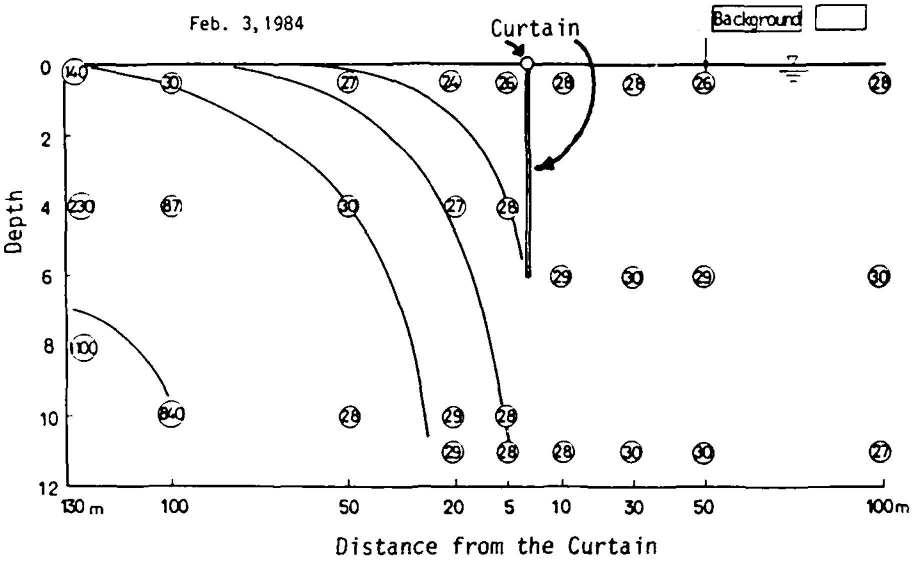

Feb. 3, 1984

Figure 18. Change in SS concentrations inside curtains, February 3, 1984

The SS concentrations of the upper layers decrease from 140 mg/ℓ (at the turbulence point) to 26 mg/ℓ (at the curtains). The concentrations of the bottom layers change from 1,100 mg/ℓ to 29 mg/ℓ.

Figure 19 shows the data on May 21, 1984. The SS concentrations change more gently than on February 3. This may be due to the differences in particle size and tide velocity.

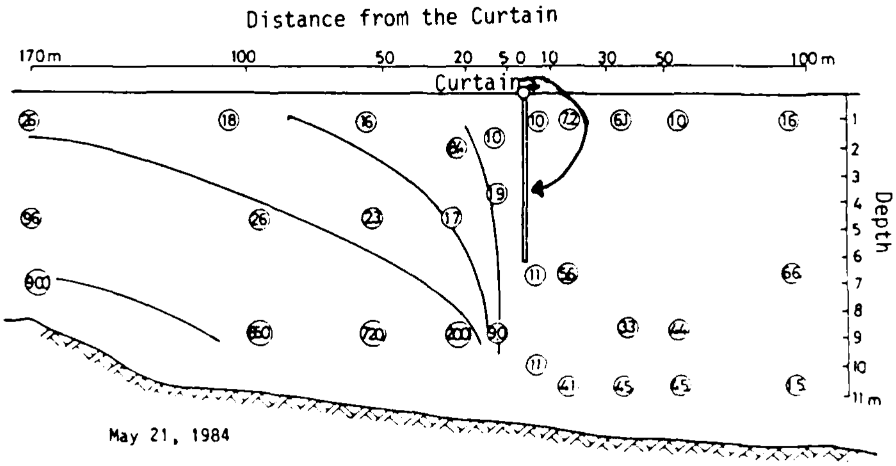
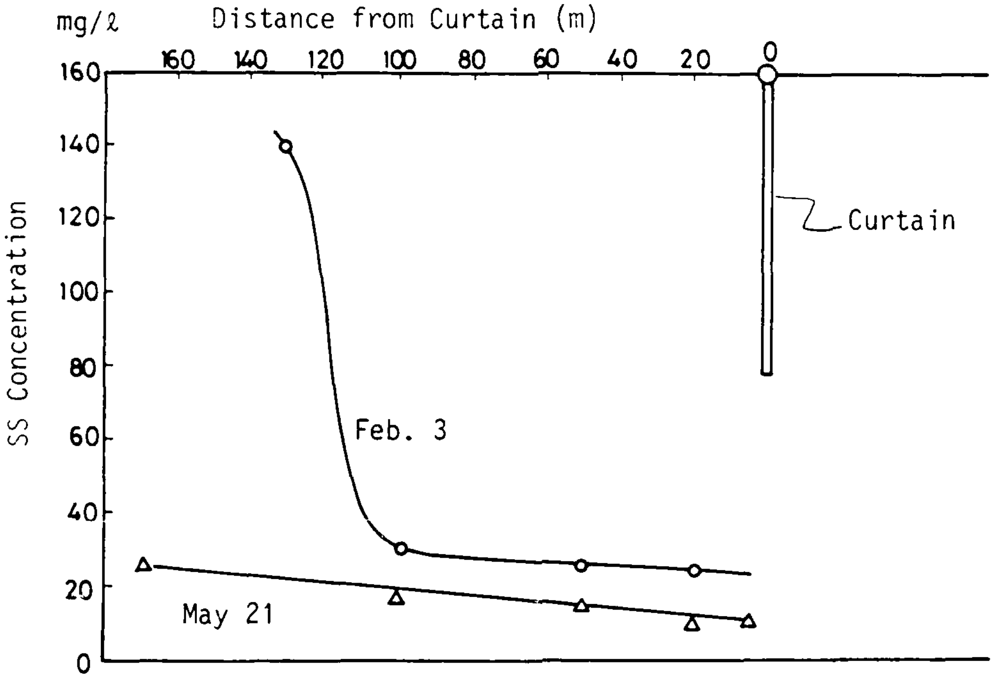
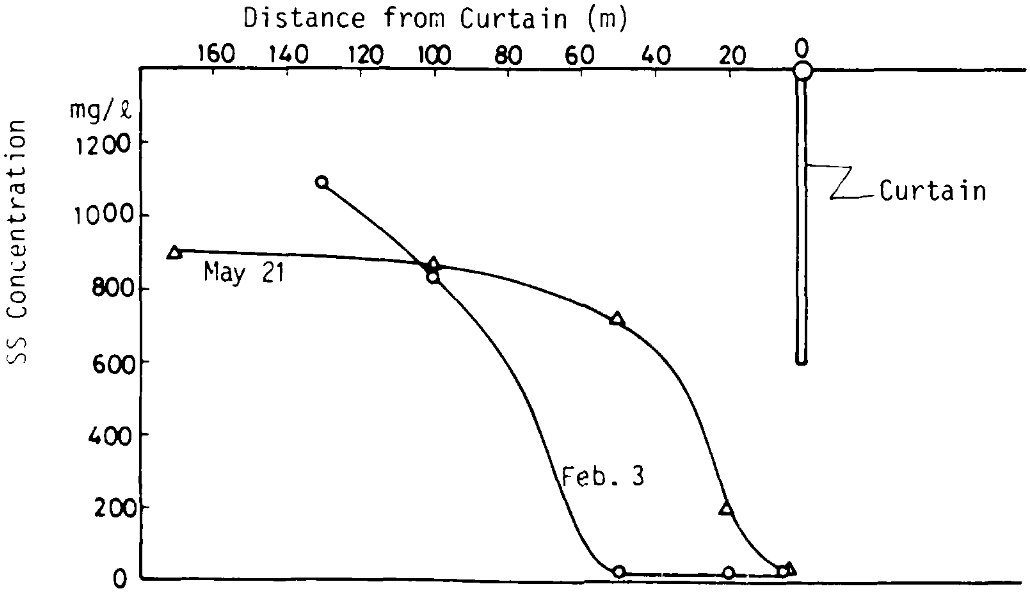

May 21, 1984

Figure 19. Change in SS concentrations inside curtains, May 21, 1984.

Figure 20 shows the horizontal change in SS concentrations at the upper and lower layers. These figures suggest that the influence of the curtains begins to act about 100 m from the curtains.



a. Upper layers



b. Lower layers

Figure 20. Horizontal change in SS concentrations

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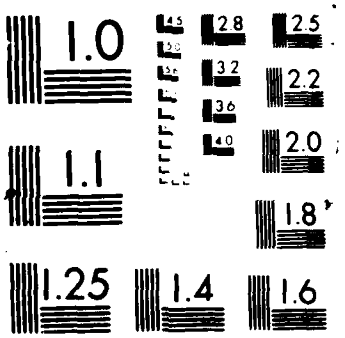
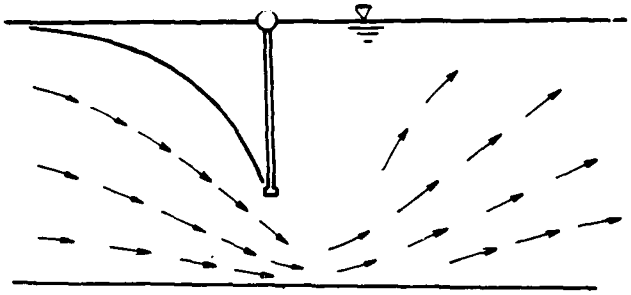
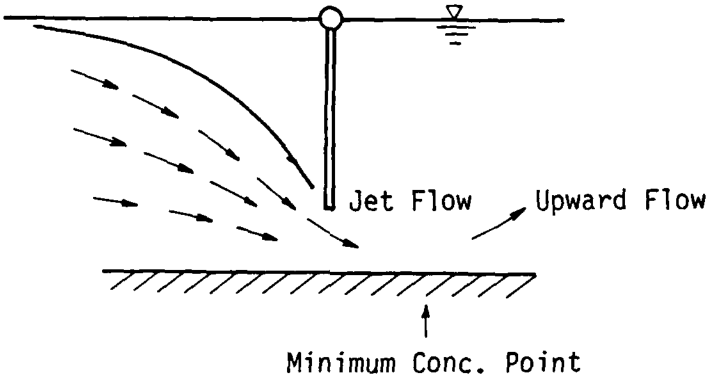

1.0
4.5
5.0
5.6
2.8
3.2
3.6
4.0
2.5
2.2
2.0
1.1
1.8
1.25
1.4
1.6

CHANGE IN SS CONCENTRATIONS OUTSIDE CURTAINS

The tide flows are changed by the curtains but return to normal after passing through the bottom openings.



The minimum concentration of about 4 mg/ℓ is obtained at about 20 m from the curtains.



At the passage directly underneath the curtains the flow velocity increases to two or three times faster then the initial flow and has a generally downward direction. Therefore at this point the separation of water and solids is also performed. After the point of minimum concentration, the flow turns upward. At that time, the solid particles are resuspended by the jet flow and the upward flow. Figure 21 shows this phenomenon clearly. The resuspension results in increased SS concentrations at a point 100 m from the curtains, indicating 16 mg/ℓ. However the resuspended solids quickly return to background values of SS concentrations in the open sea.

CONCLUSIONS

This paper has studied the behavior of SS concentrations around anti-turbidity curtains. We learned that sedimentation is substantially promoted by the curtains. Anti-turbidity curtains should be applied often for various purposes of water clarification along with sedimentation. For example, they can be applied for water treatment at settling ponds and spillways.

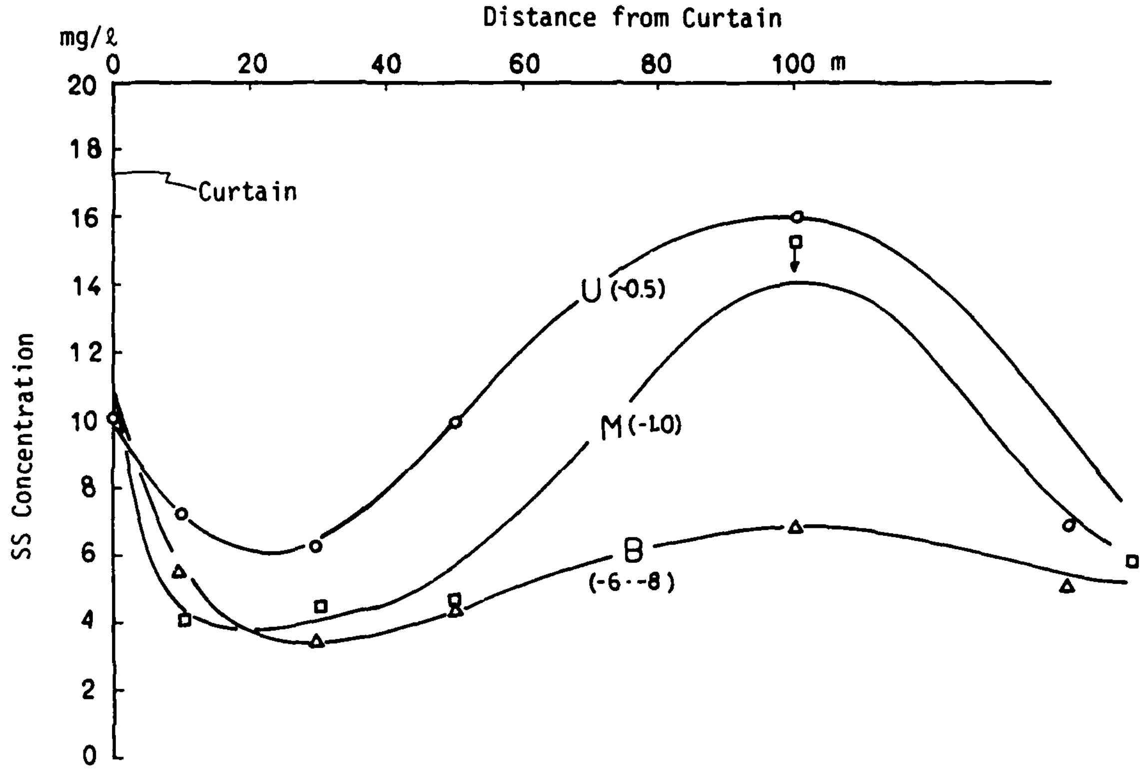


Figure 21. Change in SS concentrations outside the curtains

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