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US Army Corps
of Engineers
Water Resources
Support Center

Management of Bottom Sediments Containing Toxic Substances

Proceedings of the 12th U.S./Japan
Experts Meeting

11-14 November 1986
Yokohama, Japan

Thomas R. Patin, Editor

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PREFACE

The 12th U.S./Japan Experts Meeting on Management of Bottom Sediments Containing Toxic Substances was held 11-14 November 1986 in Yokohama, Japan. 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 (WRSC), was the U.S. Chairman. Mr. Takao Nishimura, 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. Mr. Charles C. Calhoun, Jr., was Program Manager, DOTS, at the time of the meeting. Dr. Robert M. Engler is the present Manager of the DOTS Program.



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AGENDA

12th U.S./JAPAN EXPERTS MEETING ON MANAGEMENT
OF BOTTOM SEDIMENTS CONTAINING
TOXIC SUBSTANCES

Yokohama, Japan
11-14 November 1986

November 11, 1986

- 9:00-9:30 Opening Ceremony
- 9:00-9:10 Director-General of the Ports and Harbors Bureau, MOT
- 9:10-9:20 Mayor of Yokohama City
- 9:20-9:30 Cochairman Col. Kleb
 Mr. Nishimura
- 9:30-10:00 Beach Reclamation and Dredged Disposal for Marine
 Environment Improvement
 Presented by Yutaka Matsubara, Ports and Harbors
 Bureau, Ministry of Transport
- 10:00-10:30 Research Consideration of Physical Factors Relevant
 to Disposal of Contaminated Dredged Material
 Presented by Limbarios Vallianos, U.S. Army Corps
 of Engineers, Waterways Experiment Station (WES)
- 10:30-11:00 Coffee Break
- 11:00-11:30 Dewatering and Reuse of Water Blooms of the Lake
 and Port
 Presented by Zenya Yoshino, Science University of
 Tokyo
- 11:30-12:00 Lake Suwa Is Being Restored
 Presented by Tatsuo Yoshida, Japan Sediments
 Management Association
- 12:00-13:30 Lunch
- 13:30-14:00 The Movement of the Turbid Water in Reclaiming
 Presented by Eiji Yauchi, Japan Dredging and
 Reclamation Engineering Association
- 14:00-14:30 Dredged Material Disposal Management Studies for the
 Port of New York and New Jersey: Progress Report

Presented by John Zammit, U.S. Army Engineer
District, New York

- 14:30-15:00 Method on Watch for Prevention of Secondary
Pollution Caused by Dredging Work of Bottom
Sediment Containing Mercury at Minamata Bay
Presented by Koichi Kyuma, Environment Department,
Kumamoto Prefecture
- 15:00-15:30 Coffee Break
- 15:30-16:00 Aiming at Cleaner Sediments in Hiroshima Bay -
Trial Cleaning Project by Means of Sand Covering
in Kure Inlet and Its Effects
Presented by Hideaki Kakigawa, The 3rd District
Port and Harbor Construction Bureau, Ministry of
Transport
- 16:00-16:30 Puget Sound Dredged Material Disposal Analysis
Presented by Frank Urabeck, U.S. Army Engineer
District, Seattle
- 16:30-17:00 Removal Work of Organic Deposit by Oozer Pump
Dredger
Presented by Fujio Otsu, Japan Dredging and
Reclamation Engineering Association
- 17:00-18:00 Free
- 18:00-20:00 Japan Reception
Yokohama International Conference Center Lounge

November 12, 1986

- 9:00-9:30 Assessment of Sediment Control on Lake
Eutrophication Using an Ecological Model
Presented by Masaaki Hosomi, Laboratory of
Freshwater Environment, National Institute for
Environmental Studies
- 9:30-10:00 New Technologies Applied at Meinohama Landfill Work
Presented by Masayoshi Tajiri, Japan Sediment
Management Association
- 10:00-10:30 Coffee Break
- 10:30-11:00 The Role of Microorganisms in Mobilization of
Contaminants from Sediment
Presented by Douglas Gunnison, WES-Environmental
Laboratory

- 11:00-11:30 Test on Sediment Gas Generation
Presented by Kiyoshi Nikaido, Japan Sediment
Management Association
- 11:30-12:00 Dealing with Toxic Polluted Sediments in the Great
Lakes Basin
Presented by Peter L. Wise, U.S. Environmental
Protection Agency
- 12:00-12:30 Evaluation of Chemical Pollutant by Analysis of Oil
Component in Sediments
Presented by Katsuyuki Ninomiya, Yokohama
Environmental Research Institute
- 12:30-13:30 Lunch
- 13:30-17:00 Yokohama City Excursion (Yokohama Port, etc.)
- 17:00-18:30 Movement
- 18:30-20:00 U.S. Reception (New Sanno Center, Tokyo)

November 13, 1986

- 9:00-9:30 Port of Baltimore: Present and Planned Management
of Contaminated Dredged Materials
Presented by Frank Hammons, Baltimore Port
Authority
- 9:30-10:00 Mechanism of Gas Generation in Canals of the Tokyo
Port
Presented by Yoshihiro Matsumoto, Department of
Construction, Bureau of Port and Harbor, Tokyo
Metropolitan Government
- 10:00-10:30 Coffee Break
- 10:30-11:00 Development of the Port of Long Beach: Past,
Present, and Future Projects
Presented by Charles Connors, Long Beach Port
Authority
- 11:00-11:30 Preliminary Study on the Ecological Structure of
Benthic Organisms in Tokyo Bay
Presented by Masaru Yoshida, Japan Sediment
Management Association

11:30-12:00 Impact Assessment of Inplace Contaminated Sediments
 on Water Quality
 Authored by Dr. Tom Hart
 Presented by Douglas Gunnison
 WES-Environmental Laboratory

12:00-12:30 Signing of Joint Communique, Closing Remarks

12:30 -18:00 Free

18:00-20:00 Yokohama City Reception
 (Hotel Yokohama Teishin Kaikan)

November 14, 1986

8:30 M. O. T. Excursion
 (Institute for Technical Research of Ports and
 Harbors, Kamakura)

BEACH RECLAMATION AND DREDGED MATERIAL DISPOSAL FOR
MARINE ENVIRONMENT IMPROVEMENT

AD-P006 895



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INTRODUCTION

The Bureau of Ports and Harbors, Ministry of Transport, is engaged in a number of sea bottom management operations. These operations consist of the prevention of harbor pollution, implemented primarily by dredging bottom sediment, and a study of sea bottom sediment clarification designed to eliminate organic sediment in bays and inland seas.

In implementing these projects, one very significant problem that must be solved is disposal of the dredged material. Difficulties are also encountered in disposal of the material produced from the improvement works of port facilities, fairways, and anchorages. These difficulties occasionally block the implementation of such projects.

This paper describes examples of the disposal of, or use of, certain types of dredged material and organic sediment for beach reclamation.

BACKGROUND

Japan is surrounded by the sea, and the productive activities and life styles of the Japanese people were once closely related to the sea. In particular, beaches and tidelands played important roles, both as protection against tidal waves and for fisheries and salt manufacturing. The beaches and tidelands provided the vital environmental functions of aeration, water purification, and maintenance of the ecological balance for birds and shellfish, while at the same time affording recreational and sightseeing opportunities for people to swim, fish, and surf.

However, as land reclamation and harbor facility construction increased during the period of Japan's high economic growth, the beaches and tidelands gradually disappeared along the shores of the large urban areas. Today only 10 percent of the original beach areas remain around the shores of Tokyo Bay.

Now that industrial development has peaked, the importance of these beach areas is being reevaluated. The needs of the people have diversified and their interest in the natural environment has increased.

Under these social circumstances, the Ministry of Transport began a study of methods to dispose of, or utilize, dredged material generated from sea bottom management and port and harbor work. Dredged material disposal and utilization can be placed in the overall sea environment improvement scheme as illustrated in Figure 1. Actual examples of the use of dredged material for beach reclamation are presented in the following paragraphs.

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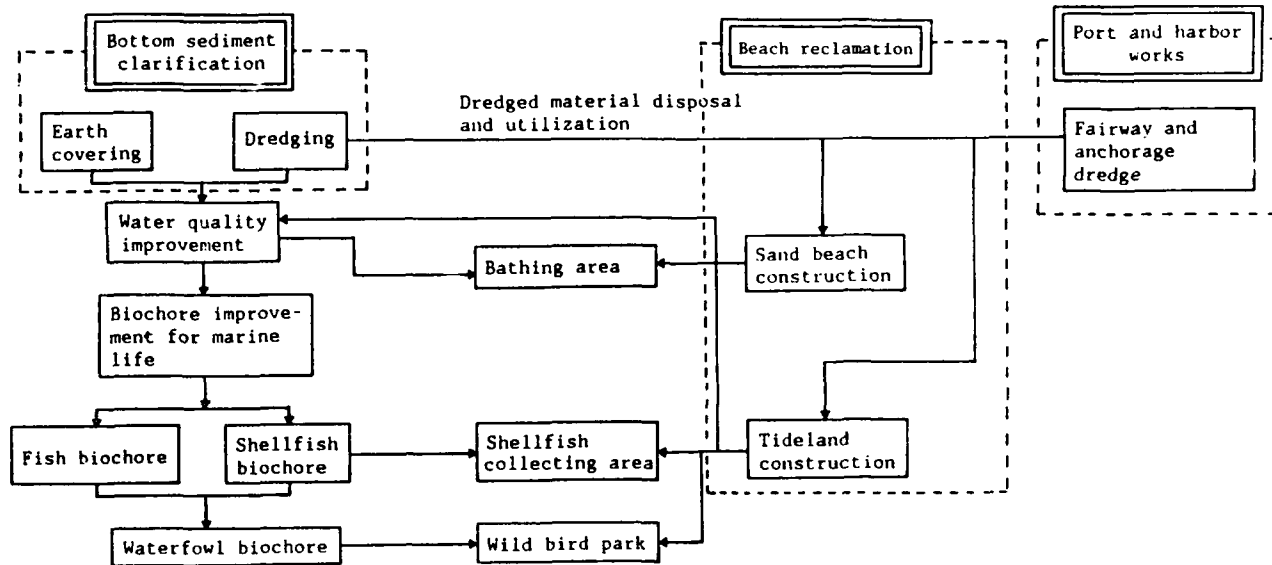


Figure 1. Utilization of dredged material for marine environment improvement

Sea Bottom Sediment Clarification Projects and Beach Reclamation

In bays and inland sea areas where there is little exchange of water between the open sea, huge amounts of organic sediment are created by residential and industrial wastes. This sediment accumulates on the sea bottom, as illustrated in Figure 2 and Table 1. In a sea of this nature, oxygen

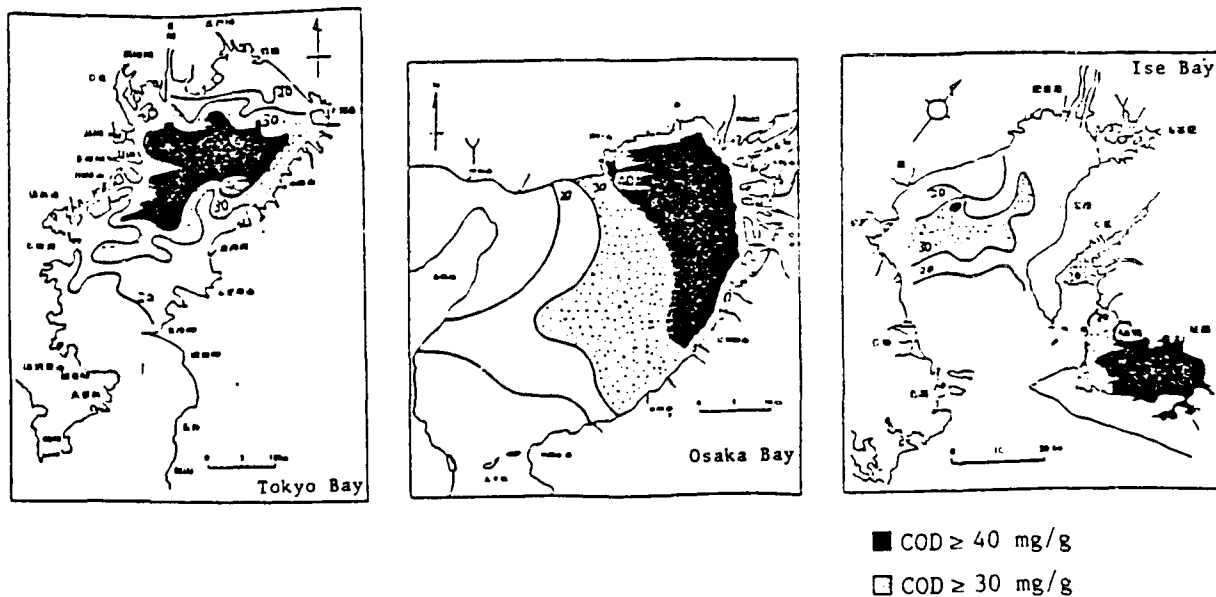


Figure 2. Organic sediment distribution (COD = chemical oxygen demand)

depletion due to elution of eutrophic salts and decomposition of organic materials accelerates water quality degradation, thereby severely damaging the fishing industry. The objective of the bottom sediment clarification project

TABLE 1. ORGANIC SEDIMENT STATUS IN TYPICAL SEA ZONES

Environmental Factor	Tokyo Bay	Mikawa Bay	Osaka Bay
Amount of organic sediment, $m^3 \times 10^3$			
COD contamination ≥ 30 mg/g	235,000	79,000	226,000
COD contamination ≥ 20 mg/g	403,000	240,000	698,000
Area of organic sediment, km^2			
COD contamination ≥ 30 mg/g	440	250	670
COD contamination ≥ 20 mg/g	610	320	1,130
Average organic sediment thickness, cm			
COD contamination ≥ 30 mg/g	53	32	34
COD contamination ≥ 20 mg/g	66	75	62
Average water depth in contaminated area, m			
COD contamination ≥ 30 mg/g	18	11	18
COD contamination ≥ 20 mg/g	20	10	21
Maximum water depth in contaminated area, m			
COD contamination ≥ 20 mg/g	32	18	45
Sediment quality, COD maximum contamination, mg/g	62	64	45
Sea zone area, km^2	1,200	600	1,500
Amount of seawater, km^3	15	6	42
Average water depth, m	15	10	28
Land water inflow, $m^3/day \times 10^7$	3.2	0.7	5.9

is to improve the seawater either by dredging the sediments or capping. Bottom sediment clarification projects have been studied since 1979 (Figure 3). Bottom sediment clarification projects are broadly divided into the dredging method and the capping method. The most difficult problem encountered in dredging projects is the disposal of the large amounts of sand and organic sediment generated. Table 2 lists the various disposal methods presently under study, together with their relative advantages and disadvantages. Further details of the beach reclamation disposal method are discussed herein.

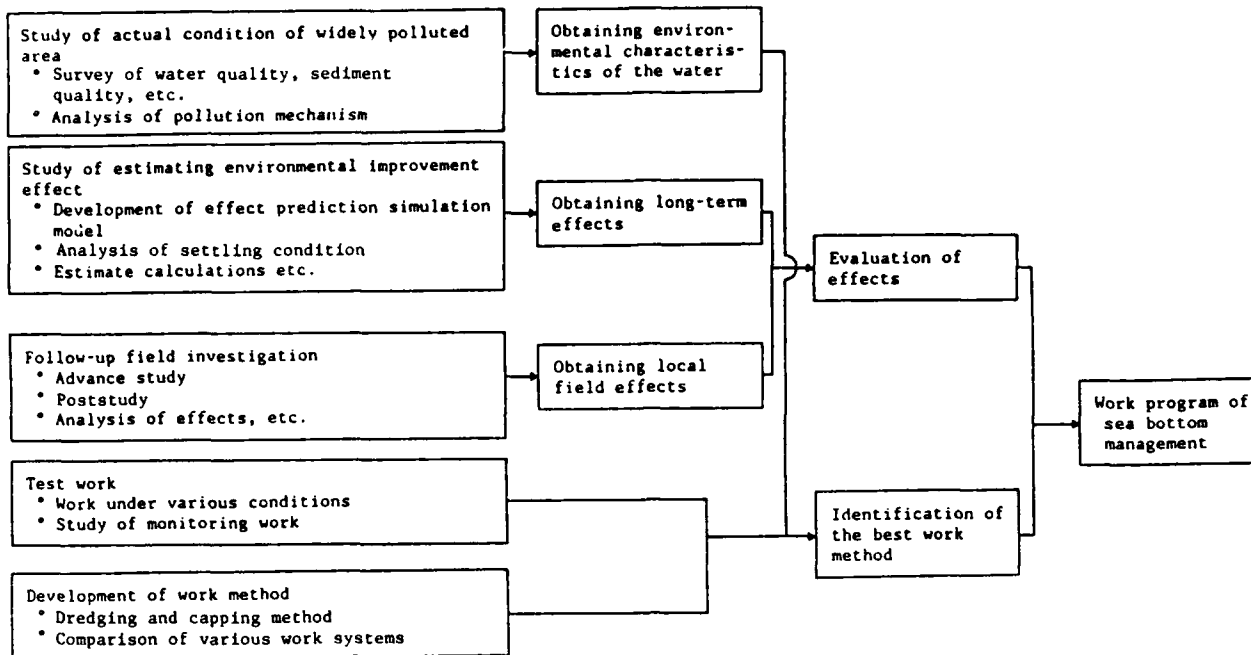


Figure 3. Process of bottom sediment clarification study

The amount of sand and organic sediment produced by dredging is generally very large, usually very soft soil with a high water content. Figure 4 shows the technical problems in constructing reclaimed beaches with dredged material. Various studies of these problems have been conducted; as a part of these studies, it was assumed that 3,000,000 m³ of sediment in Osaka Bay is to be dredged and used to create a reclaimed beach. Figure 5 shows a cross section of the reclaimed beach. By using Table 3, an estimate of the construction cost can be calculated and compared with the costs of other disposal methods. As indicated in the table, the cost to reclaim a beach is higher than that of disposing the sediment in pits in the sea bottom or in the open ocean. However, beach reclamation is worthy of consideration because no sea bottom pit exists of adequate size for 3,000,000 m³ of sediment in Osaka Bay, and it would be difficult to obtain public approval to dispose the material in the open ocean. Therefore, although the cost of beach reclamation with the sediment may be high, the cost of reclaiming land by any other means is much higher.

Port and Harbor Works and Beach Reclamation

In recent years, the production of dredged material from port and harbor works has risen to 70,000,000 m³, most of which has been disposed of as land reclamation materials. It is often difficult, however, to find locations where such large amounts of material can be placed; and some programs for the improvement of fairways and anchorages are being delayed for this very reason.

The characteristics of the dredged material for which destinations cannot be found are suitable for the construction of beaches and tidelands. The amount of suitable materials is estimated at 2,500,000 m³ per year.

TABLE 2. DREDGED SEDIMENT DISPOSAL METHODS

Disposal Methods	Outline	Advantage	Disadvantage
Land reclamation	Revetment constructed, into which wastes and dredged sediment are filled to form land	Little effect of secondary contamination, and organic sediment safely disposed	Water area to be reclaimed must be secured High project cost Substantial foundation improvement required to use the reclaimed land
Beach reclamation	Revetment constructed, and wastes and dredged sediment filled to form beach	Reclaimed land used as sand beach and tideland Water quality improvement effect expected Social consensus gained relatively easily for the formation of beach	Cost of revetment and earth covering is relatively high Consideration must be given to the possibility of secondary contamination
Sea bottom pit fill	Organic sediment dumped into deep hole or depression in seafloor and then covered with earth	Economically cheaper than any other method if the capacity corresponding to the amount of dredge is secured	Amount of disposed organic sediment determined by available seabed hole capacity Cost is high if a new hole must be provided Consideration must be provided regarding the risk of secondary contamination during the work
Open ocean disposal	Organic sediment transported by barge, etc., to open sea and disposed	Costs less than land or beach reclamation	Large secondary contamination effect, public approval difficult to obtain

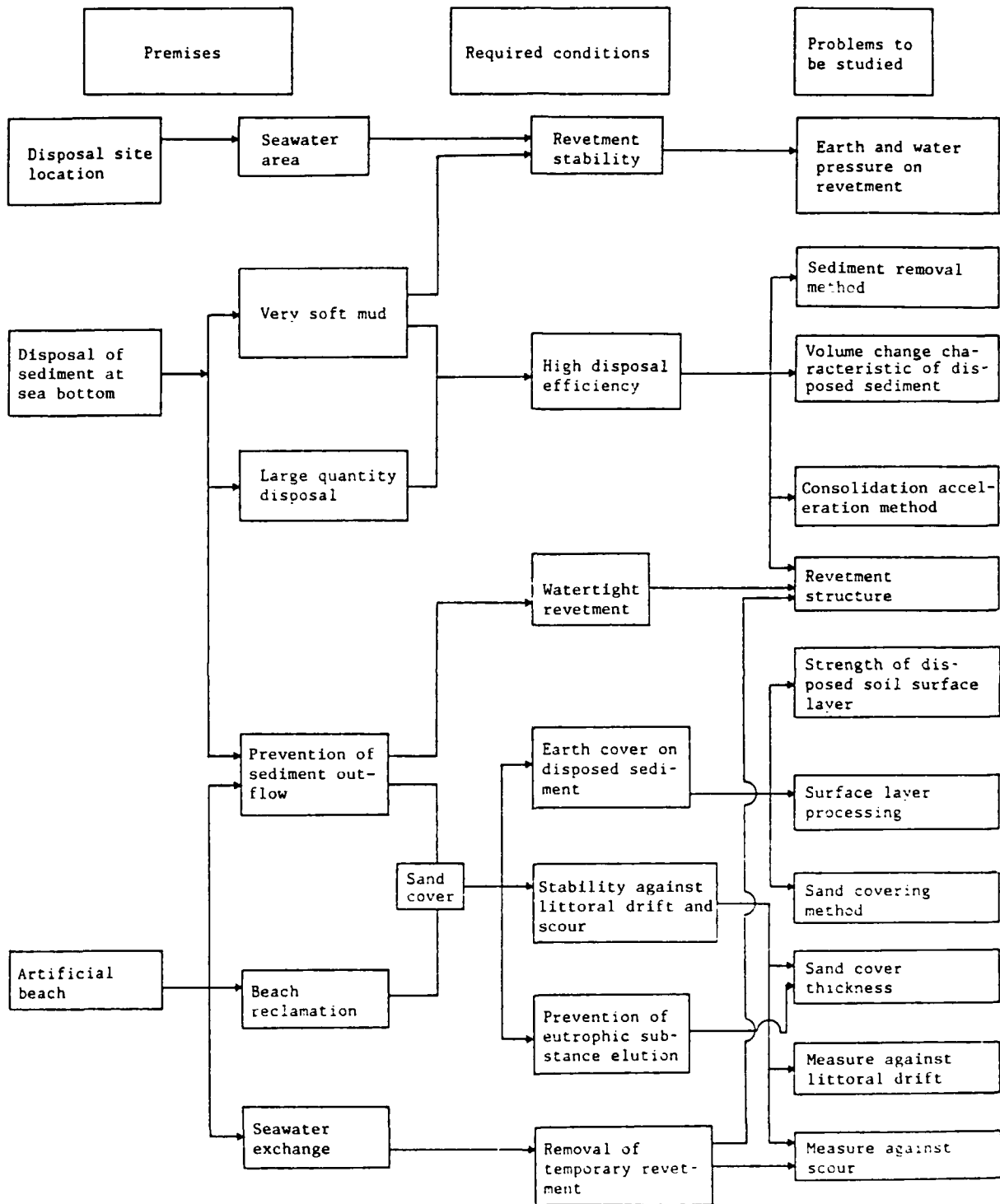


Figure 4. Flowchart of beach reclamation technical problems

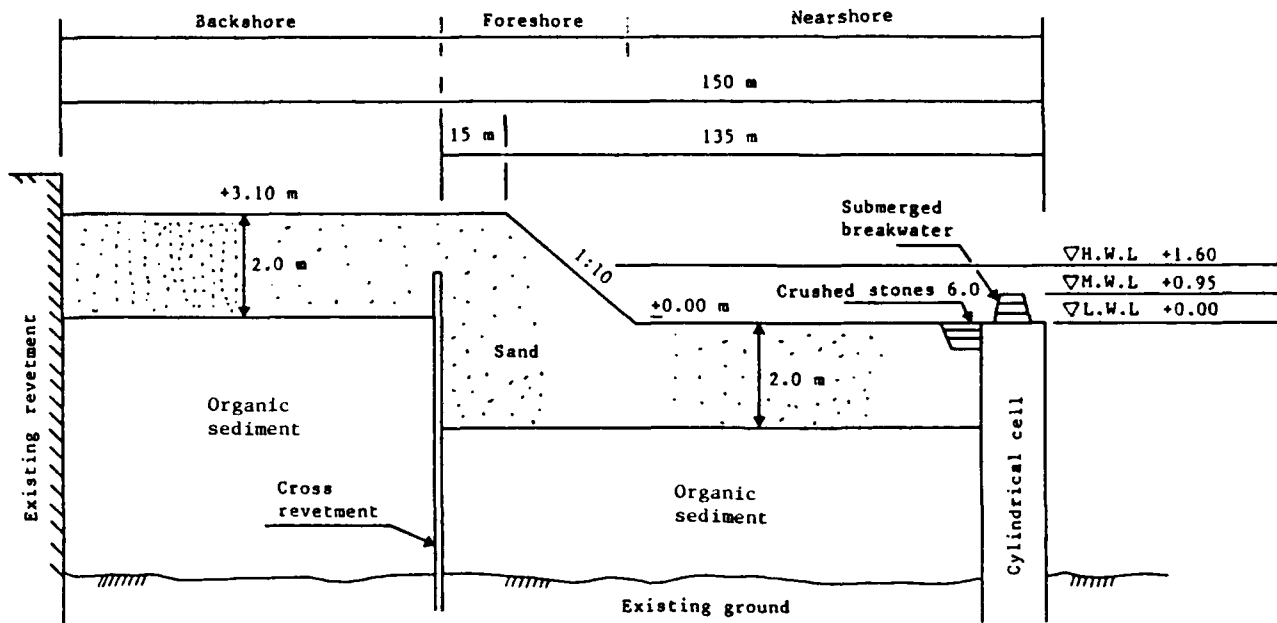


Figure 5. Cross section of reclaimed beach

TABLE 3. COMPARISON OF DISPOSAL COSTS

Method	Requirements	Cost
Land reclamation	Peripheral revetment Surface treatment Earth cover Foundation improvement	Several hundreds of billion yen (depending on required foundation improvement)
Beach reclamation	Peripheral revetment and cross revetment Surface treatment. Sand covering Foundation improvement	13 billion yen (85% for revetment)
Sea bottom pit fill	Turbidity prevention work Sheet work Covering work	1.7 billion yen
Open ocean disposal	Transport 81 nautical miles (150 km) to open ocean and dispose	1.6 billion yen

Based on these considerations, beach reclamation studies were initiated. As such, a large, experimental, artificial tideland construction program was undertaken at the Ninoshima-Nagahama district, Hiroshima Bay. In this project, 1,300,000 m³ of material produced from dredging the fairways and anchorages of Hiroshima Bay will be spread on 40,000 m² of seabed to create a tideland.

This project, which began in 1983 and is currently ongoing, includes construction of submerged breakwaters and the earth covering works. It will cover a period of 3 years at a total cost of 2.5 billion yen. The unit cost incurred for the disposal is 1,200 yen/m³, which can be regarded as reasonable when compared with the unit cost for conventional land reclamation at 1,000 yen/m³. The project outline and photographs of the construction work are provided in Figures 6-10.

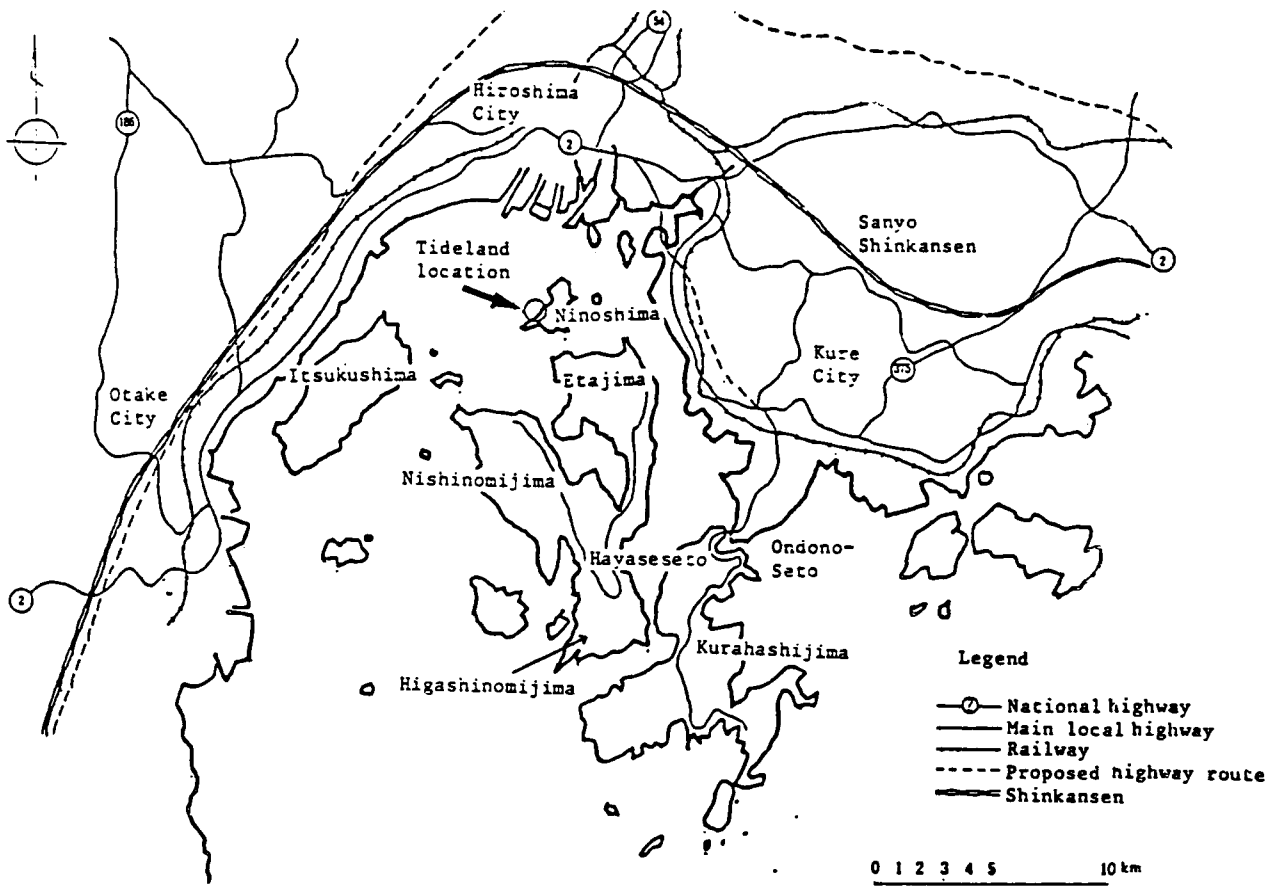


Figure 6. Location map of Ninoshima District reclaimed tideland

CONCLUSIONS

Because lifestyles are more diverse and the public's interest in the value of the natural environment is enhanced, the various functions of sea-shore and beach areas are in the process of being reevaluated. With these trends, beach reclamation projects have dual functions: create a seashore environment, and facilitate bottom sediment clarification projects and port and harbor works. It is from these viewpoints that the Ministry of Transport plans to use dredged material in the reclamation of beaches to improve marine environments.

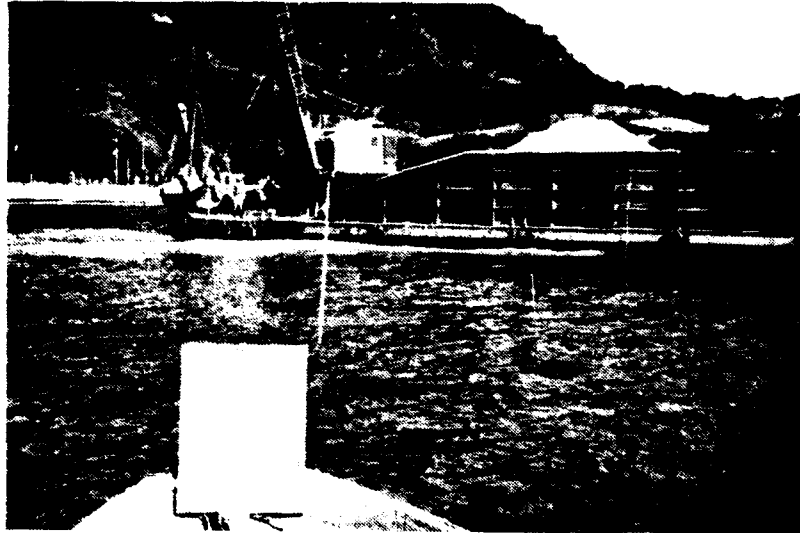


Figure 7. Sand covering (sand is transferred from the sand carrier with a grab bucket and dumped by a crane ship)



Figure 8. Completed sand cover

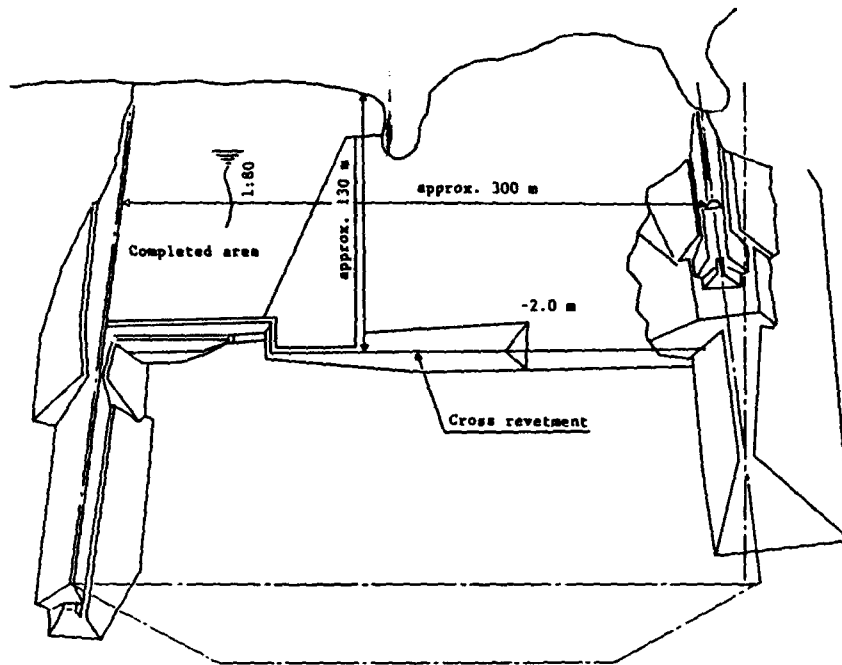


Figure 9. Plan

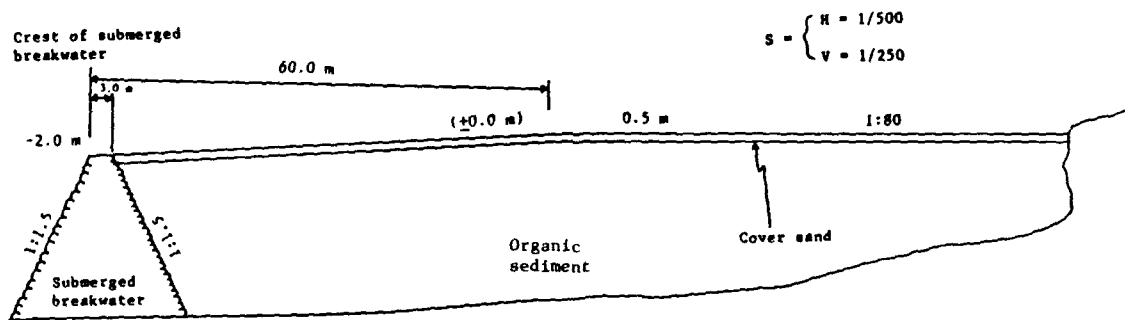


Figure 10. Standard cross section

RESEARCH CONSIDERATIONS OF PHYSICAL FACTORS RELEVANT
TO DISPOSAL OF CONTAMINATED DREDGED MATERIAL

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AD-P006 896



ABSTRACT

The US Army Corps of Engineers is considering the initiation of five research areas directed toward improving analytical methods and field monitoring procedures related to the physical aspects of disposing and confining contaminated dredged material. These research initiatives are:

a. Evaluation of critical shear stress. Theoretical and mathematical investigations will be conducted to develop the best methodology for calculating shear stress. Field measurements will be performed to obtain data sets for verification of theoretical results.

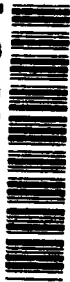
b. Fate and stability of dredged material disposed in open waters. This research will involve an ensemble of existing numerical models in developing a standard application framework for simulating the hydrodynamic/sediment interactions in offshore and estuarial environments. Standardized techniques and equipment will be developed for monitoring open-water disposal sites.

c. Suitability of material for capping contaminated dredged material. Research will combine integrated field tests and physical and numerical model investigations to establish a standard methodology for the design and placement of stable caps under varying physical, chemical, and biological conditions.

d. Technology for monitoring hopper loads. This area of research will focus on defining and assembling new instrumentation systems that integrate such components as nuclear density probes, draft indicators, and microprocessors to monitor and measure hopper loads.

e. Open-water disposal site capacity and management. Numerical and physical modeling of successful disposal sites will determine if greater capacities can be achieved by alterations in the existing

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geometries, and if such alteration would result in stable disposal mounds. In a second phase of research, representative shapes found to produce high volume and stable mound configurations will be studied with the goal of establishing general guidance for disposal site design and management.

INTRODUCTION

The US Army Corps of Engineers Directorate of Civil Works is involved in virtually every navigation dredging operation performed in the United States. Dredging is the largest single item in the Corps' Civil Works operations and maintenance budget, and amounts to approximately \$475 million (US) annually. Although dredging has always been a primary function of the Corps, the past 10 years have seen significant changes in the nature of that function and its interaction with others. Environmental concerns, a large increase in the level of contract dredging, and the oil embargo and its consequences are some of the national and international trends that have placed unprecedented demands on the Corps' dredging mission. Additionally, the Office of the President has instituted strong initiatives directed at budget deficit reductions, project cost-sharing with non-Federal interests, user fees, and support for the deepening of major US ports. These changes and the Administration's policies have created pressures to examine and improve, where possible, the way dredging operations are planned, executed, and managed. In response, the Corps of Engineers has, among other measures, continued a dynamic applied research effort spanning all aspects of dredging and the placement of dredged material. The purpose of this paper is to outline five areas of new and/or expanded research activities that the Corps is considering for near-future initiation. These are all related to the physical aspects of placing and confining contaminated dredged material in open waters. Specifically, the areas of research being considered are: (a) evaluation of critical shear stress, (b) fate and stability of dredged material disposed in open waters, (c) suitability of material for capping contaminated dredged material, (d) technology for monitoring hopper dredge loads, and (e) open-water disposal site capacity and management. Each of these research topics is outlined in the following text of this paper.

EVALUATION OF CRITICAL SHEAR STRESSES

The critical bottom shear stress (threshold shear stress) is a fundamental parameter required in calculation of water circulation and sediment movement. Every method for calculating water flow and sediment transport rate, from simple desk computations to sophisticated numerical models, requires an estimate of this pivotal parameter. The importance of an accurate calculation of the shear stress and associated friction coefficient cannot be overstated since not only sediment transport, but also the governing momentum and energy equations for wave transformation and current circulation contain dissipation contributions expressed in terms of the shear stress. In present engineering practice, highly simplified calculation procedures are used to estimate the critical shear. The basic physics of the critical shear stress are not well understood, and there is no substantive guidance available for its calculation.

For verification of theoretical developments, there are no data sets available that give reliable measurements of the shear taken under field conditions. In fact, the instruments needed to make such measurements have only recently become available. These instruments must be integrated, developed into a system capable of field deployment, and then used to acquire data under a variety of wave, current, and bottom conditions.

Work Unit 1: Calculation of Critical Shear

This work unit will entail an intensive theoretical and mathematical investigation to develop a method for calculating the critical bottom shear stress. Available methods will be critically analyzed and extended, as necessary, to cover the widest range of engineering application. All relevant field and laboratory data will be presented in three forms: (a) graphical representation for field use and desk calculation, (b) computer programs for use on personal computers (PC's), and (c) computer algorithms for adaptation in numerical models. Reports will be published to provide the full theoretical background as well as to provide guidance for engineering use.

Calculation of the critical shear requires knowledge of the value of the bottom friction coefficient. The friction coefficient depends on the bottom roughness of the bed, as well as the sediment characteristics. In dredging applications, waves and uniform (tidal) currents may coexist. Therefore, the calculation of the friction factor and shear stress must be valid for oscillatory flow (waves), uniform flow (tide), and the combination of these two flows. Results obtained for linear monochromatic waves will be extended to include a more realistic description of waves by incorporating finite amplitude and spectral effects, if necessary.

The final product is aimed toward providing the Corps with a general and accurate methodology to calculate the critical bottom shear stress under all anticipated environmental conditions encountered in dredging and dredged material disposal operations. The results of this work unit will be of immediate benefit to several other areas of interest. For example, an accurate knowledge of the shear stress is required for defining navigable depth in fine-grained sediment and evaluating the stability of disposal mounds.

Work Unit 2: Measurement of Shear Stress and Threshold of Movement

Calculation of sediment movement requires accurate quantitative knowledge of the shear stress, both at the threshold of movement and in the active transport condition. Only limited theoretical work has been done on this subject, and engineering estimation methods for the bed shear and bottom friction coefficient are extremely crude (for example, use of a constant value irrespective of actual sediment grain size, water temperature, presence of both waves and currents, presence of entrained sediment, and varying water depth; neglect of phase difference between fluid flow and stress; etc.). The main reason for inadequate understanding of this important parameter is a lack of field or full-scale laboratory measurements.

Recently, instrumentation has become available with which to measure the shear stress in the field under operational conditions in which dredging operations take place. A number of measurement methods have been developed for deeper water applications, and their capabilities can be extended to the usual dredging environment. Thus, heavy initial developmental costs can be avoided by assimilating and modifying existing new technology.

This work unit will evaluate modern techniques for measuring the bottom shear. Techniques to be investigated include the classical velocity distribution method by means of velocity probes and acoustic doppler speed profiler; direct measurement of Reynolds stresses; and shear plates. The most reliable and accurate methodology will be selected for obtaining measurements under typical site conditions. The focus of the work unit will be on technical aspects of the equipment in order to optimize measurement capabilities and actual field data collection.

A bottom stress measurement system and related flow measurement devices will be purchased and modified for nearshore use. After testing, the system will be deployed to collect data at representative locations. Numerous field deployments are anticipated in order to collect data for various sediment types, water depths, wave conditions, water temperatures, etc. It is well known that the critical stress developed under laboratory conditions (scale, sediment state, fluid state) is quite different from the stress developed under field conditions. Therefore, in situ field measurements under the widest possible range of environmental conditions are required. Due to the high volume of raw data return from the electronic instrumentation, special provisions will be necessary to properly collect, transmit, store, format, and catalog the data for analysis. The raw data will be appropriately annotated and systematically archived for use in concurrent and subsequent analysis in Work Unit 1.

The developed instrumentation will also allow measurement of the sediment shear at specific sites for optimizing dredged material disposal.

FATE AND STABILITY OF DREDGED MATERIAL DISPOSED IN OPEN WATER

Innovative ideas for placement of dredged material require extensive study before they are adopted for use in the field. At present, the Corps does not have a widely accepted, standard procedure for applying numerical models to a project to predict the ultimate fate of the dredged material. The Corps needs to develop and implement an accepted procedure that will address the questions concerning the interaction of dredged material and local processes that arise during a dredging project or the life of a dredged material disposal area. In order to be complete, this procedure must include both numerical simulation and in situ field monitoring.

Emphasis will be placed on dredged material disposed in oceanic and estuarine environments. Because these environments are hydrodynamically dissimilar, separate modeling techniques are required to provide adequate predictions of sediment transport, and ultimately the fate and stability of the disposed material. Characteristically, oceanic environments are well mixed and hydrodynamically dominated by waves and wave current interactions. Estuaries tend to be stratified, with large temperature and salinity gradients, and are hydrodynamically dominated by tidal and density currents.

In the ocean, the interaction of dredged material with nearshore (inner shelf) processes such as waves, currents, varying water depth, and wave refraction will be simulated, while modeling efforts in estuaries will address the effect on dredged material stability of such variables as current velocity, salinity, water depth, wind-induced mixing, water temperature, and bottom character. In both environments, the short-term (dredging season) and long-term (years) fate of disposed material will be numerically simulated.

The results of this work unit will allow predictions of long-term sediment transport in open-water disposal sites. This will permit updating characteristics such as material size, quantity, and percent of water column occupied. Utilizing these and other data, the results of the unified modeling study will enable Field Operating Agencies (FOA's) to determine if disposal characteristics will change over time. This will allow FOA's to address rationally a variety of potential circumstances, such as: does the area change from nondispersive to dispersive by changing sediment characteristics or occupying more of the water column due to increased disposal rates.

Field measurement procedures using accurate and cost-effective technology will be developed and implemented. Field data are needed to determine the most efficient and economic dredged material disposal methods for different conditions. Field data are also an essential part for numerical model adjustment and verification.

Work Unit 1: Simulation
Methods for Evaluation of Off-
shore Disposal of Dredged Material

Numerical models will be used to investigate the response of dredged material to nearshore (inner shelf) processes. A suite of existing numerical models will be applied, developing a standard application framework to simulate the interaction of nearshore processes on dredged material disposed in the offshore region. Modeling technology which has been and is presently being developed in other research projects will be used to investigate this problem. In order to apply the results of the numerical simulations to the largest number of locations, hypothetical project locations, geometries, sediment characteristics, and incident nearshore processes will initially be simulated. Existing models will be used to hindcast the fate of dredged material disposed offshore. The effects of waves, currents, varying water depth, wave/current interaction, and wave refraction on the fate and stability of dredged material will be simulated. Simulations will be used to hindcast the fate and stability of existing dredging projects which have been extensively monitored. Additional field data that may be required will be obtained through Work Unit 3. In order to adequately determine sediment transport potentials over the life of a dredged material disposal site, it will be necessary to implement both deterministic and stochastic modeling techniques. A fully deterministic approach would most likely be cost prohibitive. Therefore, a combination of deterministic and stochastic modeling techniques will be used to obtain predictions of sediment transport directions and quantities from the idealized dredged material disposal sites.

This work unit will produce a set of numerical models that will provide sediment transport predictions for project planning and guidelines on the fate and stability of dredged material.

Work Unit 2: Modeling of
Short-Term Sediment Transport in
Estuarine Open-Water Disposal Areas

In an estuary, dredged material is placed in either a low energy area where it is supposed to remain or a high energy area where it is dispersed. An understanding of the mechanisms that determine the stability of a site is essential to the efficient management of disposal areas. Dredged material disposal models have been developed and are presently in use to evaluate the short-term fate of dredged material associated with the disposal process. However, further expansion and enhancement of these models are needed to include sediment transport relationships into the numerical codes. The result will be numerical models that predict the final location of dredged material placed in estuarine open-water sites. With an understanding of the dynamics of a disposal area and the disposal process, decisions can be made regarding planning goals, resource storage, permanent disposal, or timed dispersal of dredged material.

This unit will coordinate with Work Unit 3 to acquire the necessary field data to modify the existing numerical codes. Case studies of different types of disposal areas will be conducted in cooperation with participating FOA's.

Work Unit 3: Field Techniques and Data
Analysis to Assess Open-Water Disposal

The Corps presently monitors open-water disposal areas in the oceans, estuaries, and Great Lakes to ensure that dredged material behaves as planned and causes no adverse physical effects. The types of monitoring and levels of detail are presently decided on a case-by-case basis. A task initiated under the Dredging Operations Technical Support (DOTS) Program is assembling state-of-the-art guidelines for monitoring disposal sites. This work unit will provide the necessary expansion and follow-up to the DOTS task to apply and improve the recommended techniques and equipment and to develop new ones where the DOTS work identified deficiencies. The work unit will:

- a. Apply state-of-the-art techniques at selected disposal sites.
- b. Apply new techniques to determine their potential.
- c. Recommend methodology or equipment modification or development.
- d. Recommend data analysis and interpretation procedures.
- e. Tailor the general monitoring guidelines developed under the DOTS Program for different disposal configurations.
- f. Develop data sets for use in Work Units 1 and 2.

- g. Develop additional data sets to verify models at a range of test sites.

The case studies of different types of disposal areas will be constructed in cooperation with participating Corps FOA's.

SUITABILITY OF MATERIAL FOR CAPPING CONTAMINATED DREDGED MATERIAL

Isolation of contaminated dredged material by subaqueous contained aquatic disposal (CAD) in open water is a viable option for many locations. A considerable body of work exists on designing caps for CAD sites to meet chemical and biological criteria, but considerably less methodology is available to allow confident prediction of cap physical stability. Stability should be addressable both from the event (storm) standpoint and from the long-term perspective.

Work under this task will combine integrated field and laboratory experiments and numerical simulation studies to determine stability of CAD cap materials under varying physical, chemical, and biological conditions. Field investigations will be performed to gather data for calibrating and verifying numerical simulation techniques and to obtain representative cap material samples for laboratory experiments. Flume tests will be conducted to determine high-stress (storm) effects on various types and conditions of cap material. Numerical simulation will include the following:

- a. Turbulence closure simulation of a CAD site to approximate the bottom boundary layer under combined current-wave interaction.
- b. Demonstration of simulation techniques to assess the long-term behavior of a CAD cap.
- c. Parametric study to establish guidelines for cap placement as a function of bathymetry and expected hydrodynamic conditions.

This work will produce an integrated methodology to both predict the physical behavior of a CAD cap under particular event conditions and assess its long-term stability in a given environment.

TECHNOLOGY FOR MONITORING HOPPER LOADS

Vertical density gradients within the sediments collected in the bins of hopper dredges and dump scows vary depending upon such factors as type and size of dredging device used, sediment characteristics, retention time in bins, overflow conditions, and loading cycles. Clamshell dredged material, especially of new work, is frequently in the form of large cohesive clumps, while hydraulically dredged material is usually dispersed throughout the hoppers. The volumetric quantity of dredged material in the hopper is presently estimated using the displacement method. The change in vessel draft, the depth (or volume) of material in the bins, some value for specific gravity, or an estimate of water content of the sediments are influencing factors to the final calculation of cubic yards of solids in the hopper.

Scientific techniques can be utilized to describe more accurately the density strata within the hoppers, thus enabling exact calculations of cubic

yards of material dredged. The primary function of this work is to define and assemble new systems that integrate such components as nuclear density probes, draft indicators, and microprocessors to monitor and measure hopper loads.

A major benefit of this program will be to determine more accurately how much material has been placed in the hopper and the amount spilled in the overflow process. Improvements will also be realized in the hopper dredge economic loading procedure.

OPEN-WATER DISPOSAL SITE CAPACITY AND MANAGEMENT

Selection of open-water disposal sites involves the careful analysis of many factors such as location, environmental effects, and long-term fates. Much progress has been made in providing managers with effective and functional design guidance on site selection. However, after a suitable site has been selected, a determination of site capacity is made using somewhat limited technology. If the engineer's estimate of disposal site capacity is low, the result might be the abandoning of the site before it has reached its full capacity, which may result in disposal activities being shifted to a less desirable location, often at a greatly increased expense. If the engineer estimates a capacity greater than the site should contain, then the overfilling of the site may cause undesirable effects such as large amounts of sediment being resuspended or moved out of the immediate disposal area. Placement of material within the area often follows no overall plan, resulting often in discrete piles or in large accumulations near the boundary closest to the dredge site.

In years past, when dredging activities were less frequent than today and when operations costs were not as high, accurate determination of open-water disposal site capacity was not a strong economic factor. Consequently, little research was undertaken to address this question. In today's economic climate, it is essential that managers have the technology available to select a potential disposal site, to determine an appropriate and stable three-dimensional shape for the site which gives the maximum capacity, and to specify placement of the material within the site to produce the desired shape and consolidation to maximize site capacity.

This work aims to provide engineers and managers modern methods for determining open-water disposal site capacity and to produce guidance on managing the placement of material within the site. The research direction will build upon past experience by examining those sites that are considered more successful than others. A recently completed DOTS management task dealing with initial considerations on estimating site capacity and documentation of deficiencies with existing methods will form a basis for this work. Postconstruction surveys of these sites will help to isolate potential geometries which hold promise for design. The relationship of these configurations to the regional environmental conditions will be investigated. Naturally occurring bottom features such as linear shoals, reefs, and nearshore bars will be included as potential disposal site configurations. Numerical and physical modeling of successful disposal sites will focus on determining if a greater disposal capacity exists at these specific sites by the design of more appropriate disposal geometries. Models also will determine if new geometries produce a stable mound. Immediate benefits may be realized if some of these selected sites prove to have a greater capacity than previously thought.

In the second phase of this research, representative shapes found to produce high volume, stable mound configurations in the site-specific examples will be studied with the goal of producing design guidance. It is expected that some shapes will be successful only in a regional extent, being closely tied to the prevailing environmental conditions.

The final outcome of this work will be greatly improved guidance for disposal site design and management. This guidance will be in the form of manuals which will help the manager match a potential site with an appropriate shape and placement technique, considering local and regional processes. Prediction of actual site capacities for selected designs will be available in the form of software fitting into the framework of computer-aided design.



DEWATERING AND REUSE OF WATER BLOOMS
IN THE LAKE AND THE PORT

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ABSTRACT

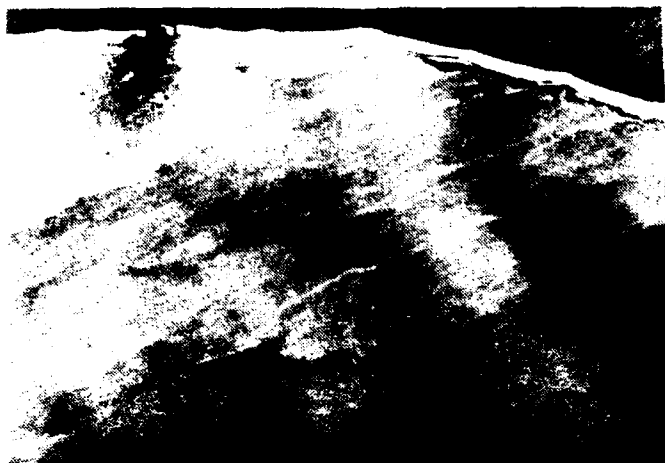
The dewatering process for the recovery and reuse of bottom sediments and water blooms has been developed over the past several years by the Department of Chemical Engineering at the Science University of Tokyo. This paper describes how to introduce and apply the continuous vacuum precoated filter and how this filter relates to environmental conservation of lakes and ports.

The experimental results with fundamental apparatus and pilot plants are described, as well as aspects of the bottom sediments and water blooms used in both wet and dry forms.

INTRODUCTION

Water blooms, called AOKO in Japan, have increased in nutritious lakes and ports such as Lake Kasumigaura and a pond, Teganuma, as shown in Figure 1. Water blooms tend to accumulate locally as water temperature rises (Figure 2), such as in the height of summer. Ordinarily these water blooms are dominated by *Microcystis aeruginosa*, which are smaller than several microns, as shown in Figures 3 and 4. Figure 5 shows the present treatment process of water blooms used by the city of Tsuchiura and the Ministry of Construction. However, the number of water blooms that are treated is much smaller than the total amount of *M. aeruginosa* that can be found during the height of summer. After several





a. Aug 1984



b. Aug 1985

Figure 1. Proliferation of water blooms

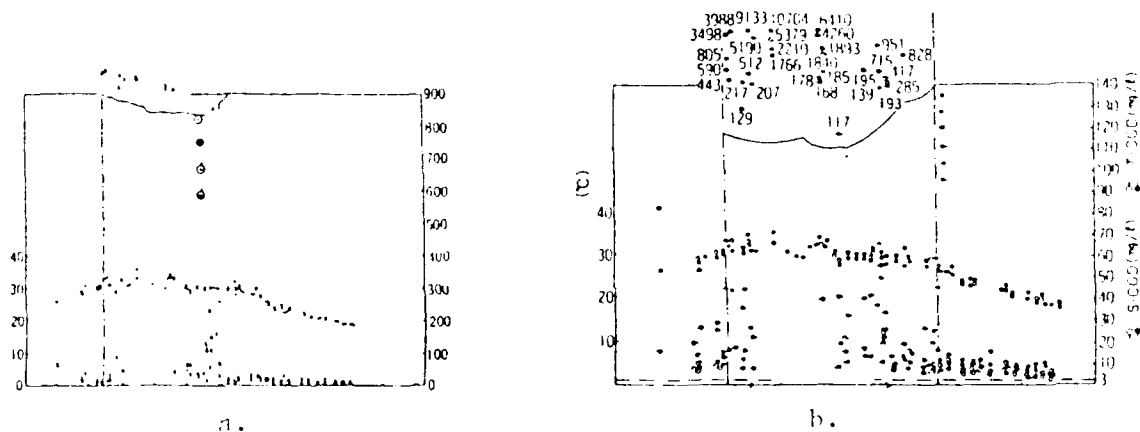


Figure 2. Progress of proliferation

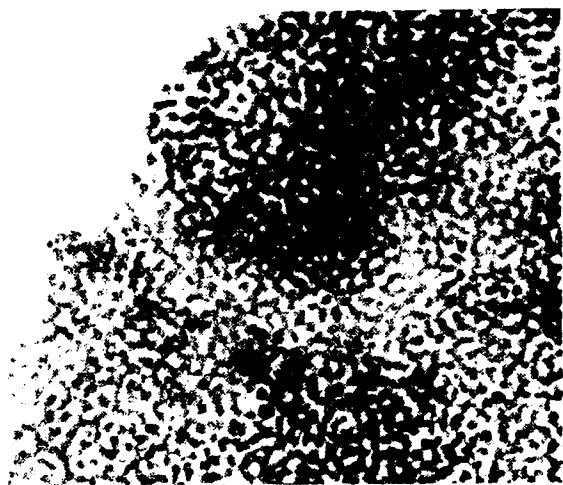


Figure 3. Water sample (unfiltered)

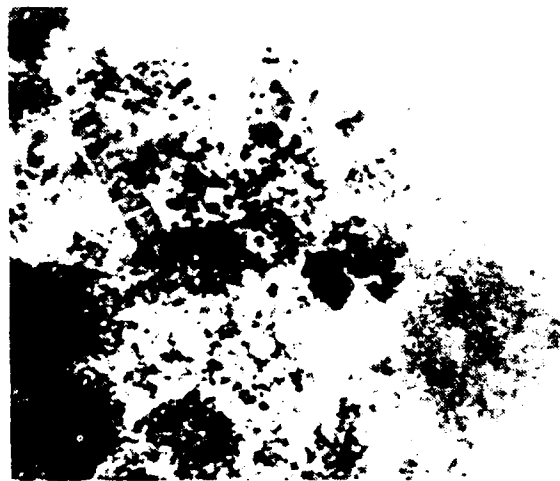


Figure 4. Water sample (unfiltered & decanted)

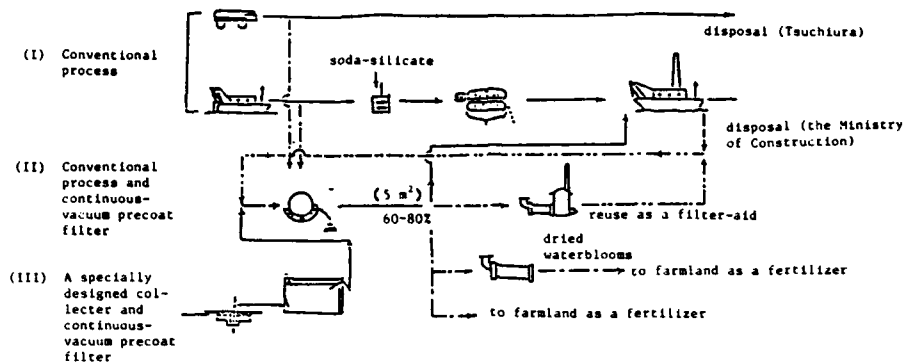


Figure 5. Dewatering and reuse process of water blooms

days, the untreated water blooms rot and begin to smell (Figure 1). Therefore, the development of a process for dewatering and beneficial use of water blooms is an urgent problem in Japan.

Theoretical and experimental methods of dewatering and reuse of water blooms have been investigated for several years in our laboratories and in Lake Kasumigaura. These methods are discussed herein.

FUNDAMENTAL CONSIDERATIONS

As mentioned above, water blooms are dominated by *M. aeruginosa*, which are smaller than several microns (Figure 2) and troublesome to dispose of, as discussed below. In general, the dewatering characteristics of filtrate capacity of suspensions may be expressed by Equation 1.

For a gravitational dewatering process such as a sand filter:*

$$\begin{aligned}
 u &= (1/A)dV/dt = \epsilon^3/k'(1 - \epsilon)^2(1/S)^2\rho g\Delta h/L_s \\
 &= \epsilon^3/k'(1 - \epsilon)^4(x/\phi)^2\rho g\Delta h/L_s
 \end{aligned} \tag{1}$$

For a mechanical dewatering process such as pressure filter or vacuum filters:

$$\begin{aligned}
 u &= (1/A)dV/dt = \epsilon^3/k'(1 - \epsilon)^2(1/S)^2\Delta p \cdot g_c/\mu \cdot L_s \\
 &= \epsilon^3/k'(1 - \epsilon)^4(x/\phi)^2\Delta p \cdot g_c/\mu \cdot L_s
 \end{aligned} \tag{2}$$

$$t/V = (1/K)V + (2/K)V_m \tag{3}$$

* See notation at end of paper for explanation of variables.

where

$$K = 2A^2 \Delta p g_c (1 - mw) / \alpha w \mu \rho \quad (4)$$

$$\begin{aligned} \alpha &= 2A^2 \Delta p g (1 - mw) / K w \mu \rho \\ &= (1 - \epsilon)^3 / \epsilon^3 (1 / \rho_s) k (\phi/x)^2 \end{aligned}$$

$$V_m = AR_m (1 - mw) / \alpha w \rho \quad (5)$$

From Equation 3, the following relation can be derived:

$$V/t + v_m/t = \left[K(1/t) + (v_m/t)^2 \right]^{1/2} \quad (6)$$

Then, as the filtration time t increases, filtration rate V/t decreases in proportion to $t^{-1/2}$ (Yoshino 1984).

Figure 6 compares the solid-liquid separation capacity of conventional dewatering processes. The continuous vacuum drum precoat filter (hatched area in Figure 6) is the most effective dewatering technique of the conventional processes (Yoshino 1984).

EXPERIMENTAL APPARATUS AND PROCEDURES

Fundamental experimental tests on the thickening and filtration characteristics of various water blooms from ponds (Shinobazu and Teganuma) and Lake Kasumigaura were conducted in the laboratory at the Science University of Tokyo, supported by the Ministry of Education, Japan. These tests employed a batch testing apparatus using a pressure float (Figure 7), a vacuum test leaf, and small-scale continuous vacuum filters (Yoshino 1984).

Following the laboratory investigations, field experiments were conducted at Lake Kasumigaura with the continuous vacuum precoat filters (surface area of 2 and 5 m²) (Figure 8) from July 1983 to October 1986.

Figure 8 outlines the experimental plant used at Lake Kasumigaura: a water bloom collector (Figure 9) to pump the water blooms out of the water (2), a continuous pressure float to concentrate the dilute suspension of water blooms (Figure 10), the continuous vacuum precoat filters as dewaterers (Figures 11 and 12), and a rotary dryer (Figure 13). The collected water blooms are pumped to the pressure float (3) and the continuous vacuum precoat filters ((6), (7) in Figure 8).

The thin cake formed on the filter (e.g. diatomite or perlite for the filter) is removed by an advancing knife while the filtrate is pumped back into the lake. The filter cakes are dried naturally to return to the farm. The cakes are sent to the rotary dryers ((8) and (9) in Figure 8) and dried down to grain particles, which are reused as organic fertilizer.

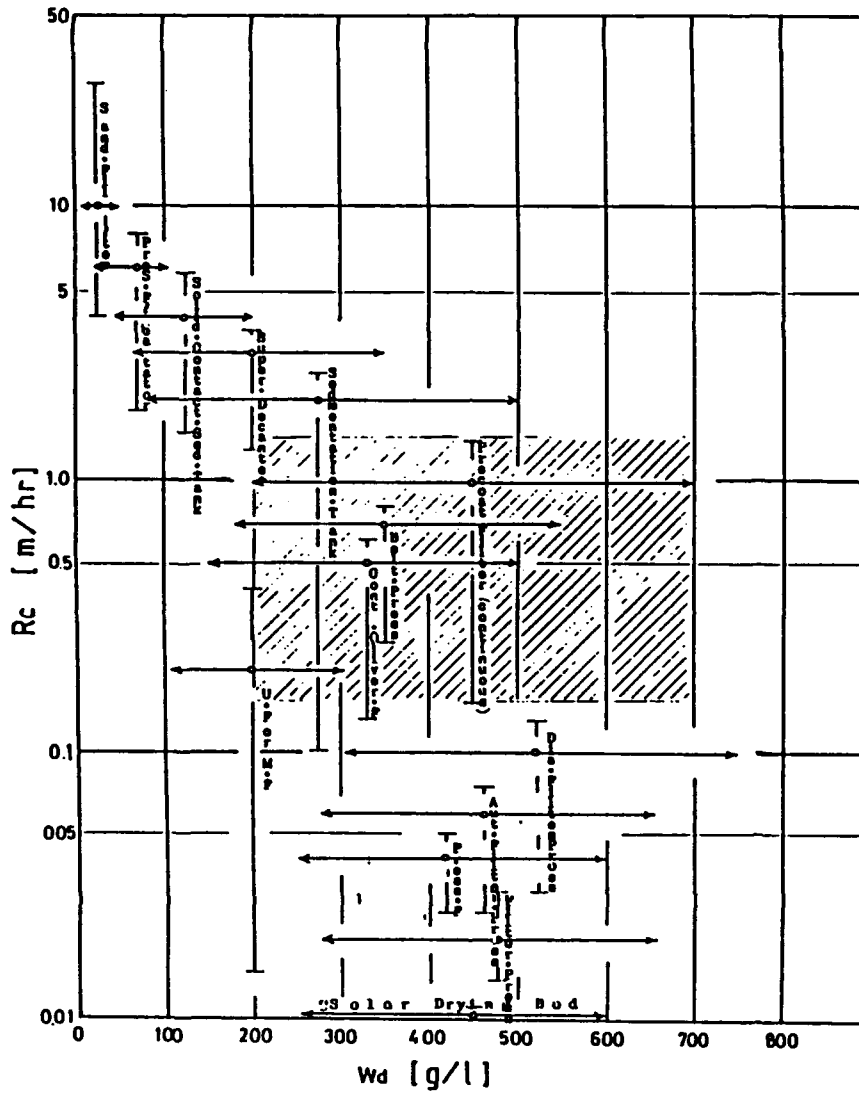


Figure 6. Solid-liquid separation process

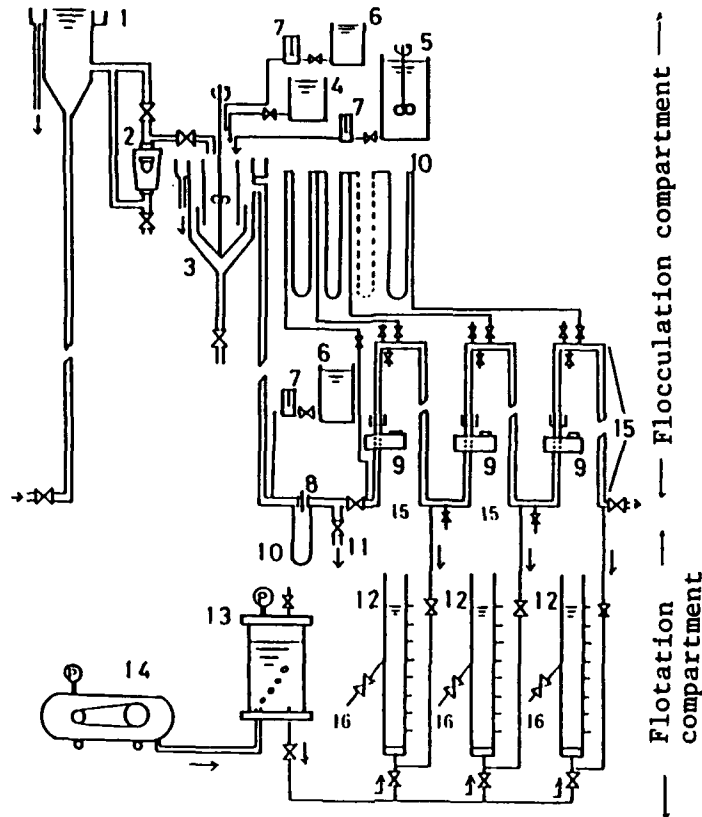


Figure 7. Experimental apparatus of flocculation and pressure flotation

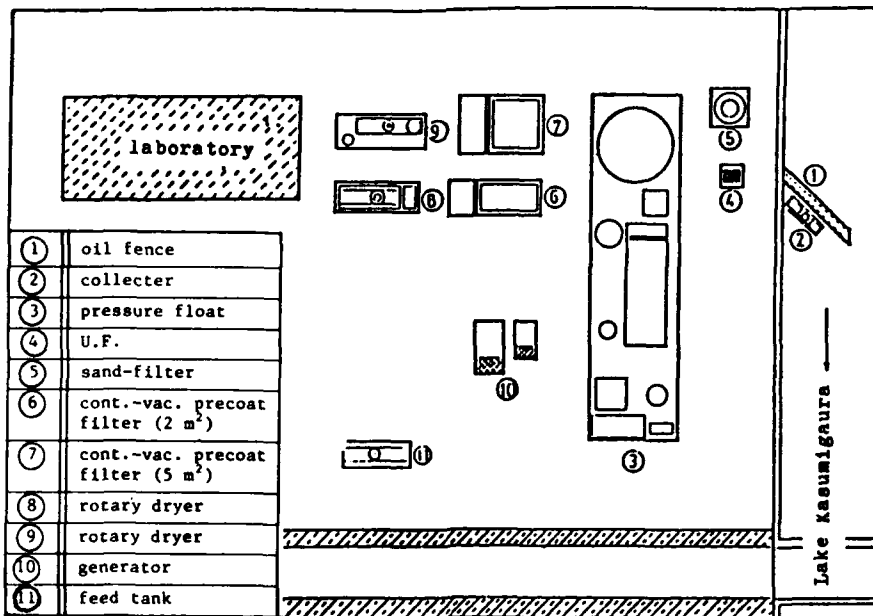


Figure 8. Experimental plant at Lake Kasumigaura

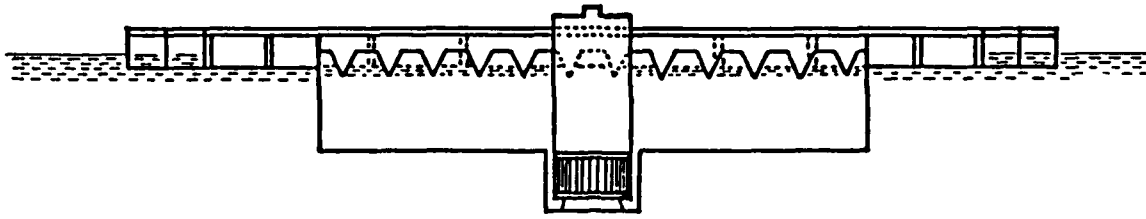


Figure 9. Water bloom collector

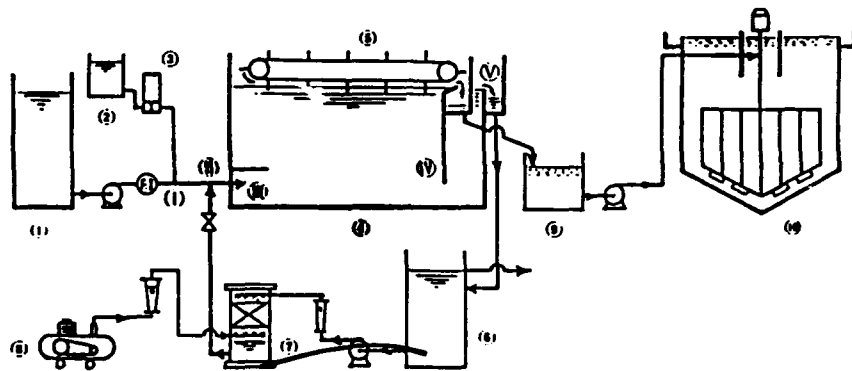


Figure 10. Continuous pressure float

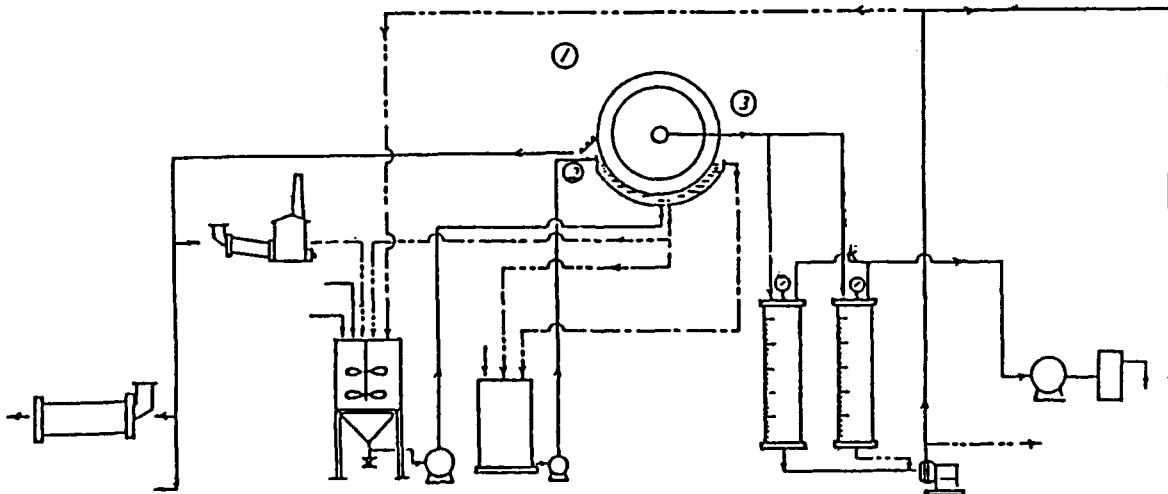


Figure 11. Continuous vacuum precoat filter (2 m²)



Figure 12. Continuous vacuum precoat filter (5 m²)



Figure 13. Rotary dryer and dried water blooms

EXPERIMENTAL RESULTS AND CONSIDERATIONS

Dewatering Characteristics

Figure 14 indicates the relation between filtration pressure Δp and specific resistance α of the water blooms measured by the leaf test. It was found that water blooms are very difficult to dewater by conventional dewatering processes such as filter press and belt press because of the extremely high specific resistance and excessive compressibility (Figures 6 and 14).

Recovery of Water Blooms

The experiments confirmed that water blooms can be easily collected by the specially designed collector shown by Figures 9 and 15.

Filter Capacity

Figure 16 indicates the filtrate capacity of the continuous vacuum precoat filter facility. It was found that even troublesome water blooms could be efficiently dewatered. Filtrate capacity was approximately $0.8 \text{ m}^3/\text{m}^2/\text{hr}$ for the water blooms of $4 \text{ g}/\ell$ solid content, and $0.3 \text{ m}^3/\text{m}^2/\text{hr}$ for the waterblooms of $25 \text{ g}/\ell$ solid content.

Since the slope of the dotted line expressing experimental results is $-1/2$, the fundamental equation (7) is applicable to these dewatering phenomena.

Clarity of Filtrate

Filtrate discharged from the continuous vacuum precoat filter was clear enough (less than $20 \text{ mg}/\ell$ suspended solids) to be pumped back into the lake and/or reused as process water.

Moisture Content of Filter Cake

The moisture content of the cake removed from the continuous vacuum filter was less than 78 percent, and therefore eligible for disposal or reuse as a fertilizer without any difficulty.

In addition, since the cake was very thin (less than 1.0 mm), it could be dewatered until the moisture content reached 40 percent by subsequent solar drying for 1 or 2 days (Figure 17).

Drying Characteristics of Filter Cake

Figure 18 indicates the drying characteristics of the wet cake (75-percent moisture content). It was found that dried water blooms containing a small quantity of diatomite (filter aid) were discharged from the rotary dryer as grain particles (Figure 18) in the proportion of $0.1\text{--}1$ heavy oil/ ℓ of dried products.

Figure 14 is missing.
Not provided to WES.

Figure 15 is missing.
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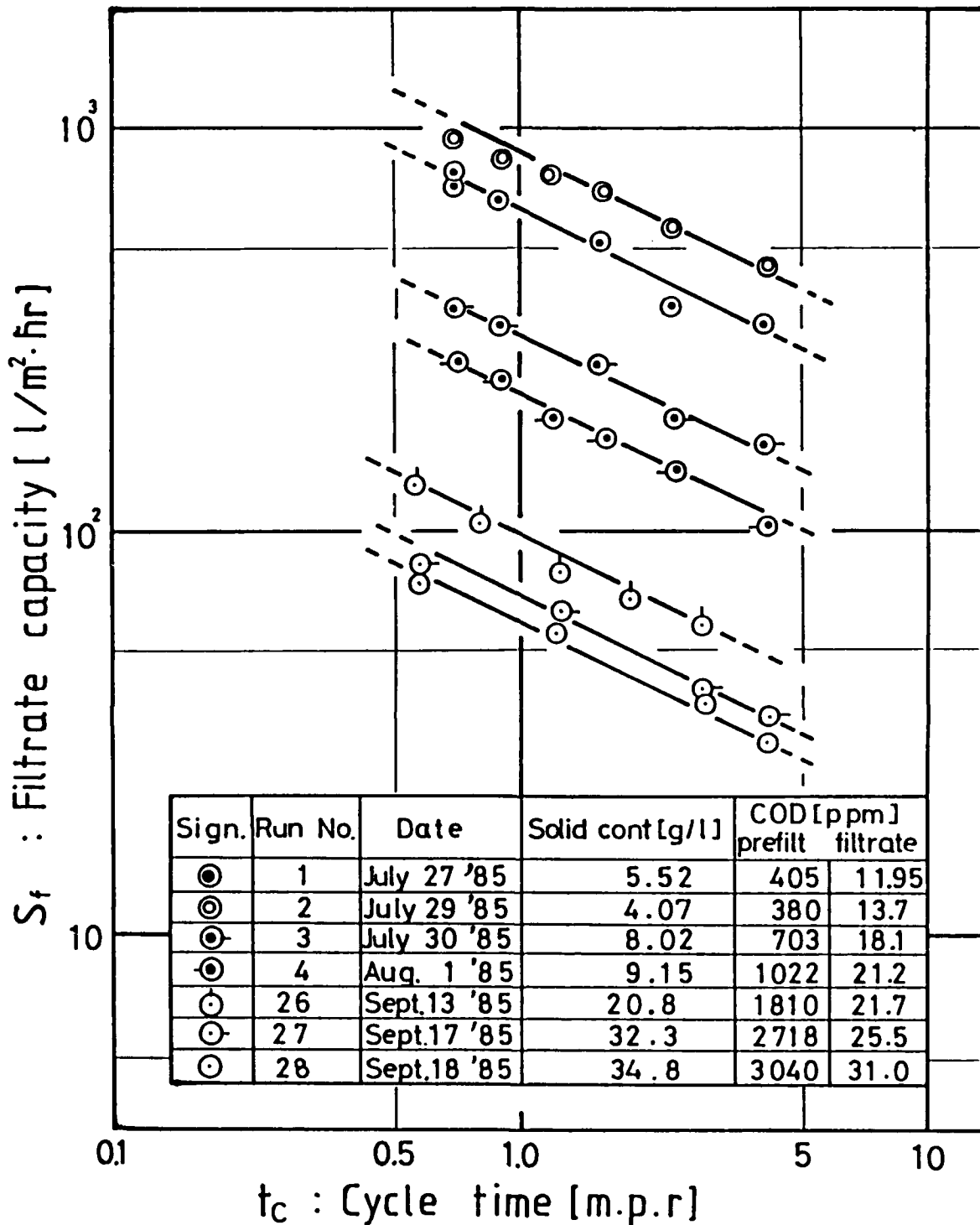


Figure 16. Filtrate capacity of continuous vacuum precoat filter for the water blooms of Lake Kasumigaura

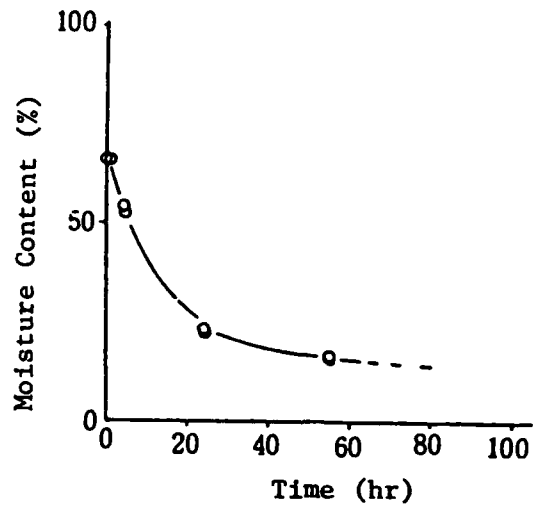


Figure 17. Dewatering curve of the filter cake by solar drying

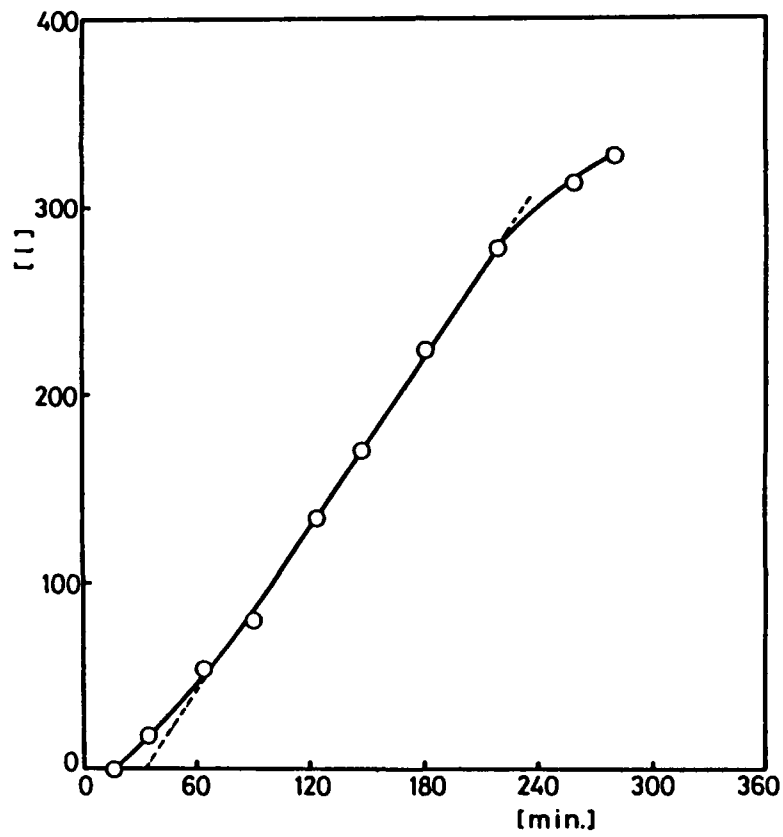


Figure 18. Dewatering the filter cake with a rotary dryer

Reuse of Dried Water Blooms

Figures 19 and 20 indicate the results of growing test vegetables such as Komatsuna using a farm soil and a mixture of farm soil with dried water blooms. The growth rate with the mixture of farm soil and dried water blooms was superior beyond comparison. Furthermore, because the dried water blooms contain more than 3 percent available nitrogen, the material can be reused as organic fertilizer.



Figure 19. Growth tests using komatsuna in farm soil

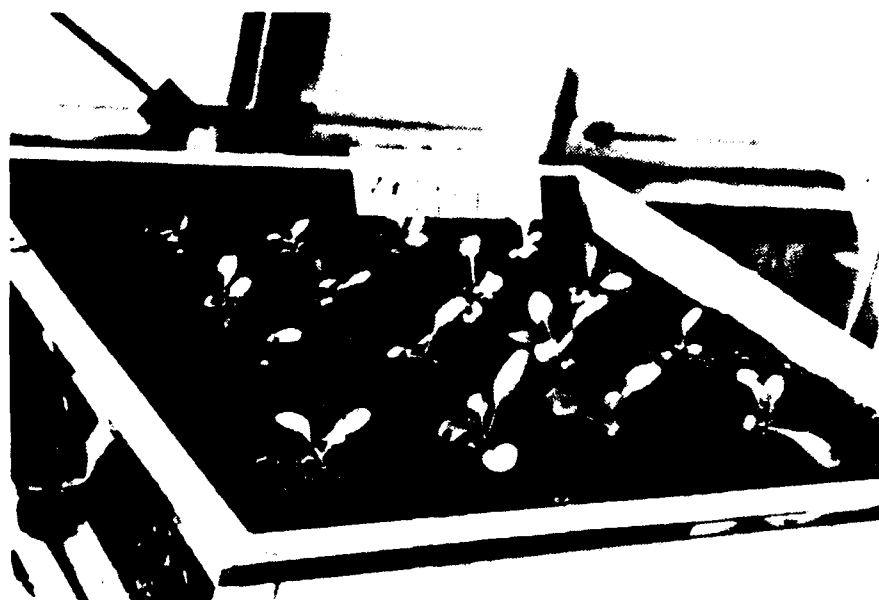


Figure 20. Growth tests using komatsuna using a mixture of farm soil and dried water blooms

Recycle of Filter Aid

Because the dried water blooms retain more than 2,000 kcal, they can be burned without additional fuel (Figure 21). The ashes remaining after combustion can be reused as a filter-aid because the specific resistance α was nearly equal to the specific resistance of a fresh filter aid as shown by Figures 22 and 23.

Figure 21. Combustion of dried water blooms (without fuel)

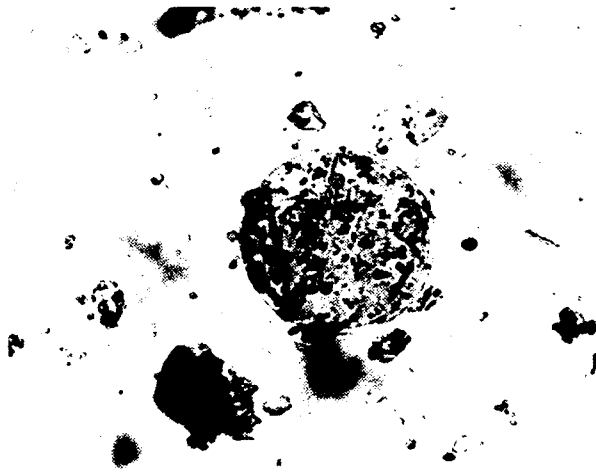
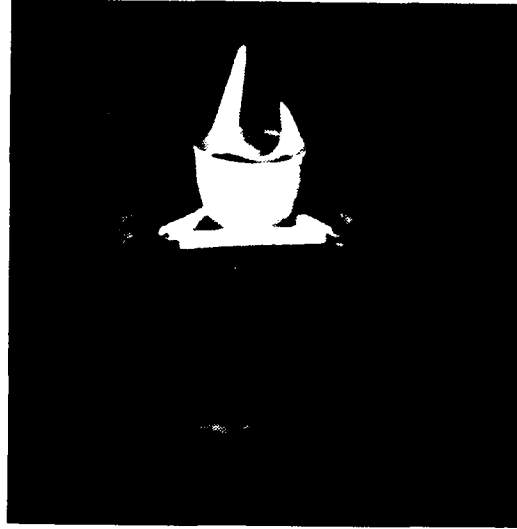


Figure 22. Micrograph of filter aid (diatomite)

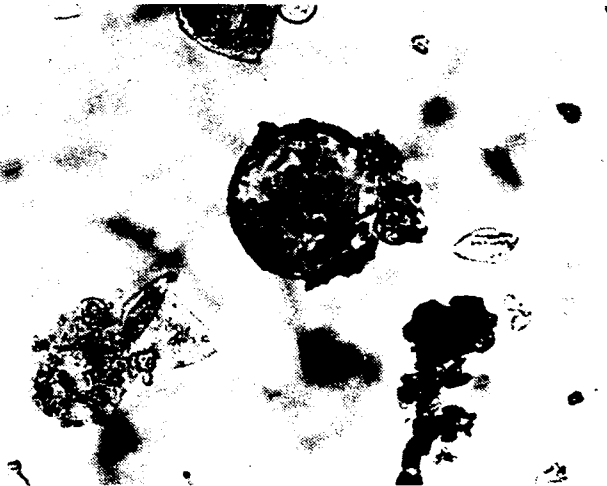


Figure 23. Micrograph of filter aid (ashes after combustion, dried water blooms)

CONCLUSIONS

The following conclusions can be drawn from the experimental results:

- a. From the properties and dewatering characteristics of water blooms, it was determined that water blooms are very difficult to dewater by conventional methods because of the extremely high specific resistance and the excessive compressibility.
- b. By using the specially designed collector, water blooms in the lake or ports can be easily collected.
- c. The comparative studies performed on a dewatering process found the continuous vacuum precoat filter to be the most effective.
- d. The field experimental investigations at Lake Kasumigaura found the following:
 - (1) Troublesome suspensions such as water blooms and bottom sediments can be efficiently dewatered with the continuous vacuum precoat filter.
 - (2) The filtrate capacity can be expressed by the following equation:
$$S = V/(A \cdot t) \approx K \cdot w^{-1/2} \cdot t^{-1/2}$$
 - (3) Filtrate discharged from the continuous vacuum precoat filter contains suspended solids less than 20 mg/l, clear enough for pumping back into the lake or reusing as process water.
 - (4) Moisture content of the cake removed from the continuous vacuum precoat filter is less than 78 percent, and therefore is eligible for reuse as fertilizer or recycling as filter aid.
 - (5) Filter cakes can be further dried by using a rotary dryer with a very low consumption of fuel such as 0.1- ℓ heavy oil/ ℓ of dried water blooms.
- e. Growth of vegetables using the dried water blooms revealed an extremely rapid growth rate as compared with farm soil only.
- f. The combustion tests of the dried water blooms that were carried out because the blooms had more than 2,000 kcal found that spontaneous combustion continued without any fuel, and the ashes after combustion could be reused as a filter aid to precoat.

The continuous vacuum precoat filter may be the best method available for the treatment and reuse of bottom sediments and water blooms, from both a technical and an environmental viewpoint.

ACKNOWLEDGEMENTS

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NOTATION

A	Filtration area	cm^2 or m^2
g	Acceleration due to gravity	cm/sec^2
g_c	Conversion factor	$\text{kg}\cdot\text{m}/\text{kg}\cdot\text{sec}^2$
k	Empirical constant	--
K	Filtration constant	m^2/sec
L_s	Depth of filter medium	cm or m
m	Weight ratio of dry cake to wet cake	--
S	Specific surface area	cm^2/cm^3
S_f	Filtrate capacity	$\ell/(\text{m}^2\cdot\text{hr})$
t	Filtration time	sec or hr
t_c	Filter cycle time	min or hr
u	Filtration velocity	$\text{cm}^3/\text{cm}^2/\text{sec}$
V	Volume of filtrate	cm^3 or m^3
V_m	Imaginary filtrate volume corresponding to the filter medium	cm^3 or m^3
w	Solid content	g/ℓ or kg/m^3
x	Particle size	μm or cm
α	Specific cake resistance	cm/g or m/kg
Δh	Head	cm or m
Δp	Filtration pressure	kg/cm^2
ϵ	Porosity	--
μ	Viscosity	$\text{g}/\text{cm}/\text{sec}$
ρ	Density	g/cm^3
ϕ	Shape factor	--

LAKE SUWA IS BEING RESTORED

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AD-P006 898



ABSTRACT

Lake Suwa is the only lake in our country in which sediment removal has been conducted consecutively for its restoration. Dredging commenced in 1969 and is still continuing. The total amount of sediment dredged to date is about 1.6 million m^3 (16 years), and the scheduled volume remaining to be dredged is about 5 million m^3 .

Although a considerable amount of material has been dredged, doubt still exists on whether the sediment removal will result in a restorative effect or not. Sediment removal, as it affects lake restoration, is our greatest concern. This paper describes our study of these effects.

HISTORIC BACKGROUND OF POLLUTION

Chemical oxygen demand (COD) concentrations at Lake Suwa's center change from year to year (Figure 1). Prior to 1960, COD concentrations were about 2 mg/l, making the lake oligotrophic. After 1960 the concentration increased rapidly and reached maximum values of 9 to 10 mg/l in 1977 and 1978, and then

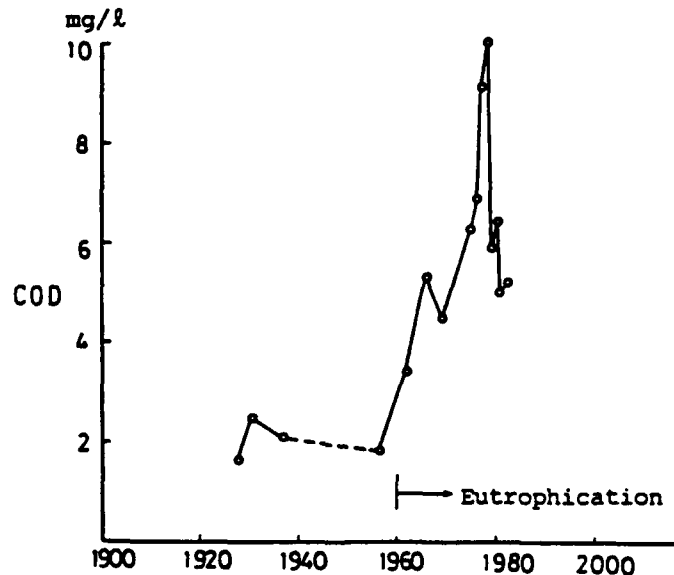


Figure 1. Yearly variation of COD

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decreased to its present state, about half the maximum value. From this, we can see that the eutrophication began in 1960 and is an occurrence which spanned over the last 20 to 25 years.

However, COD data obtained over the last 10 years from Lake Suwa has indicated a trend toward restoration (Figure 2). A comparison between the COD

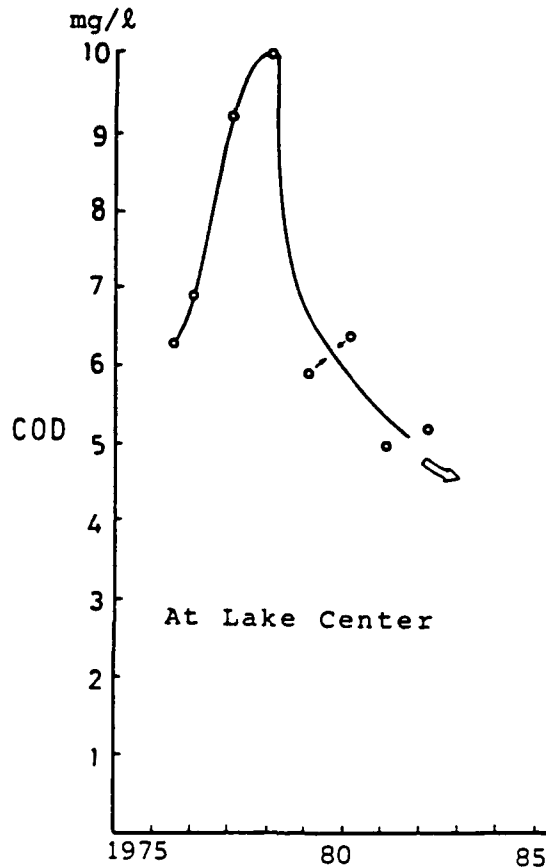


Figure 2. Variation of COD over the last 10 years

concentration of Lake Suwa and Lake Teganuma, where there is no restoration measure being applied (Figure 3), reveals that COD concentrations at the center of Lake Teganuma are increasing, while the COD concentrations at the center of Lake Suwa are decreasing. This seems to indicate that sediment removal is having a positive restorative effect on Lake Suwa.

DREDGING

Dredging, approved by the government as a lake restoration technique, began in 1969. The first stage of the Lake Suwa restoration program involved dredging that portion of the littoral area less than 2.5 m in depth (Figure 4).

The portion of the littoral zone that was dredged had a total area of 2.5 million m^2 and occupied 24 percent of the Lake Suwa total bottom area of 10.4 million m^2 . The total quantity of dredged material removed from Lake Suwa from 1969 to 1981 was approximately 1.4 million m^3 .

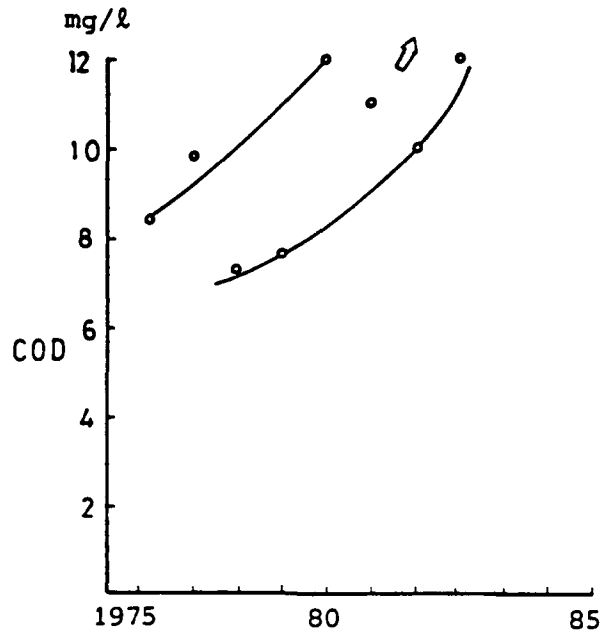


Figure 3. Variation of COD in Lake Teganuma over the last 10 years

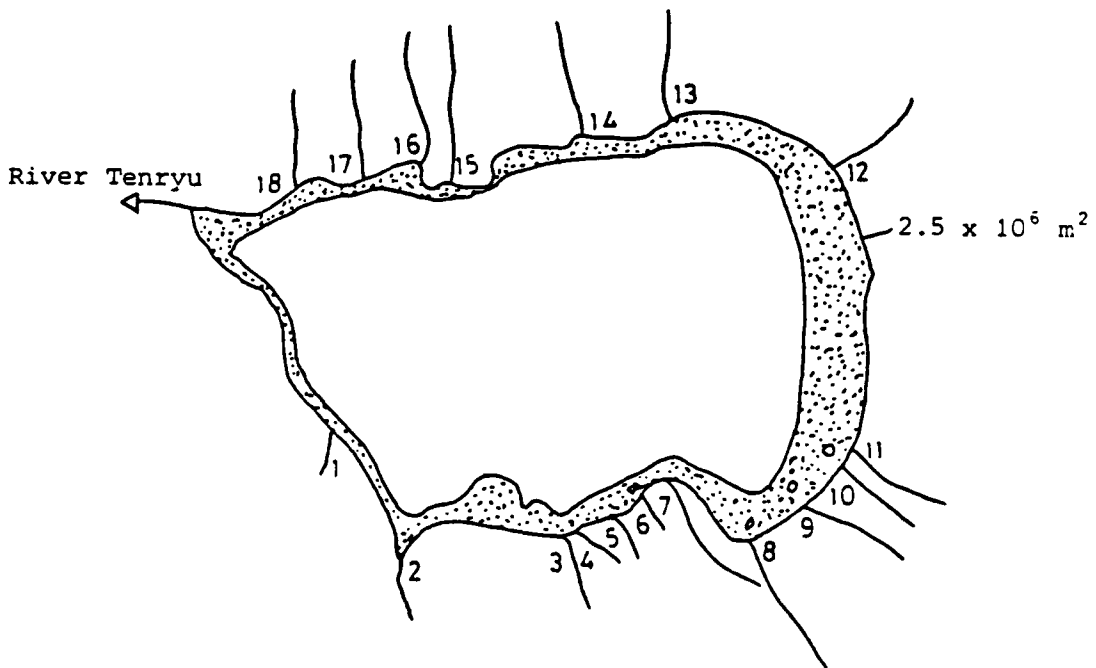


Figure 4. Dredged area of the first stage

The fact that the dredging began in the shallow area allows room for criticism. As Figures 5 and 6 show, the sediment in the shallow areas is weakly polluted, whereas the deeper areas are strongly polluted.

The dredging should have started in the deeper areas with the highly polluted sediment; this might have brought about the beneficial effect of sediment removal more rapidly than did the dredging of shallow areas.

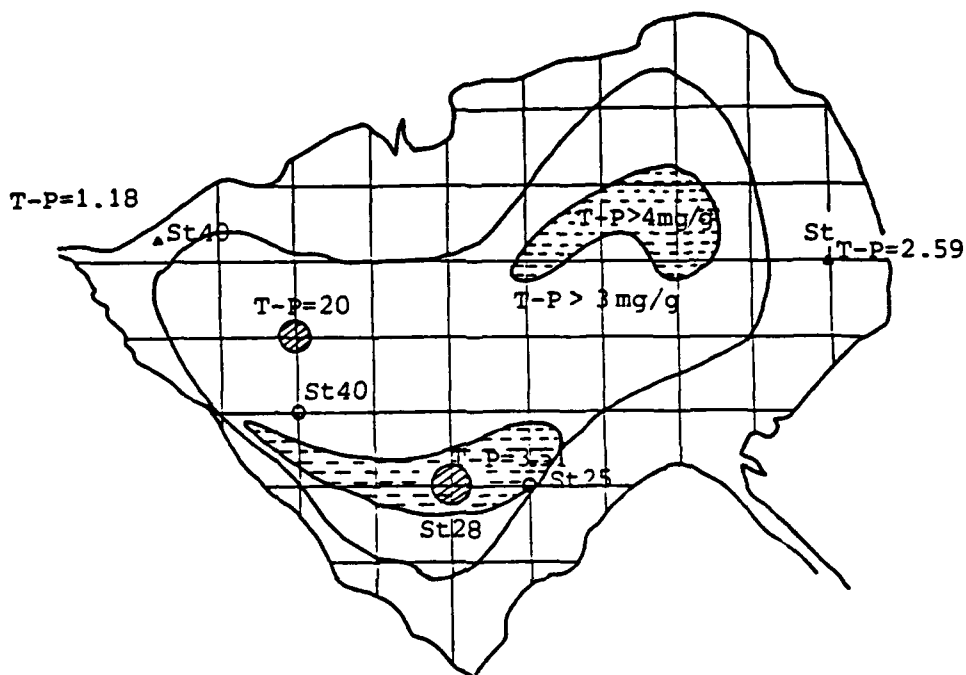


Figure 5. Plane distribution of total phosphorus (T-P) in sediment

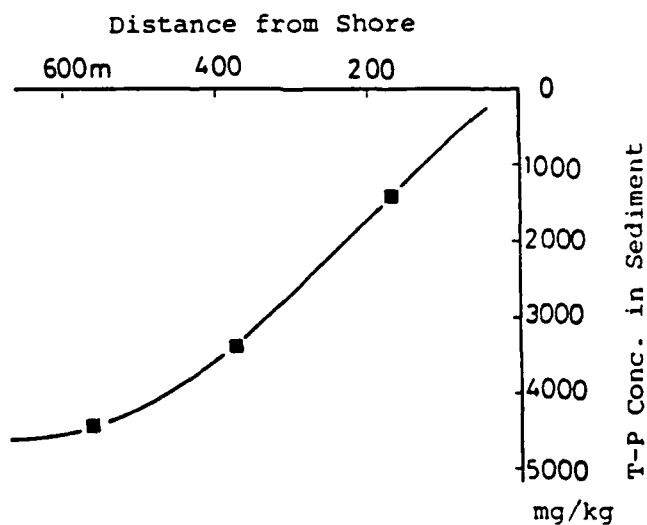


Figure 6. Pollution levels

Therefore, for the second stage (dates,), it was decided to begin dredging in the most highly polluted section of the remaining area. If the remaining area is divided into four blocks (Figure 7), the first block is the worst, with mean concentrations of 7,206 mg/kg T-P and 5,611 mg/kg total nitrogen (T-N).

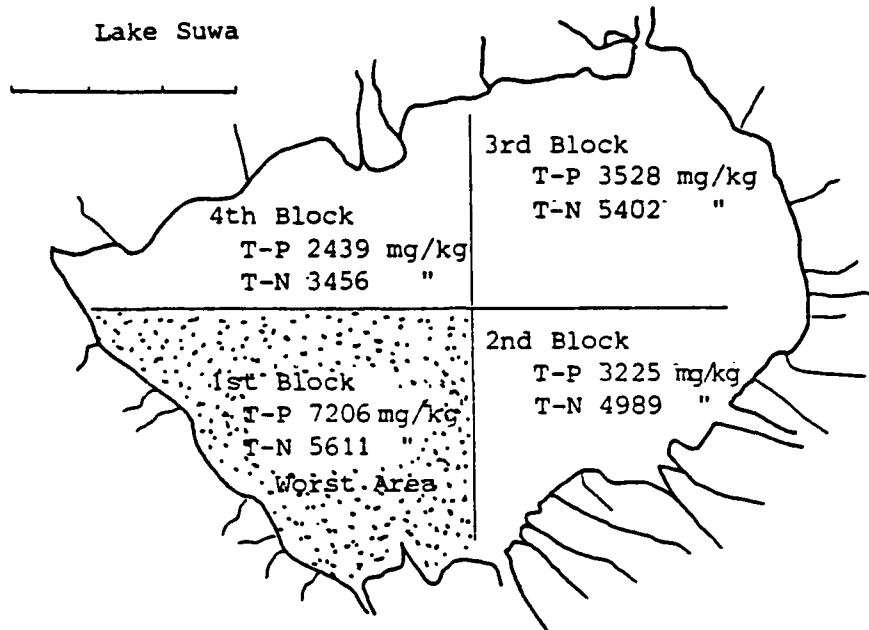


Figure 7. Division of lake area

CHANGE OF SUMMER PEAK

A summer peak is a thermal occurrence in a nonstratified lake caused by increased material released from sediment resulting from the higher water temperatures in summer. Consequently, the summer peak can be illustrated as a diagram having a parameter of water temperatures (Figure 11).

Figure 8 shows the summer peak concentration of T-P for 1973 at the center of Lake Suwa. The T-P concentration increased from 100 $\mu\text{g}/\ell$ in March to a peak concentration of 462 $\mu\text{g}/\ell$ in August.

As Figure 9 shows, the summer peak in 1982 was much smaller, and the basic concentration also dropped. The summer peak was 170 $\mu\text{g}/\ell$ (September), the basic concentration was 65 $\mu\text{g}/\ell$, and the peak ratio was 2.62.

The irregular valley in August is presumably the result of a cool summer and an abnormally large rainfall. A comparison of the two peaks in 1973 and 1982 shows a much improved lake water quality. To verify the irregularity of 1982, the summer peak of 1984 is indicated in Figure 10. The same pattern is evident except for water temperature differences.

Figure 11 clearly shows the significant improvement of water quality from the viewpoint of summer peak.

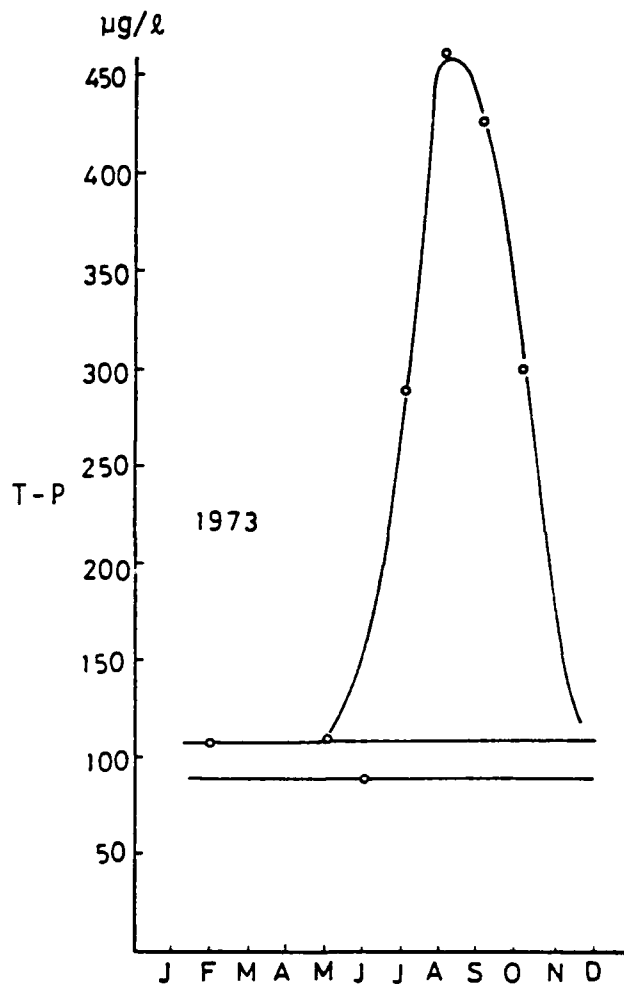


Figure 8. T-P summer peak in 1973

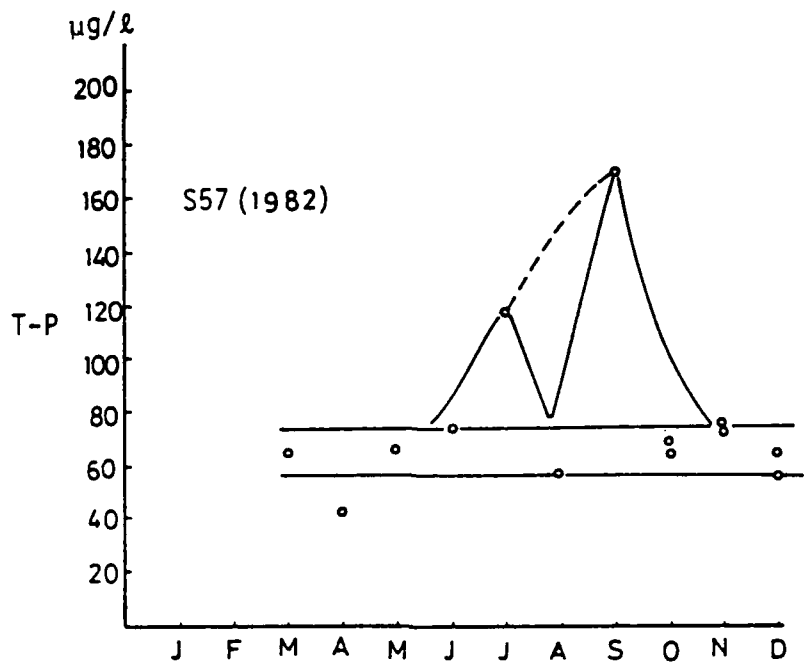


Figure 9. T-P summer peak in 1982

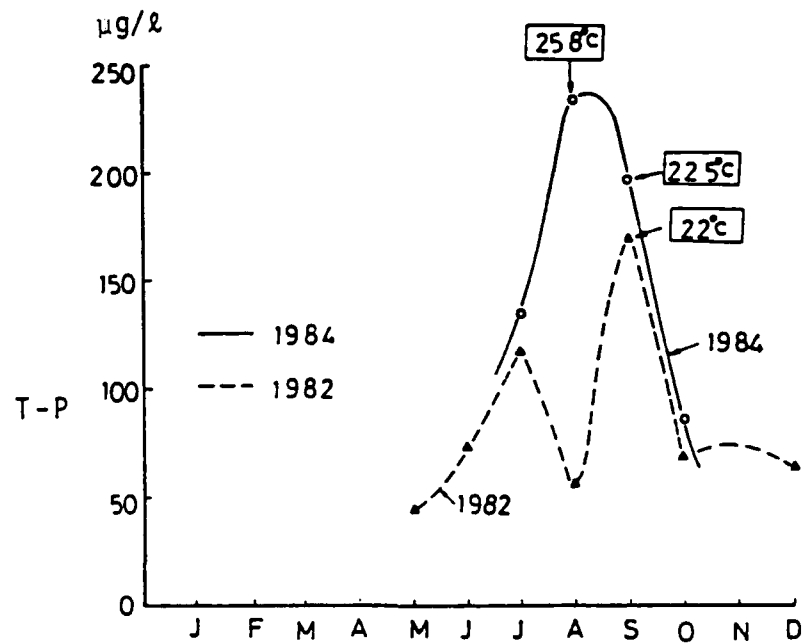


Figure 10. T-P summer peaks in 1982 and 1984

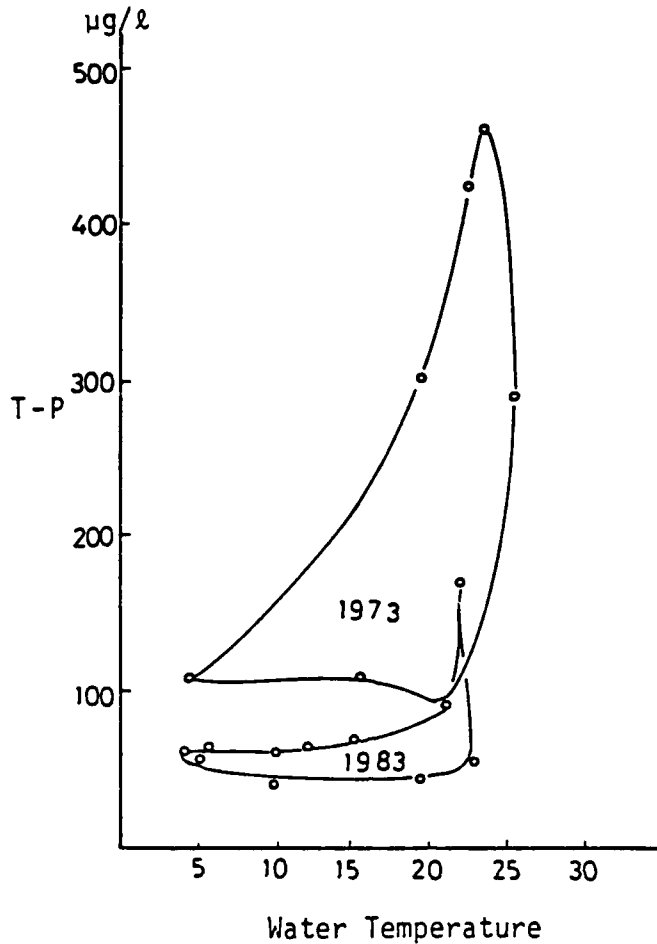


Figure 11. Summer peak versus water temperature

VARIATION IN PEAK CONCENTRATIONS

As mentioned before, in any polluted lake a phenomenon of summer peak occurs. Therefore, to restore a lake is to diminish its summer peak. In Figure 12, the peak concentrations are reviewed over the last 15 years.

It is worthwhile to notice that peak concentrations dropped considerably after 1979. This means that the effects of sediment removal became apparent at that time.

The quantity of sediment dredged until 1979 was 1.3 million m³, which corresponds to 25 percent of the total amount of sediment to be dredged. This suggests that the effect of dredging does not appear immediately, but begins to manifest itself after considerable quantities of sediment are removed. At Lake Suwa, the realization of the effect occurred at a dredged percentage of 25 percent and 10 years (Figure 13).

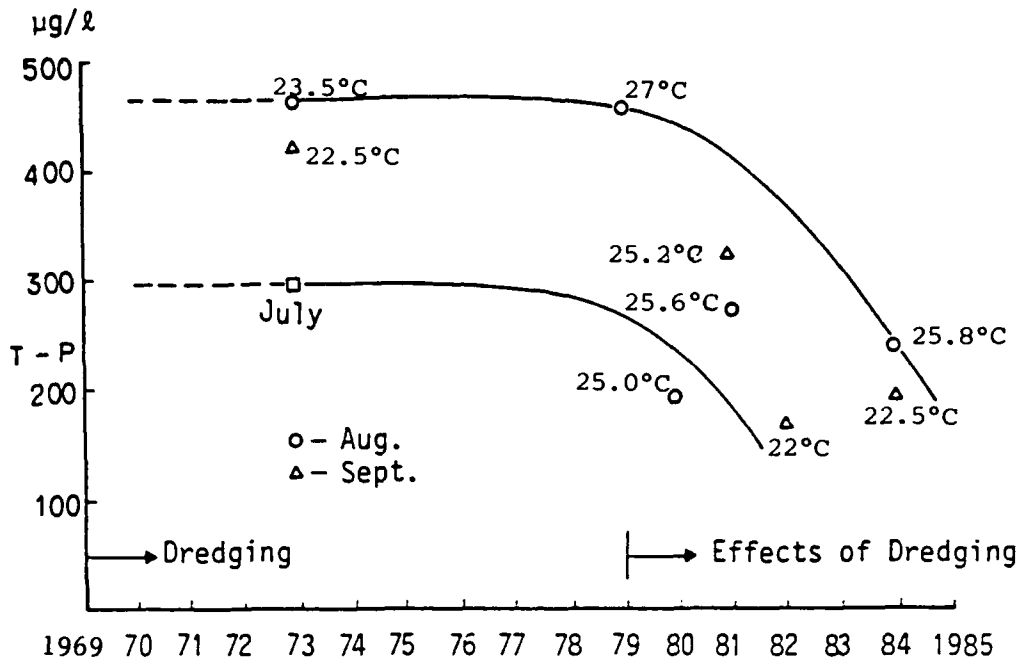


Figure 12. Yearly variation in peak concentrations

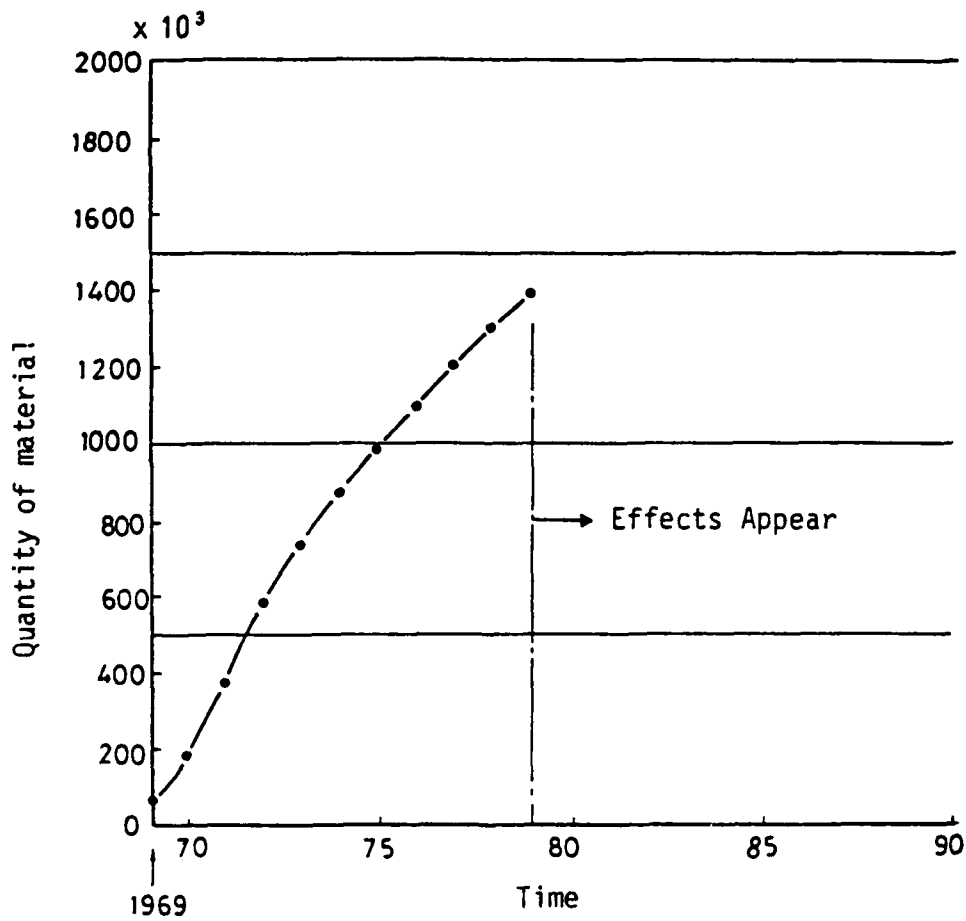


Figure 13. Quantities of dredged material

VARIATION OF CHLOROPHYLL a CONCENTRATIONS

In the worst year, 1973, the maximum chlorophyll a concentration was 300 mg/m³ (Figure 14). At that time there was an enormous crop of green algae.

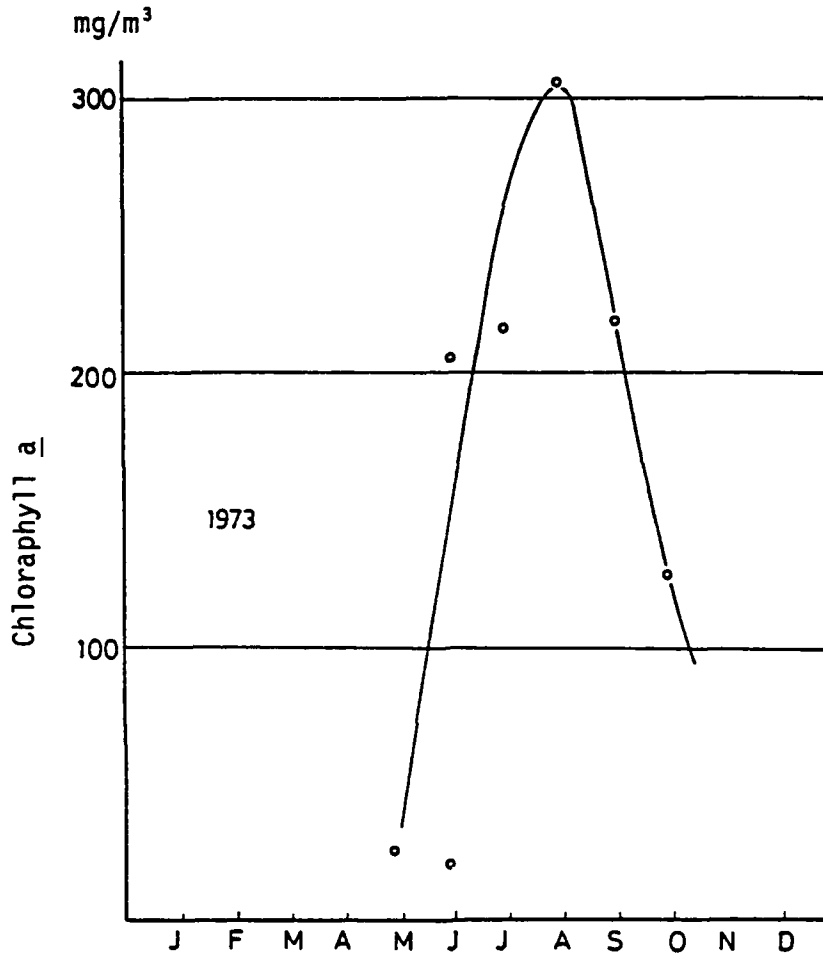


Figure 14. Summer peak of chlorophyll a

As Figure 15 shows, the peak concentrations of chlorophyll a in 1982 and 1984 were both about 100 mg/m³. This is a significant decrease in chlorophyll a concentrations. From a biological point of view, it is clear that the lake is being restored.

RELATIONSHIP BETWEEN CHLOROPHYLL a AND COD CONCENTRATIONS

The relationships between chlorophyll a and COD concentrations in Lake Suwa are indicated in Figure 16.

As Figure 16 shows, the relationship curves are linear with a high correlation. It is interesting that the curves are changing from a steep slope to a gentle one, in other words, rotating in a clockwise direction. The transition of these curves seems to provide evidence for the progress of lake restoration (Figure 17).

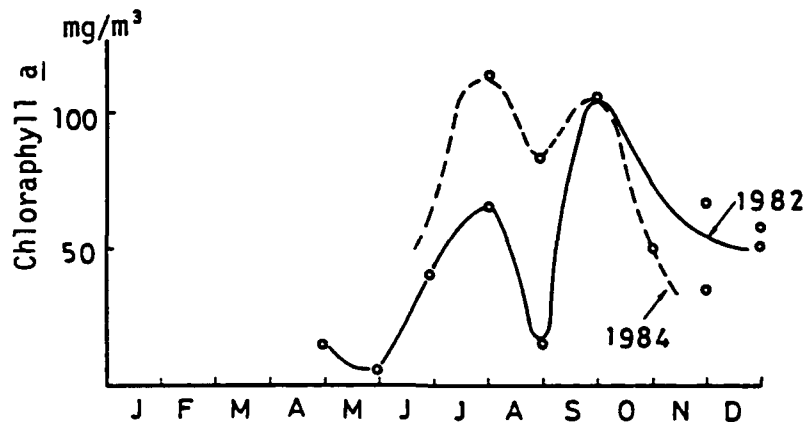


Figure 15. Variation of chlorophyll a concentrations

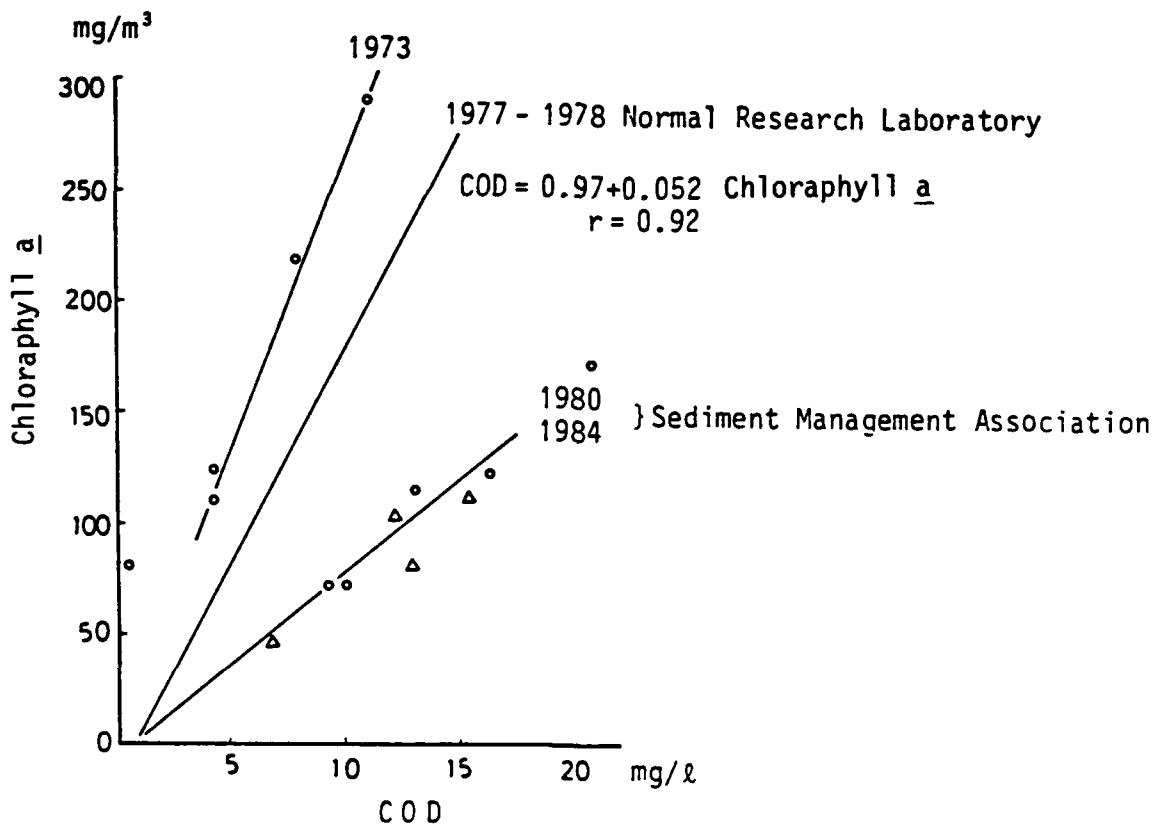
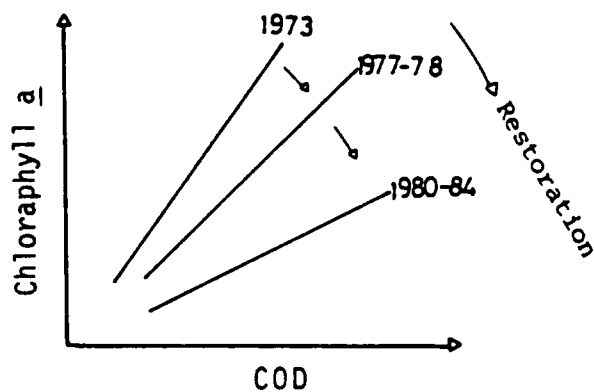
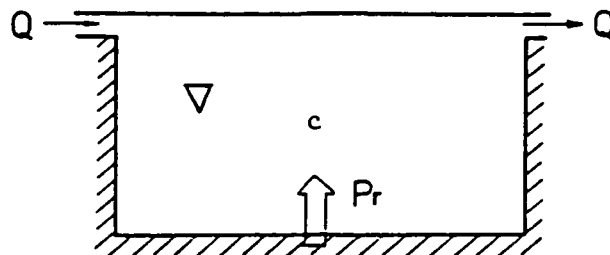


Figure 16. Relationship between chlorophyll a and COD concentrations

Figure 17. Transition of chlorophyll a and COD curves



MATHEMATICAL APPROACH TO SUMMER PEAK



The change in lake water concentrations is represented as follows:

$$V \frac{\partial c}{\partial t} = Pr(t)A - cQ \quad (1)$$

where

V = water volume of lake

c = lake water concentration (for example, T-P)

t =

Pr = release rates of nutrients (in this case, T-P)

A = bottom area of lake

Q = inflowing quantities

In this case, we assume that the influx loading is steady and does not contribute to the change in lake water concentrations against time.

Using $\frac{Q}{V} = F$, $\frac{A}{V} = H$, Equation 1 can be rewritten as follows:

$$\frac{\partial c}{\partial t} + \frac{Q}{V} c = Pr \frac{A}{V} = \frac{Pr}{H}$$

$$\frac{\partial c}{\partial t} + \epsilon c = \frac{Pr}{H} \quad (2)$$

where

ϵ = reciprocal of retention time

H = mean depth of the lake

The solution of Equation 2 is obtained as follows:

$$C\epsilon^{-t} = \frac{Pr}{H} \int \epsilon^t dt + C \quad (3)$$

The seasonal change in water temperature T can be approximated to the following parabolic curve.

$$T = T_{max} - \frac{t^2}{2Pt} \quad (4)$$

$$2Pt = \frac{\tau^2}{T_{max} - T_{min}} \quad (5)$$

where

τ = time from the maximum temperature in autumn to the minimum temperature in winter

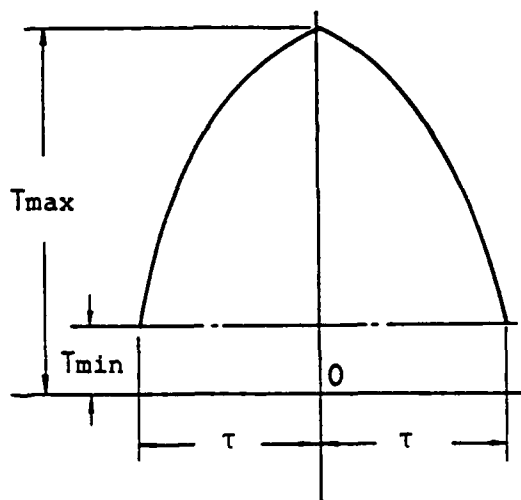


Figure 18 shows the seasonal variation in water temperature in 1973. The release rate is strongly dependent on water temperature. From our research the following equation is obtained:

$$Pr(t) = Pr(20^\circ C) a^{\epsilon} e^{T-20} \quad (6)$$

$$a^{\epsilon} = 1.11$$

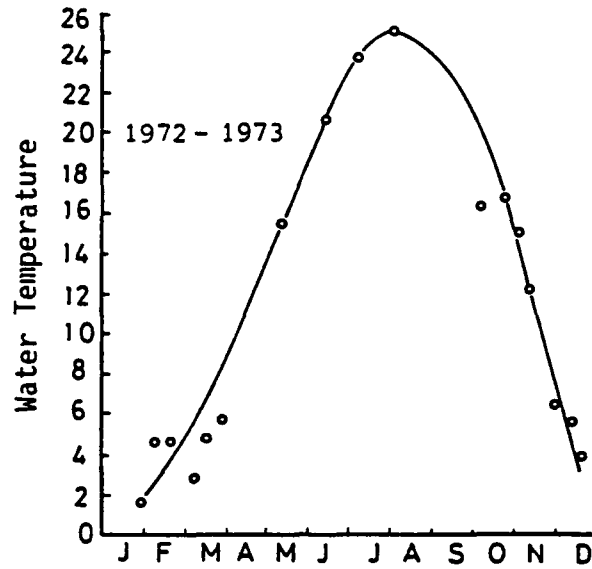


Figure 18. Water temperature versus time (months) for 1973

From the above two assumptions, Equation 3 can be rewritten as follows:

$$C_e e^{Et} = \frac{Pr(20^\circ\text{C})}{H} a e^{T_{\max}-20} a e^{-\frac{t^2}{2Pt}} e^{Et} dt \quad (7)$$

$$\begin{aligned}
 a e^{-\frac{t^2}{2Pt}} &= (1 + 0.11) e^{-\frac{t^2}{2Pt}} = 1 + (0.11) \left(-\frac{t^2}{2Pt}\right) + (0.11)^2 \frac{\left(-\frac{t^2}{2Pt}\right) \left(-\frac{t^2}{2Pt} - 1\right)}{2.1} \\
 &+ (0.11)^3 \frac{\left(-\frac{t^2}{2Pt}\right) \left(-\frac{t^2}{2Pt} - 1\right) \left(-\frac{t^2}{2Pt} - 2\right)}{3.1} = 1 - 0.05218 \frac{t^2}{Pt} \\
 &+ 0.001363 \frac{t^4}{Pt^2} - 0.00002315 \frac{t^6}{Pt^3} + \dots \quad (8)
 \end{aligned}$$

$$e^{\epsilon t} = 1 + Et + \frac{1}{2} E^2 t^2 + \frac{\epsilon^3}{6} t^3 + \frac{\epsilon^4}{24} t^4 + \dots \quad (9)$$

$$\begin{aligned}
F(t) = a \epsilon^{-\frac{t^2}{2Pt}} e^{Et} dt = t + \frac{1}{2} \epsilon t^2 + \left(-\frac{0.05218}{Pt} + \frac{\epsilon^2}{2} \right) \frac{t^3}{3} \\
+ \left(-\frac{0.05218\epsilon}{Pt} + \frac{\epsilon^3}{6} \right) \frac{t^4}{4} + \left(\frac{0.001363}{Pt^2} - \frac{0.05218}{2Pt} \epsilon^2 \right) \frac{t^5}{5} \\
+ \left(\frac{0.001363}{Pt^2} \epsilon - \frac{0.05218}{6Pt} \epsilon^3 \right) + \frac{t^6}{6} + \dots
\end{aligned} \tag{10}$$

where $F(t)$ is a function of time and the reciprocal of retention time. The solution of Equation 2 becomes as follows:

$$C = \frac{Pr(20^\circ C)}{H} a \epsilon^{T_{max}-20} F(t) e^{-\epsilon t} + C e^{-\epsilon t}$$

As a boundary condition $t = 0$, $C = C_0$, $F(0) = 0$, then $C = C_0$. Therefore, we get

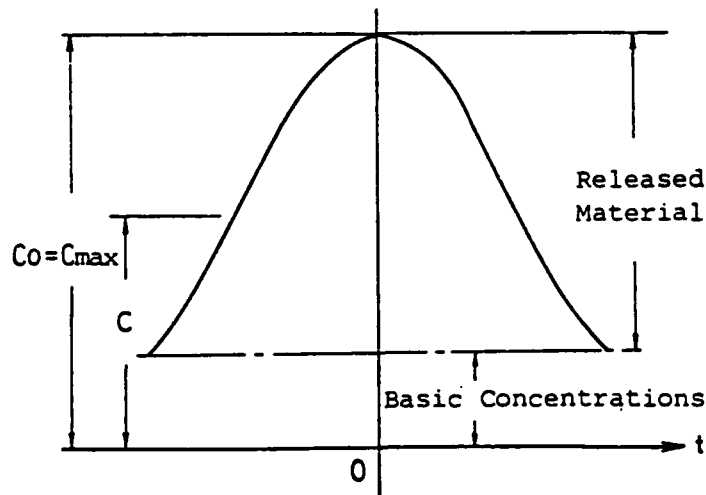
$$C = \left[\frac{Pr(20^\circ C)}{H} a \epsilon^{T_{max}-20} F(t) + C_0 \right] e^{-\epsilon t} \tag{11}$$

This is the equation of the summer peak. From this it can be seen that the summer peak is caused by the temperature dependence of release rates and retention time. Here we reform the following equation:

$$\begin{aligned}
Pr(20^\circ C) a \epsilon^{T_{max}-20} &= Pr(20^\circ C) a \epsilon^{T_{max}-20} - \frac{t^2}{2Pt} + \frac{t^2}{2Pt} \\
&= PR(20^\circ C) a \epsilon^{T-20} a \epsilon^{\frac{t^2}{2Pt}} \\
&= Pr(t) a \epsilon^{\frac{t^2}{2Pt}}
\end{aligned}$$

Then, Equation 11 can be rewritten once more as follows:

$$C = \left[\frac{Pr(t)}{H} a e^{\frac{t^2}{2Pt}} \cdot F(t) + C_0 \right] e^{-\epsilon t}$$



It is worthy of notice that the actual release rates can be calculated from the form of summer peak observed by using this equation as follows:

$$\begin{aligned} Pr(t) &= \frac{C - C_0 e^{-\epsilon t}}{\frac{t^2}{1/H a e^{\frac{t^2}{2Pt}} F(t) e^{-\epsilon t}}} \\ &= \frac{(C e^{\epsilon t} - C_0) H}{\frac{t^2}{a e^{\frac{t^2}{2Pt}} F(t)}} \end{aligned}$$

Figure 19 shows the measurement of water temperature in 1984. From this it is presumed that the maximum temperature would have been 26° C and would have occurred on 15 August, which is taken as an origin of the diagram.

$$T_{max} = 26^\circ \text{ C}$$

$$T_{min} = 4^\circ \text{ C (assumed)}$$

$$Pt = \frac{\tau^2}{2(T_{max} - T_{min})} = \frac{73^2}{2(26 - 4)} = 121.11 \text{ day}^2/\text{°C}$$

$$C_0 = C_{max} = 250 \text{ } \mu\text{g}/\ell$$

$$H = 216 \text{ m}$$

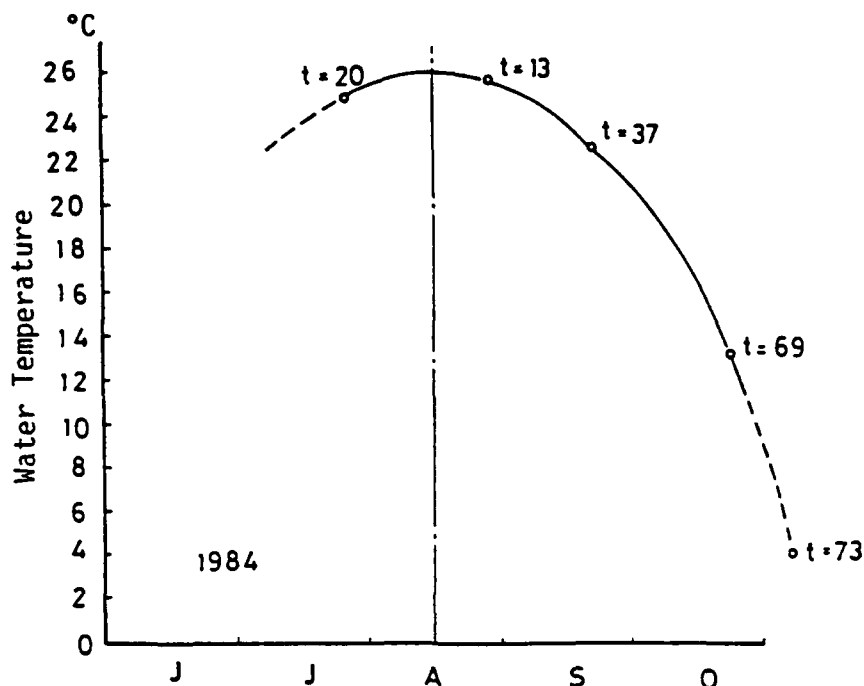


Figure 19. Water temperature measured in 1984

Figure 20 shows the release rates calculated in Table 1. By comparing the calculated release rates in 1973 and 1984 (Figure 20), it is obvious that the release rates in 1984 decreased significantly when compared with 1973. This is caused by sediment removal. Consequently, if dredging is continued, restoration should progress remarkably, as Figure 21 shows.

CONCLUSIONS

Sediment removal in Lake Suwa has been performed over the past 16 years and will continue for 20 years or more.

For such long-term dredging it is very important to know how the work done so far has contributed to restoring the lake. From our study it is clear that the lake is on a steady path toward restoration. This lends confidence to the present restoration techniques.

From this study we have learned the importance of the inner loading of pollutants from sediment. The release rates in Lake Suwa are extraordinarily high and cause a large summer peak. In such a case, there is no effective countermeasure other than sediment removal. It is reported with pleasure that this is being verified in Lake Suwa.

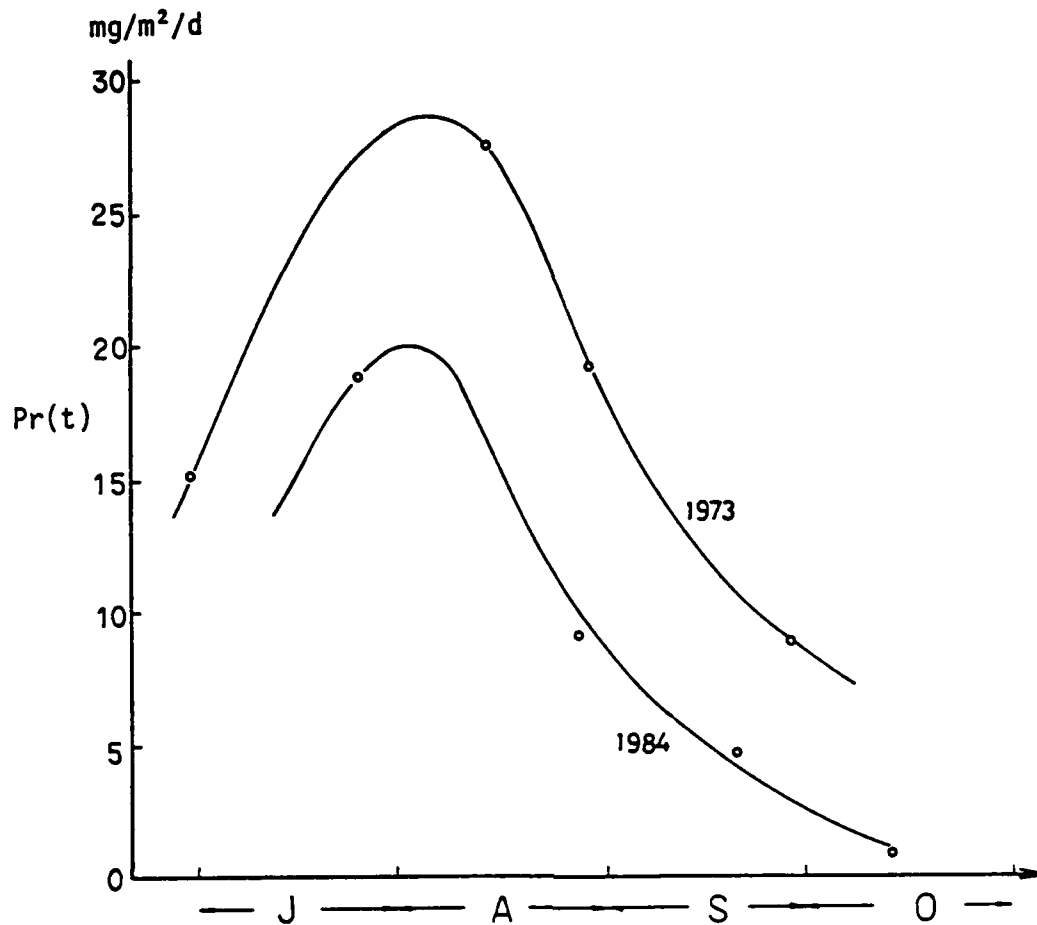


Figure 20. Calculated release rates

TABLE 1. CALCULATION OF RELEASE RATES

t day	F(t) day	$\frac{t^2}{2Pt}$ °C	$\frac{t^2}{\alpha 2Pt}$	$e^{\epsilon t}$	C µg/l	$Ce^{\epsilon t}$ µg/l	$Ce^{\epsilon t} - C_0$ µg/l	Pr mg/m ² /d
-20	-15.90	1.651	1.188	0.698	169	118.0	-132	18.2
13	14.27	0.698	1.076	1.264	240	303.4	53.4	9.04
37	41.17	5.652	1.804	1.946	196	381.4	131.4	4.6
69	80.22	8.249	2.365	3.460	88	304.5	54.5	0.75

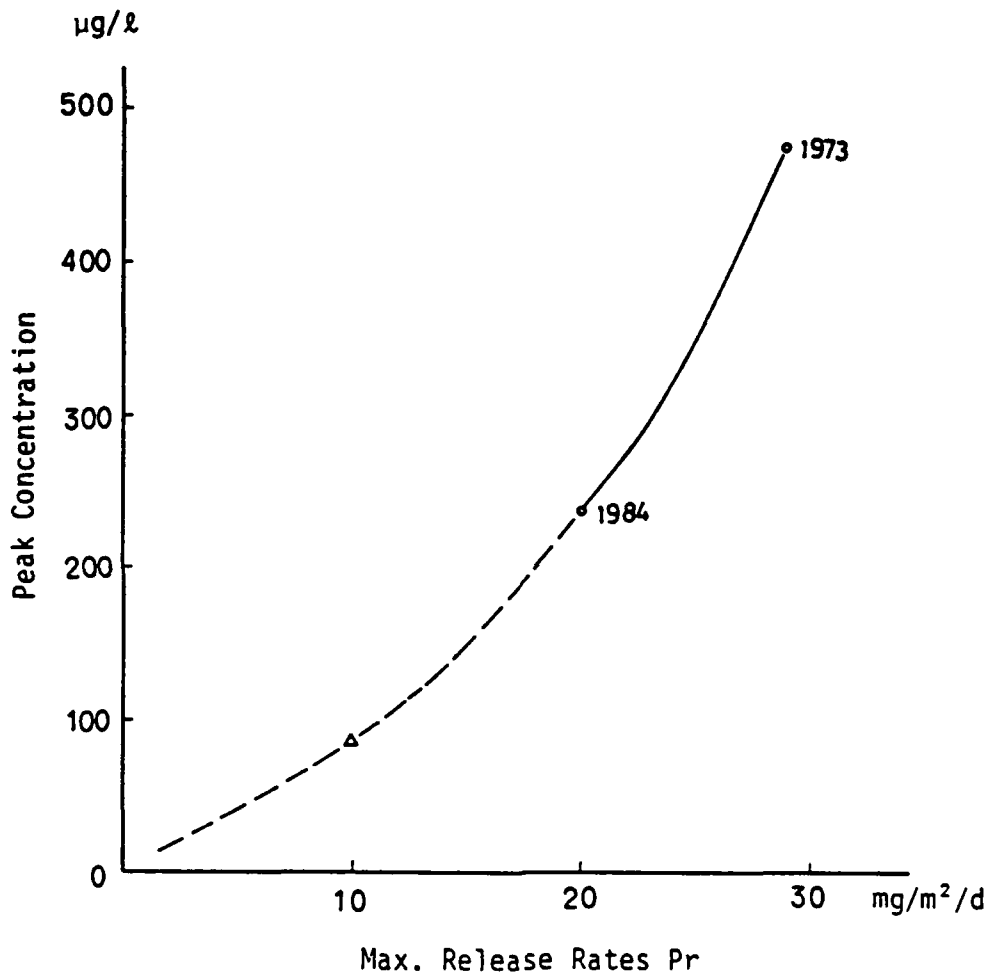


Figure 21. Transition of release rates

MOVEMENT OF TURBID WATER IN RECLAIMING

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Japan Dredging and Reclamation Engineering Association

ABSTRACT

This paper presents experimental investigations on the hydraulic movement of a turbid water in reclaiming to produce an analytical model for efficient spill water treatment. The main findings of the study are as follows:

- a. Land is reclaimed by reciprocation of turbidity currents.
- b. Density in the spillway is highest in the tank independent of initial density.
- c. Density in the spillway decreases in the last stage of the experiment.
- d. The turbidity current is divided into several layers.

INTRODUCTION

When land is reclaimed from the sea with a pump dredger, the spill water flows into the sea from the pond. However, a suitable treatment for the spill water is necessary to satisfy environmental controls. Therefore, it is necessary to know the movement of the turbid water in reclaiming, in particular, the change in density of a spill water before reclamation in order to plan a spill water treatment.

This paper presents experimental investigations on the hydraulic movement of turbid water in reclaiming to produce an analytical model for efficient spill water treatment.

EXPERIMENTAL APPARATUS AND PROCEDURE

The tank used in this test was 4 m long, 0.6 m deep, and 0.5 m wide (Figure 1). One side wall of the tank was a glass wall; the spillway was at one end opposite a turbidity nozzle. The tests were conducted using an artificial soil. The spherical particles of the artificial soil were composed of vinyl chloride having a grain-size distribution curve as shown in Figure 2. Their specific weight and median size were 1.38 and 14 μm , respectively.

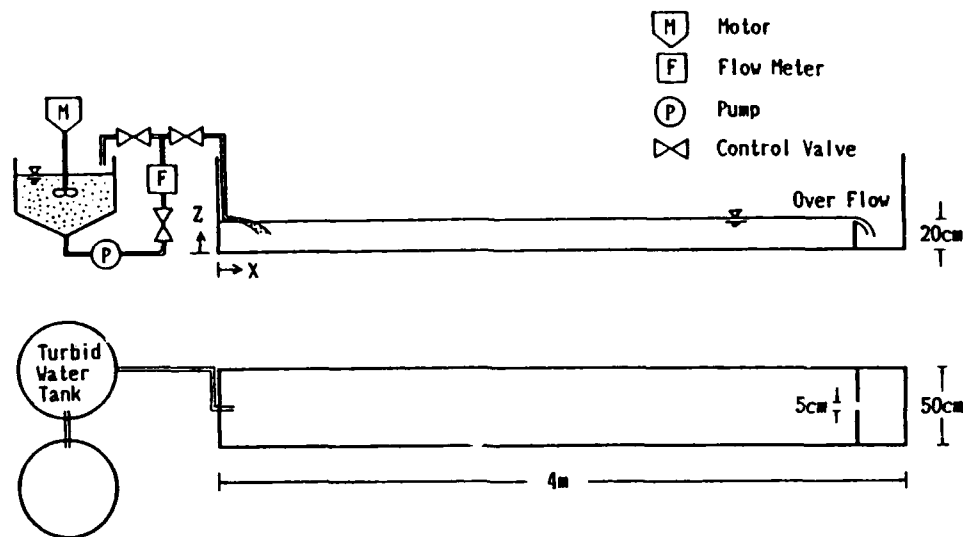


Figure 1. Experimental setup of the tank

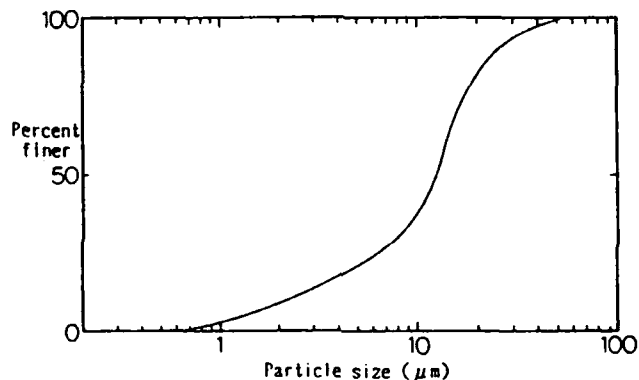


Figure 2. Particle-size distribution curve

The experiment was conducted with different densities. The nozzle diameter, the turbid discharge, and the water discharge remained constant. The turbidity nozzle position was set on the water surface, as shown in Figure 1. The reclamation was judged complete when the turbid water could not advance. Experimental conditions are listed in Table 1. The turbidity in the pond and the spillway, the profile of the reclamation, and the velocity profile were measured. The turbidity was measured at the center of the tank, the spillway, and the median point of the two, using a turbidimeter (transmissometer type).

TABLE 1. EXPERIMENTAL CONDITION

Run	Density(%)	Time	Discharge(l/min)
1	4.7	45h 00m	1.0
2	20.1	18h 20m	1.0

The reclamation profile was measured with a camera and digitized by a digitizer. The two-dimensionality of the reclamation profile was confirmed by a preliminary experiment.

TURBIDITY AND DENSITY

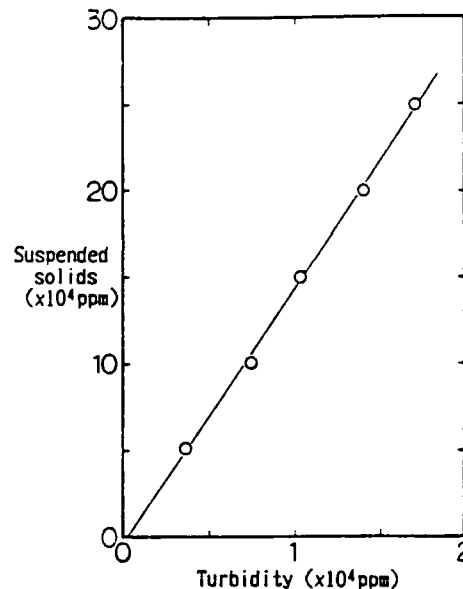
Turbidity is an optical property of a suspension (measured in percent transmittance or NTUs) different from density (usually measured as suspended solids (SS) concentration). Miyazaki, Saotome, and Masuda (1978)* indicated that the turbidity was dependent on the particle size, and finer particles produced measurements higher in density.

We had previously checked the relationship between turbidity and density. The result is shown in Figure 3. Consequently, Equation 1 was obtained using the method of least squares:

$$C_{SS} = - 8058 + 15.1 \times C_{tur} \text{ (ppm)} \quad (1)$$

From then on, we used this SS value which was transformed by using Equation 1.

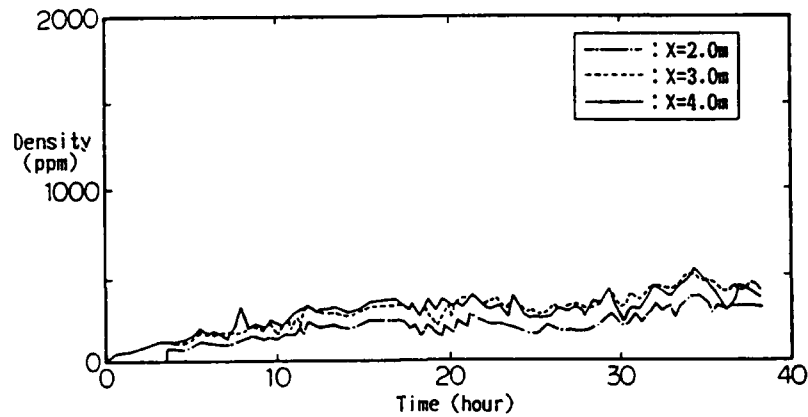
Figure 3. Relationship between turbidity and density



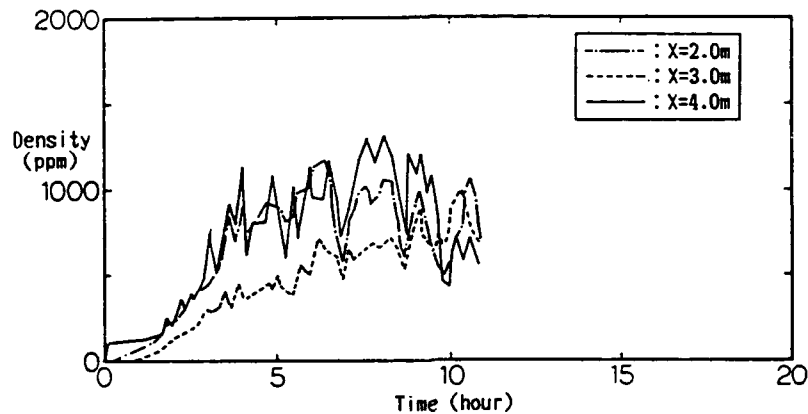
CHANGE OF DENSITY

Figure 4a shows the result of 5-percent density in the nozzle. The density was first observed at the spillway. The reason is explained in the following paragraphs. As the turbid water flows from the nozzle, it settles vertically, and then advances toward the spillway as a turbid density current. This turbidity current reaches the spillway without an increase in its thickness. Furthermore, a part of the turbidity current flows out from the spillway, and the rest returns in the direction of the nozzle. Thus, the turbidity current has alternate layers. These alternate layers reach the water surface

* Miyazaki, S., Saotome, Y., and Masuda, K. 1978. "Influence of Particle Diameter on the Measuring of Turbidity," Technical Note of Port and Harbour Research Institute, Ministry of Transport, No. 284 (in Japanese).



a. 5-percent density



b. 20-percent density

Figure 4. Change of density versus time

after making a return trip in the tank, finally reading the spillway. In Figure 4a, the density value in the spillway is the highest of the three measuring points from beginning to end in reclaiming. The reason is considered the same as above.

Figure 4b is the result of 20-percent density. The time spent in reclamation is a fourth of that for 5 percent because the density was quadruple that for 5 percent. The movement of the turbid water was again a turbid density current. The turbidity changed only moderately for 1 hr. Thereafter, it increased for 5 hr. It then remained almost constant from 5 to 10 hr. After that, it decreased gradually. The result of 20 percent differs from that of 5 percent as follows. In 5 percent, the least density was observed in the nozzle side, while it was in the middle position in 20 percent. This is true because in 5 percent, since almost all of the turbid water from the nozzle settles, the thick turbid water does not flow on the water surface. On the other hand, in 20 percent a part of turbidity from the nozzle does flow on the water surface and settles vertically.

It was found that the progress of the experiment can be divided into three processes: an early stage of increasing density, a second stage of remaining constant, and a final stage of decreasing. However, it was not clearly defined in 5 percent.

TURBIDITY SPECTRA

Figure 5 shows the results of the turbidity spectra to investigate the periodicity of the turbidity movement. In Figure 5a, a spectrum peak exists at the frequency of 2×10^{-4} Hz (83 min); hence, the turbidity movement is very slow in 5 percent. The energy in a high-frequency regime remains almost constant. Thus, it is thought that movement of a turbidity current in a reclaimed pond is random, with little periodicity.

Figure 5b shows the results at 20 percent. A spectrum peak exists at 9×10^{-4} Hz (19 min). In a high-frequency regime, the spectrum energy is very small, and more periodic than 5 percent.

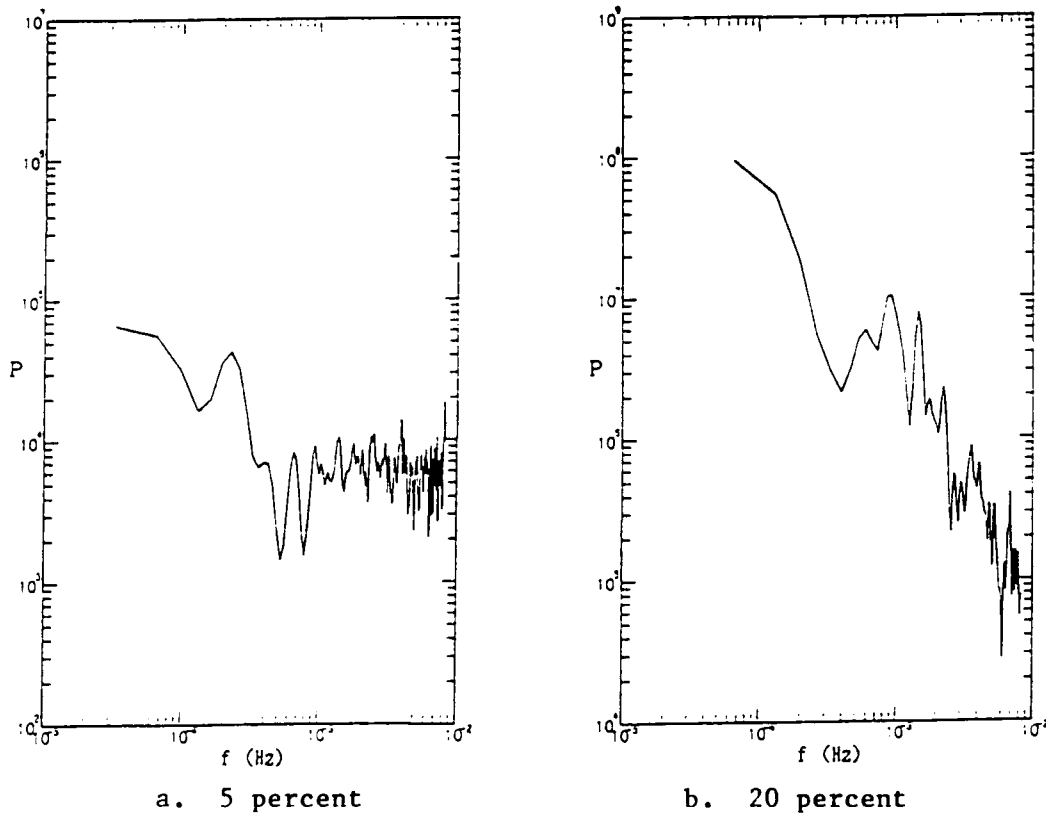
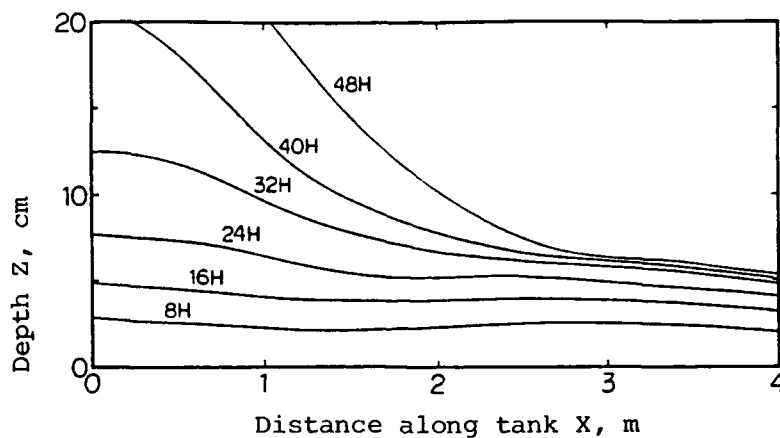


Figure 5. Turbidity spectra

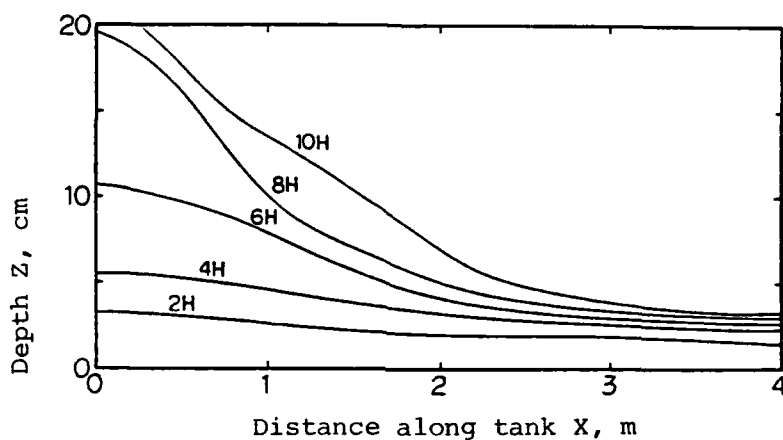
CHANGE OF RECLAMATION PROFILE

Figure 6a shows the change in reclamation profile at 5 percent. In this figure, the particle profile is flat in the early stage. The accumulation forms an incline after 20 hr, and then becomes an acute angle. Further, the thickness changes only slightly at the 3- or 4-m point near the spillway after 28 hr. When the angle becomes steep, particles accumulate near the nozzle.

Similarly, Figure 6b shows the results at 20 percent. In this figure, the process of reclamation is similar to a fourth of that in 5 percent. In



a. 5 percent



b. 20 percent

Figure 6. Change in the reclamation profile

comparison with Figure 4b, it is found that the particles accumulate beyond the water surface when the density decreases. At this time, few particles reach the spillway because they stop in the middle of the slope after slipping.

VELOCITY PROFILE

Figure 7 shows the vertical velocity profile near the glass wall at 20 percent after 100 min. In this figure, it is found that the layer is divided in 5, and the middle layer is thick, while the layer near the water surface is thin. The velocity of the return current reaches a maximum near the water surface. The turbidity current advances strongly on the water surface and on the bed. On the other hand, it returns strongly in the middle depth zone. This means that the movement of the turbidity current in reclamation is not explained by a settling-basin model or a diffusion-equation model. The density front of turbidity current just after flow initiation moved at a rate of 0.92 cm/sec. This velocity is almost equal to the dominant frequency of the turbidity spectrum because it takes 15 min for this turbidity current to go and return once.

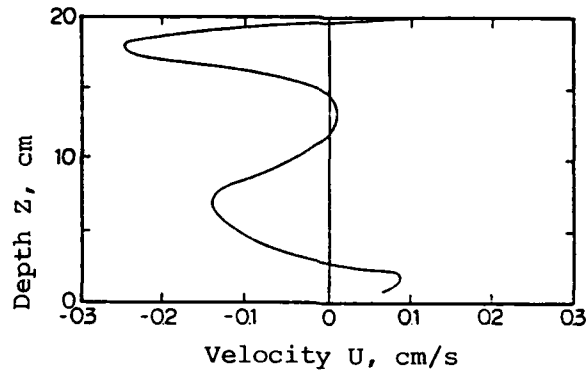


Figure 7. Vertical velocity profile
(after 100 min)

CONCLUSIONS

The main findings of the present study are as follows:

- a. Land is reclaimed by reciprocation of turbidity current.
- b. Density in the spillway is highest in the tank independent of initial density.
- c. Density in the spillway decreases in the last stage of the experiment.
- d. The turbidity current is divided into several layers.

DREDGED MATERIAL DISPOSAL MANAGEMENT STUDIES FOR THE
PORT OF NEW YORK AND NEW JERSEY: PROGRESS REPORT

AD-P006 899



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ABSTRACT

Recent progress is presented of ocean disposal activities and studies of alternatives to ocean disposal for contaminated dredged material from the Port of New York and New Jersey. The capping studies conducted in 1980-81 and subsequent monitoring have shown that ocean disposal of dredged material in the New York Bight can be managed effectively by pin-point disposal and capping when necessary to mitigate potential adverse environmental impacts. Disposal in subaqueous borrow pits with capping has been determined to be a technically feasible and environmentally suitable disposal alternative. Environmental clearances are currently being pursued to designate a borrow pit disposal site in New York Harbor, to be reserved only for dredged material not suitable for unrestricted ocean disposal. Siting studies are still being performed for locating a large containment island, and three potential sites have been identified in the Lower Bay of New York. Beach nourishment, upland disposal, use of dredged material as sanitary landfill cover, small containment areas and wetlands stabilization with dredged material are all possible in special cases on a project-by-project basis but not on a regional basis at this time. The results of these studies have demonstrated there is continued need for ocean disposal of the majority of dredged material from the Port, that ocean disposal can be managed effectively in an environmentally sound manner, and that there are a few potential disposal alternatives for the limited volumes of dredged material which are not suitable for unrestricted ocean disposal.

92-17632



INTRODUCTION

Among the many duties of Operations Division in the New York District is facilitating navigation in the Port of New York and New Jersey by dredging the over 380 km of Federal navigation channels and managing the disposal of the 6 to 7.5 million cubic metres of dredged material generated each year.

Historically, 90 percent or more of this material has been disposed each year in the ocean at the "Mud Dump Site" (see Figure 1). However, because the majority of the dredged material is predominantly silt and clay and is contaminated to varying degrees with heavy metals, petroleum hydrocarbons, and organic compounds such as PCB's, studies of alternatives to ocean disposal and the environmental impacts of ocean disposal were started in 1980.

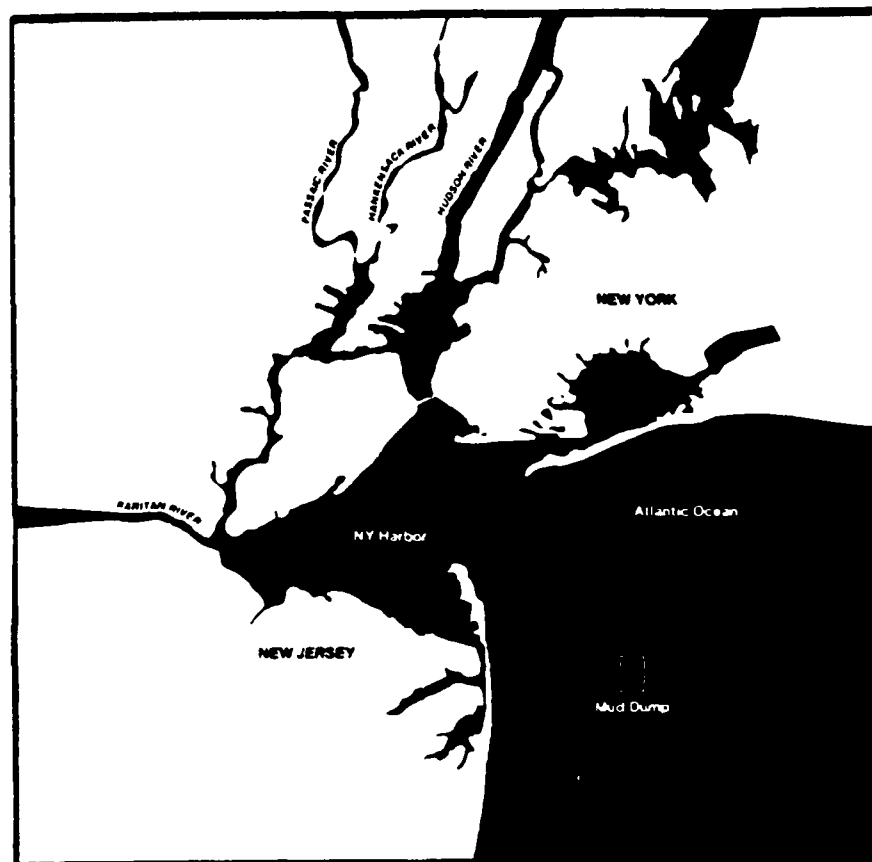


Figure 1. New York Harbor showing major rivers and Mud Dump Site

Previous papers presented by other authors at earlier U.S./Japan Conferences described some of these studies of dredged material disposal alternatives and ocean disposal impacts. Studies of "capping" at the Mud Dump Site and in subaqueous borrow pits were discussed in the 1981 and 1982 conferences (Suszkowski 1981, Suszkowski and Mansky 1981, Mansky 1982). Studies of alternatives to open water disposal were discussed at the 1983 conference (Coch et al. 1983). These were the construction of large containment islands, construction of small containment areas or wetlands stabilized areas adjacent to shore, confined upland disposal, use of dredged material as sanitary landfill cover, and beach nourishment. This paper will describe the status and results to date of the studies of each of these disposal alternatives. Although the study of alternatives to ocean disposal will not be completed until the end of 1987, we have progressed to a point where conclusions have begun to crystallize.

CAPPING STUDIES

Capping at the Mud Dump Site

Capping is the covering over of subaqueous disposed contaminated dredged material with relatively uncontaminated dredged material in order to isolate the contaminated material from the marine environment. Previous capping projects have been successfully completed in Hiroshima Bay and at several sites in the Long Island Sound (O'Connor and O'Connor 1983). Based on the success of these projects, a capping pilot project was performed at the New York Bight ocean disposal site in 1980. Approximately 650,000 cu m of contaminated dredged material was disposed at a previously unused portion of the Mud Dump Site and capped with 224,000 cu m of relatively uncontaminated fine-grained dredged material and 1.2 million cubic metres of dredged sand (Suszkowski 1981).

Chemical, geological, and biological monitoring was performed to determine the effectiveness of capping in minimizing the environmental effects of ocean dumping. A sediment budget study was performed in order to estimate the amount of contaminated dredged material which could be dispersed into the environment during the dredging and disposal process (Tavolaro 1984). Estimates were made of the dry mass of dredged material before dredging (in-place), during transport (in the barges), and after disposal in the ocean using a variety of measurement techniques. After disposal, a sediment cap stability study was performed to estimate the surficial sediment characteristics of the cap over a 1-year period, determine the erosional thresholds of the cap material using in situ measurements, and measure aspects of the physical oceanography over the Mud Dump Site which affect the stability of the cap (Freeland et al. 1983). A mussel bioaccumulation study was performed after capping was completed to determine if caged mussels at the cap site showed higher levels of bioaccumulation of contaminants of concern than mussels placed at control sites in the New York Bight (Koepp et al. 1982). Also performed was a geochemical study designed to determine if capped dredging projects could be chemically discerned from the cap material, the degree of variability in sediment chemistry of individual barge loads of a dredging project, and whether individual dredging projects could be identified after disposal by unique chemical signatures (New York University Medical Center 1982). The results of all these studies, and an overall review of the effectiveness of the capping project in the New York Bight, were prepared (O'Connor and O'Connor 1983).

The capping studies conducted in 1980-1981 and subsequent monitoring have shown that ocean disposal of dredged material in the New York Bight can be managed effectively by pin-point disposal and capping when necessary to mitigate potential adverse environmental impacts. The chemical, geological, and biological monitoring performed concluded that:

- a. Capping can be effectively conducted in the open ocean conditions of the New York Bight. Through pin-point disposal techniques it is possible to place a 1-m cap of uncontaminated material over contaminated dredged material.
- b. Less than 4 percent (by weight) of the dredged material to be capped is dispersed during the disposal operation.

- c. The sand cap is stable under normal weather conditions.
- d. The cap is an effective barrier to the migration of metals and organic compounds from the contaminated dredged material into the water column.
- e. Bioaccumulation of contaminants from the dredged material is not likely to occur. Mussels placed on the capped deposit showed no increase in body burden of contaminants above mussels placed at reference sites.

Since 1983, we have continued to monitor the cap site by periodically coring through the deposit, noting cap thickness, and chemically analyzing the sediment layers. This monitoring shows that the cap is still present with no signs of major loss of cap material and that the cap has continued to be an effective barrier to contaminant migration. Because capping has proven to be an effective environmental mitigation technique for dredged material disposal in the New York Bight, it is used on an operational basis when determined necessary.

Capping in Subaqueous Borrow Pits

Capping of dredged material in subaqueous borrow pits was also studied as a separate effort. Subaqueous borrow pits are underwater pits that remain from sand and gravel mining operations. In the Lower Bay of New York Harbor, many of these pits occur, and there is the potential that additional borrow pits would be dredged in future sand mining operations. These existing borrow pits are acting as sediment "sinks," rapidly accumulating fine-grained sediments in contrast to the surrounding sandy bottom. Also, the dissolved oxygen concentrations and benthic community assemblages present at certain times of the year in these borrow pits indicate a highly stressed and variable environment (Bokuniewicz, Cerrato, and Hirschberg 1986). This makes them potentially very desirable sites for the disposal of highly contaminated dredged material.

At the 1981 conference, a planned pilot project was described involving the filling of one small portion of an existing borrow pit in a three-phase operation (Suszkowski and Mansky 1981). However, concern over the environmental impact of the project to fin fish populations led to further studies of fish distribution in the area and a delay in the pilot project (Pacheco 1983; National Marine Fisheries Service 1984; Conover, Cerrato, and Bokuniewicz 1985). The fisheries surveys showed that all deepwater, muddy-bottomed areas in the Lower Bay could be characterized by the same type of fish populations and that these populations were higher in catch, number of species, and weight of trawl than populations in shallower, sandy-bottomed areas. Because no unique fisheries utilization of borrow pits was observed and because there are many deepwater, muddy-bottomed areas in the Lower Bay, no significant impacts to overall fish populations were anticipated by filling in a borrow pit.

While these fisheries studies were being performed, the ocean disposal capping studies described previously, as well as laboratory studies of the effectiveness of capping, were completed (Brannon et al. 1985). Also, a successful capping operation in a borrow pit in Puget Sound was completed (Sumeri 1984). The completion of all these studies led us to the conclusion that disposal of dredged material in a borrow pit with capping is

operationally feasible, environmental impacts to fisheries could be minimized to the point where they were not environmentally significant, and therefore the small pilot project originally planned was no longer needed.

Efforts were then turned towards designing an operational borrow pit disposal program using either newly dug or existing borrow pits. Separate screening criteria have been developed to look at the suitability of existing borrow pits and potential areas to excavate new pits for the disposal of dredged material (Figure 2). Environmental clearances are currently being pursued to designate one or more borrow pit disposal sites in New York Harbor. However, based upon the current sociopolitical climate concerning disposal of dredged material in the estuarine waters of the New York/New Jersey area, it is not likely that more than one borrow pit disposal area will be designated at any time. Because of the limited volume available in a borrow pit, it will be reserved only for dredged material not suitable for unrestricted ocean disposal, as determined by the ocean disposal testing criteria.

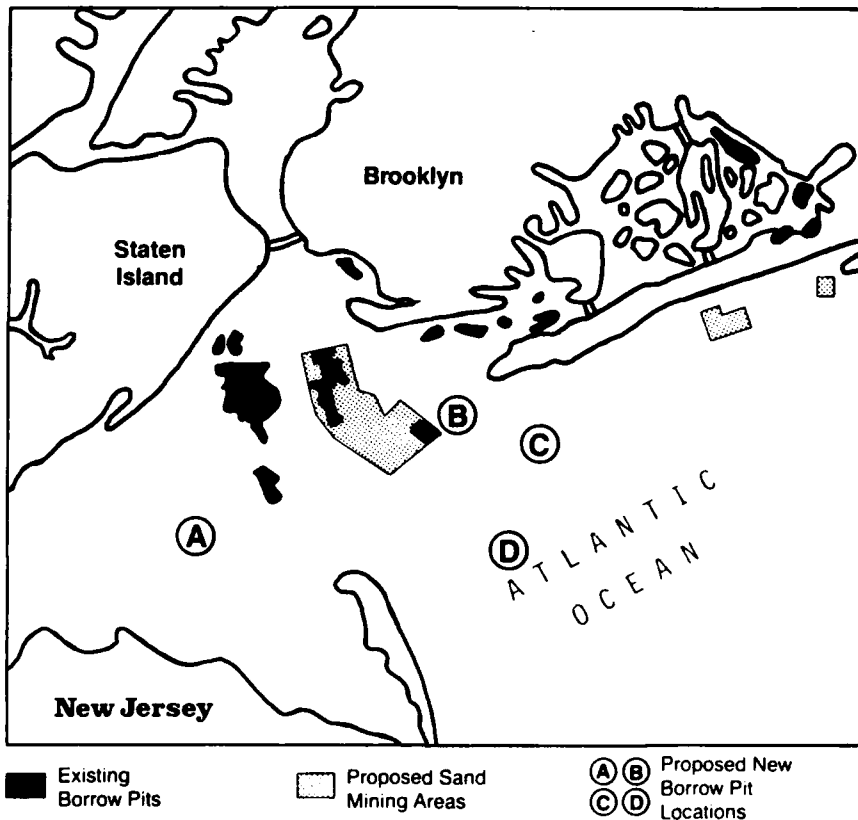


Figure 2. Locations being studied for the potential siting of a borrow pit disposal site

CONTAINMENT FACILITIES/WETLANDS STABILIZED AREAS

Containment facilities are diked disposal areas located in offshore shallow waters. As discussed at the 1983 conference, two types of containment facilities are being studied in the New York Harbor area: large containment islands of up to 200 ha and small containment areas attached to the shoreline (Coch et al. 1983). It was also determined to consider the creation of tidal

wetlands using dredged material at any site that might be suitable as a small containment area. Siting criteria were very similar for small containment areas and wetlands stabilized areas, so the siting studies were combined. The studies primarily involved locating an environmentally appropriate site for the facility.

Small Containment Areas/
Wetlands Stabilized Areas

Preliminary siting studies for small containment areas and wetlands stabilized areas involved delineating waterways in the New York Harbor area that are highly biologically productive and exclude them from further screening (US Fish and Wildlife Service 1982). Criteria were then developed to screen the remaining waterways for suitable sites for containment areas and wetlands stabilized areas (Allen 1983), and nine separate sites were identified (US Army Engineer District (USAED), New York 1983c).

Further site-specific coordination with local civic groups, environmental groups, and government agencies led to the elimination of sites 1, 3, 6, 7, and 8, based upon environmental considerations. Four sites remained for further consideration in greater detail, site 2 - Raritan Bay, site 4 - Bowery Bay, site 5 - Flushing Bay, and site 9 - Newark Bay (Figure 3). A preliminary engineering and economic analysis of containment area and wetlands stabilized area construction was performed at each site (Poindexter 1985). A preliminary benthic faunal survey was performed at each site to estimate the nature of the benthic community that would be affected if these facilities were built (Cerrato and Bokuniewicz 1985). Computer modeling was also performed to determine the degree that tidal elevations and tidal currents would be impacted by containment area or wetlands stabilized area construction (Bokuniewicz, Bowman, and Visser 1985). In addition, an analysis of potential impacts from odors, dust, and bird populations was performed by surveying other Corps of Engineers Districts to determine if these factors had been a problem elsewhere, and if so, the means by which the problems were solved (Shafer 1984, Landin 1984).

As a result of these studies, it was determined that site 4 - Bowery Bay, site 5 - Flushing Bay, and site 9 - Newark Bay should no longer be considered as wetlands stabilized sites. Sites 4 and 5 flank La Guardia Airport, and it was determined not to be prudent to attract bird populations so close to this heavily used airport by constructing wetlands. Collisions between birds and jets have been documented to cause extensive damage to jet engines and in some cases have caused airline accidents. Site 9 - Newark Bay could not be attached to the shoreline because of the adjacent Port Newark and Elizabeth facility and could only be considered as a small containment island site. Constructing a wetlands island was not considered appropriate due to the port and industrial facilities surrounding the site.

Site 2 - Raritan Bay is contiguous to undeveloped county parkland which contains extensive wetland areas. It was therefore determined that this site should no longer be considered as a containment area site, but only as a wetlands stabilized site, which is more compatible with adjacent land use. No significant adverse environmental impacts to tidal currents or elevations were determined for any of the four sites by the computer modeling. Also, no

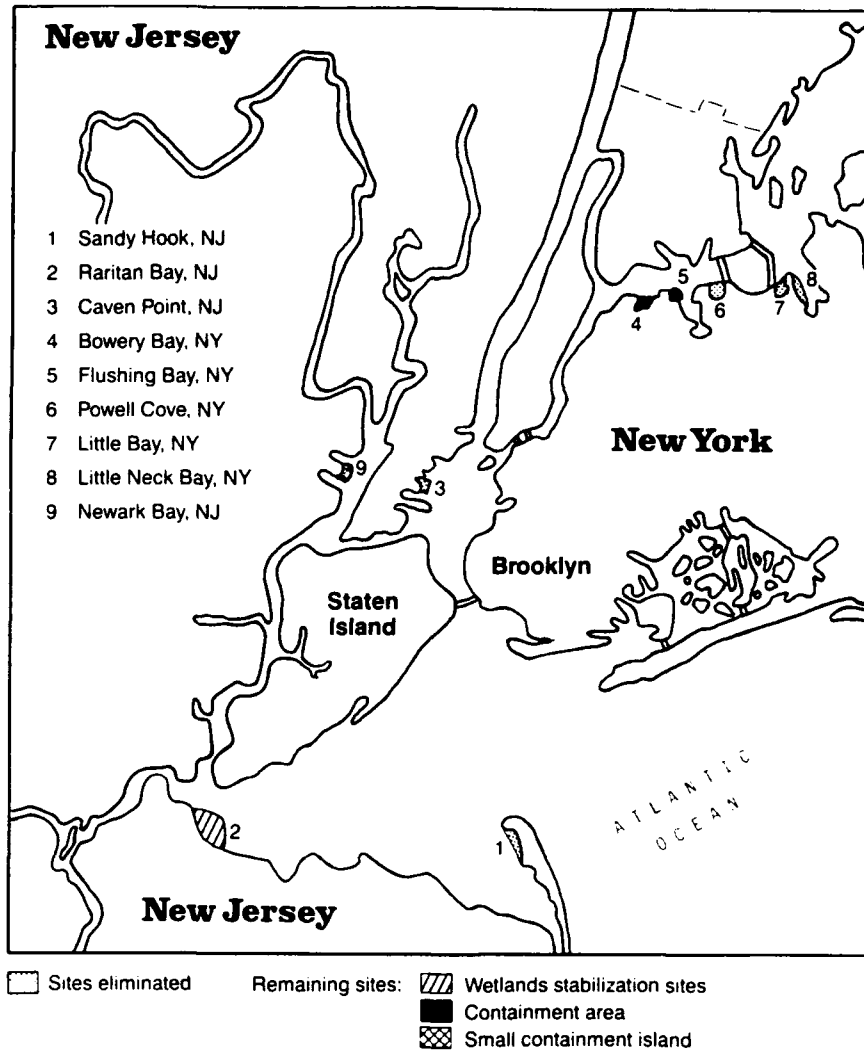


Figure 3. Potential small containment area/wetlands stabilized area sites

significant adverse environmental impacts were expected from odors and dust, or to overall benthic populations due to the creation of containment areas or wetlands stabilization areas at any of the four sites.

The results of the engineering and economic analysis determined that retaining dikes were possible to build at each of the four sites. In addition, capacity of each site, service life, total cost of dike construction, and cost per cubic metre were estimated for several scenarios: disposal of contaminated dredged material, disposal of uncontaminated dredged material, and wetlands stabilization (Table 1). It should be noted that the cost per cubic metre does not reflect dredging, transport, and rehandling costs, but only cost of dike construction divided by site capacity. Actual cost per cubic metre would be higher.

The overall conclusions reached were that these four sites have a very limited disposal volume and would be relatively expensive to construct and

TABLE 1. SUMMARY OF THE RESULTS OF THE ENGINEERING AND ECONOMIC ANALYSIS OF POTENTIAL CONTAINMENT AREA AND WETLANDS STABILIZED AREA SITES (FROM POINDEXTER 1985)

Disposal Site	Capacity million cu m	Service Life years	Total Cost million \$	Cost/cu m* \$
Containment area				
Bowery Bay	1.2-2.8	9-20	3.1	1.13-2.39
Flushing Bay	0.9-2.2	6-16	8.2	3.74-8.75
Newark Bay	3.1-5.2	23-40	18.4	3.56-5.93
Wetland area				
Raritan Bay	4.6	13	13.9	2.66

* Cost does not include dredging, transportation, or rehandling costs of dredged material.

operate. Also, a wetlands stabilized area could only receive uncontaminated dredged material in order not to distribute contaminants through the food chain. The disposal of uncontaminated dredged material does not pose the same types of environmental concerns as contaminated dredged material disposal and therefore does not warrant such expensive special treatment. For these reasons, small containment areas and wetlands stabilized areas were determined not to be feasible as regional disposal options but may be possible in special cases or on a project-specific basis.

Large Containment Islands

Preliminary siting studies for large containment areas were included in a report by the US Fish and Wildlife Service (1982). Originally, a 400-ha island or larger was the focus of the survey, and the preliminary screening concluded that the Lower Bay of New York Harbor was the only waterway large enough to be able to contain such a large structure. A position paper was developed by the New York District in conjunction with an accompanying draft report on the environmental benefits and detriments of siting a large containment island in the Lower Bay (Bokuniewicz 1983). At that time New York District recommended that siting a large (400 ha) containment island should no longer be pursued in light of the large area of productive subaqueous habitat that would be eliminated in order to construct one. However, it was decided that rather than eliminate the disposal alternative altogether, siting studies should continue for a smaller version of a containment island, i.e., 200 ha. Also, in order to maximize the use of a smaller containment island, it should be reserved only for dredged material that could not be disposed in the ocean without capping and/or for dredged material that did not meet ocean disposal

testing criteria. This would reduce the potential volume demand on a containment island from several million cubic metres per year to several hundred thousand cubic metres per year at a maximum.

Secondary site screening assessed existing benthic and fisheries information in a computer model to identify the least productive areas in the Lower Bay (Bokuniewicz and Cerrato 1984). Three potential sites were identified: the Raritan Bay Site, West Bank Site, and East Bank Site (Figure 4). Planning level cost and site capacity estimates were then performed for three separate dike construction scenarios for each potential site: sand dikes, rock dikes, and sheet pile cofferdams (US Army Engineer Waterways Experiment Station 1985). These are summarized in Table 2. Also, a general estimate of the real estate value of each site was performed looking at several end use scenarios (Fowler 1985).

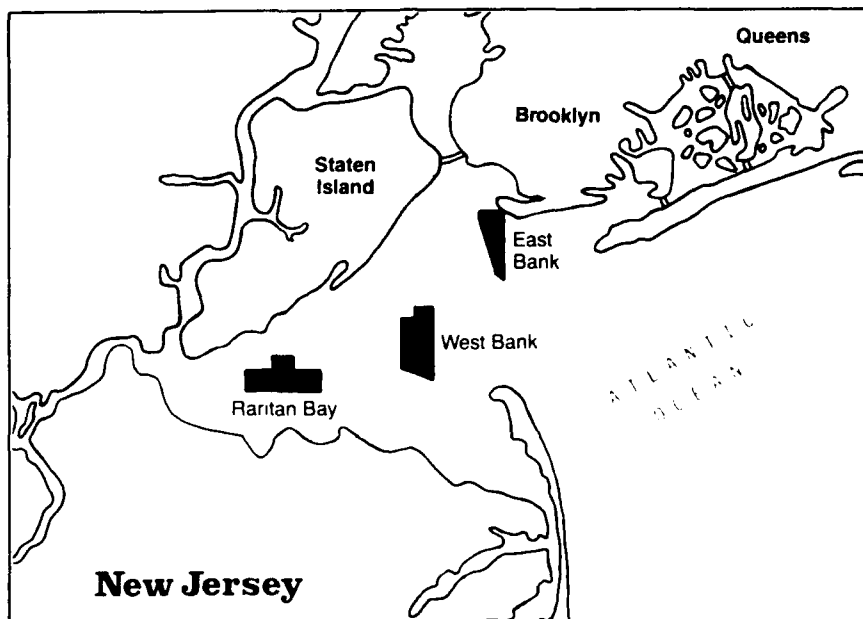


Figure 4. Potential large containment island sites

Several studies are still in preparation. Site-specific benthic and fisheries surveys are being performed over a 1-year cycle. Computer modeling studies are being performed to determine the degree that tidal elevations and tidal currents would be impacted by containment island construction at these sites. Also, a study is being performed to determine if effluent control measures are necessary, using samples of dredged material that are representative of the type of contaminated dredged material proposed for the sites. All these studies should be completed by August 1987.

The results of all studies so far have shown that it is possible to construct a large containment island in the Lower Bay of New York Harbor without unacceptable adverse environmental impacts. It is estimated that a 200-ha island could be used for 20 to 40 years if it were reserved for dredged material that needed to be capped if ocean disposed, and/or dredged material that did not meet ocean disposal testing criteria. When the results of the remaining studies have been reviewed, final decisions on possible site selection will be made.

TABLE 2. SUMMARY OF COST AND VOLUME ESTIMATES FOR SEVERAL DIKE CONSTRUCTION SCENARIOS FOR THREE POTENTIAL CONTAINMENT ISLAND SITES IN THE LOWER BAY OF NEW YORK HARBOR

Volume/Cost	Sand Dike*			Rock Dike	SPC**
	Site A	Site B	Site C		
Total cost					
\$ millions	129	160	192	463	265
Treatment costs					
\$ millions	8.6	8.6	8.6	8.6	8.6
Storage volume					
million cu m	5.0	5.0	5.0	17.4	21.6
Unit storage cost					
\$/cu m					
Without treatment	24	31	38	26	13
With treatment (settling basin)	26	33	39	27	14

* Site A = east bank, site B = west bank, site C = Raritan Bay.

** Sheet pile cofferdam.

UPLAND DISPOSAL/SANITARY LANDFILL COVER

Upland Disposal

Upland disposal facilities are diked upland areas usually located near the shoreline. As for containment facilities, most of the studies for upland disposal sites involved locating an environmentally appropriate site. Preliminary screening identified all undeveloped vacant land greater than 2 ha within 160 km of New York Harbor (Leslie et al. 1980). Undeveloped land was divided into three general categories: cropland and pastureland, wetlands, and barren areas (less than one-third vegetative cover). It was determined that cropland and pastureland and wetlands should not be included in further screening because of their high environmental and/or social value in the highly urbanized greater New York area. This limited further site screening to 295 barren areas ranging in size from 2 ha to thousands of hectares. General screening criteria were then developed by the states of New York and New Jersey to be used to determine the environmental suitability of each barren area.

After applying this criterion to each site, 13 potential sites were identified to warrant more detailed screening (Coch et al. 1983). A report was prepared which outlined the site screening process, identified the 13 potential sites along with what specific information was known about the

sites, and solicited for any further information which would assist in the site selection process (USAED, New York 1983b). This report was distributed to owners of the sites; adjacent owners; local, state and Federal government officials; community groups; and any other potentially interested or involved parties. An updated report was also distributed in the same manner when information became available from these sources (USAED, New York 1984).

Public forums were also held at five separate locations near these sites to provide additional opportunity for public comment. Based on the site-specific information gathered in this manner and on further developments arising from detailed site-specific screening, four sites remained in 1984: N37 (Port Elizabeth), N61 (Belford, N. J.), L13 (Hutchinson River, N. Y.) and Raritan Center, N. J. (Figure 5). Three separate disposal site sizes were studied at the Raritan Center site because the actual area available was not

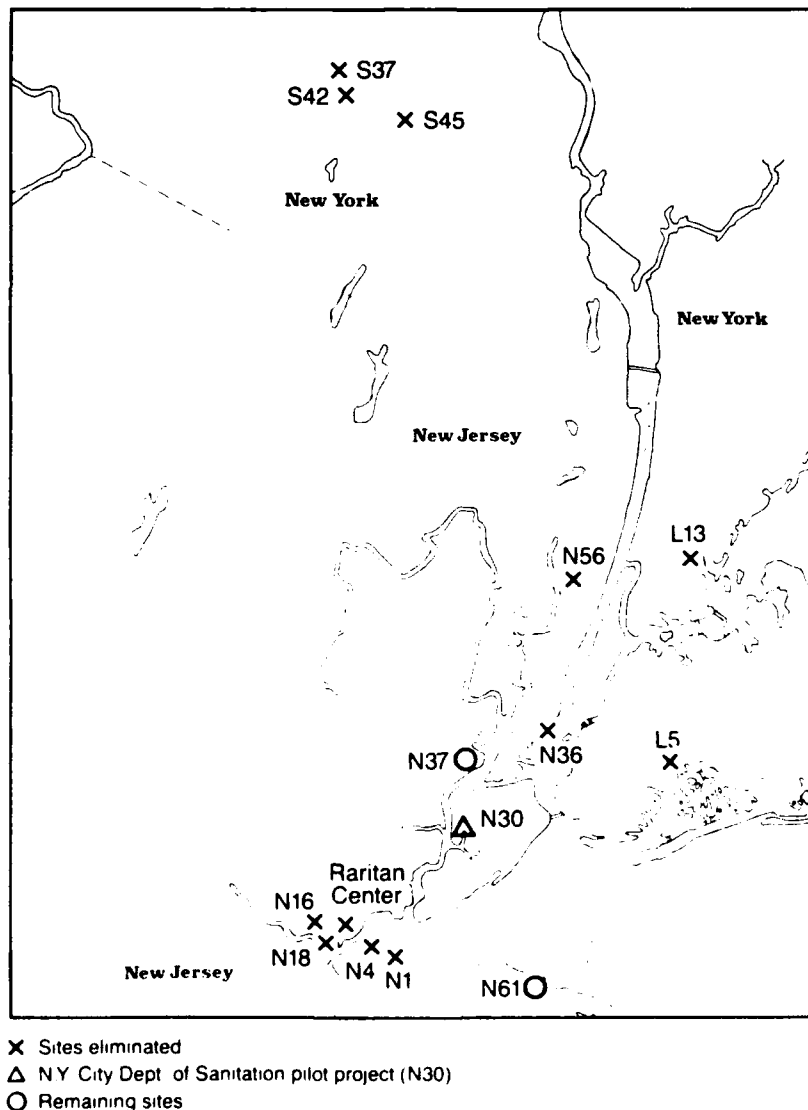


Figure 5. Potential regional upland disposal sites

certain. Site N30 (Fresh Kills) is a special case study being performed by the New York City Department of Sanitation and will be described later. Table 3 shows the capacity and cost associated with each upland disposal site.

TABLE 3. UPLAND DISPOSAL SITE CAPACITIES AND COSTS (FROM MALCOLM PIRNIE, INC. 1986)

Site	Capacity ha	Total Cost*		Total Capacity million cu m
		8 km	16 km	
N37 (Port Elizabeth)	42	12.06	12.88	6.5
N61 (Belford, N. J.)	13	6.51	7.35	2.0
L13 (Hutchinson River)	11	26.43	27.27	1.8
Raritan Center, N. J.				
Configuration 1	51	5.18	6.12	7.8
Configuration 2	71	5.05	5.87	10.9
Configuration 3	168	4.90	5.69	25.8

* Two representative transport distances were used: 8 and 16 km.

Because of the limited long-term capacity available in the few upland disposal sites remaining, it was determined that a regional upland disposal site could be feasible only if the dredged material were removed after dewatering so that the site could be used again. For this reason, the studies of upland disposal sites were combined with studies of the use of dredged material as sanitary landfill cover.

Sanitary Landfill Cover

Sanitary landfills are upland disposal sites for normal household refuse. At the end of each day, recently placed refuse is covered over with clean earth fill for health reasons (Figure 6). Thicker layers of fill are also required if portions of the site remain unused for any significant period of time or for final site closure. Since there is a great demand for this earth fill, dewatered dredged material was determined to be a potential source of cover material, and so the general feasibility of the use of dredged material as sanitary landfill cover was studied. The four upland disposal sites described above were also studied in detail to determine if they were suitable dewatering areas.

The overall feasibility studies were divided into two phases (Coch et al. 1983). Phase I involved leachate and physical testing of dredged material which represented the entire range of chemical contamination of dredged

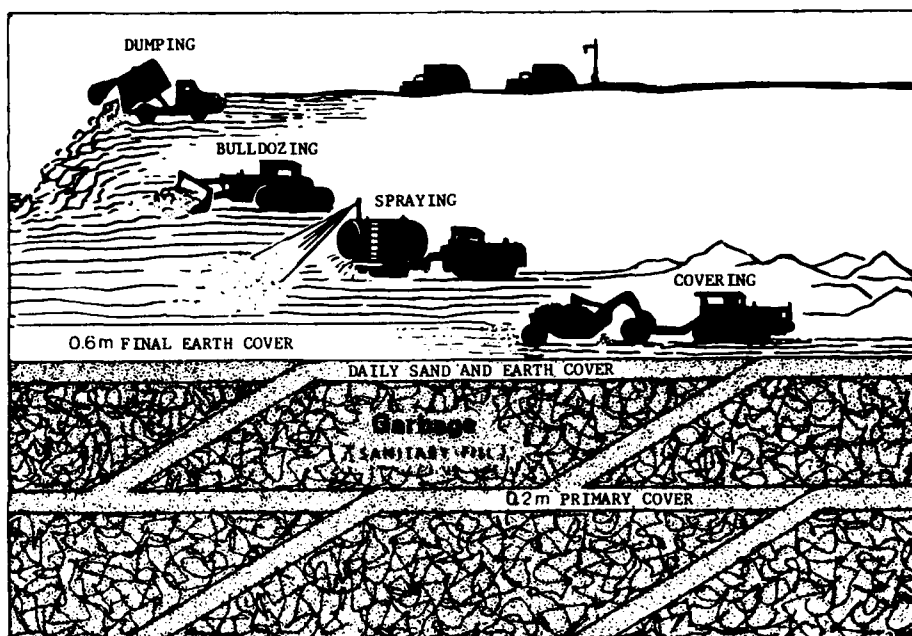


Figure 6. Schematic view of a landfill with the various operations and arrangement of "garbage cells" underground

material in the New York Harbor area. The Phase I analysis determined that all dredged material tested, regardless of the degree of contamination, met New York and New Jersey state criteria for daily and intermediate landfill cover. Approximately 70 percent met the criteria for final cover material (Malcolm Pirnie, Inc. 1982). Phase II originally investigated dewatering techniques and application of dredged material as landfill cover in two specific landfills, DeKorte Park and Fresh Kills (Malcolm Pirnie, Inc. 1983). However, due to comments received on the Draft Phase II report, the entire scope of the study was modified to eliminate studying dredged material as landfill cover in DeKorte Park and expanded to include more chemical testing and the use of the upland disposal sites described above as dewatering areas. Specifically, Phase II studied dewatering techniques in greater detail (thick lift versus thin lift), methods to reduce costs associated with dewatering, a comparison of landfill leachate to dredged material leachate, and at the request of the states of New York and New Jersey, additional bulk chemical testing for base neutral organic compounds, additional heavy metals and organic compounds, and testing of permeability and pH (Malcolm Pirnie, Inc. 1986). In addition, Site N30 (Fresh Kills) was included due to interest by the New York City Department of Sanitation in the possibility of using dredged material from their own marine transfer station dredging projects as cover material for the Fresh Kills Landfill. Table 4 shows the cost of landfill cover, yearly capacity of dredged material, and yearly yield of cover material using the upland sites as dewatering areas.

While Phase II was being performed, two of the upland sites included in this study were dropped from further consideration. Site L13 (Hutchinson River) was dropped because the owners of the site sold it to developers whose planned use of the site was not compatible with dewatering of dredged

TABLE 4. YIELD AND COST ASSOCIATED WITH USING UPLAND DISPOSAL SITES AS DEWATERING AREAS FOR DREDGED MATERIAL TO BE USED AS SANITARY LANDFILL COVER (FROM MALCOLM PIRNIE, INC. 1986)

Site	Total Cost* \$/cu m	Capacity cu m/year	Yield cu m/year
N30 (Fresh Kills)	21.21	143,000	63,000
N37 (Port Elizabeth)	36.50	259,000	114,000
N61 (Belford, N. J.)	32.59	79,000	34,000
L13 (Hutchinson River)	50.92	70,000	31,000
Raritan Center			
Configuration 1	19.06	310,000	135,000
Configuration 2	18.76	432,000	190,000
Configuration 3	18.32	1,023,000	450,000

* Total cost includes cost of dewatering, stockpiling, land acquisition, and transport to the nearest landfill.

material. The Raritan Center Site was also dropped because detailed investigations showed that the vast majority of the site contained wetlands regulated under Corps of Engineers jurisdiction, and these wetlands could not be avoided by simply redefining the study area boundaries. Since one of the original site screening criteria was that sites containing Corps-regulated wetlands should not be considered for upland disposal site development unless the wetlands areas could be avoided, the Raritan Center Site was eliminated from further consideration.

The results of these studies have shown that the use of dredged material as daily, intermediate, and final sanitary landfill cover is technically and environmentally feasible in the majority of cases if the dredged material is dewatered before applying to the landfill. The upland disposal sites studied were determined to be technically feasible to use as dewatering areas, although they are of limited value if used solely as upland disposal sites due to their limited site capacities. Although use of dredged material as sanitary landfill cover was determined to be technically feasible, at this time dredged material is not economically competitive with conventional cover material unless the dewatering site is located immediately adjacent to the landfill. This situation exists only for the New York City Department of Sanitation at this time, who not only have a dewatering area adjacent to a sanitary landfill but also generate dredged material from their marine transfer stations. Site N30 is the location of their dewatering area and adjacent

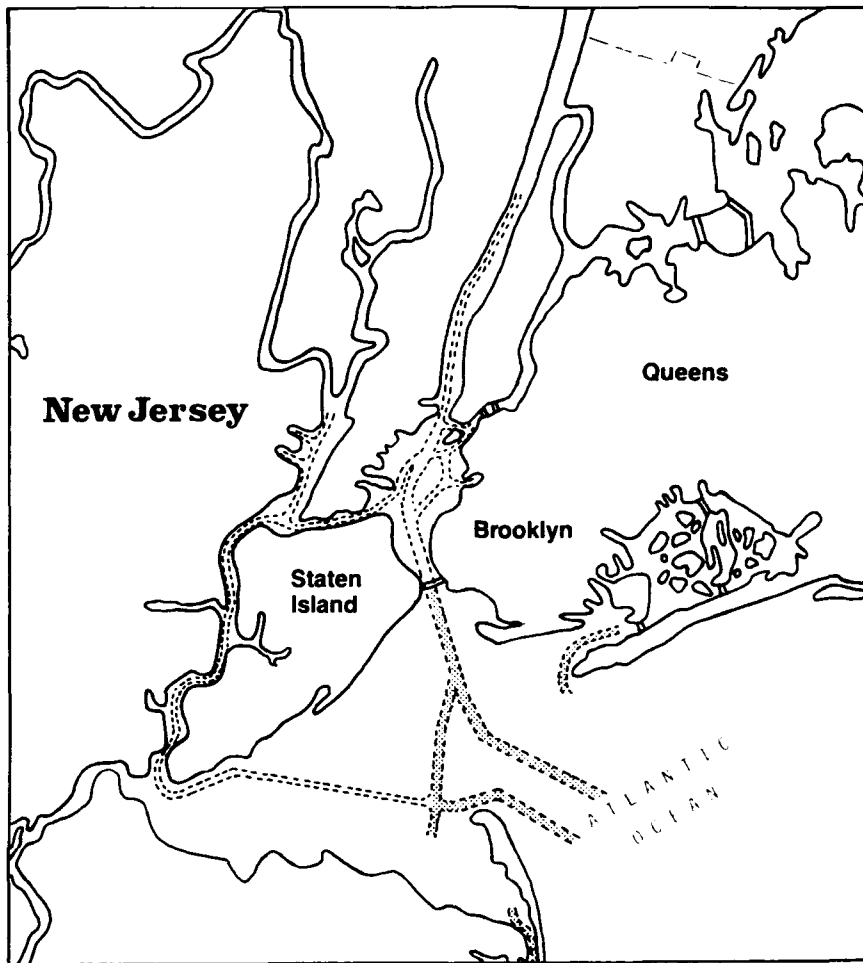
sanitary landfill (see Figure 6). They are currently beginning a pilot project using their own dredged material as sanitary landfill cover, building upon the results of the Corps' feasibility studies. Overall, the use of dredged material as sanitary landfill cover was determined to be possible in special cases, but not feasible as a regional disposal alternative for dredged material at this time.

BEACH NOURISHMENT

Beach nourishment is a common practice in the United States in which dredged sand is placed on recreational beaches to replace the sand lost by erosion. It is generally performed by hydraulically pumping the sand from an offshore borrow area. Beach nourishment, in this case, means using sand dredged from Federal navigation channels to replenish the eroding beaches. Beach nourishment is usually technically feasible if the dredged material is approximately 90-percent sand or greater.

A grain-size survey of the Federal navigation channels revealed that only a relatively few channels contain dredged material that is predominantly sand, amounting to no more than approximately 10 percent of the average annual quantity of dredged material generated from the Port of New York and New Jersey (Figure 7). The sand that is dredged from these channels is usually finer grained than that optimally desired for beach nourishment. Also, the cost of placing sand from these channels to nearby recreational beaches would be approximately \$2.60 to \$3.90/cu m above the cost of ocean disposal (USAED, New York 1983a). From a cost and technical perspective, beach nourishment using sand dredged from these navigation channels is environmentally acceptable but is not always the most efficient use of the money involved. At times, it may be more practical to use this money to dredge sand from an offshore borrow area where there is more choice available in the quality of sand. There are also certain limitations placed upon Corps of Engineers involvement in beach nourishment where navigation projects are the source of sand. Public Law 94-587, passed in 1976, states that sand generated from Federally authorized navigation projects can be placed on beaches only when it is in the public interest and if the increased cost of beach nourishment over the most cost-effective method of disposal is paid for by the local interests who desire the sand. This law was recently modified so that local interests are only responsible for contributing 50 percent of the increased costs of beach nourishment. However, even a 50-percent contribution can add up to a significant sum of money for governments to account for.

Because of these factors, beach nourishment using sand dredged from navigation channels is possible only in special cases. Beneficial use of this sand is still potentially possible and encouraged whenever feasible. For example, there is currently a demand for sand as fill and construction aggregate material that exceeds the supply available from upland sources. Private business concerns are currently performing a form of "sand mining" in some portions of the sandy entrance channels to New York Harbor to meet this demand. This not only puts the sand to some beneficial use, rather than disposing it in the ocean, but the Federal navigation channel is dredged in the process at no real cost to the Federal government, the state collects revenue in the form of royalty fees from the dredging company, and the dredging company makes a profit from sale of the sand.



--- Predominantly sand

Figure 7. Federal navigation channels which contain sand potentially suitable for beach nourishment

CONCLUSIONS

The results of these studies have demonstrated that:

- a. There is a continued need for ocean disposal for the majority of dredged material from the Port of New York and New Jersey. None of the alternatives to ocean disposal that were studied, either singly or in combination, would be able to accommodate all dredged material generated from the Port of New York and New Jersey for more than a few years.
- b. Ocean disposal can be managed effectively and in an environmentally sound manner through techniques such as pin-point disposal, capping, and environmental monitoring.
- c. Any alternative to ocean disposal must be able to accept the limited volumes of dredged material which are not suitable for unrestricted

ocean disposal in order to be of any real use to the port. There appear to be two potential regional disposal alternatives which meet this criterion. Subaqueous borrow pits could satisfy short-term needs and a large containment island could satisfy long-term needs. All other alternatives seem to be possible only in special cases and could be implemented if specific local cooperation is provided.

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PREVENTION OF SECONDARY POLLUTION CAUSED BY DREDGING BOTTOM
SEDIMENT CONTAINING MERCURY IN MINAMATA BAY

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ABSTRACT

In Minamata Bay, sediment containing 25 ppm mercury (about 1,500,000 m³) has accumulated on the bottom (about 2,110,000 m²), and the fish and shellfish in Minamata Bay are now polluted by mercury. Since October 1977, we have been dredging to provide a safe environment to ensure the inhabitants' health. We have restored Minamata Bay by dredging a part (about 1,500,000 m²) and using the other part (about 600,000 m²) as a disposal area.

During dredging, we have guarded against secondary pollution. We have made sure that water quality did not deteriorate due to stirring of the bottom sediment or by soakage of mercury from the disposal area and that marine products were not further polluted. We have established several stations along the border of the working area and measured the concentration of mercury in seawater collected at these stations. The equipment for the analysis is located onsite so that we can get the results of the analysis immediately. If the analysis shows concentration over the allowed level, we stop or slow down the work at once and find the cause of the pollution. However, since the analysis for mercury is slow, we use the estimation method. This method measures the turbidity in seawater collected from stations in the work area for use as indicators of mercury; we then estimated the diffusion of mercury.

In the first disposal area (dredging was carried out from June 1981 to March 1984), the estimation method was used and satisfactory results were



obtained. In the second disposal area (dredging began in October 1982), we are also using the estimation method for the diffusion of mercury by dredging.

INTRODUCTION

Minamata Bay is a useful port located near the southern tip of Japan and Port of Minamata in Minamata Bay, which is on the Yatsushiro Sea. Chisso Chemical Co., located on Minamata Bay, used mercury as a catalyst in making acetaldehyde and vinyl chloride for the 40 years from 1932 to March 1971. A part of the mercury used as the catalyst was converted into methyl mercury by a side reaction in the plant. Methyl mercury in the wastewater discharged from the Chisso Chemical Co. accumulated in the fish in Minamata Bay, and many people who ate the polluted fish were poisoned by methyl mercury. It is said that 70 to 150 tons or more of mercury was discharged to Minamata Bay, polluting most of the bay. The polluted sediment containing mercury at a rate greater than 25 ppm is 4 m thick at some points in the inner bay.

Mercury concentration in the fish in Minamata Bay has been investigated for many years. The total mercury concentration of the fish was about 20 ppm or more (as wet base) in 1960. After the suspension of using mercury in the Chisso Chemical Co., the total mercury concentration in the fish decreased to about 1 ppm or less. But this mercury concentration in the fish from Minamata Bay is higher than that in the fish from other districts, and the total mercury in the same fishes exceeds the safety guideline for fishes (0.4 ppm).

TREATMENT OF SEDIMENT

To solve the mercury problem, we will dredge the sediment which contains mercury levels higher than 25 ppm (the required level for removal is 25 ppm). The area of removal is shown in Figure 1, and the working area which includes the removal area is shown in Figure 2. The inner part of the Bay (582,000 m², 726,000 m³) where the bottom has a thick layer of sediment containing the high concentration of mercury was chosen as a disposal area for the dredged sediment. The area for removal (1,510,000 m², 784,000 m³) was dredged and pumped directly to the disposal area.

Most of the mercury in the sediment is thought to be mercury sulfide, which does not dissolve in seawater. However, the mercury sulfide in the sediment would transform to methyl mercury under the aerobic condition resulting from the irradiation of sunlight. We decided to use the safe working method which does not stir the bottom sediments as much. While cleaning the polluted bay, we have made sure that water quality did not deteriorate due to stirring of the bay bottom or by mercury soakage from the disposal area, and that marine products were not further polluted.

First of all, the defense net for fish was set along the border of the working area to inhibit the movement of fish from the bay to outside or from outside to the bay. A temporary separation revetment (430 m long) was built between Koiji Island and Myojin Cape to prevent diffusion of pollutants by decreased current in the bay working area. Then, at the proposed site of the disposal area bulkhead, the bottom was covered with a layer of sand 1 m thick

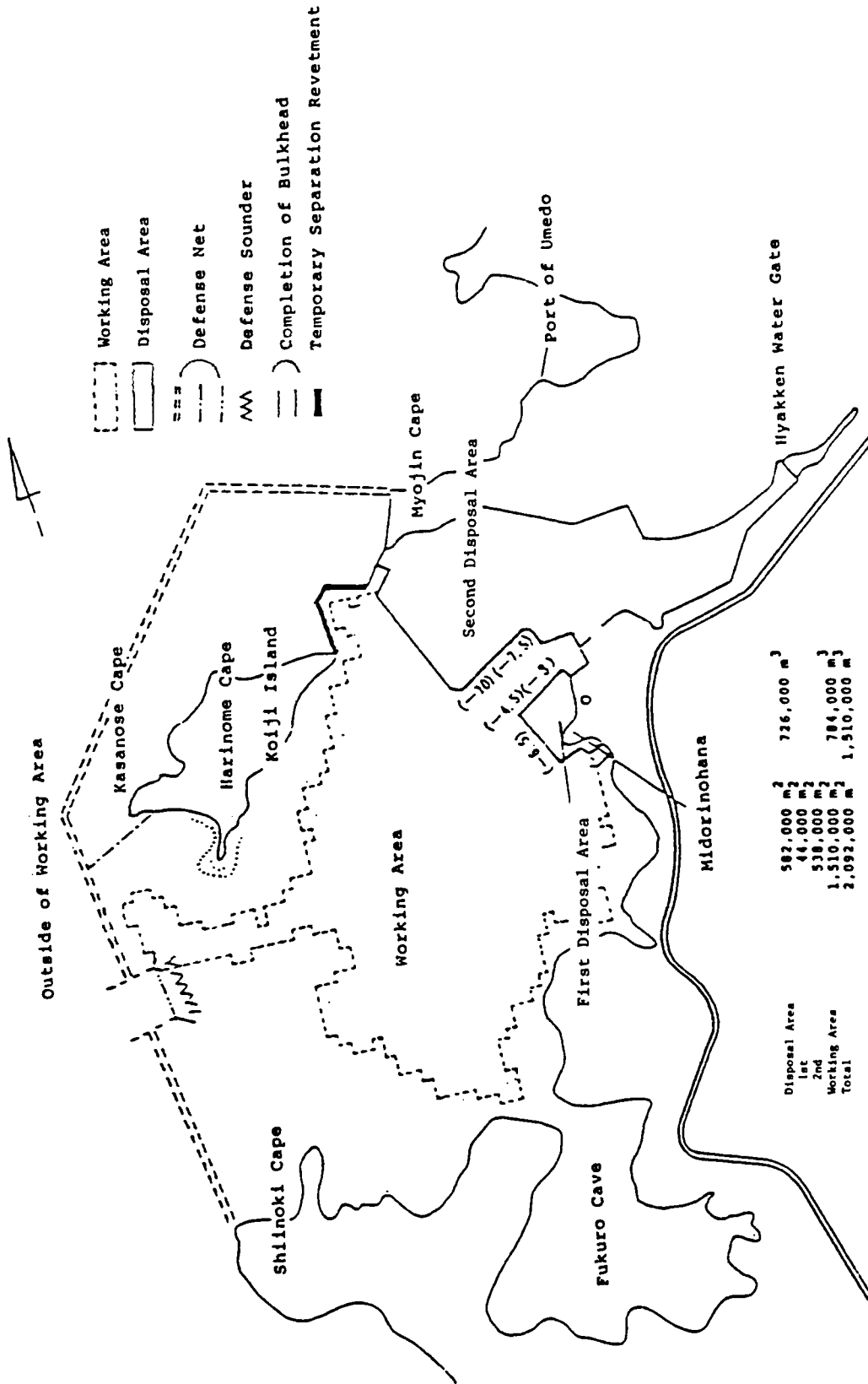


Figure 1. Minamata Bay working area

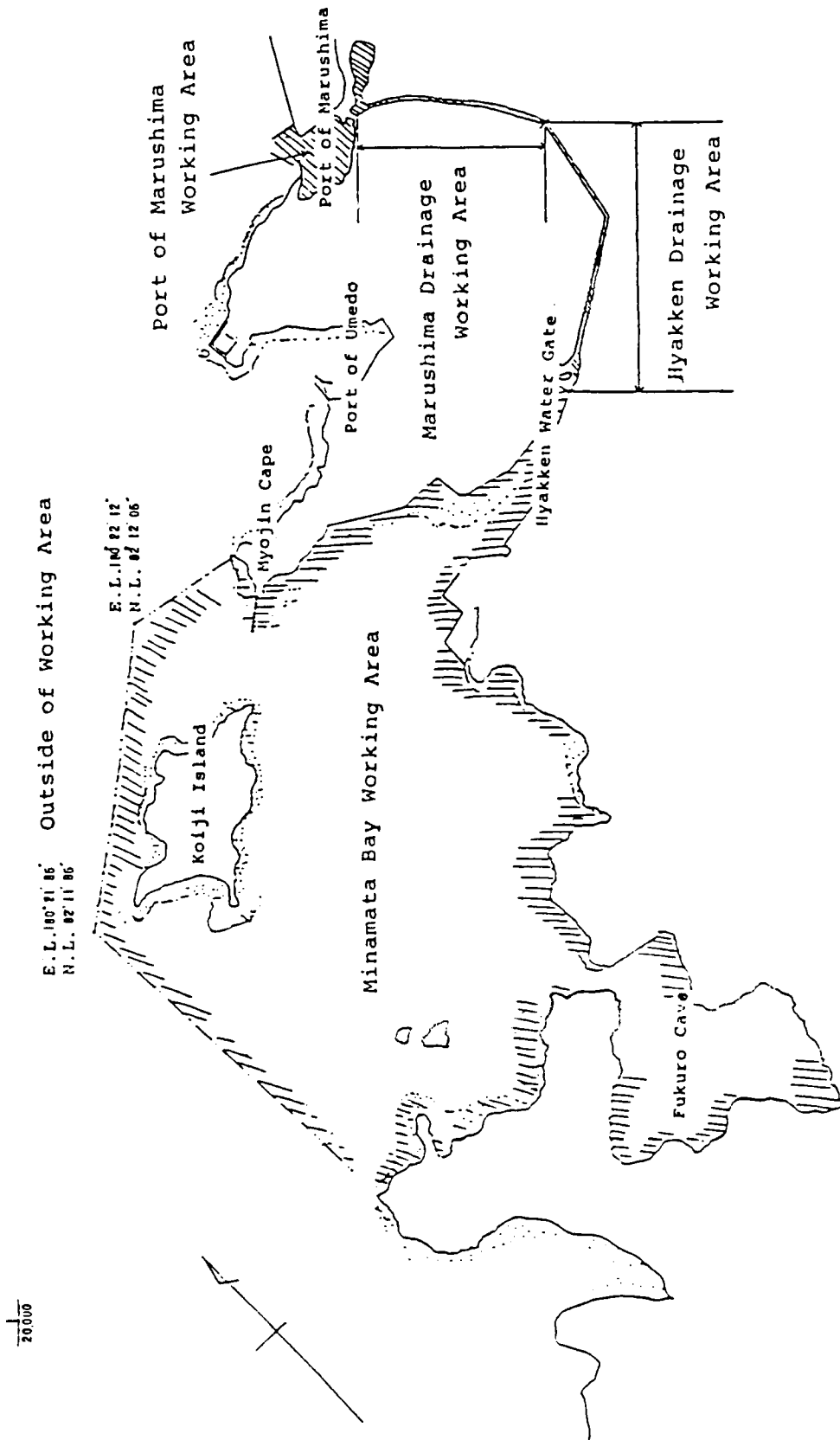


Figure 2. Map of working area

to prevent diffusion of the sediment, and the sand-drain work and sand-compaction work were put into operation. After the sand-compaction work, the bulkhead (762.5 m long) was built by the method of the steel sheet pile cell.

For the dredging, we used the cutterless pump so as not to stir the bottom sediment too much. After the primary sedimentation in the disposal area, the effluent was pumped into the treatment plant and treated with coagulant. After treatment, the effluent was discharged into the bay. At the disposal area, the sediment was covered with seawater 0.5 m thick or more during the dredging in order to inhibit the formation of methyl mercury. After dredging was completed, seawater on the sediment was discharged into the treatment plant, and the surface of the sediment in the disposal area was covered with soil from mountains or sand.

During the dredging in the first working area (experimental dredging work area), 1980 to 1985, no further pollution caused by the dredging work was found. Therefore, we used the same work method in the second working area (main dredging work area). At present, the sand-compaction work, the bulkhead, and the water treatment plant are complete, and the dredging work is under way in the main working area. All of the work in Minamata Bay is scheduled for completion in March 1990.

MONITORING

As we carried out the work to clean the polluted bay, we monitored the water and fishes collected from the bay in order to ascertain the level of environmental pollution caused by stirring the bottom sediment containing mercury; we found no secondary pollution. The monitoring plan is shown in Tables 1 and 2. The monitoring stations are shown in Figure 3.

Water Quality

Primary Stations

Several primary stations were established on the border of the working area to confirm that the water quality outside the working area was not deteriorated by stirring the bottom sediment. If the analysis at these stations showed concentrations over the allowed level of mercury content, we had to stop or slow down the work at once and find the cause of the pollution.

In the sample collected from the primary station, total mercury concentration and turbidity were measured three times per day, and pH, COD, DO, and oil were measured once per day, and arsenic and lead were measured once per week.

Secondary Stations

Several secondary stations were established between the primary stations and the dredging area in order to estimate changes in water quality and to determine if a work suspension would be necessary.

Samples collected here (upper, middle, and bottom layer), measured turbidity every 2 hr during the dredging work. We chose water turbidity as the indicator for mercury content because mercury in Minamata Bay is adsorbed to the bottom sediment, and we could estimate mercury concentration in water

TABLE 1. WATER QUALITY STATIONS

Stations	Parameter	T-Hg	As, Pb	Cd, CN, O-P	Cr ⁶⁺ , M-Hg, PCB	Turbidity	pH	COD	DO	Oil	Nutritive Salts
Primary stations	Frequency	3/day	1/week	--	--	3/day	1/day	1/day	1/day	1/day	--
	Standard quality	0.0005 ppm	As: 0.05 Pb: 0.1 ppm	--	--	--	7.8-8.5	3 ppm	5 ppm	not detectable	--
Secondary station	Frequency	--	--	--	--	5/day	--	--	--	--	--
	Standard quality	--	--	--	--	7 ppm	--	--	--	--	--
Effluent station	Frequency	1/day	1/week	1/week	Continually	Continually	--	--	--	--	--
	Standard quality	0.005 ppm	As: 0.5 Pb: 1 ppm	Cd: 0.1 CN: 1 O-P: 1	--	--	--	--	--	5 ppm	--
Ground water station	Frequency	1/month	--	--	--	--	--	--	--	--	--
	Standard quality	not detectable	--	--	--	--	--	--	--	--	--
Additional stations	Frequency	1/week	--	--	--	--	--	--	--	--	--
	Standard quality	not detectable	--	--	--	--	--	--	--	--	--
Working area	Frequency	2/month	--	--	--	--	--	--	--	--	--
	Standard quality	not detectable	--	--	--	--	--	--	--	--	--
Working area	Frequency	1/week	--	--	--	1/week	1/week	1/week	1/week	--	2/month
	Standard quality	not detectable	--	--	--	--	--	--	--	--	--

TABLE 2. FISH AND PLANKTON STATIONS

Station	Area	Contents
Fish barrier	Defense net	Nets were set along the border of the working area
	Catch net	Five catch nets were set in the working area
Fish station	Outside working area	11 species of fish Total mercury: 4 times/year
	Working area	Mebaru, marble rock fish, perch, gillthead, sasanohabera, ishimochi Total mercury: 1 time/month
Plankton station	Outside working area	Zooplankton Total mercury: 6 times/year
	Working area	Zooplankton Total mercury: 6 times/year
Fish rearing test	Working area	Red sea bream, Mejina Total mercury: 3 times/month

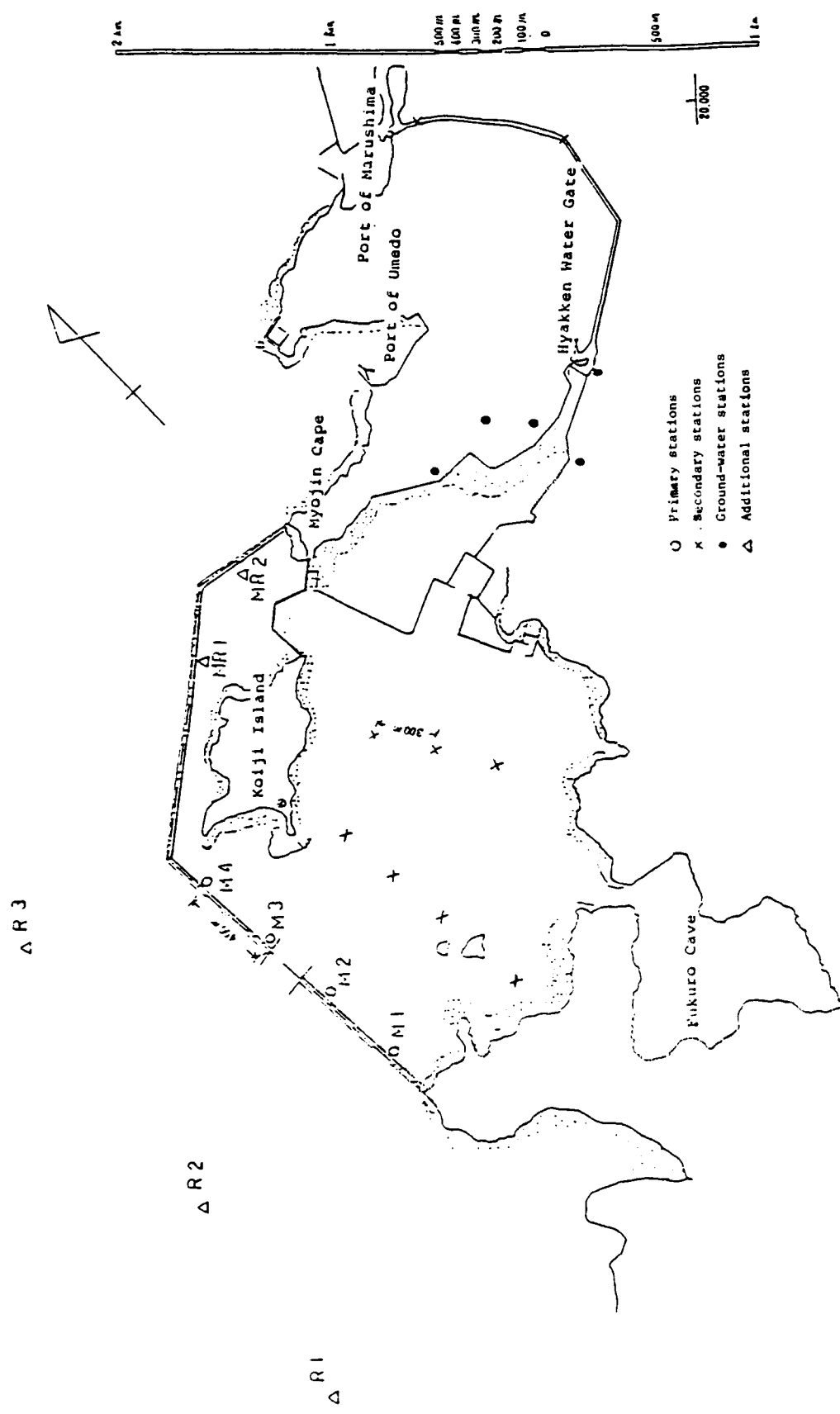


Figure 3. Station locations

using the result from the turbidity measurement. Moreover, the analysis of mercury takes time, but turbidity can be measured in a short time. Therefore, a judgment on the suspension of dredging can be made as soon as possible based on turbidity results.

Before the dredging, the correlation between total mercury concentration and water turbidity of the bottom sediment collected from Minamata Bay was not known. The correlation between total mercury concentration and water turbidity was investigated in an experiment in which bottom sediment collected from Minamata Bay was suspended in seawater (Figure 4). Therefore, turbidity can be used to monitor for deterioration of the water quality caused by stirring of the bottom sediment. The allowable level of water turbidity was flexible, according to the correlation with the mercury concentration in the dredging area.

Effluent Stations

Effluent monitoring in the disposal area was conducted using the self-registering analyzer of the continuous system for the water turbidity at the wastewater treatment plant. We estimated mercury concentration in the effluent by using the turbidity results. Total mercury was measured once per day, and other pollutants were measured once per week.

Working Area Station

Turbidity and oil at the surface in the dredging area were monitored from a boat during the work.

Ground-Water Stations

These stations monitored mercury soaking into the ground water. Total mercury in the ground water was measured once per 3 months. After disposal was completed, total mercury was measured once per month.

Additional Stations

Supplementary analyses were made for mercury in samples collected from outside the working area, and for nutritive salts in seawater from inside the working area.

Marine Products and Plankton

Outside Working Area

Mercury concentration in the fish collected from outside the working area was measured four times per year. If the total mercury concentration in the same kinds of fishes averaged 0.4 ppm or more (guideline, 0.4 ppm), methyl mercury concentration in the fish was measured.

Working Area

In order to determine changes in mercury concentration level in fish, several kinds of fish were caught using the net, and mercury concentrations were measured.

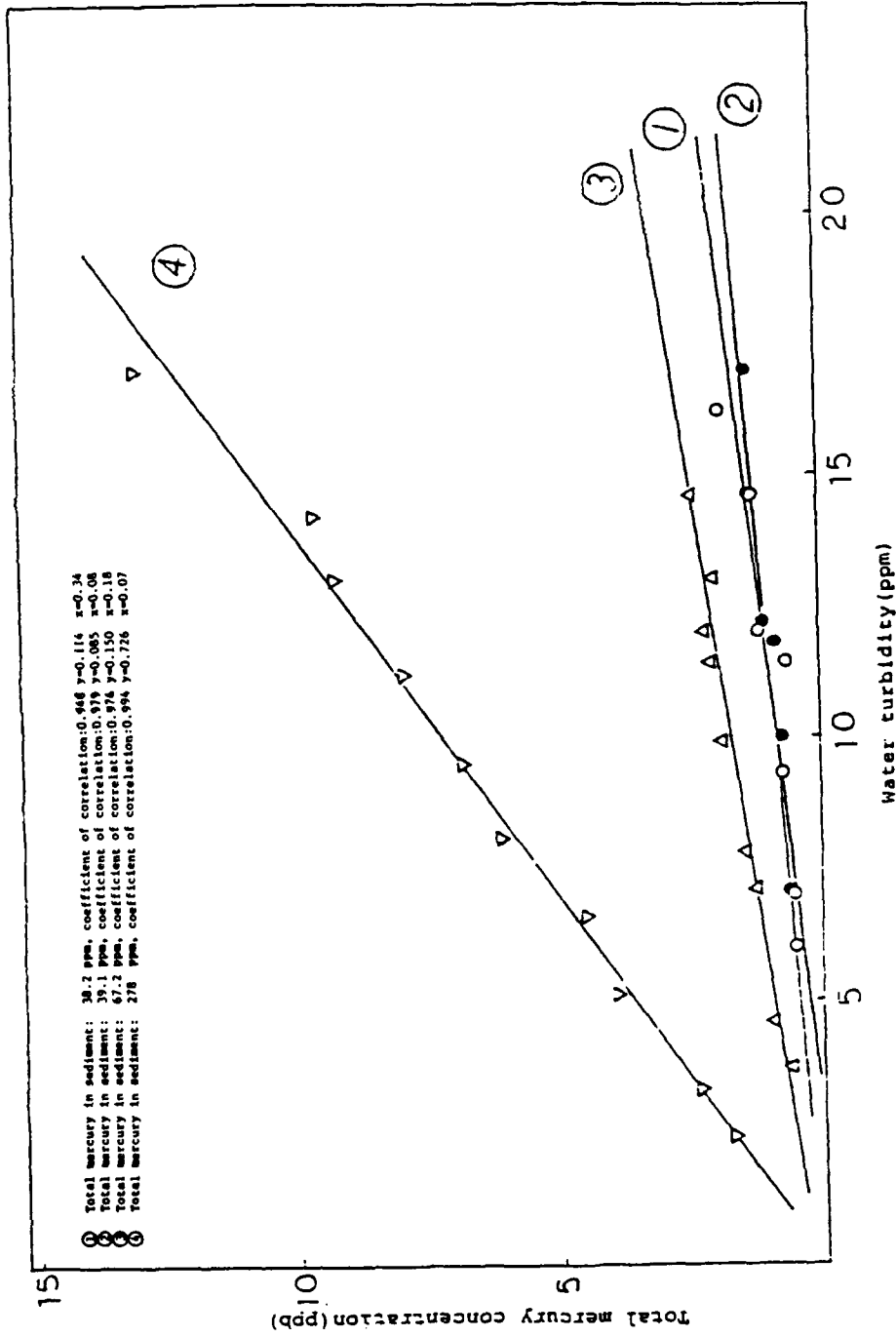


Figure 4. Correlation between total mercury concentration and turbidity of bottom sediment collected from Minamata Bay

Rearing Test of Fish

In order to know the precise effect (mercury bioaccumulation) of dredging on the fish as soon as possible, red sea bream and mejina were reared in a cage in the sea near the dredging work area. Total mercury concentration in the fish collected in the rearing cage was measured every 10 days.

Plankton

Zooplankton was collected in the working area and outside the working area every 2 months, and the total mercury concentration in the zooplankton was measured.

MONITORING RESULTS

Water Quality

Results from the primary stations are as follows:

- a. Total mercury. Total mercury in 3,600 to 3,700 samples every year was measured, and mercury concentration was not detectable in all samples.
- b. Turbidity. Changes in turbidity were caused by rain or red tide. The change caused the dredging work was not observed. The transition in monthly average turbidity in the past several years is shown in Figure 5.
- c. Other pollutants. Oil was not detected in any sample. Other pollutants showed the transition caused by natural phenomena, but no change caused by the dredging work was observed. The results are shown in Figure 6.
- d. Arsenic. The concentration of arsenic was 0 to 0.002 ppm.

The turbidity results at the secondary station are shown in Table 3 and Figure 5. A change in turbidity caused by rain or red tide was seen, but no change caused by dredging was observed.

The turbidity changes caused by rain and red tide can be seen in Figure 7. When it rains, large amounts of sediment-laden water stream into the upper layer of seawater; therefore, the turbidity of the upper layer increases and the chlorine ion in upper layer decreases. Red tide is formed when phytoplankton and DO in the upper layer increase; phytoplanktons increase in the upper layer, which further increases turbidity in the upper layer.

The results from all samples at the effluent station show concentrations and water turbidity less than the required level.

Total mercury concentrations in all ground-water samples were not detectable, and the transition of chlorine ion was not observed.

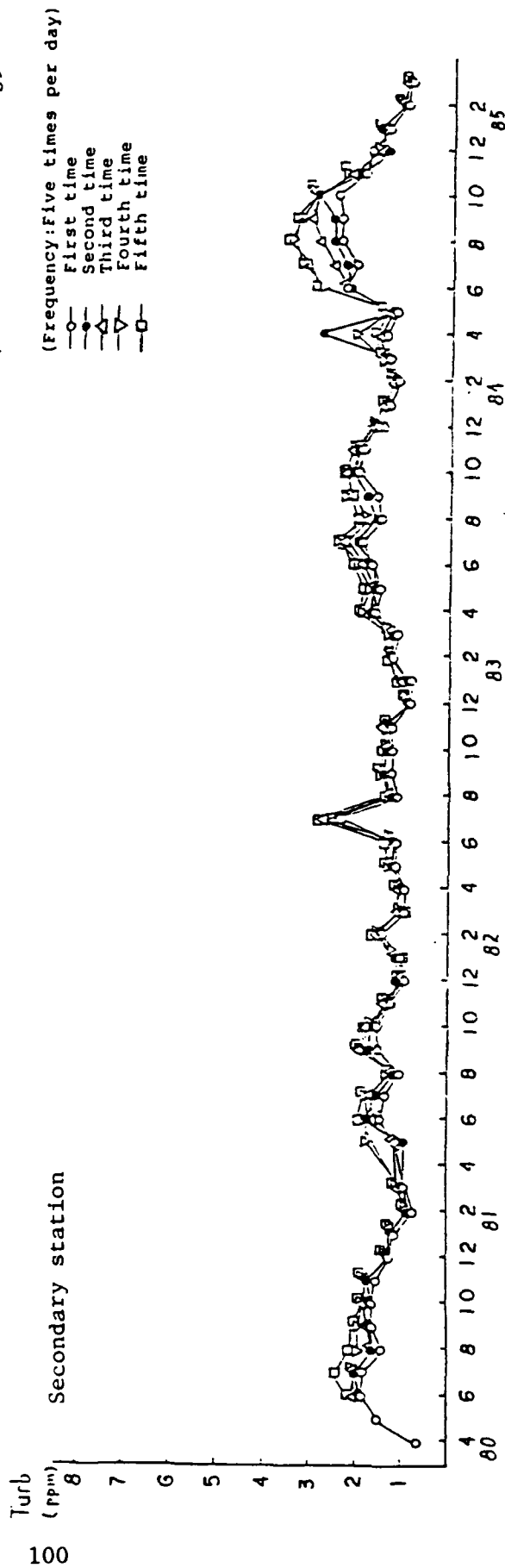
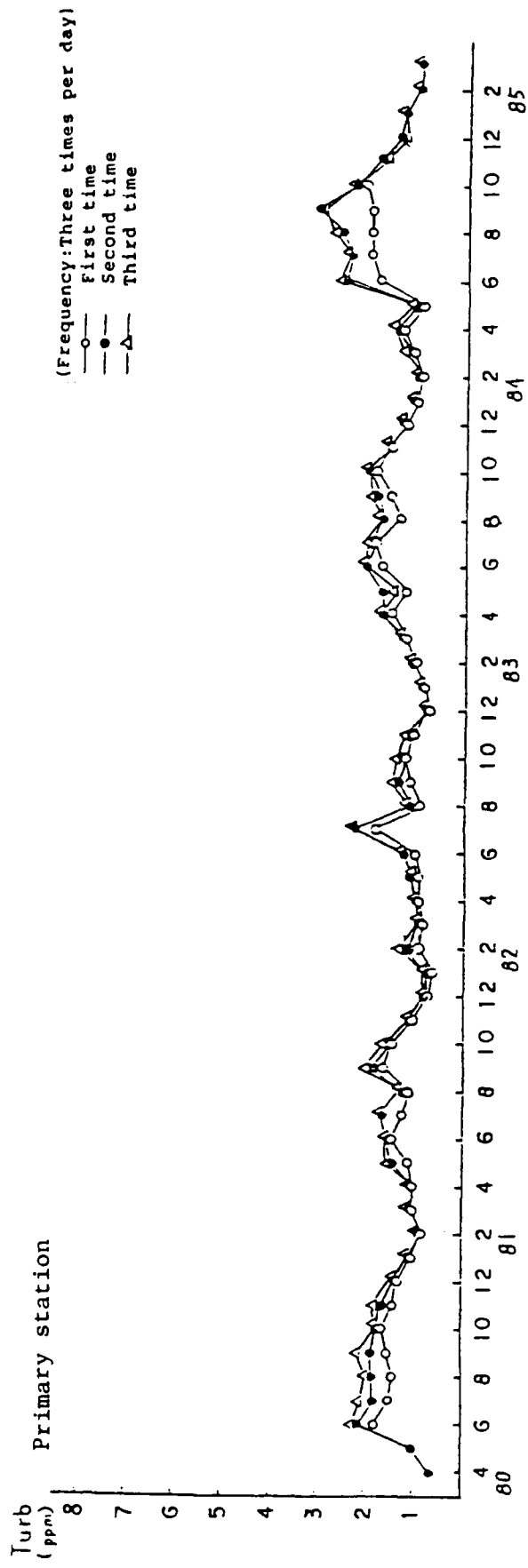


Figure 5. Water turbidity at primary and secondary stations

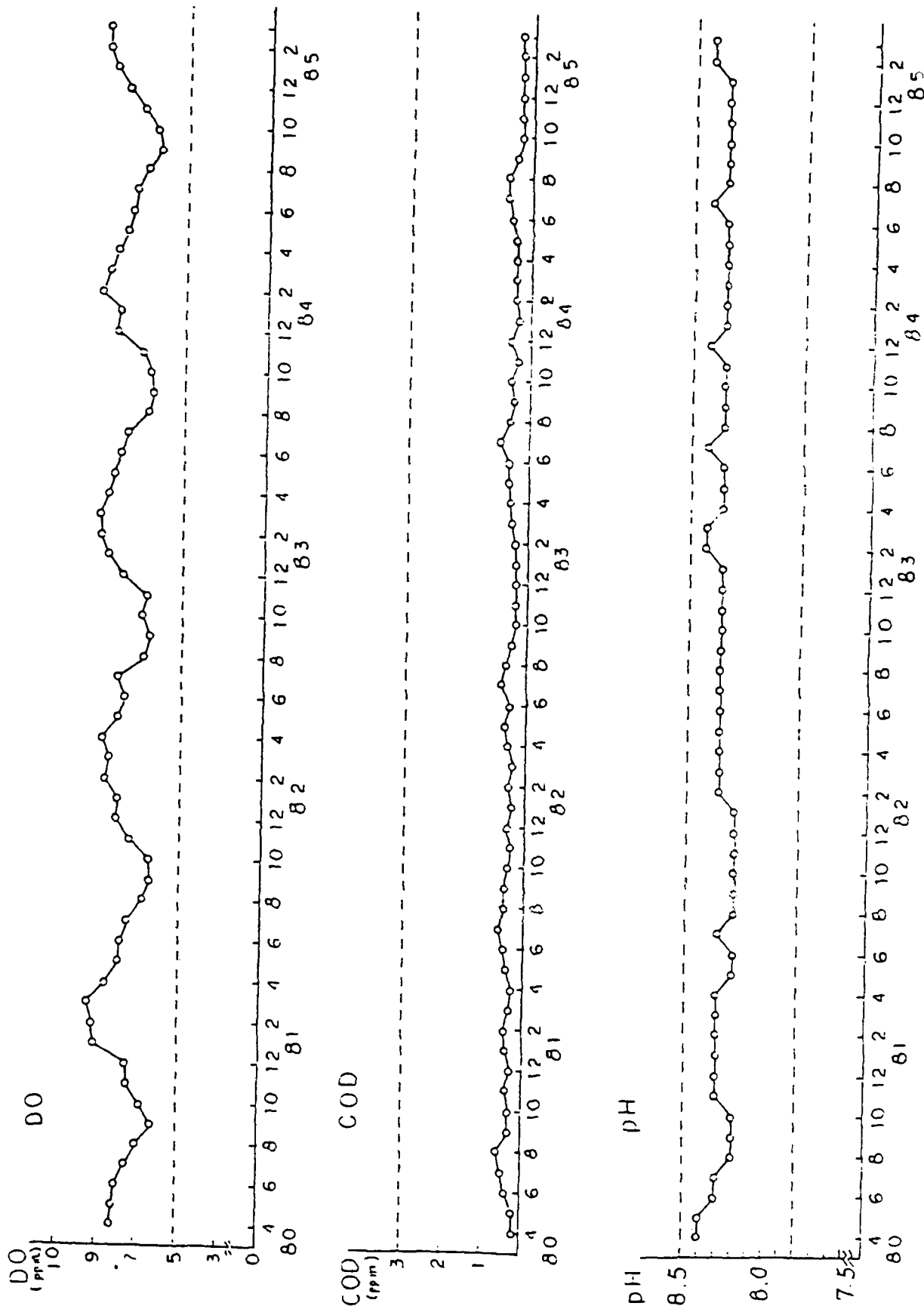


Figure 6. DO, COD, and pH at primary station M-2

TABLE 3. WATER TURBIDITY RESULTS IN SECONDARY STATION S-2

Year Times	1980					1981					1982					1983					1984				
	1	2	3	4	5	6	1	2	3	4	5	6	1	2	3	4	5	6	1	2	3	4	5	6	
Apr. Max.	1.0																								
Apr. Min.	0.3																								
Apr. Ave.	0.6																								
May Max.	4.6																								
May Min.	0.5																								
May Ave.	1.5																								
Jun Max.	3.9	4.5	3.6	3.9	4.7																				
Jun Min.	0.7	0.7	0.9	0.7	0.9																				
Jun Ave.	1.8	1.8	1.9	1.9	2.0																				
Jun Max.	2.7	2.6	2.5	2.7	3.3																				
Jul Min.	1.1	1.2	0.9	1.0	2.0																				
Jul Ave.	1.8	1.9	1.9	2.1	2.4																				
Jul Max.	2.4	2.7	3.5	4.0	4.1																				
Aug Min.	0.7	0.8	0.9	0.9	0.8																				
Aug Ave.	1.4	1.6	1.6	1.9	2.1																				
Aug Max.	3.0	2.7	2.4	2.6	2.8																				
Sep Min.	0.6	0.5	0.7	0.5	0.7																				
Sep Ave.	1.6	1.6	1.7	1.9	1.9																				
Sep Max.	2.2	2.2	2.4	2.4	2.4																				
Oct Min.	0.7	0.6	0.8	0.7	0.8																				
Oct Ave.	1.6	1.6	1.8	1.8	1.7																				
Oct Max.	2.0	3.1	2.5	3.6	2.3																				
Nov Min.	1.2	1.2	1.1	1.3	1.2																				
Nov Ave.	1.5	1.7	1.7	1.8	1.6																				
Nov Max.	1.9	2.3	1.9	1.9	1.9																				
Dec Min.	0.7	0.7	0.7	0.8	0.8																				
Dec Ave.	1.2	1.2	1.2	1.2	1.2																				
Dec Max.	1.7	1.7	1.9	1.7	1.7																				
Jan Min.	0.6	0.6	0.6	0.6	0.6																				
Jan Ave.	1.1	1.1	1.1	1.1	1.1																				
Jan Max.	1.1	1.2	1.3	1.2	1.1																				
Feb Min.	0.5	0.5	0.4	0.3	0.5																				
Feb Ave.	0.7	0.8	0.8	0.8	0.8																				
Feb Max.	1.2	1.2	1.3	1.5	1.6																				
Mar Min.	0.6	0.7	0.7	0.7	0.7																				
Mar Ave.	0.9	0.9	1.0	1.0	1.1																				
Mar Max.	4.6	4.5	3.6	4.0	4.7																				
Year Min.	0.3	0.5	0.4	0.3	0.5																				
Year Ave.	1.3	1.4	1.5	1.6	1.6																				

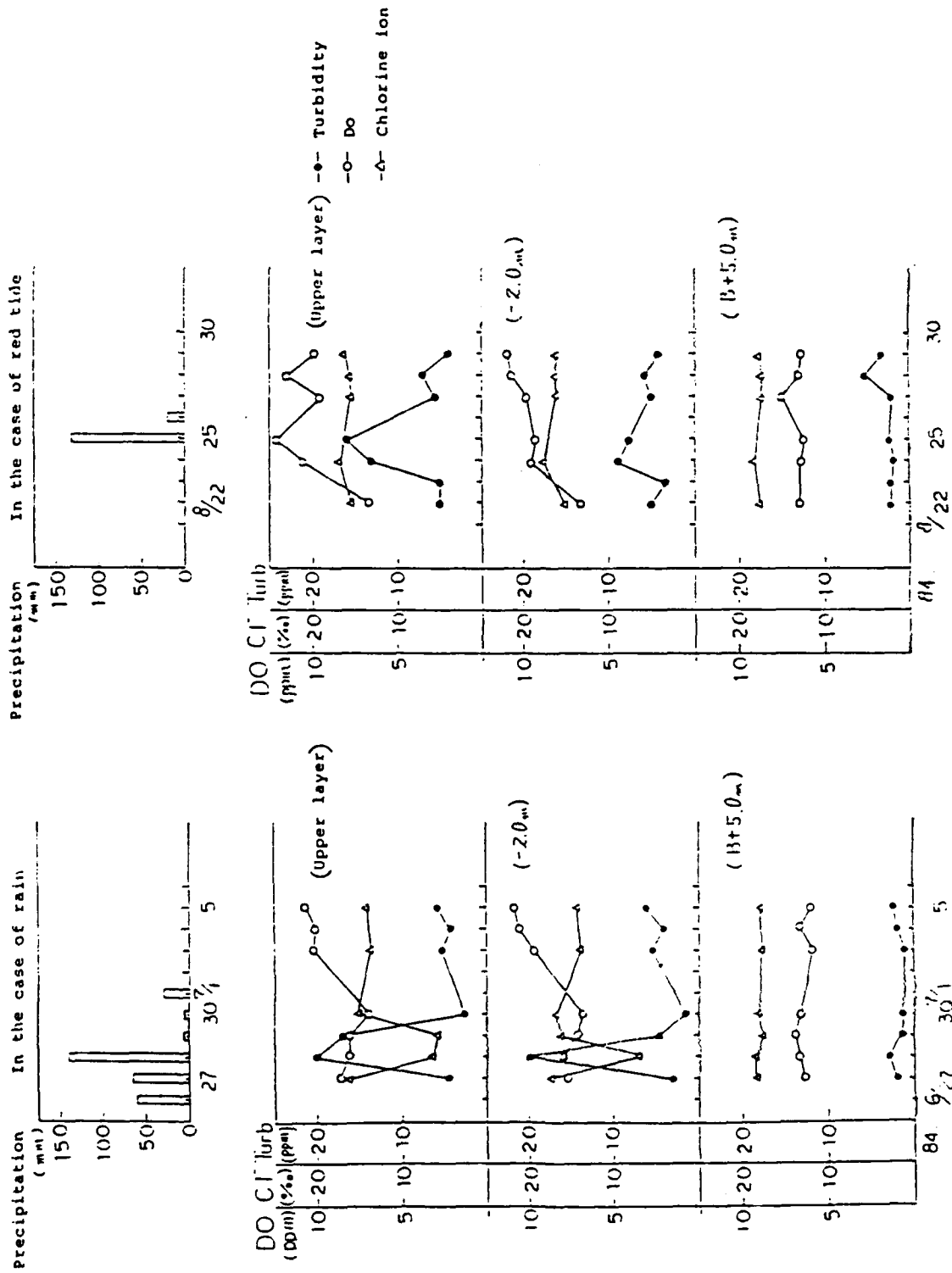


Figure 7. Water quality changes caused by rain and red tide

Marine Products and Plankton

Outside Working Area

Total mercury in all samples was less than the guideline (0.4 ppm).

Working Area

The variation of mercury concentration in fish correlated with the body length of the fish. A standard body length was selected based on data obtained from each species of fish.

Mercury concentration data in the fish were corrected to the body length, and the corrected datum (standardized datum) was used in the monitoring results.

Overall, the effect caused by the dredging was not found on fish in the working area (Figure 8).

Fish-Rearing Tests

Rearing tests of fish were carried out every 6 months for 4 years. The yearly test results were then compared. From the results of these tests, no effect caused by dredging was found (Figure 9).

Plankton

Monitoring results showed the effect caused by dredging.

CONCLUSIONS

As we dredge Minamata Bay, it is important that water quality not deteriorate because of the stirring of the bottom sediment and the environment not be polluted. We must monitor for secondary pollution and decide immediately whether or not to suspend dredging. Therefore, turbidity was chosen as the indicator for mercury content because the analysis for mercury takes a long time and we could measure the turbidity in a short time.

Since dredging began, no increase in mercury concentration was detected, and in the fish-rearing (red sea bream and mejina used), no change was observed in mercury concentration in the fish by bioaccumulation. Also, in the fish and plankton from the working area, no effect caused by the dredging was observed.

These results showed that dredging Minamata Bay did not form secondary pollution of the environment, and the work was carried out very safely.

It can be concluded from this study that the use of turbidity as the indicator of water quality and the use of fish as the indicator of bioaccumulation grasp exactly the effect caused by the dredging work.

At present, the main dredging is under way, and we will continue to follow the monitoring program described herein.

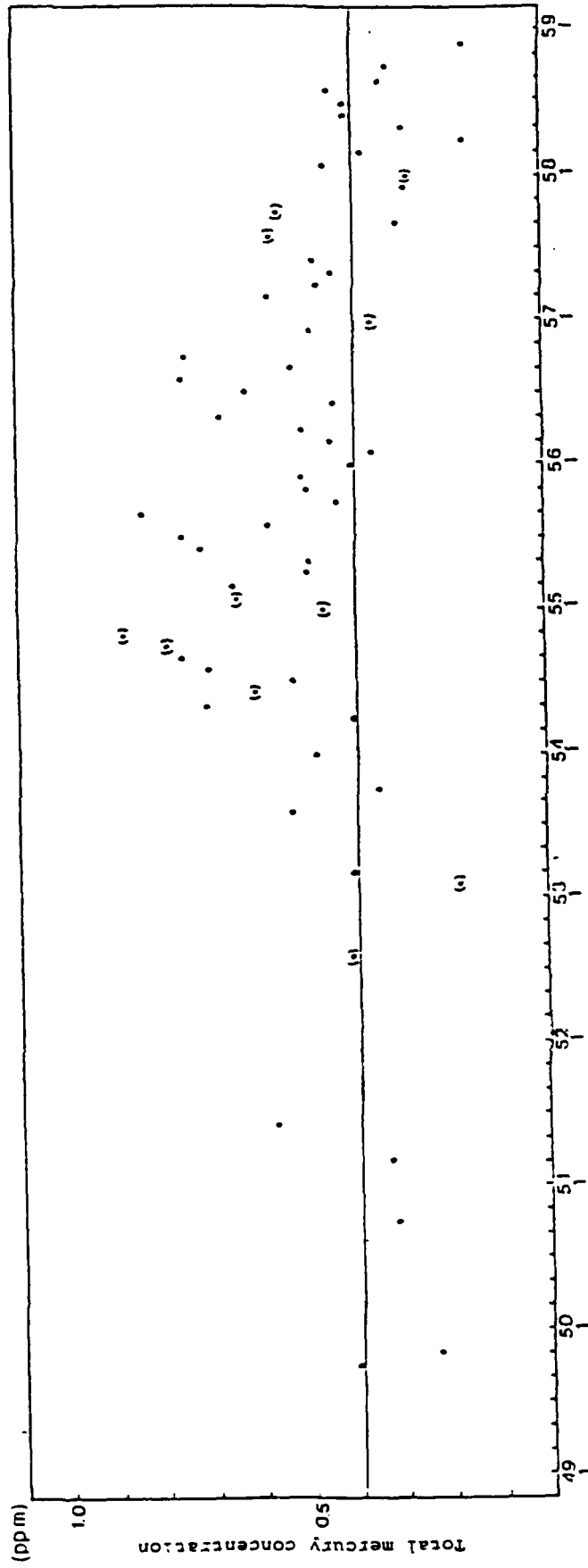


Figure 8. Total mercury concentration in gilthead

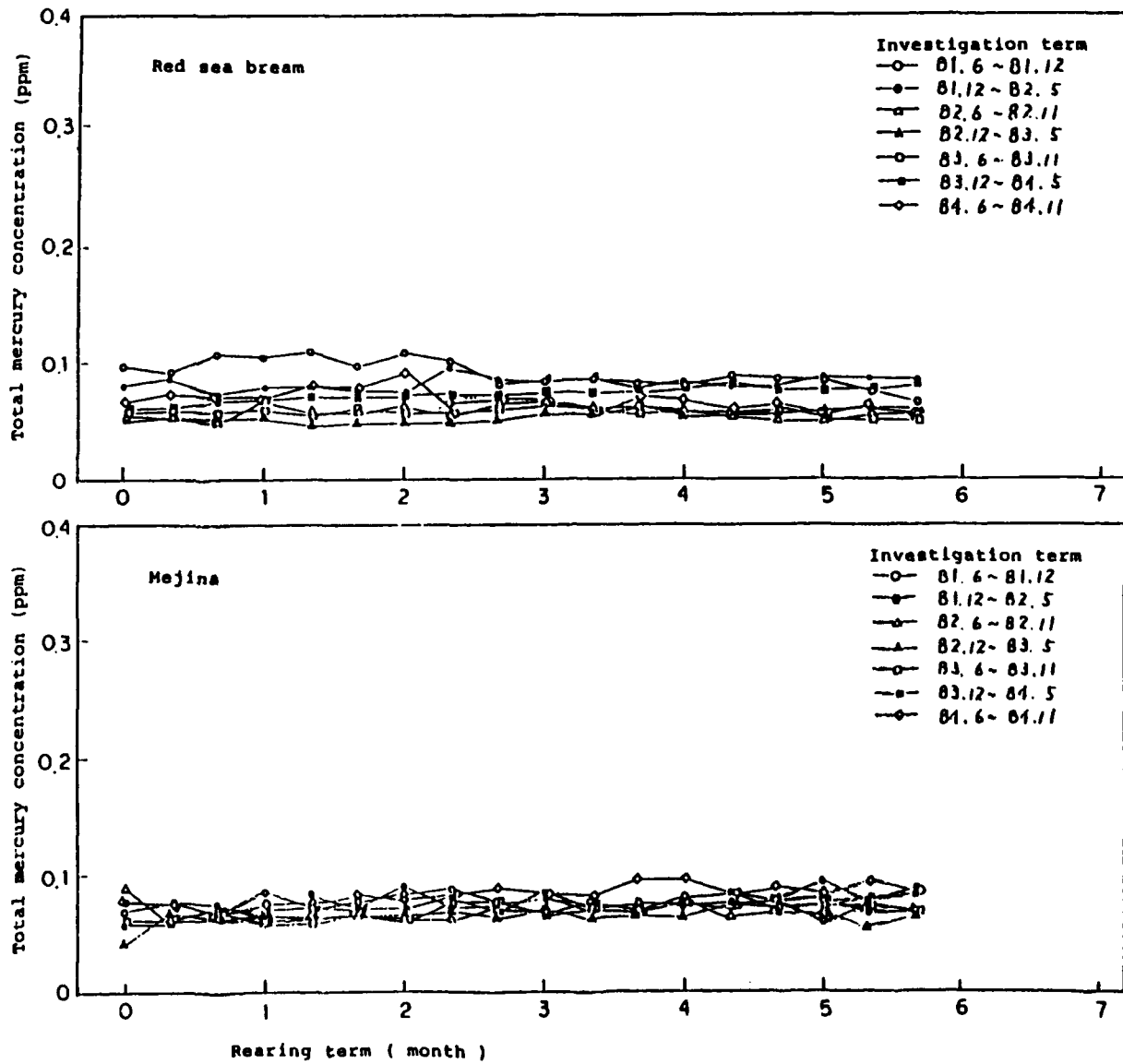


Figure 9. Results of fish-rearing tests

CLEANER SEDIMENTS IN HIROSHIMA BAY--A TRIAL CLEANING PROJECT
USING SAND COVERING IN KURE INLET

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AD-P006 901

INTRODUCTION

In recent years, the quality of water and benthic sediments in many of Japan's enclosed bays has been deteriorating, partly because of the topography which results in infrequent replacement of bay water with fresh outside seawater, and partly because of the inflow of organic and other objectionable substances from inland areas. These conditions often cause serious problems in utilizing the bay as fishery grounds and in maintaining an environmentally sound bay.

In the mid-1960's, strict regulations were enforced controlling the discharge of pollutants into the sea. At the same time sewage and waste treatment facilities were improved. Thus, the inflowing pollutant loads were reduced; however, this does not mean that bay water quality has been definitely improved because red tides are still reported very often.

In view of this, the 3rd District Port Construction Bureau, Ministry of Transport, set up a site to test the sand covering method. The test site is in Hiroshima Bay (which includes Kure Cove), one of the highly polluted enclosed bays. Sand covering is believed to be one of the best methods for achieving environmental improvement of sea areas. Its effects are being carefully studied in this experimental project.

This paper introduces the outline of the experimental work implemented during 1979 and 1980 and its effects on cleaning the bay sediments.

POLLUTION IN HIROSHIMA BAY

As shown in Figure 1, Hiroshima Bay is a sea area surrounded by Yashirojima Island and Kurahashijima Island. It has an area of 946 km², a volume of 24.2 km³, and a mean water depth of 25.6 m. Many small islands are scattered in the bay, and many rivers (the largest being the Ohta River with flow rates of 100.6 m³/sec in summer and 76.1 m³/sec in winter) flow into the bay. Thus, pollutant loads generated by the inhabitants and the industries concentrated in the watersheds of these tributary rivers are carried to the bay.

The complicated topography of Hiroshima Bay is reflected in its current flow. The current flows into the bay through Hashirajima Channel, around the bay clockwise, and out toward Kure Cove when the tide is rising; however, during ebb tide, the current flows through Kure Cove, around the bay counterclockwise, and out through Hashirajima Channel. The prevailing current, however, comes into the bay through the channel between Hashirajima Island and



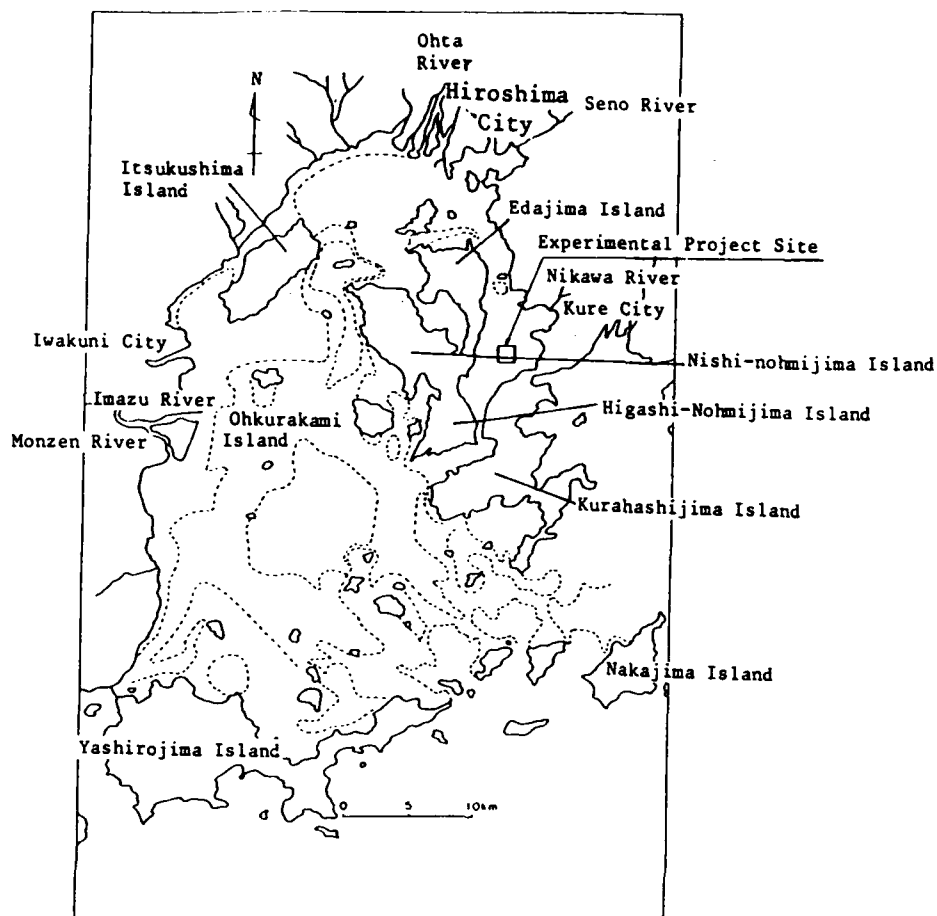


Figure 1. Hiroshima Bay area

Kurahashijima Island and flows northward in the bay. In the innermost area of the bay and in Kure Cove, the prevailing current is comparatively weak and tends to flow eastward.

The prevailing winds in Kure are from the northeast or north-northeast and from the west-southwest. Wind velocity is almost always less than 5 m/sec under normal weather conditions.

Pollution in Hiroshima Bay is discussed in the following paragraphs.

Water Quality

Water quality characteristics of the surface of Hiroshima Bay are indicated in Figure 2 and Table 1. The transparency of the bay water off the innermost coast is as low as 2 m or less, apparently indicating that the bay water is very polluted. In this area of low transparency, COD is as high as 3 mg/l, and T-P and T-N are also high, 0.03 mg/l and 0.4 mg/l, respectively, and higher concentrations are observed at locations nearer to the innermost coast.

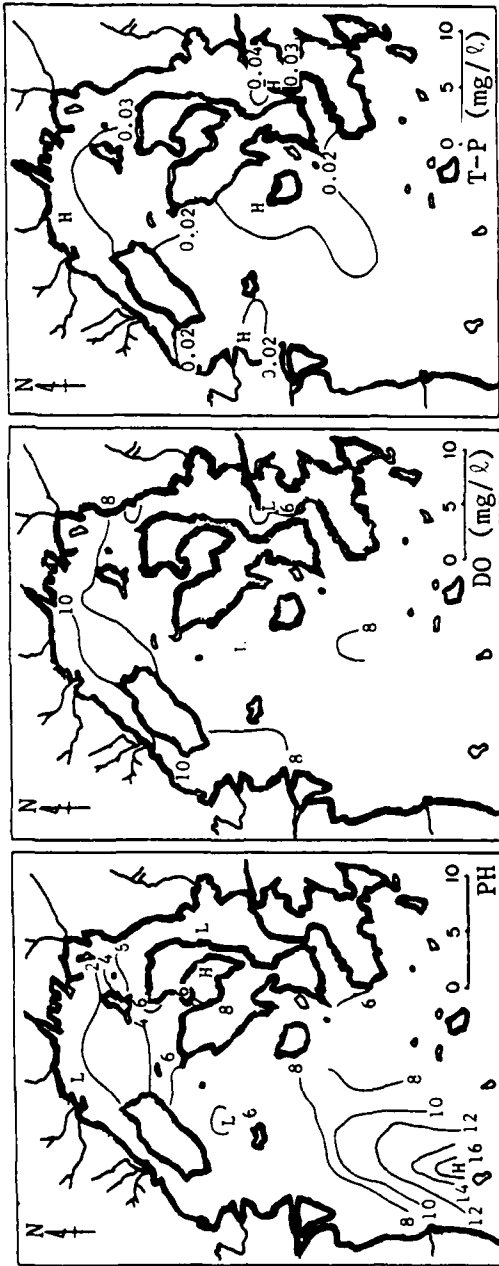
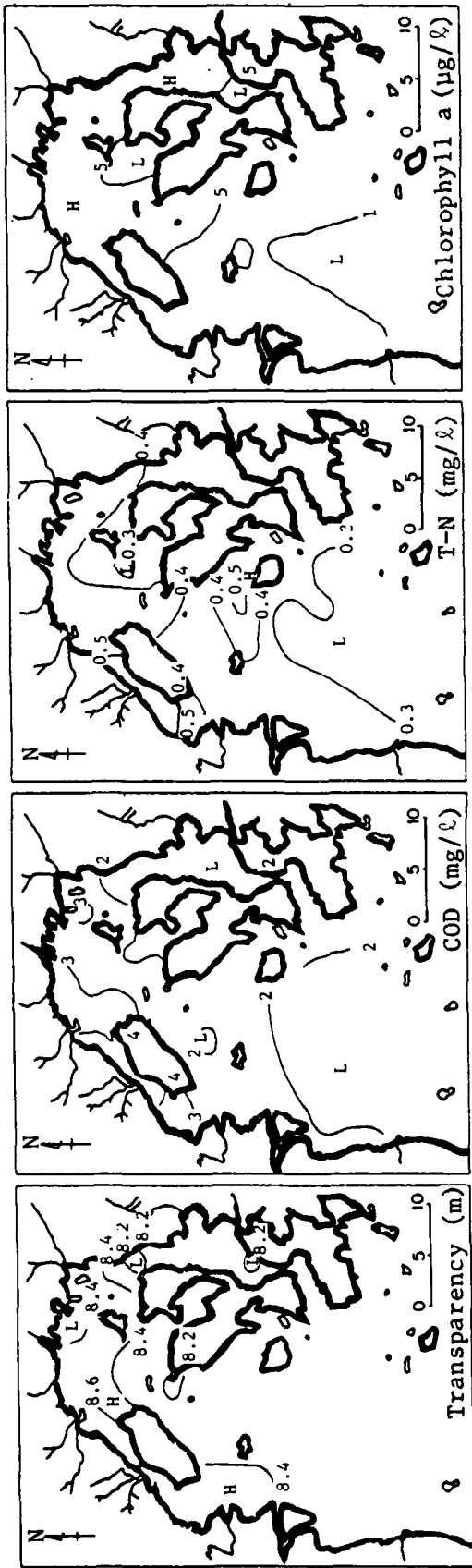


Figure 2. Water quality in the surface stratum (2 m below sea level), Aug 23-28. (Survey conducted for entire area of Hiroshima Bay in 1982.)

TABLE 1. SURFACE STRATUM IN WATER QUALITY

Characteristic	Sea Area					
	Innermost Area	Kure Bay Channel	Edajima Cove	Kure Cove	Entire Area	
					Average	Range
Transparency, m	2.4	5.9	3.0	4.8	8.3	2.8 ~ 14.5
pH	8.5	8.2	8.3	8.2	8.3	8.3 ~ 8.5
DO, mg/l	9.3	6.0	7.1	5.6	8.6	8.0 ~ 10.0
COD, mg/l	3.4	1.5	2.0	1.4	1.8	1.0 ~ 3.5
T-N, mg/l	0.44	0.37	0.37	0.34	0.21	0.14 ~ 0.39
T-P, mg/l	0.031	0.035	0.016	0.039	0.018	0.010 ~ 0.045
Chlorophyll a, $\mu\text{g/l}$	7.9	5.5	1.6	3.7	3.2	0.2 ~ 23.5

Chlorophyll a, the index of primary production, has a concentration not less than 5 $\mu\text{g/l}$ in the bay area surrounded by Miyajima Island and Nishi-Nohmijima Island; the concentration tends to become higher at the sites nearer to the innermost coast, as in the case with nutrient salts.

Benthic Sediments

The conditions of the surface layer of the benthic sediments in Hiroshima Bay, as indicated in Table 2, show the existence of sediments with high organic matter content in Edajima Cove and Kure Cove.

TABLE 2. QUALITY OF THE SURFACE LAYER OF SEDIMENTS

Characteristic	Sea Area					
	Innermost Area	Kure Bay Channel	Edajima Cove	Kure Cove	Entire Area	
					Average	Range
Ignition loss, %	10	11	12	10	9.6	2.7 ~ 12
COD, mg/g	41	42	55	54	36	3.1 ~ 62
Sulfide, mg/g	0.28	0.17	0.65	0.82	0.24	0.04 ~ 0.85
T-P, mg/g	0.67	0.77	0.74	0.79	0.71	0.43 ~ 0.86
T-N, mg/g	2.2	2.4	2.8	2.8	2.1	0.48 ~ 2.8

To illustrate the distribution of surface layer areas of the sediments polluted by organic matter, the bay areas with COD of 4 mg/g or more, ignition loss (IL) of 10 percent or more, and sulfide content of 0.5 mg/g or more are detailed in Figure 3. The areas with identical pollution conditions are, as shown in Figure 4, Kure Cove, Edajima Cove, and the innermost coastal water area of the bay. Pollution is particularly evident in these areas.

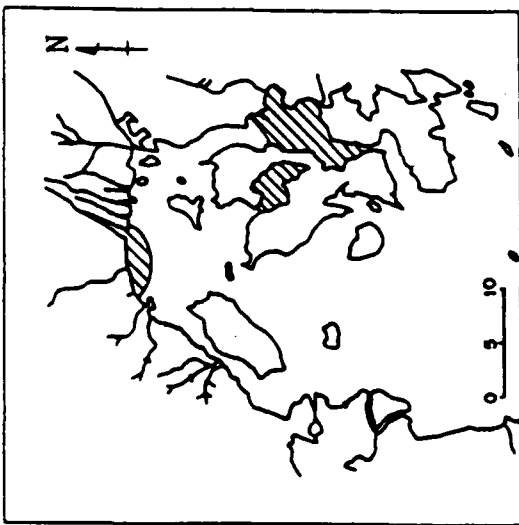
Biota

Figure 5 shows the summer plankton levels in Hiroshima Bay. The quantitative distribution of phytoplankton cells indicates that a large number of cells are produced in Edajima Cove, off the innermost coast of the bay, and in Kure Cove. No less than 20 species of phytoplankton cells were found off the innermost coast and in Kure Cove, and no more than 10 species were found in Edajima Cove. Of the 200 plankton organisms, a total of 56 species were found, and the average population was 3.5×10^4 organisms/m³; the population tended to increase in the water area around Miyajima Island. In the water area off the innermost coast of the bay, the species of emerging plankton were limited, indicating that this water area is less diversified.

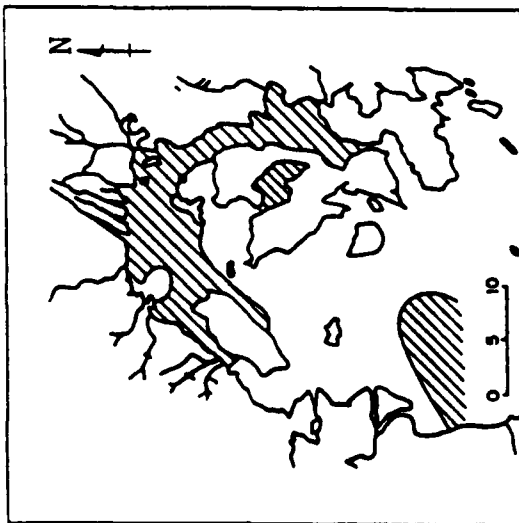
The summer distribution of benthic biota in Hiroshima Bay is shown in Figure 6. While the figure indicates a tendency for the biomass to increase in the area, the fact that most of the biomass can be accounted for by polychette worms suggests that the bay area is in an advanced stage of organic contamination.

Red Tide

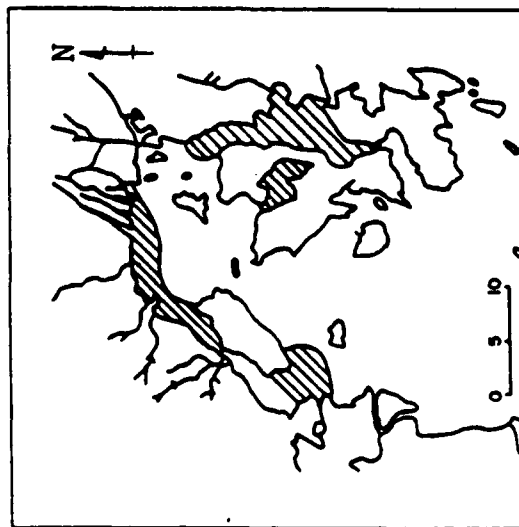
The number of red tide occurrences in Aki Channel is indicated in Figure 7, and the number of cases where damages were caused by the red tide is indicated in Table 3. The occurrences are less frequent today than in the late 1960's; however, approximately 10 occurrences a year are still being reported.



a. Areas with COD
≥ 40 mg/g



b. Areas with IL
≥ 10%



c. Areas with sulfide
content ≥ 0.5 mg/g
or more

Figure 3. Distribution of sediments in surface layer (0-5 cm)

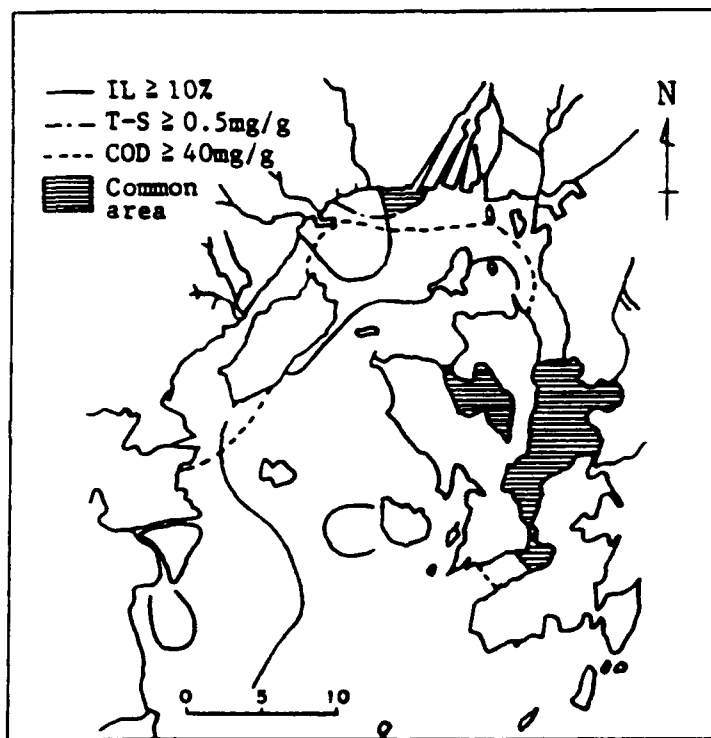
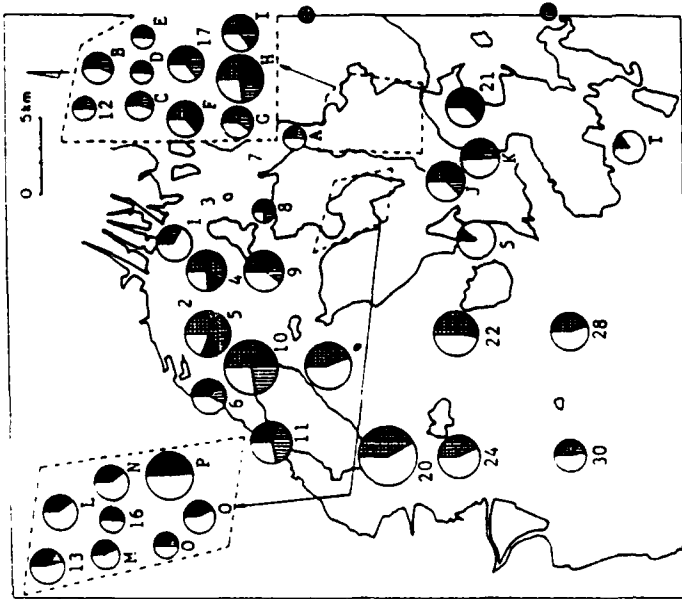
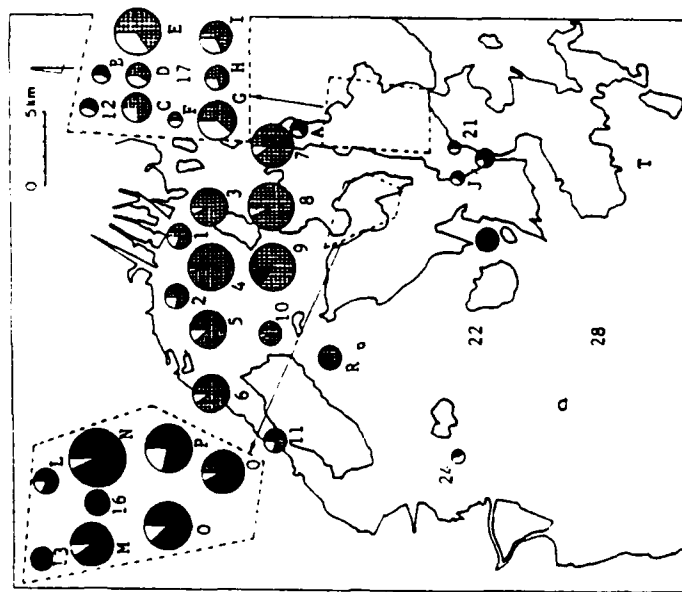
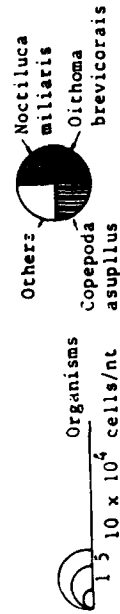


Figure 4. Areas where sediments are polluted by organic matter



The species whose organism number ratios at all survey points totaled 10% or more were indicated in the figure. Where the ratio of any species was 10% or higher at each survey point, the ratio is indicated in a pie chart. Φ means 10,000 organisms/m³ or less.



The species whose cell number ratios at all survey points totaled 10% or more were indicated in the figure. Where the ratio of any species was 10% or higher at each survey point, the ratio is indicated in a pie chart. Φ means 100 cells/ml or less.

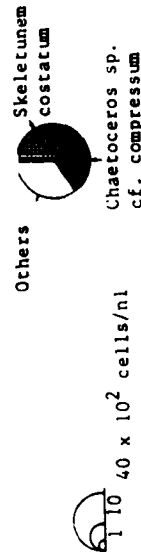


Figure 5. Horizontal distribution of plankton (Hiroshima Bay, Aug 1983)

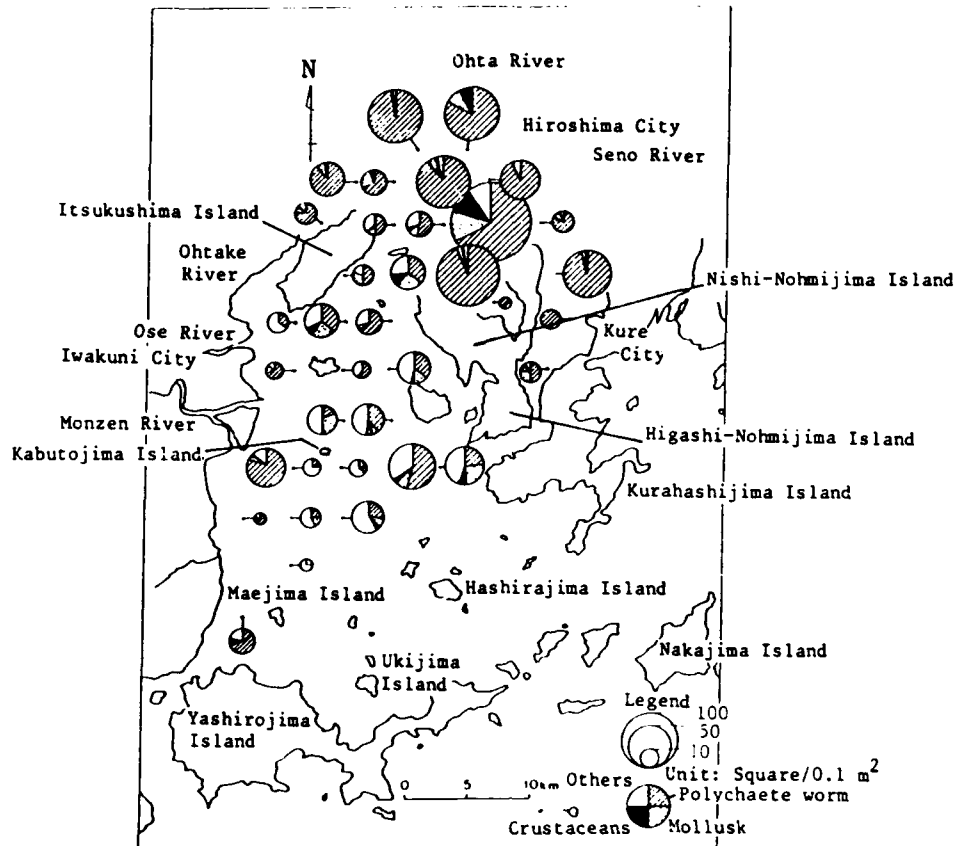


Figure 6. Distribution of benthic biota (Aug 1982)

Figure 7 indicates the monthly distribution of red tide occurrences in Hiroshima Bay. No red tide occurred in those months not shown in the figure, i.e., January, February, November, and December (survey for the year 1982).

The area in which red tide occurred extended from the innermost coast to Kure Cove, and red tide occurrences were seen every year in this specific part of the bay.

BENTHIC SEDIMENT CLEANING PROJECT

As mentioned earlier, environmental pollution has been worsening in Hiroshima Bay. To alleviate this situation, it is essential that the inflowing pollutant load be reduced and at the same time the self-cleaning ability of the bay itself be restored. Benthic sediment cleaning was conceived as a means to break the cycle of nutrient salts and benthic sediment (which serves as a cyclic medium) and thereby reduce the release rate of nutrient salts. To be more specific, benthic sediment cleaning removes or covers eutrophic sediment deposited at the sea bottom in order to suppress the release of nutrient salts, prevents oxygen depletion in seawater at the bottom water layer, and thus restores the self-cleaning ability of the bay area. Before implementation of a full-scale benthic sediment cleaning project, a small-scale

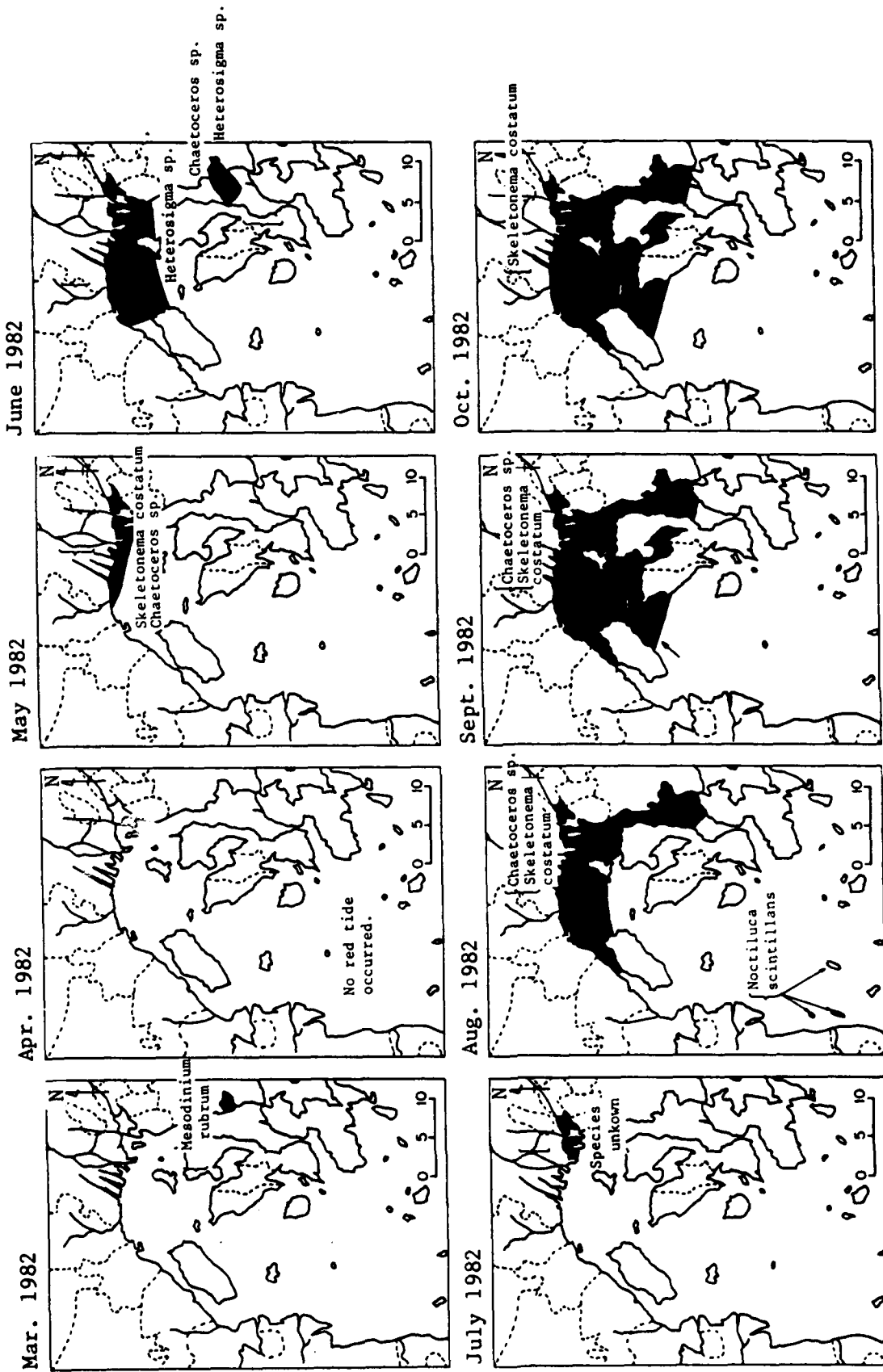


Figure 7. Monthly distribution of red tide occurrences in Hiroshima Bay

TABLE 3. RED TIDE OCCURRENCES IN AKI CHANNEL*

	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985
No. of red tide occurrences	11	11	14	24	18	14	18	8	12	8	13	11	14	10	8
No. of cases where damage was caused			1	2	4		2		1						

* After "Environmental Preservation in Seto Inland Sea."

experimental project using the sand covering method was undertaken in Hiroshima Bay; its effects on environmental improvement are still being studied. This experimental project will be described in the following paragraphs.

Experimental Project

Outline of the project. The outline of this experimental project is indicated in Table 4.

TABLE 4. HIROSHIMA BAY EXPERIMENTAL PROJECT

	1979	1980
Site	Off the southeastern of Edajima Island (-21 m)	
Project period	Oct 1979 - Nov 1979	July 1980 - Aug 1980
Work implemented	Sand covering 1.92 ha (120 m × 160 m) Thickness of sand cover: 0.5 m	Sand covering 4.48 ha (200 m × 200 m + 80 m × 60 m) Thickness of sand cover: 0.3 m
Ships used	Conveyor barge (2,000 m ³)	Unloader ship (260 m ³ /hr)
Covering material	Sea sand	Sea sand

Project site. The portion of the bay chosen for the site of this experimental project was a tightly enclosed area located off the southeastern coast of Edajima Island (Figure 8). Pollution was particularly evident in this area. Figure 9 shows the scope and profile of the experimental project.

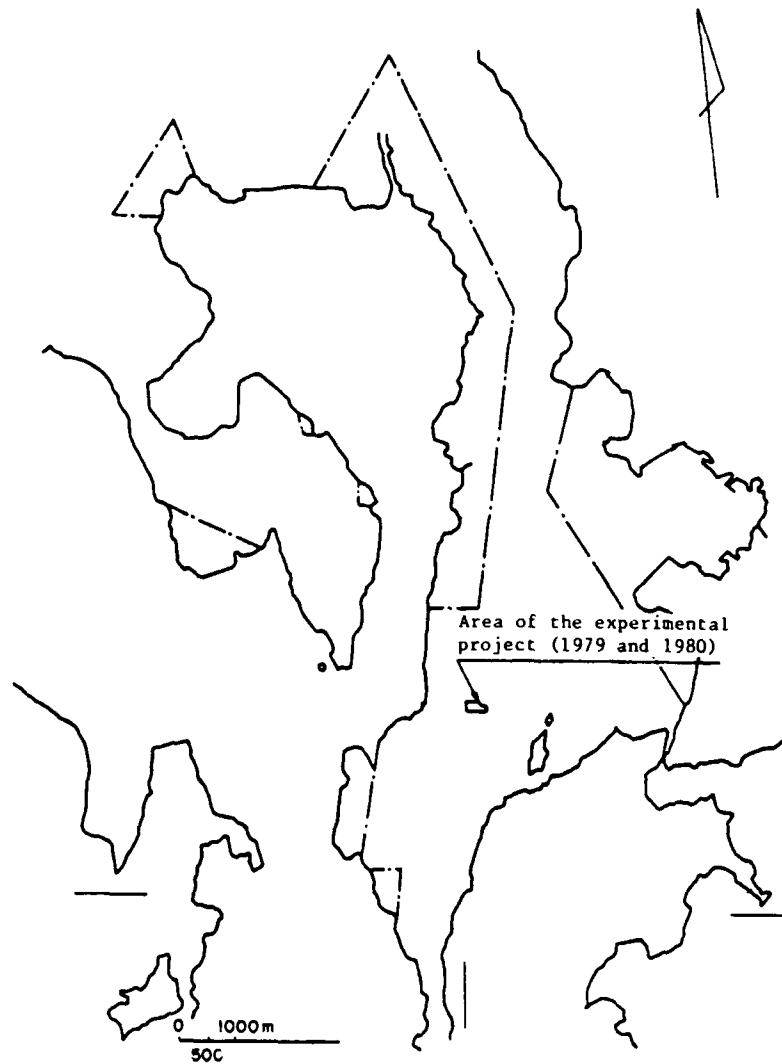


Figure 8. Site of the experimental project

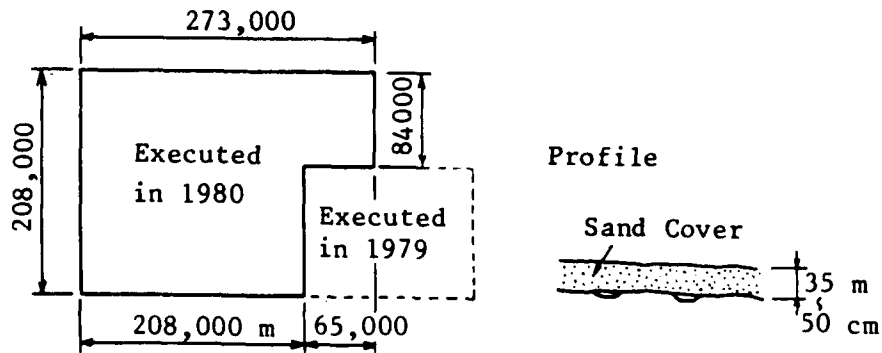


Figure 9. Scope and profile of the experimental project

Working method and precision of the work. The work was conducted using two different methods: one using a conveyor barge, and the other using a barge unloader (Figure 10). After the work was completed, cylindrical specimens were sampled and measured by the pole penetration method. This measurement indicated that the actual sand covering thickness was 99 to 108 percent of the design. From this, it may be stated that the design sand thickness was well satisfied. (Sand taken from the sea bottom about 50 km away from Kure Cove was used as covering material.)

Follow-Up Surveys of Effects

In order to determine the effects of the experimental project, a series of follow-up surveys was conducted. These surveys were used to determine how the release rates of organic substances and nutrient salts from the benthic sediment were reduced and to what extent the benthic biota was recovered. Figure 11 shows the layout of the survey stations.

Benthic sediments and interstitial water. In order to determine changes in the benthic sediments, COD, T-P, T-N, and SC contents in the benthic sediments and COD, PO_4 -P, and NH_4 -N contents in the interstitial water were investigated.

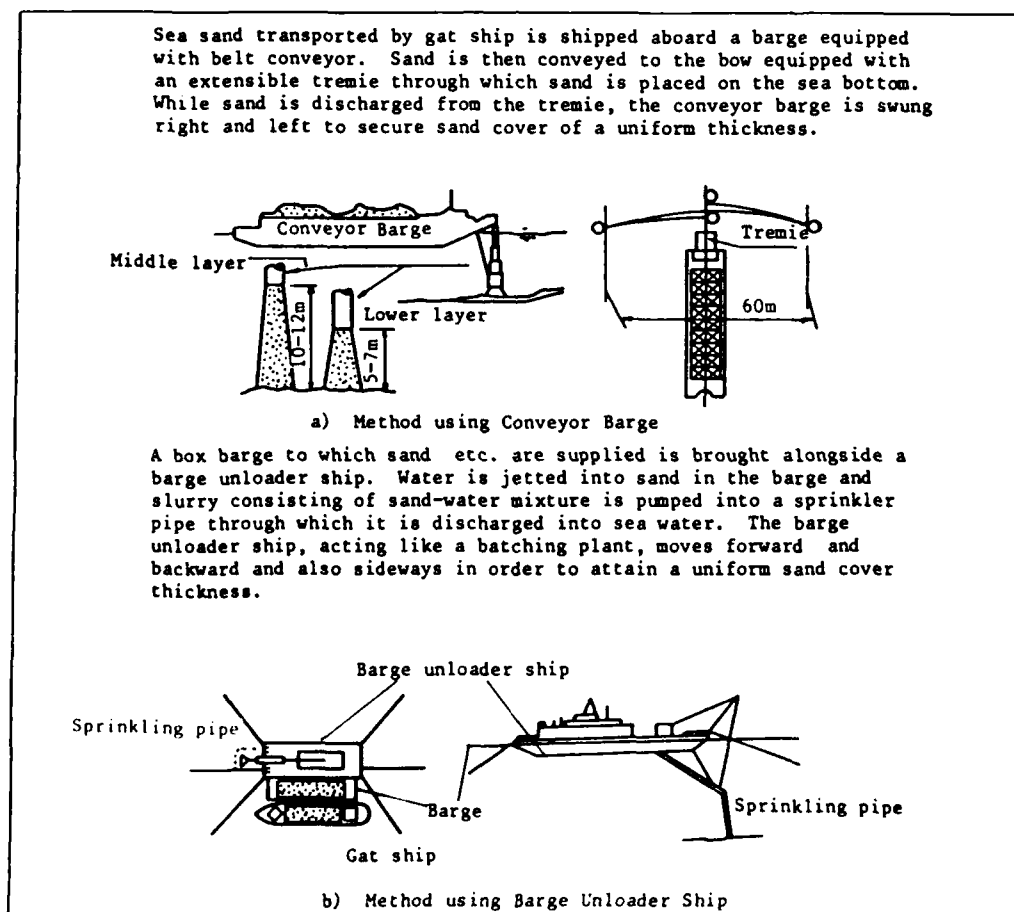


Figure 10. Methods used for implementation of experimental project

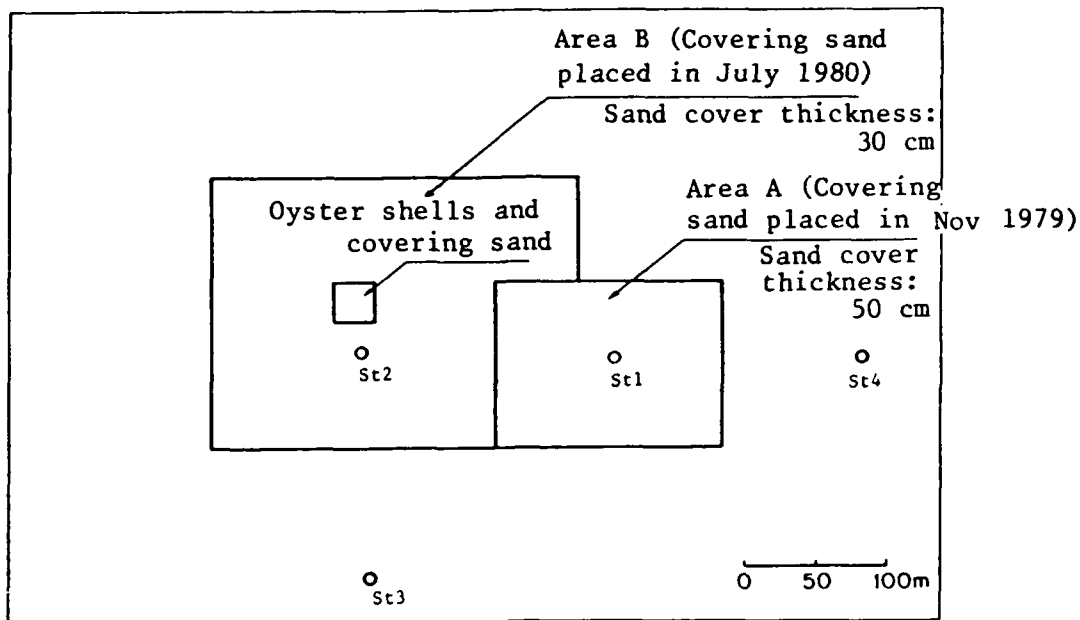


Figure 11. Follow-up survey stations located within and outside of the experimental sand covering project site

Release rates. To determine changes in the release rates of organic substances and nutrient salts (COD, PO_4 -P, and NH_4 -N) from the benthic sediment, in situ and laboratory tests were conducted.

Benthos. To determine changes due to the experimental work in the benthic biota, the species and the population of benthos were investigated.

Survey Results

Benthic sediments and interstitial water. Figure 12 indicates changes in the benthic sediments in the surface sediment layer sampled from areas within and outside of the project site. Compared with the benthic sediments sampled from the area outside the project site, those sampled from the sand-covered areas (Areas A and B) were characterized by considerably lower contents of organic substances and nutrient salts. The analysis proved that the effects of sand covering were well maintained even after a period of about 6 years.

Figure 13 shows the vertical distribution with time of the benthic sediments inside and outside of the experimental project site. At the time of surveying, a 10-cm-thick layer of loose sediment had accumulated on the sand cover. Because of this accumulation in the several years after project implementation, the benthos content was somewhat higher than that measured immediately after project execution; however, these values were still lower than those of the surface sediment layer outside the experimental project site.

The variation of quality of interstitial water in the benthic mud layer with time is shown in Figure 14. These results showed that the pollutant concentration directly under the sand cover had a tendency to increase with time; however, all the items (i.e., COD, PO_4 -P, and NH_4 -N) were maintained at comparatively low levels in the experimental project site.

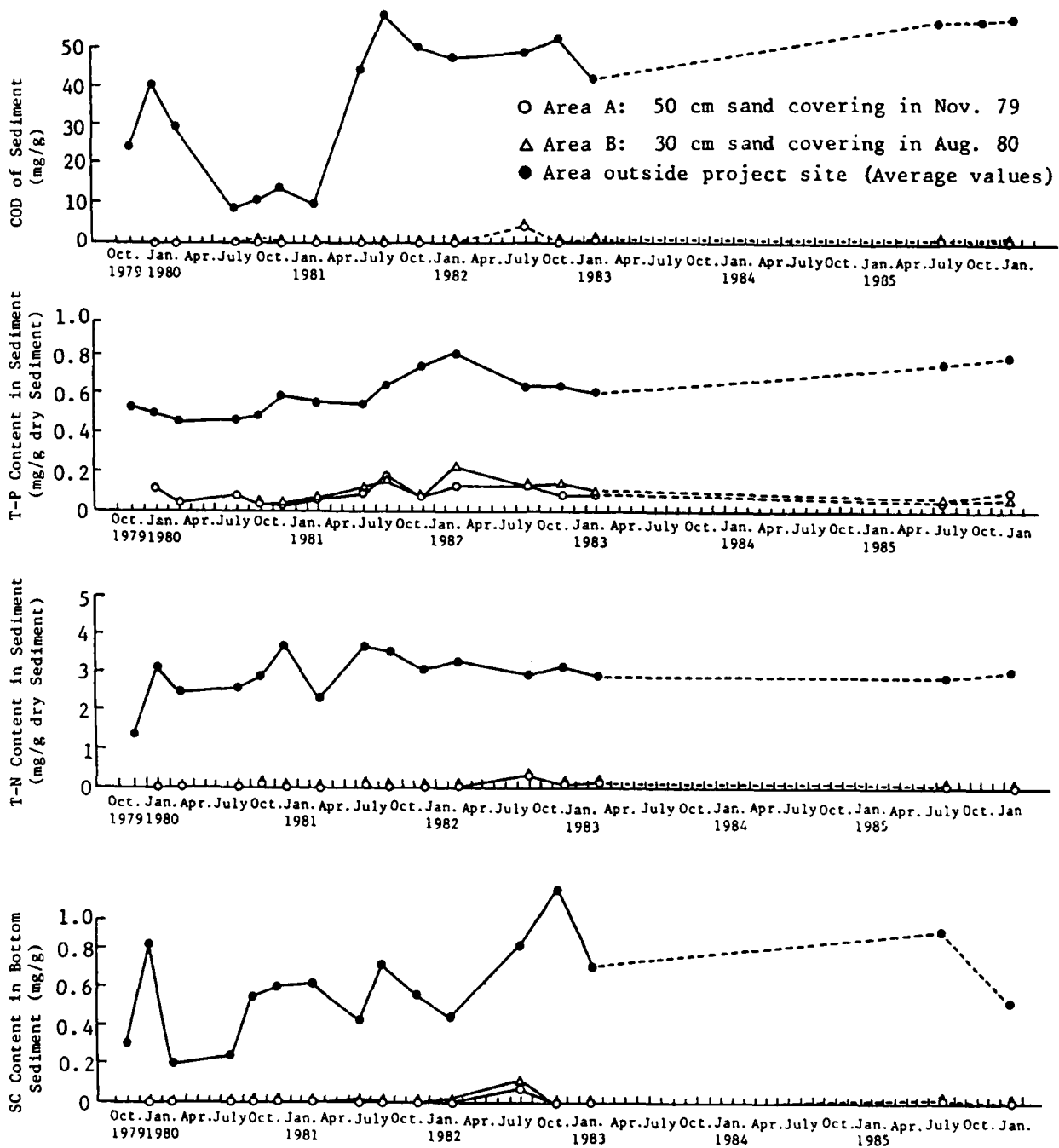


Figure 12. Variation of benthic sediments inside and outside of the sand-covered site

Release rates. Figure 15 indicates the variation with time (as determined by the in situ test) of the pollutant release rates from the benthic sediment inside and outside of the sand-covered area. While no obvious effects were recognizable for COD, the effects of sand covering on the reduction of the release rates were clearly noticeable for the nutrient salts. The release rates in areas outside the sand-covered site were considerably higher than those inside the sand-covered area. In particular, the sand covering proved to be remarkably effective for reducing the release rate of P.

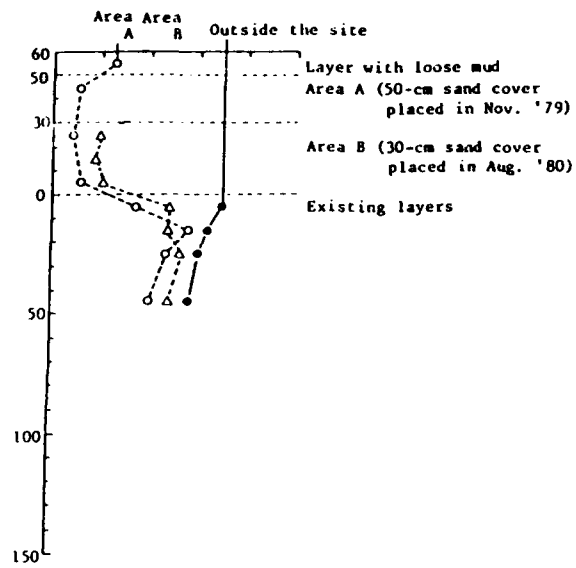


Figure 13. Vertical distribution of chemical contents in benthic sediments with time inside and outside of the sand-covered area (Sheet 1 of 4)

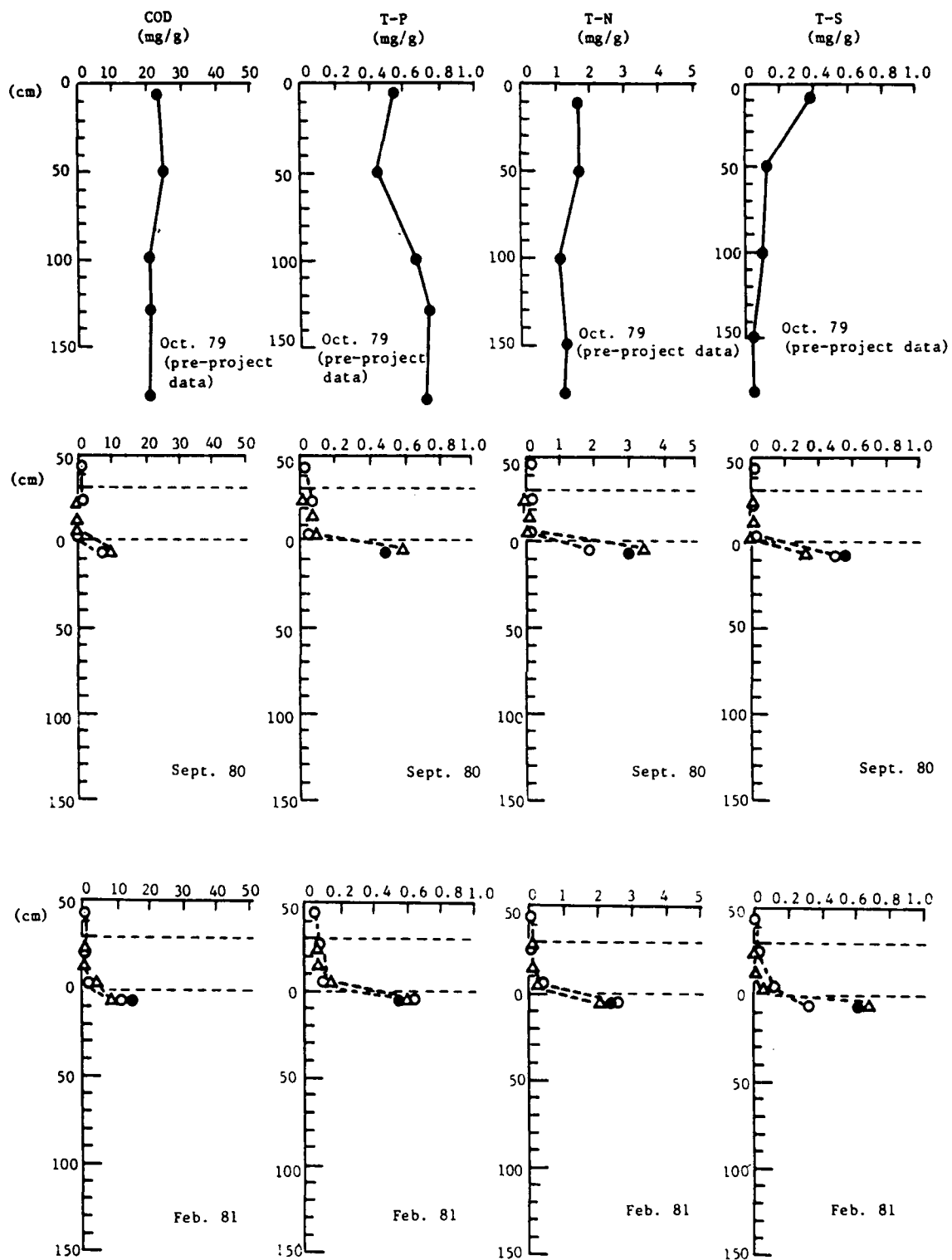


Figure 13. (Sheet 2 of 4)

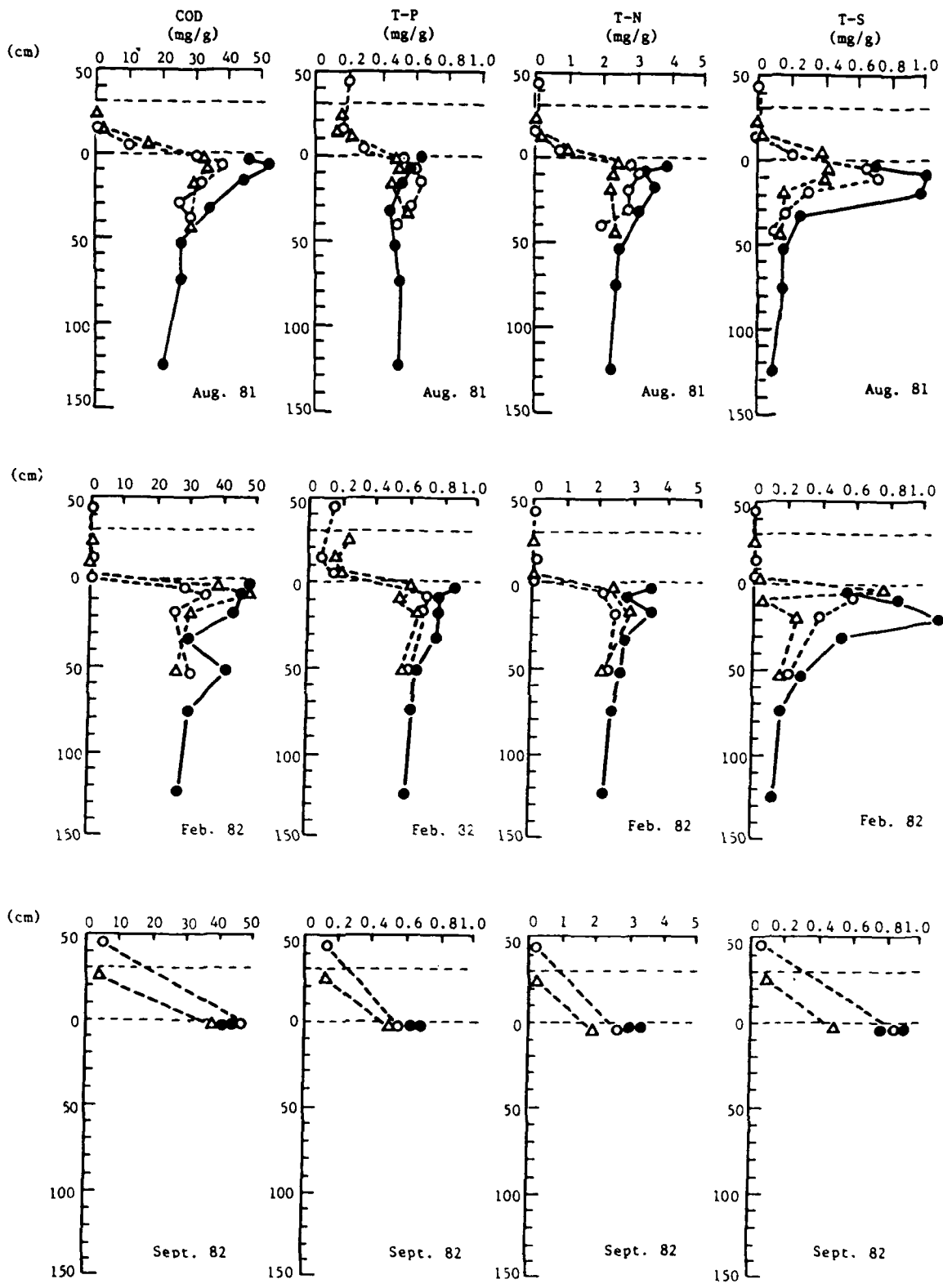


Figure 13. (Sheet 3 of 4)

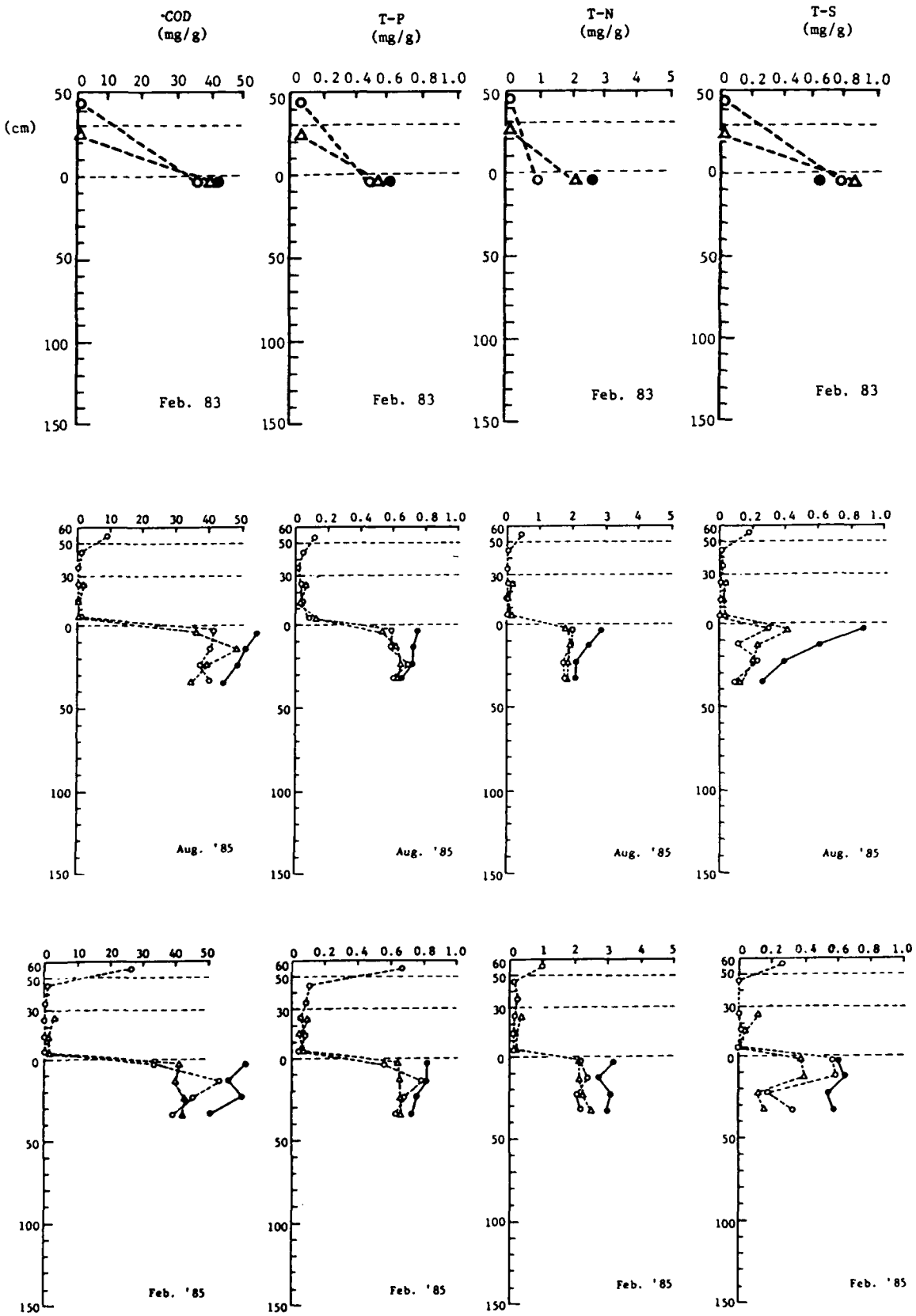


Figure 13. (Sheet 4 of 4)

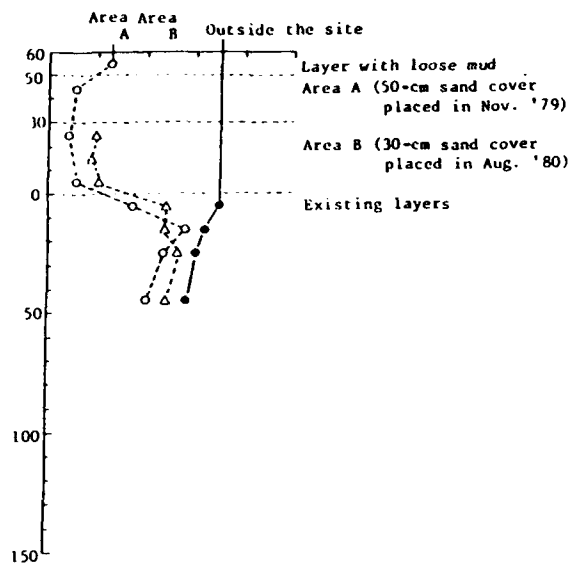


Figure 14. Vertical distribution of quality of interstitial water in benthic sediment layer with time inside and outside of the sand-covered area (Sheet 1 of 4)

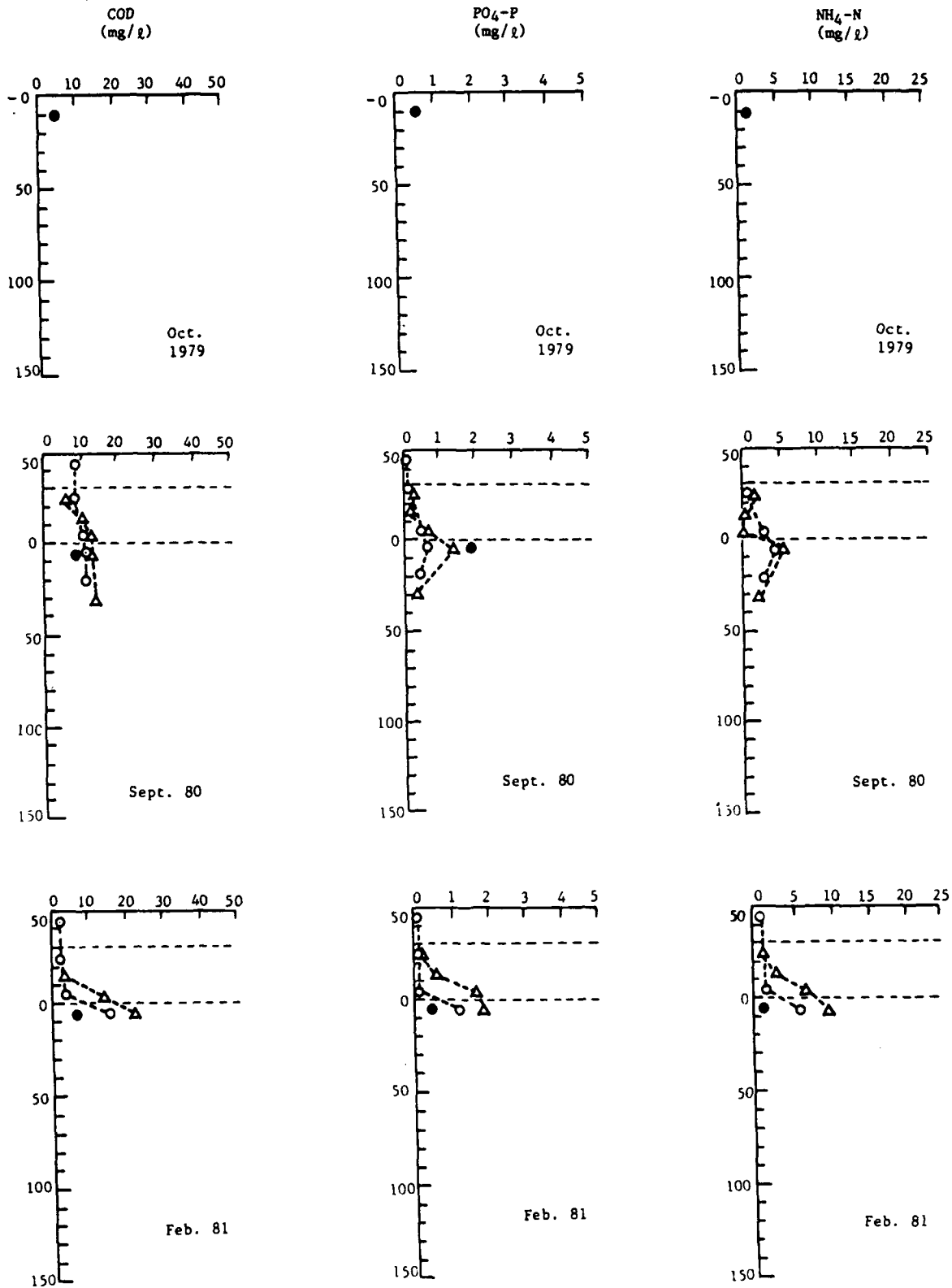


Figure 14. (Sheet 2 of 4)

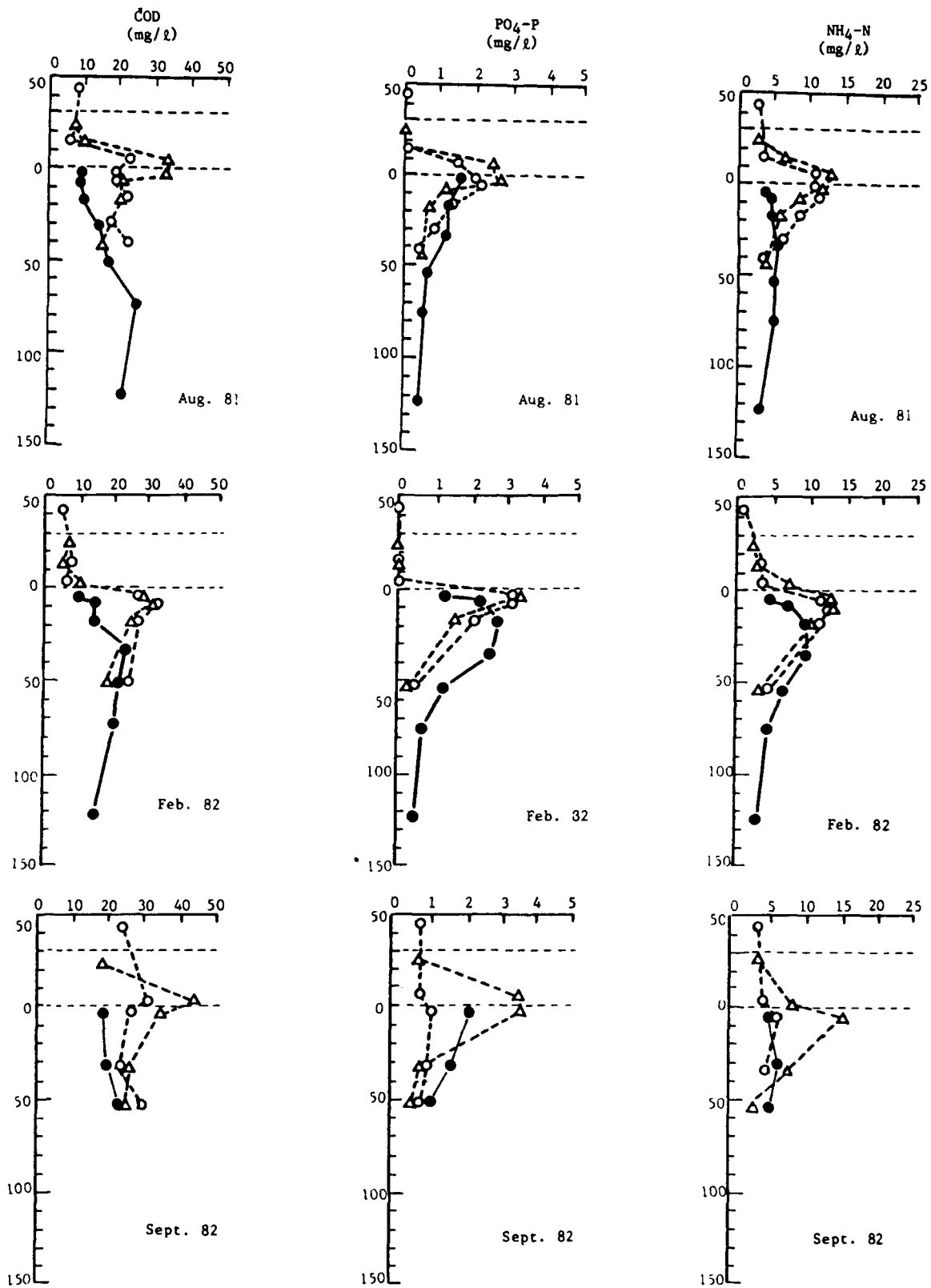


Figure 14. (Sheet 3 of 4)

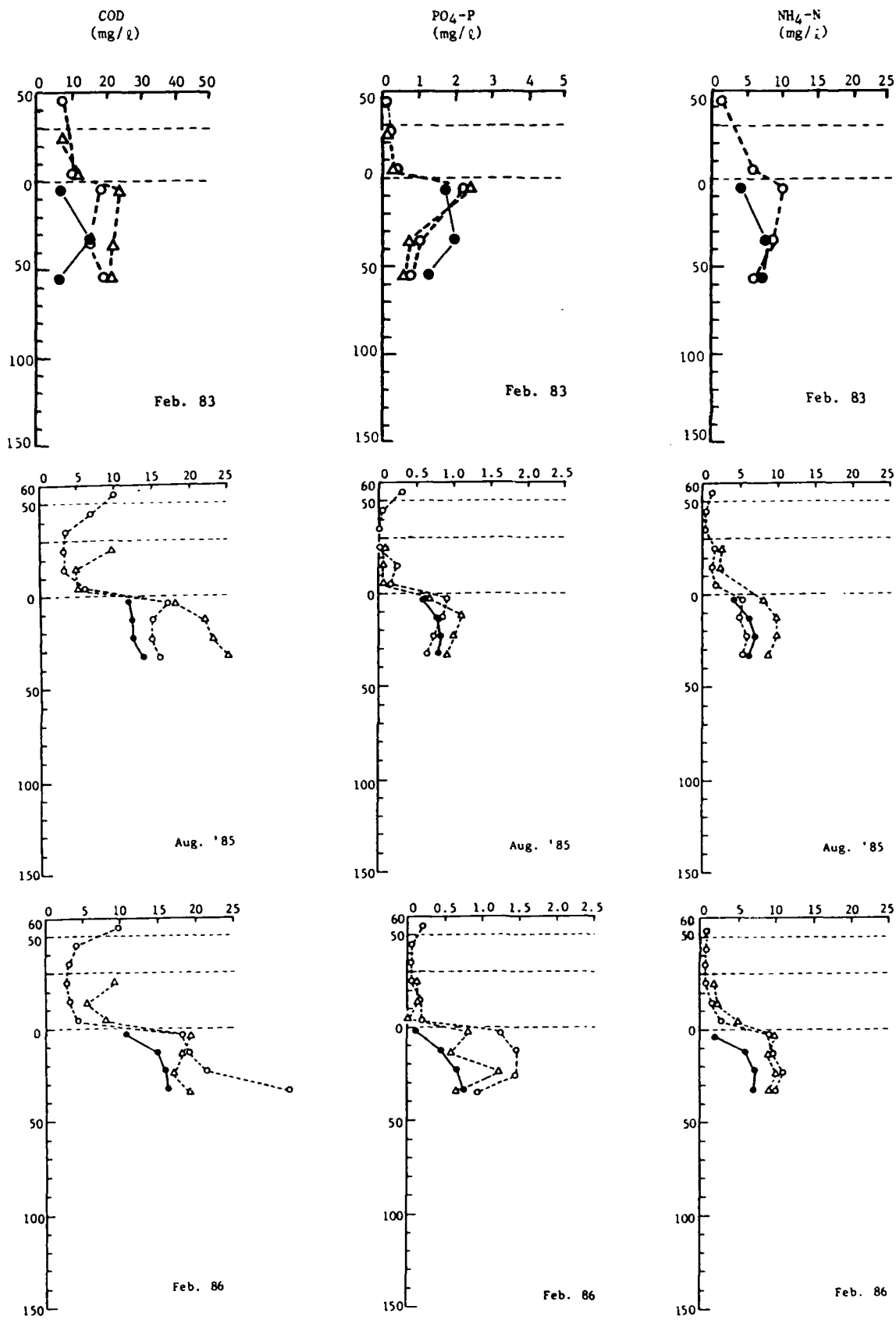


Figure 14. (Sheet 4 of 4)

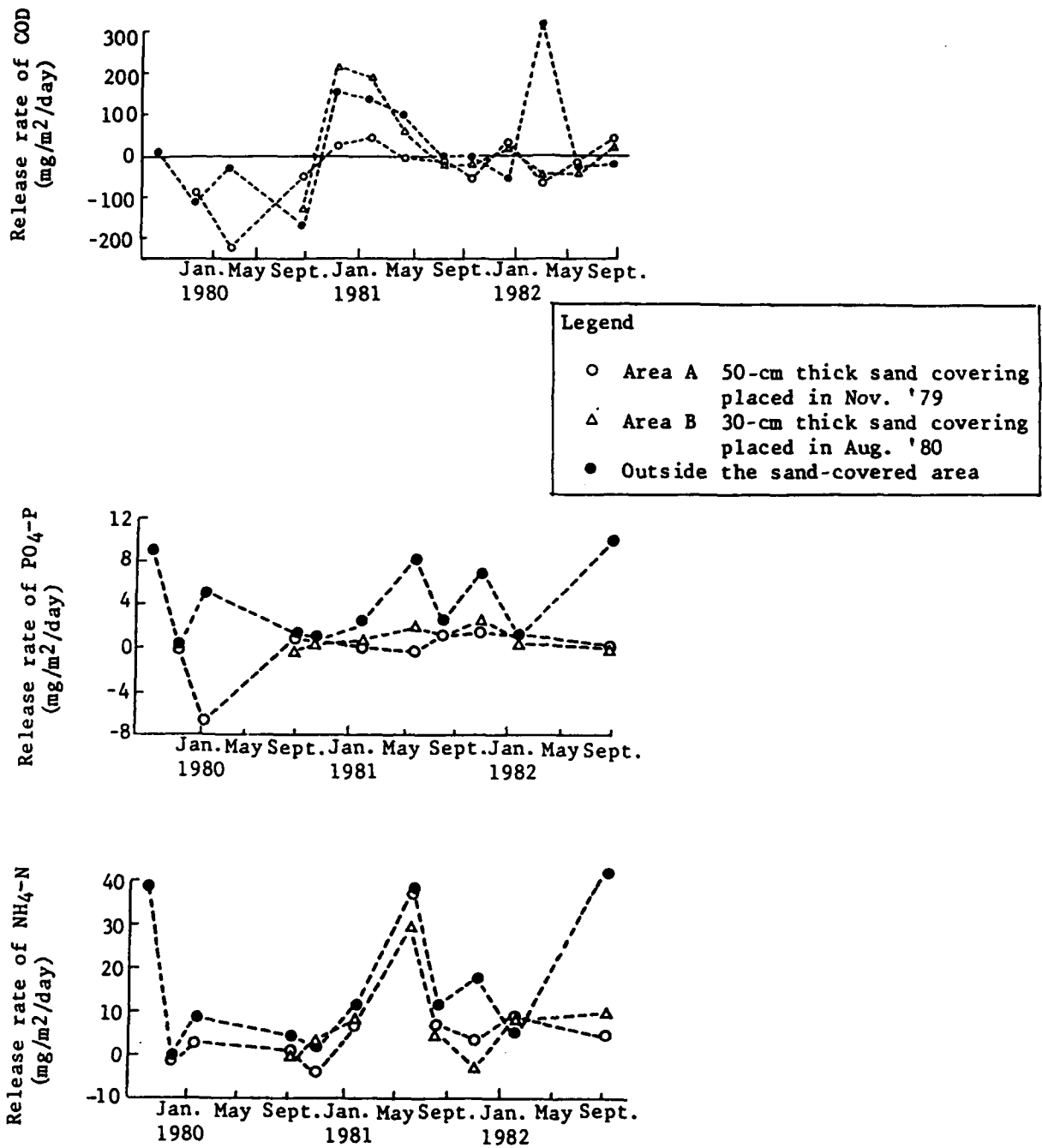


Figure 15. Release rates of pollutants in benthic sediment inside and outside of the experimental sand-covering project site (in situ test results)

Benthos. Figure 16 shows the variation with time of the number of species and the total population of benthos inside and outside of the sand-covered area. The number of species of benthos and their respective populations in the sand-covered area were larger than those in the outside area, and

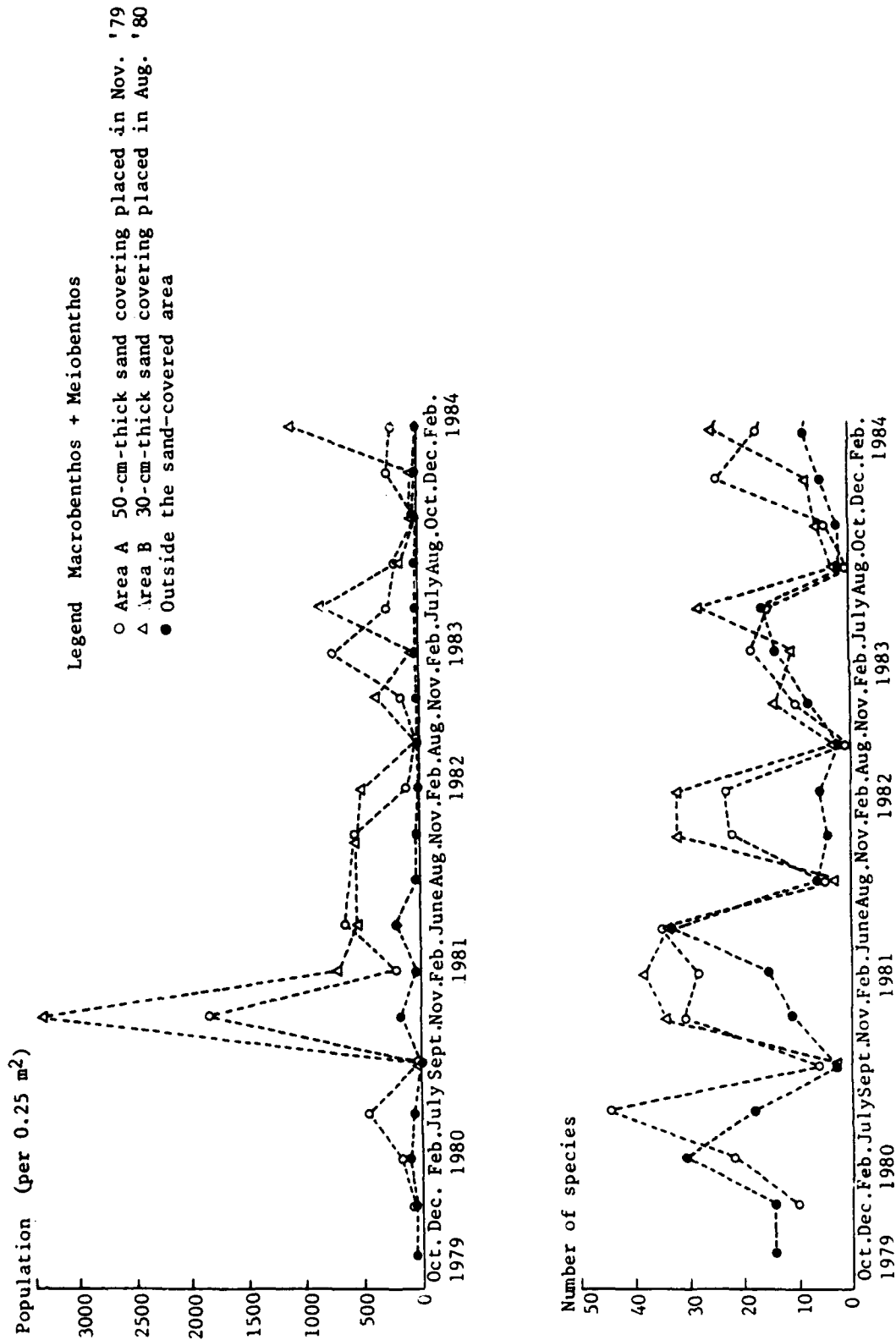


Figure 16. Variation of number of species and total population of benthos with time

although they tended to be lower in summer, they showed a quick recovery in autumn. From this, it can be assumed that the bioecologic environment was improved by the sand covering.

SUMMARY

The effects of sand covering in Kure Cove and their sustainability may be summarized as follows:

- a. The concentrations of pollutants in benthic sediments and interstitial water became somewhat higher with time, but as a whole maintained low values.
- b. The release rates of pollutants in the benthic sediment in the sand-covering project site were lower than those in areas outside the site.
- c. A remarkable recovery of benthos was apparent immediately after placement of the sand covering. The recovery rate was subsequently reduced; however, the recovery rate remained more significant than that found in areas outside the experimental site.

The effects of sand covering on improving the benthic environment, reducing the pollutant release rate, and bringing about the recovery of biota, as well as sustaining these effects, were generally proven.

In implementing any future benthic sediment cleaning project, accurate appraisal of the project's effects on improvement of the water quality and the ecosystem, and studies on measures for preventing inflow of suspended sediment will constitute the main problems that will have to be solved. Furthermore, it is essential that experimental projects of a still larger scale be implemented to develop more effective ways to deal with the actual conditions of bay area pollution.

PUGET SOUND DREDGED DISPOSAL ANALYSIS

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AD-P006 902



ABSTRACT

Future disposal of dredged material in the Puget Sound estuary of the State of Washington is of major interest to Federal, state, and local governmental regulatory agencies, as well as those responsible for maintaining existing waterways and harbors. Elevated levels of toxic chemicals exist in bottom sediments of all the urban bays, with tumors and other biological abnormalities found in bottom fish associated with these water bodies. Public awareness of this situation has been heightened by extensive media coverage of recent government investigations of environmental conditions in Puget Sound. These investigations and public concerns have led to three ongoing regional planning efforts, all of which deal with Puget Sound water quality and marine bottom sediments. This paper reports on the Puget Sound Dredged Disposal Analysis (PSDDA), a 3-year joint Federal-state study primarily focusing on unconfined, open-water disposal of material dredged from Federal and non-Federal navigation projects. Study objectives include (a) selection of unconfined, open-water disposal sites; (b) development of sampling, testing, and test interpretation procedures to be used in evaluating the suitability of dredged material for disposal in Puget Sound waters; and (c) formulation of disposal site management plans. Preliminary findings for each of these objectives are discussed for central Puget Sound, which includes the ports of Seattle, Tacoma, and Everett.

INTRODUCTION

The Puget Sound Dredged Disposal Analysis (PSDDA) is a 3-year study with primary focus on unconfined, open-water disposal of material dredged from Federal and non-Federal navigation projects. It is a joint study by the Seattle District, Region 10 of the Environmental Protection Agency (USEPA), and the Washington Departments of Natural Resources and Ecology. The study began in April 1985.

Well over one half of the state's population is located in the Puget Sound area (Figure 1) and about 2.2 million people reside in the metropolitan



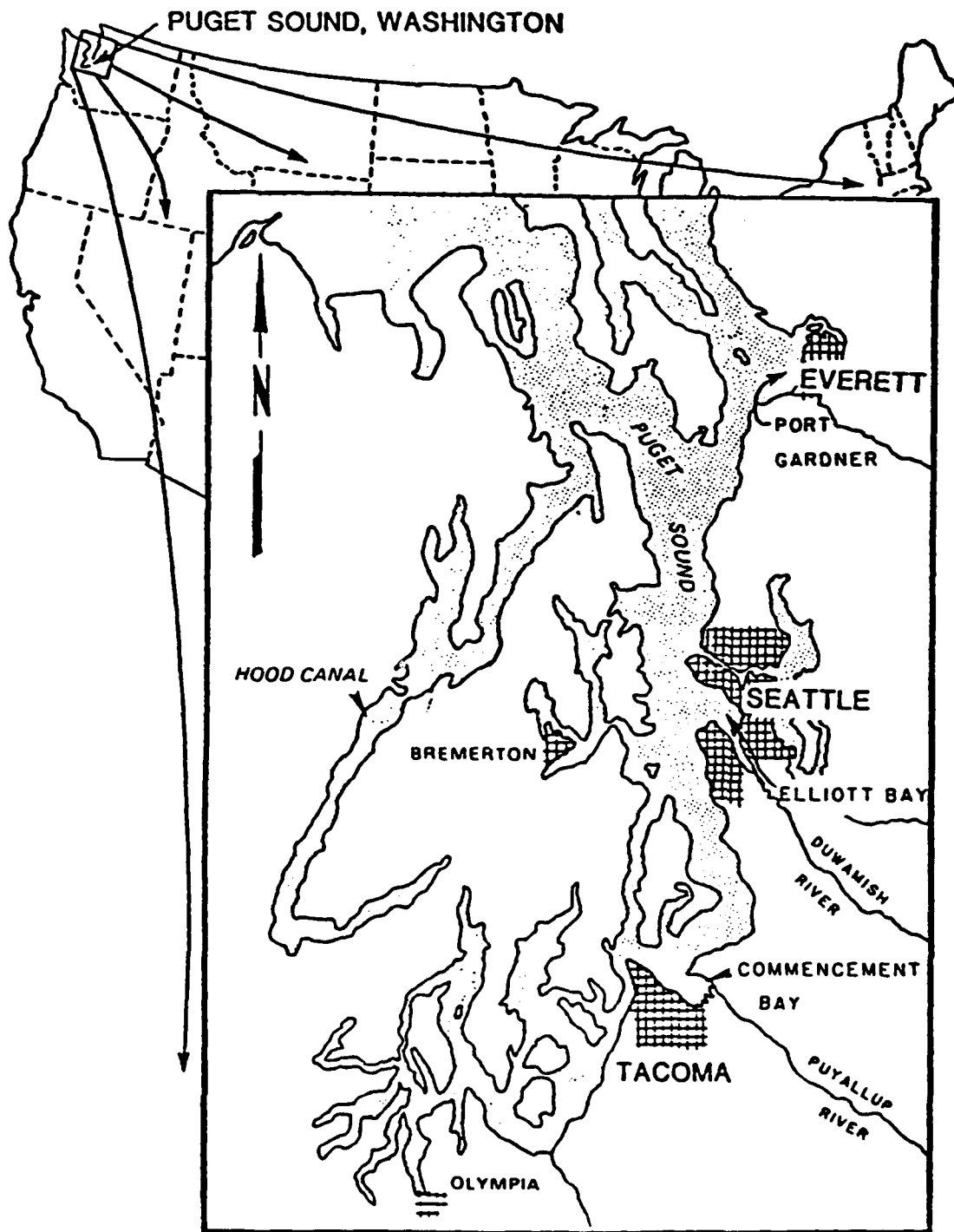


Figure 1. Puget Sound, Washington

corridor of Tacoma, Seattle, and Everett. The Sound is treasured because of its varied 2,000-mile shoreline and 2,500 sq miles of popular boating waters used for sailing, power cruising, and fishing. Also, the water body is very important to the economic well being of the region with foreign and domestic trade made possible by naturally deep harbors. Throughout the sound nearly

2 million cubic yards per year have been dredged over the past 15 years, with about 40 percent of the dredged material placed in state-designated, open-water disposal sites.

Three main areas of concern have focused public attention on the Sound's water quality, resulting in public demand for action. First, high levels of potentially harmful chemicals have been found in marine sediments and organisms, especially in the urban bays near Seattle, Tacoma, and Everett due to uncontrolled as well as regulated discharges (Malins et al. 1984). Many see these chemicals as a causal factor of tumors and other biological abnormalities found in bottom-dwelling fish. Second, oceanographers estimate 60 to 80 percent of the water flowing out of the central and south Sound on outgoing tides is recycled (via tidal "refluxing") back into the system. Most contaminants released into the Sound through point discharge, urban runoff, or an atmospheric fallout appear to never leave and end up bound to the bottom sediments. Third, there are uncertainties about the long-term health of the Sound in light of limited knowledge on the cumulative effects of pollutant discharges from human activities.

In the fall of 1984, a Seattle citizen appealing to the Washington State Shoreline Hearing Board protested a disposal site-use permit, previously issued to the Department of Natural Resources. This resulted in the closure of Fourmile Rock, a major long-term, open-water disposal site in Seattle's Elliott Bay. Other disposal sites have also been closed.

Television, radio, and newspaper coverage of concerns over Puget Sound water quality have been extensive, leading to a highly sensitized public which in turn has demanded investigations of perceived problems and corrective actions on the part of regulatory agencies.

Both Federal and state governments have responded to public and agency demands for action. In early 1984, the Federal Government, through Region 10 of USEPA, developed the Puget Sound Estuary Program (PSEP). This is a long-term effort that has two primary objectives: (a) identification of water quality problems, and (b) promotion of cleanup actions.

Problem identification activities of PSEP include:

- a. Mapping pollutant loading sources.
- b. Evaluating tidal transport of contaminants.
- c. Identifying bottom sediment areas of concern defined by either elevated levels of toxic chemicals or depression of benthic biological communities.

Problem-solving activities of PSEP include the following:

- a. Establishing improved chemical and biological testing procedures for marine sediments.
- b. Formulating contaminated sediment cleanup plans for Elliott Bay, Everett Harbor, Commencement Bay and other areas where contaminated

sediments have been identified as being a major concern. Source control (that is, reducing or eliminating the release of harmful chemicals into the Sound) is a key feature of all cleanup plans.

- c. Encouraging a Sound-wide, comprehensive examination of dredging and the disposal of dredged material to deal with what has been perceived as one possible reason for the spread of contaminants into deeper waters.

Public feelings about Puget Sound are so strong that protection and cleanup of its waters became an issue in the 1984 fall elections. After the elections, the new Governor and the State Legislature established a special state agency, the Puget Sound Water Quality Authority (PSWQA). This new agency has considerable powers. Through development of a comprehensive water quality plan, the agency can require that other state agencies and local governments take corrective actions leading to pollution abatement and improvement of water and bottom sediment quality. The draft comprehensive plan, distributed for public review in September 1986, covers the full range of water quality issues including dredging, disposal, and cleanup of contaminated sediments.

PUGET SOUND DREDGED DISPOSAL ANALYSIS

Goal

The goal of PSDDA is to provide environmentally safe and publicly acceptable guidelines governing unconfined, open-water disposal of dredged material, improving consistency and predictability in the decisionmaking process.

Objectives

Study objectives are to:

- a. Identify acceptable unconfined, open-water disposal sites.
- b. Define consistent and reasonable evaluation procedures for dredged material to be placed at those sites.
- c. Formulate site use management plans that will ensure adequate controls and public accountability.

Scope

The PSDDA study is a Sound-wide effort being accomplished in two geographic phases, each about 2 years in length. The phases overlap in time; the second phase started a year after the first.

Figure 2 shows the geographic coverage of Phase I and Phase II. The Phase I effort covers the critical central Puget Sound industrial core, where several major Corps of Engineers' navigation projects are proposed and where major Federal and state permit decisions are pending, such as the US Navy homeport at Everett. Phase II of the study covers the north and south Sound. The phasing has allowed the study to focus on the area having the greatest need for dredged material disposal sites. Each phase includes an

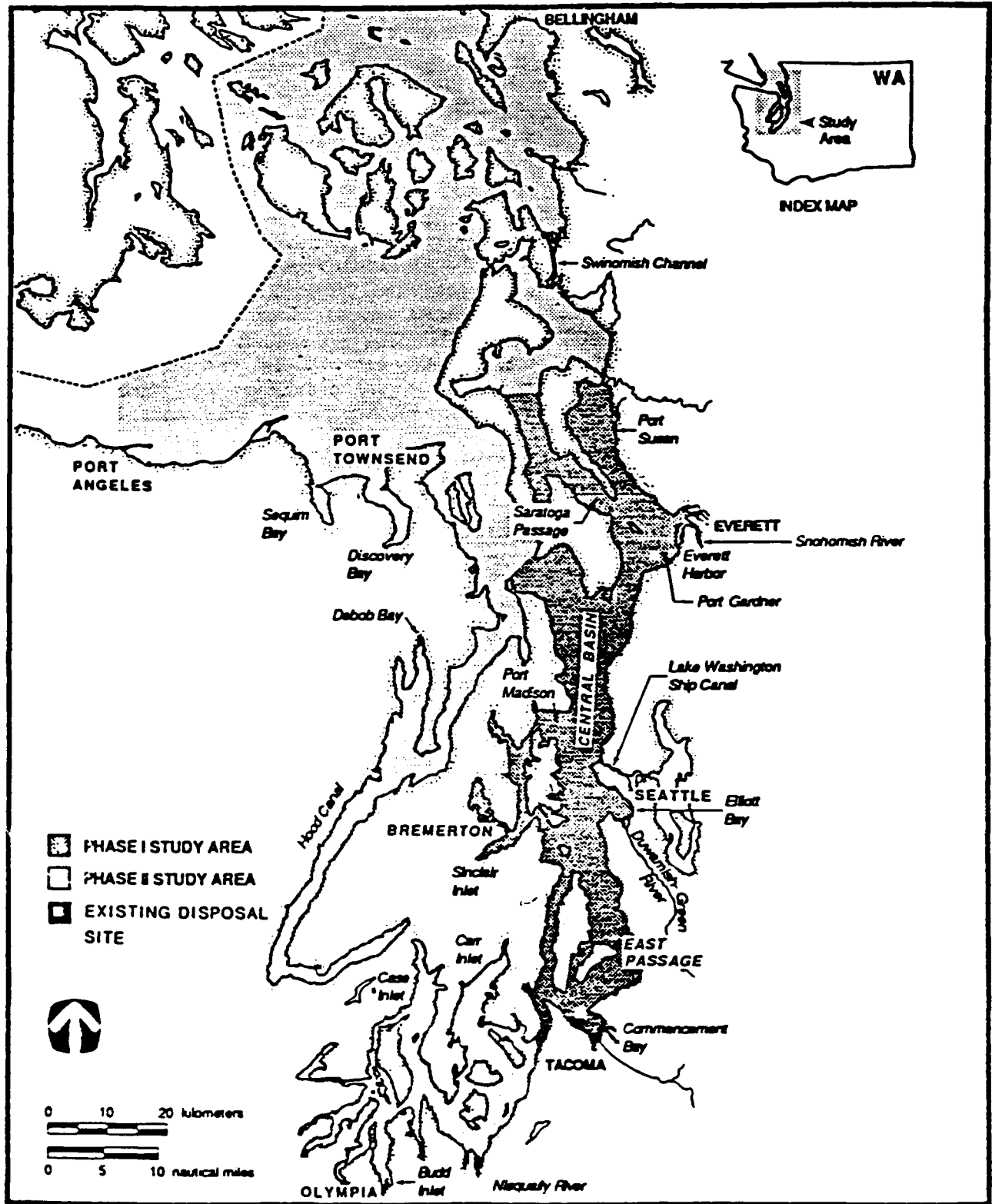


Figure 2. Study area--Puget Sound Dredged Disposal Analysis

Environmental Impact Statement (EIS) to support subsequent implementing actions. These implementing actions include adoption of evaluation procedures that define the appropriate tests for dredged sediments and disposal guidelines that will be used to decide what is acceptable for unconfined, open-water disposal.

Participants and Organization

The study is being accomplished using a consensus approach among the four principal agencies, with close participation by Puget Sound ports, local governments, Indian tribes, environmental groups, and other Federal and state agencies. Three technical work groups have been established, one for each of the study objectives (Figure 3). Also shown are the Policy Review Committee,

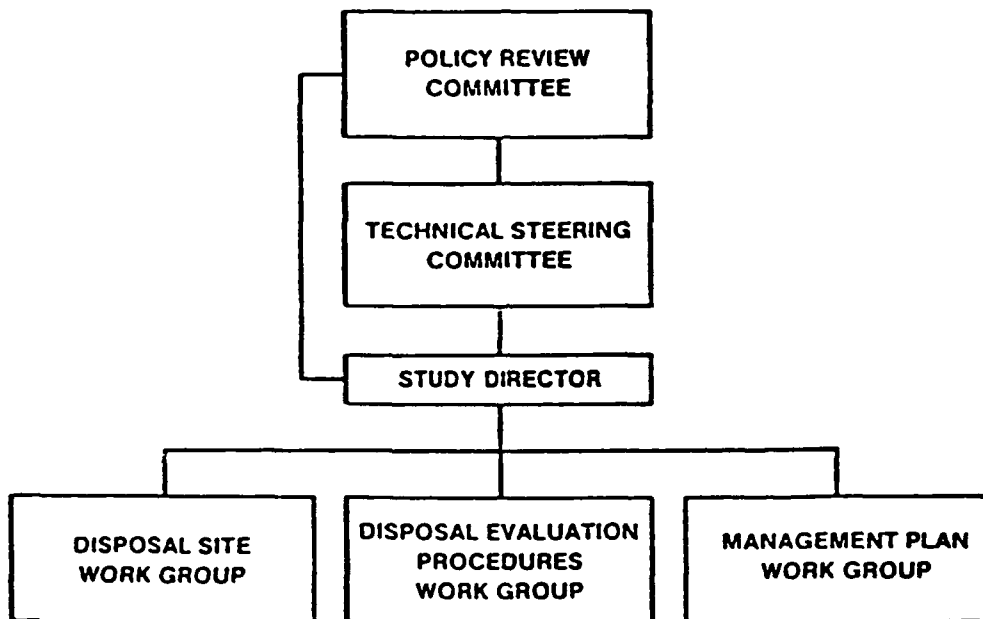


Figure 3. Organization--Puget Sound Dredged Disposal Analysis

comprised of the heads of the four principal agencies, and the Technical Steering Committee, comprised of middle managers from the agencies. The committees meet periodically with the Study Director to review study progress and discuss policy and technical issues.

CURRENT DREDGED MATERIAL MANAGEMENT

A description of current management practices for dredging and dredged material disposal in Puget Sound is needed to place PSDDA in perspective.

In comparison to many areas of the United States, annual dredging volumes in Puget Sound are relatively small. This is attributable to the naturally steep relief and resulting deep waters. Dredging volumes in the central portion of the Sound (the Phase I study area of PSDDA) for the period 1970-1985 are shown in Table 1.

The table indicates that of the 17.3 million cubic yards dredged between 1970 and 1985, approximately 40 percent was discharged in unconfined,

TABLE 1. PUGET SOUND DREDGED MATERIAL INVENTORY, PHASE I AREA
(SEATTLE, TACOMA, EVERETT), 1970-1985

Total volume dredged, cu yd				17,320,000
Total volume disposed to open water, cu yd				6,758,000
Total volume, cu yd, disposed at				
Port Gardner				692,000
Elliott Bay				4,598,000
Commencement Bay				782,000
	Corps of Engineers Projects	Port Projects	Other Projects	
Total volume dredged, cu yd	5,755,000 (4,400,000 m ³)	4,635,000 (3,544,000)	6,930,000 (5,298,000)	
Total volume disposed to open water, cu yd	2,167,000 (1,657,000 m ³)	1,389,000 (1,062,000)	3,202,000 (2,448,000)	
Total volume disposed upland or nearshore, cu yd	3,588,000 (2,743,000 m ³)	3,246,000 (2,482,000)	3,727,000 (2,850,000)	
	<u>Disposal Methods for Corps of Engineers Projects</u>			
	<u>1970-1979</u>		<u>1980-1985</u>	
	Volume cu yd	Percent	Volume cu yd	Percent
Water	818,214	24	1,027,227	54
Upland/nearshore	2,544,766	76	887,274	46

open-water disposal sites in Puget Sound. This represents about 450,000 cu yd per year to the Sound. The rest was placed in upland and nearshore disposal sites. Volumes dredged by the Corps of Engineers, the Puget Sound ports, and others (the private and municipal navigation interests) each constitute about one third of the total during the period. The ports placed about 70 percent of their dredged material in upland and nearshore sites, while the Corps and other dredgers placed only 62 and 54 percent, respectively. These figures suggest that smaller dredgers (not the ports or Corps) have relied more heavily on unconfined, open-water disposal due to fewer opportunities for land development (fill) projects.

Another consequence of the steep relief of the Puget Sound shoreline is that there are few easily accessible upland or nearshore fill areas that do not conflict with human habitation, industrial activity, and environmental

resources (wetlands and fish habitat). As upland and nearshore sites have become increasingly scarce, the reliance on unconfined, open-water sites has increased. As can be seen from Table 1, while only 24 percent of the Corps' dredged material went to open-water sites in the 1970's, over 50 percent has been going to open water in the 1980's. (This trend has also resulted in several recent uses of the confined aquatic disposal (CAD) (capping) technique. A major test of this technique is planned by the US Navy for their Homeport project at Everett, Washington.)

Dredging forecasted to occur over the next 15 years (through the year 2000) is shown in Table 2. A 31-percent increase in potential dredging is anticipated relative to the last 15 years, primarily due to several large proposed navigation improvement projects. The trend of increasing reliance on unconfined, open-water disposal is expected to continue, as well as increased use of confined aquatic disposal. Given the relatively high cost of alternatives to unconfined disposal, the disposal guidelines that determine what material will be considered acceptable for unconfined disposal are recognized as centrally important to future dredging.

TABLE 2. PUGET SOUND DREDGED DISPOSAL ANALYSIS, PHASE I AREA
DREDGING FORECASTS 1985-2000

Total forecasted dredging volumes, cu yd	23,697,000
Volumes of material that might consider use of PSDDA sites, cu yd	
Total volume*	19,397,000
Estimated volume that would be acceptable for PSDDA sites**	11,218,000
Minimum volumes that would require use of PSDDA sites†	5,460,000

* One large project has been excluded because of a recent determination that its dredged material will be placed in a confined disposal site. The volume for this project is shown in the total dredging volume, not in the total volume that might consider use of the PSDDA sites.

** Sediment chemistry data were used to estimate the volumes of material that might exceed and pass proposed PSDDA disposal guidelines. These volumes are being used by PSDDA to examine cost impacts of the alternative disposal guidelines and to determine needed site capacity for the PSDDA sites.

† This volume represents the material that is estimated to pass the proposed PSDDA disposal guidelines, without including large (and potentially speculative) proposed port improvement projects, and maintenance projects that expect to continue to rely on upland and nearshore disposal. These are purposely conservative estimates of the anticipated volumes, which are being used by PSDDA to analyze user fees (per cubic yard). These fees and possibly state appropriations are expected to pay for proposed site environmental monitoring.

Interim Disposal Criteria

A major factor in the City of Seattle's closure of the Fourmile Rock Disposal Site was the lack of defined dredged material evaluation procedures. Historically, testing requirements and test interpretation standards ("disposal guidelines") have been developed on a case-by-case basis for each project. Many past projects were exempted from testing or detailed assessment. There were no agreed-upon, consistent standards or guidelines for determining what material was acceptable or unacceptable for unconfined, open-water disposal.

Water column testing for release of dissolved contaminants during dredging and disposal was primarily relied upon to ensure water quality standards were being met. However, this testing usually did not reveal any problems, as little of the contamination was released from the sediment to the water. Multiple tests conducted using sediment elutriation procedures did not indicate any reason for concern. As we now know, contamination associated with dredged material is predominantly bound to sediment particles and does not substantively dissolve during the short-term disturbances of dredging operations. The discovery of elevated levels of chemical contaminants in the disposal site sediments, and that these sediments were toxic in the solid phase to bioassay species, underscored the need for redesign of the dredged material evaluation procedures.

It was in this context that the USEPA and the State of Washington promulgated "interim criteria" for dredged material disposal at unconfined, open-water sites (USEPA 1984). Developed initially for the Fourmile Rock Disposal Site as a stipulation of the City of Seattle's shoreline permit, a similar concept is now in place for all state-designated unconfined, open-water disposal sites in Puget Sound. These criteria were proposed as interim, pending development of better decision guidelines through a Sound-wide approach such as PSDDA. The evaluation procedures under development by PSDDA are intended to replace the interim criteria.

The interim criteria require both biological and chemical testing of the dredged material; and standards for each must be met before unconfined, open-water disposal is approved. Types of tests required vary slightly from area to area based on the degree of historic contamination potential, but all urban and industrial areas require similar testing.

With the interim criteria, dredged material sampling requirements vary from project to project. However, concurrent with promulgation of the Fourmile Rock interim criteria, the City of Seattle specified the requirement of having one core for every 4,000 cu yd to be dredged. The 4,000-cu-yd figure was selected as it represented two large barge loads of material, a volume considered sufficiently large to potentially warrant a separate decision on acceptability for unconfined disposal.

Chemical testing requirements of the interim criteria cover a fairly limited range of chemicals. Six heavy metals (Pb, Zn, As, Hg, Cu, Cd), 16 polynuclear aromatic hydrocarbons (6 low molecular weight, 10 high molecular weight), polychlorinated biphenyls (1016, 1232, 1242, 1248, 1254, 1260), and DDT's (4-4 DDD, 4-4 DDE, 4-4 DDT) are the only decision chemicals, though

other base neutral priority pollutants are often requested for analysis. Biological testing of the sediments is conducted via a 10-day, solid phase exposure of amphipods (*Rhepoxynius abronius*) (Swartz et al. 1984).

Interpretation of test results under the interim criteria is based on comparison with conditions that exist at the disposal site (for Fourmile Rock) or in the central portion of the Sound (for the rest of Puget Sound). The decision criteria are based on the concept of "nondegradation relative to reference," i.e., "in the interim, let's not let it get any worse than it already is." A major shortcoming of this approach is that the cause-effect relationship between chemistry and biological effects is not considered in setting the bulk chemical "pass/fail" values. Also, these pass/fail criteria are established for all contaminants present at the disposal site, without regard for whether they are instrumental in the effects of concern.

Bulk sediment chemical criteria for Fourmile Rock are based on no single chemical exceeding 125 percent of reference, and no three chemicals exceeding 110 percent. Acute bioassay responses cannot exceed the 25-percent absolute mortality observed using Fourmile Rock sediments (no corrections relative to test controls are allowed). The use of these values in disposal decision-making has been questioned by both economic development and environmental protection interests--the former noting them as devoid of effects significance, the latter claiming that an already adverse situation is being perpetuated.

Site Management

The state owns the bottom lands in the Sound. As such, they manage the existing sites for the Phase I area (see Figure 2). These sites have been in use since the early 1970's. The state collects a disposal site use fee from non-Federal users. Federal users, including the Corps, cooperate in the program by using the designated sites and participating in an interagency siting committee chaired by the state.

Originally, the state planned to monitor the sites. However, disposal fees were not set sufficiently high enough to provide needed revenue after administrative expenses were covered. As a result, most of the sites have received no environmental monitoring. A few sites have been examined via special research programs.

DISPOSAL SITE SELECTION

The Disposal Site Work Group (DSWG) agreed early in their deliberations that disposal sites should be reasonably accessible to the harbor projects. For the Phase I area this resulted in a strategy of having a site in each of the major urban harbors of Seattle, Tacoma, and Everett. These multiuser sites would service all nearby dredging areas.

Strategy

A three-step, tiered screening process was used to select preferred and alternative disposal sites by application of an approach similar to that outlined in a USEPA/Corps of Engineers (CE) workbook on ocean disposal siting (USEPA/CE 1984). These steps, shown in Figure 4, were as follows:

STEP 1: SELECTION FACTORS OF ZONES OF SITING FEASIBILITY (ZSF)

1. Human Uses (12)
2. Biological Resources (4)
3. Physical Parameters (3)



EVALUATION FACTORS

1. Human and Environmental Health
2. Sediment Containment
3. Biological Resources Health
4. Non-interference with Human Activities
5. Shoreline Buffer Zone
6. Vulnerable Resource Buffer Zone
7. Water Depth Constraints
8. Avoidance of Utility Corridors



RANKING FACTORS

1. Priority 1 ZSF - Minimal Dispersion
- Proximity to Major Dredging Activity
2. Priority 2 ZSF - Transport Economics



STEP 2: ZSF CHECKING STUDIES

1. Site-Specific Field Studies
2. Recommended Preferred/Alternative Sites



STEP 3: FINAL DISPOSAL SITE SELECTION

1. Assimilated Steps 1 and 2
2. Final DSWG Selection and Public Input

Figure 4. Siting evaluation and selection strategy

Step 1: Develop site selection factors based on USEPA's "Guidelines for Specification of Disposal Sites for Dredged or Fill Material," and Washington State Law, including, among others, the following criteria:

- Avoid high-energy areas that would disperse dredged material significantly beyond the disposal site area.
- Avoid significant adverse impacts on foodfish, shellfish, marine mammals, and marine birds.

- Reduce interference with human aquatic and upland uses to the lowest practicable level.

Step 2: Review existing mapped information and produce map overlays displaying information based on 19 selection factors. Following the map overlay process, potential disposal siting areas called "Zones of Siting Feasibility" (ZSF's) were selected that appeared to have minimal conflicts with human uses and natural resources. This process identified priority ZSF's within Saratoga Passage, Port Gardner, Elliott Bay, and Commencement Bay. Priority ZSF's were those sited in low current areas, greater than 2,500 ft from shorelines and "vulnerable" natural resources, between 120- and 600-ft depths, and located within 10 n.m. of the nearest center of dredging activity (see Figure 5). Existing data gaps within chosen ZSF's were evaluated by DSWG, and limited field studies were initiated to further characterize and check the ZSF's. These field studies are discussed later in this paper.

Step 3: Tentatively select preferred and alternative disposal sites. Within the ZSF's, this was accomplished using data from steps 1 and 2. Data from computer simulations of typical disposal operations in deep water provided an assessment of the size of the potential bottom impact area. Computer studies indicated that, for the sites under consideration, the impact areas could vary from about 0.4 to 0.5 sq mile, depending upon water depths and tidal currents. Additional information from site selection studies described below was used to refine the locations of the preferred site(s) by validating and verifying the relatively nondispersive qualities of each site, and the status of biological resources within and adjacent to the site boundaries.

Modeling Studies

A key evaluative factor in choosing a site is the ability to accurately place material within defined boundaries without significant dispersal beyond predetermined acceptable limits. Two computer simulation studies addressing these concerns were undertaken: disposal modeling and current modeling.

Disposal Modeling

Using Puget Sound data, the US Army Engineer Waterways Experiment Station, Vicksburg, Miss., performed a simulation study depicting dredged material disposal in an unconfined, open-water environment. To determine the behavior of material during actual operations, the model simulated passage of dredged material through the water column for varying water depths, current speeds, and sediment type. The conditions applied were representative of those in Puget Sound--depths ranging from 100 to 800 ft and currents ranging from 0 to 2 knots. If a circular, 900-ft radius target is assumed, with maximum flood and ebb tide velocities of 0.5 knot, the material would remain within an area described in Table 3 that is deemed to be the disposal site. Figure 6 provides a perspective on a ZSF and a disposal site within the ZSF, which is established from multiple bottom "footprints," or multiple derived discharges from the computer studies.

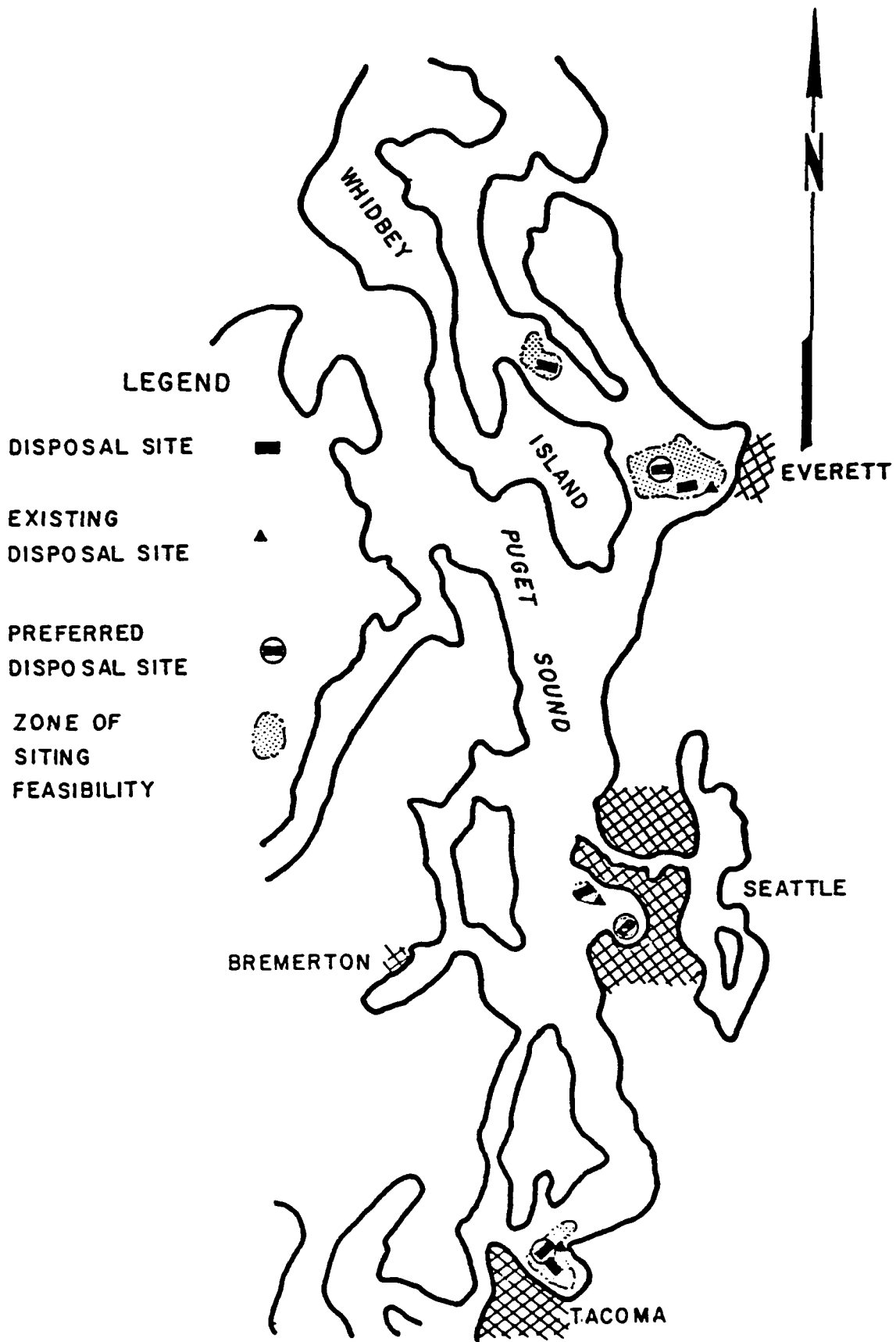


Figure 5. ZSF's and alternative disposal sites

TABLE 3. DISPOSAL SITE AREA

Amount of Material cu yd	Current knot	Depth ft	Footprint ft
1,500	0.5	400	3,000 × 3,800
1,500	0.5	600	3,000 × 4,600

Current Modeling

A two-dimensional computer simulation model was used to predict tidal elevations and horizontal currents in the Phase I area. Spring and neap tide periods were simulated to obtain average, net, and peak current velocities. Generally low current speeds were predicted within the Phase I ZSF's.

This modeling was supported by low current velocities measured by current meter arrays placed at the existing Port Gardner (Everett) and Fourmile Rock (Seattle) disposal sites.

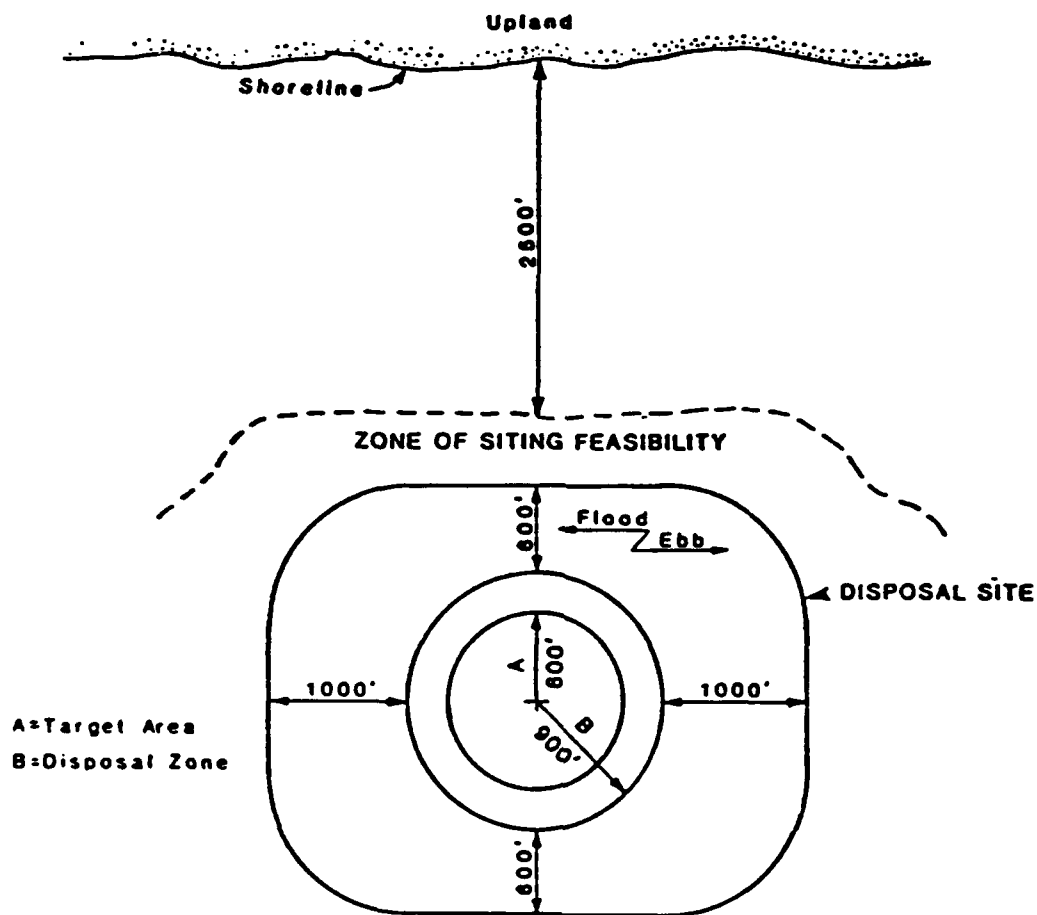
Field Checking Studies

Field studies were conducted during the fall of 1985 in the Port Gardner ZSF, Saratoga Passage ZSF, and the two Elliott Bay ZSF's, to provide supplemental baseline biological, current, and sediment transport information. These data were combined with other available information to aid the DSWG in selecting preliminary preferred and alternative sites within the ZSF's at each of the four areas.

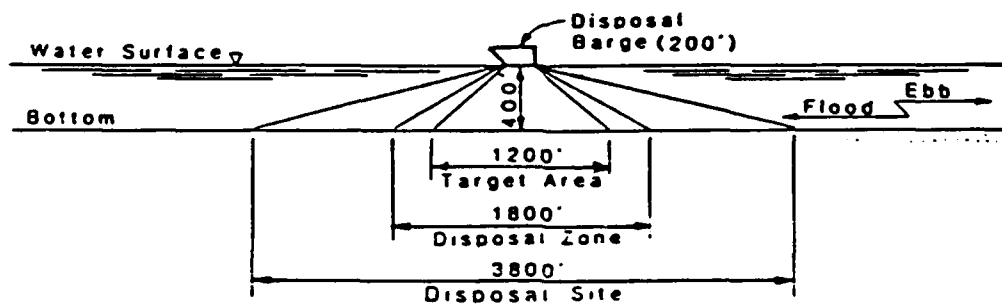
Due to weather and scheduling problems, no field studies were performed at the Commencement Bay ZSF. Existing information was used to select the Commencement Bay ZSF and the preliminary preferred and alternative disposal sites.

Other studies conducted during this period included:

- a. MANTA survey. A submersible remote operational vehicle (ROV) collected physical bottom data with 90-kHz sidescan sonar and sought data on biological resources through a low incident light video camera and a 35mm stereo still camera.
- b. Remote ecological monitoring of the seafloor (REMOTS) survey. A sediment profiling camera and computer imaging analysis system were used to provide information on physical and biological (infaunal benthos) characteristics.
- c. Van Veen grab samples. These were collected at selected stations within ZSF's to ground truth REMOTS observations, provide sediment samples for grain-size determinations, and archive benthic infaunal samples for possible future study.



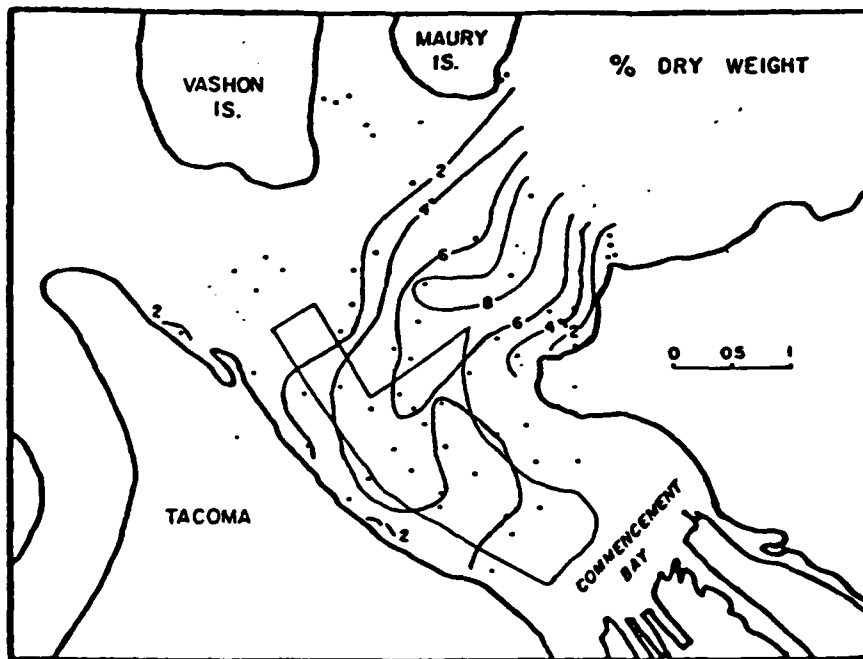
PLAN VIEW



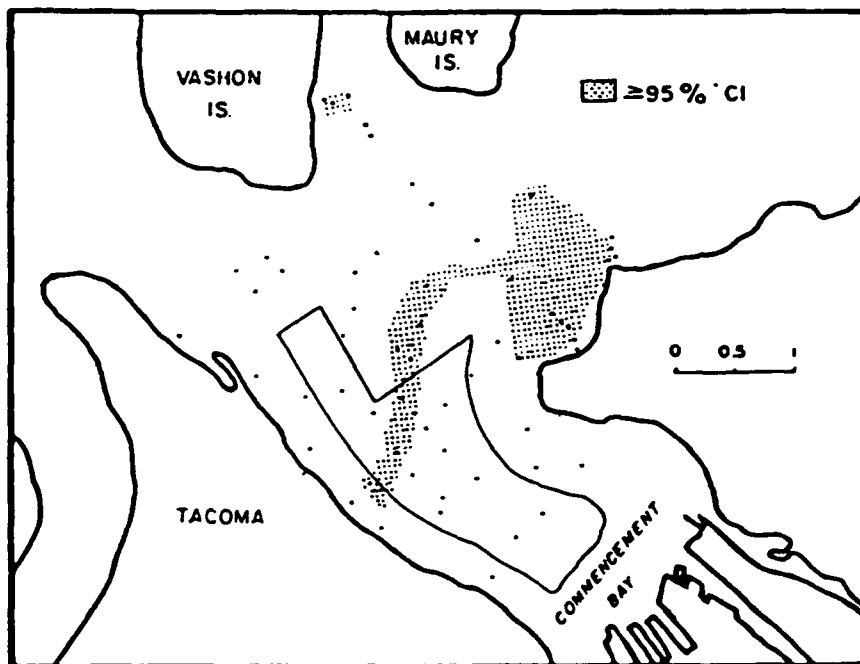
ELEVATION VIEW

Figure 6. Disposal site parameters

- d. Current meter studies. These were conducted at the existing Fourmile Rock (Elliott Bay) and Port Gardner disposal sites to provide additional information on the current regimes of these two areas. Less than 2 percent of the time were current speeds measured above approximately 0.5 knot.
- e. Fish and benthic resource studies. Sites within the ZSF's were tentatively selected by March 1986, subject to further verification and validation from site-specific studies. Site-specific studies of bottom fish and shrimp and crab resources were conducted over four seasons, from February through September 1986 within each of the selected sites and surrounding areas. Additionally, questions concerning benthic habitat quality for bottom-feeding fishes in the sites and adjacent areas were addressed during the summer field studies using a procedure called the benthic resources assessment technique (BRAT). This technique provided quantitative information concerning the benthic habitat quality for bottom-feeding fishes using the site during summer.
- f. Depositional analysis technique. A relatively new approach, known as depositional analysis, was used to assess whether dredged material deposited at a site would tend to stay there (Striplin et al. 1986). For the Phase I area, the work group chose to locate the disposal sites in relatively nondispersive areas. This was done to (1) reduce the complexity and cost of field studies, (2) minimize the potential for adverse impacts to important fishery resources, (3) facilitate follow-up environmental monitoring of the site, and (4) gain agency and public acceptance because of program accountability. An assessment of the depositional potential of an undisturbed area can be determined from a careful analysis of the bottom sediments in that area. The PSDDA field studies relied on sediment samples from over 200 stations with laboratory analysis used to determine organic loads and grain size. Previous work by Word et al. (1984) indicated that, for a given water depth, sediments in Puget Sound tend to accumulate where three criteria are met: (1) abnormally small grain size, (2) abnormally high volatile solids, and (3) abnormally high biological oxygen demand. The abnormally high determinations were made for the PSDDA data based on whether or not a sediment sample value for total volatile solids, biochemical oxygen demand, and percent water exceeded the 95-percent confidence interval (CI) for the depth contour on which the station was located. Figures 7a and 7b show the results for one of the ZSF's based on a analysis of total volatile solids. Figure 7a is a mapping of the percent of dry weight values from the stations, and Figure 7b shows the areas where values exceed 95-percent CI.
- g. Site selection. Preferred disposal sites were selected (see Figure 5) after considering the results of modeling and field studies, and public and agency views on the alternative sites. A preliminary findings report was issued to obtain public feedback, and public workshops and meetings were held to solicit public opinion. In Commencement Bay, the preferred site is very close to the existing site.



a. Percent dry weight contours



b. Areas with ≥95-percent CI

Figure 7. Depositional analysis - total volatile solids

However, in Elliott Bay and Port Gardner, the new sites are nearly 3 miles and 2 miles, respectively, from the old sites. To date, general acceptance has been given to the preferred sites by the public.

EVALUATION PROCEDURES

The dredged material evaluation procedures proposed by PSDDA are primarily focused on deciding when sediment is acceptable for unconfined, open-water disposal. As such, these procedures address sediment sampling requirements, chemical and biological testing, test interpretation, and disposal decision-making ("disposal guidelines").

Fundamental to dredged material management, the decision of what is acceptable for unconfined, open-water disposal must consider the consequences associated with disposal of the material that is not acceptable for unconfined, open-water disposal. Therefore, PSDDA is also developing evaluation procedures and design guidelines for dredging and disposal of contaminated sediments that are not acceptable for unconfined, open-water or conventional land/shore, disposal.

Sampling Requirements

The number of samples and analyses required for assessment of a dredging project is significant both to the cost of testing and to the confidence in resulting characterization of the impact potential of the material. Yet sampling intensity has too often in the past received limited attention. To assist in improving decisions on dredged material sampling, the Evaluation Procedures Work Group (EPWG) developed sampling and analysis guidelines that define the maximum volume of material to be represented by each sample and by each analysis (composite of several samples). These guidelines are shown in Table 4. The maximum volumes vary by the location of the project (relative to its potential for contamination due to source discharges) and by the depth of the cut. While cut depth will depend on the project, PSDDA has provided "location rankings" for most of the dredging projects in the Phase I area. Surface sediments in areas with many contaminant sources are therefore sampled more intensively than subsurface sediments in areas with few sources. Compositing is done such that the maximum volume per analysis represents a "dredged material management unit"--a dredgeable, separable volume of material upon which a decision on acceptability for unconfined, open-water disposal can be made.

The guidelines do not consider project size. This is due to the high variability of contaminant distribution within most dredging prisms in Puget Sound. As an option to the guidelines, the dredger can conduct a pilot study to determine if the prism variability is such that fewer samples will adequately characterize the material (from a statistical variability perspective). It is anticipated that this option will be useful primarily to new dredging projects that will be removing large volumes of well-sorted, coarse-grained material.

Testing Requirements

The EPWG has recommended several changes in the types of chemical and biological tests needed for dredged material assessment. First, the work group

TABLE 4. SAMPLING AND ANALYSIS REQUIREMENTS

Area Ranking*	Sampling**		Analysis†	
	Maximum 1,000 cu yd/Sample		Maximum 1,000 cu yd/Analysis	
	Surface <4-ft cut	Subsurface >4-ft cut	Surface <4-ft cut††	Subsurface >4-ft cut
Low	8 (6)	8 (6)	48 (37)	72 (55)
Low-moderate	8 (6)	8 (6)	32 (24)	48 (37)
Moderate	4 (3)	4 (3)	16 (12)	24 (18)
High-moderate	4 (3)	4 (3)	4 (3)	12 (9)
High	4 (3)	4 (3)	4 (3)	12 (9)

* Location of project relative to potential for contamination is expressed in terms of one of five area rankings, e.g., "Low" refers to areas where sediments are known to be relatively uncontaminated, with few discharge sources of contamination. These location rankings have been predescribed by PSDDA for the central Sound. Many of the industrial waterways of the central Sound are ranked "high."

** The minimum number of samples is calculated by dividing the project volume by the value in the table.

† The minimum number of analyses is calculated by dividing the surface and subsurface volumes by the table values. Samples are composited to form analyses. Compositing plan is determined for each project. Analyses should represent discrete dredged material management units.

†† In calculating analysis requirements, project volume located above a cut depth of 4 ft is considered separately from volume below this depth. The 4-ft cut represents a dredgeable lift (± 2 ft vertical precision).

reviewed available data on chemicals discharged to Puget Sound, which of these were present in Sound sediments, and what toxicological implications each had, and recommended adding several chemicals to the chemical analysis list. Chemicals to be added include other metals (Cr, Ni, Ag, Sb) and chlorinated hydrocarbons (e.g., chlorinated butadienes). The work group also standardized the list of conventional sediment parameters to be measured.

For biological testing, the work group has recommended the use of multiple species in the acute toxicity testing to better represent the diversity of aquatic species of concern during disposal. Tests to be included are the 10-day amphipod mortality test, the 4-day oyster larvae abnormality, and the 15-min EC50 luminescent bacteria photoinhibition test (microtox) (Schiewe et al. 1985). Additionally, for more contaminated materials, the 30-day bivalve bioaccumulation test is needed.

While none of the above recommendations are necessarily new, EPWG has proposed to link the previously separate chemical and biological testing steps. Rather than always conduct both chemical and biological testing, the

work group has recommended that the need for biological testing be based on the results of chemical testing. At low levels of chemical contamination (below screening level chemical concentrations), no biological testing is required in order to recommend unconfined, open-water disposal. At very high levels of chemical contamination (above a maximum level chemical concentration), biological testing is not necessary to indicate that unconfined, open-water disposal would be inappropriate. Interpretation of chemical and biological tests (the disposal guidelines) has been developed using this linked, effects-based approach.

Disposal Guidelines

The basis of the disposal guidelines developed under PSDDA is the avoidance of adverse biological effects due to sediment contamination. This contrasts with the existing approach of relying on comparison to reference conditions (per the interim criteria).

Early in the planning process, PSDDA recognized that the decision on what degree of dredged material contamination is acceptable for unconfined, open-water disposal was not a simple, black and white decision. Many of the study participants held very different views on the degree of acceptable adverse effect at or around the disposal site. Uncertainties in scientific information further emphasized this gray area in decisionmaking. To address this issue, PSDDA defined four alternatives describing the boundary between acceptable and significant adverse environmental effects on biological resources due to sediment chemical contamination. The alternatives described four degrees of increasing effect on resources at the disposal site, from "no effects due to sediment contamination" to "major effects due to sediment contamination." Each of these alternatives were further defined by representative laboratory test results that were considered indicative of the potential effects in the field.

Though preliminary, PSDDA has recommended the alternative labeled "minor adverse effects on biological resources due to sediment contamination." This alternative allows adverse effects within the site boundaries that are predominantly sublethal and developing only after chronic exposure. In laboratory terms, dredged material meeting this definition does not exhibit significant acute mortality in sensitive test species exposed to the sediment to be dredged.

PSDDA chemical test interpretation is based on effects information obtained from field samples where chemistry is compared with biological response. The concept utilized is the apparent effects threshold (AET) (Tetra Tech, Inc. 1986). Illustrated in Figure 8, the AET relies on field and laboratory effects data to calculate chemical-specific sediment quality values. By definition, the AET is the highest concentration observed in a biological response database that did not evidence significant adverse response. The resulting values do not necessarily indicate causal response for each chemical, but rather are indicators of combined chemical effects present in the database. In support of this approach, there are no significant adverse biological responses (occurring below an AET) that are not above an AET for another chemical. Therefore, applied as a set, the AET system has a high efficiency in correct identification of sediments with adverse biological response.

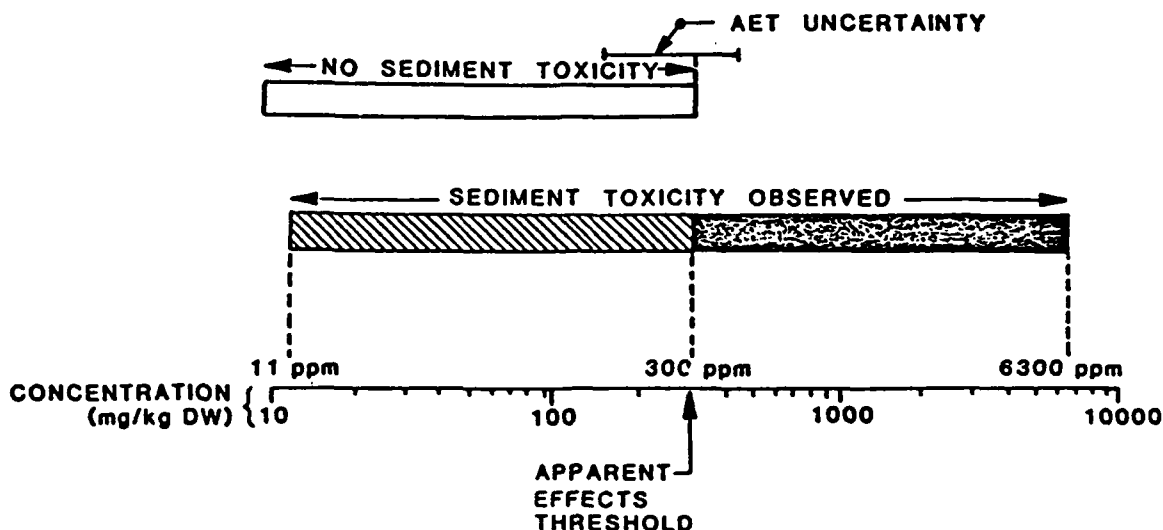


Figure 8. Apparent effects threshold determination

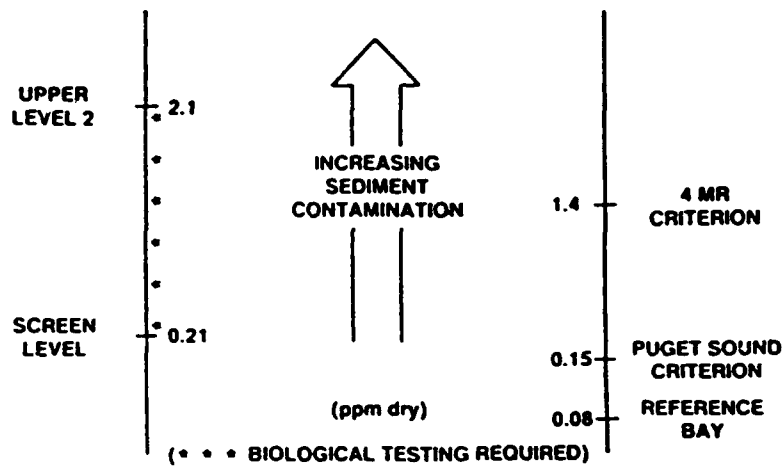
AET's can be (and have been) developed for any number of biological response indicators, lab and field. The maximum level proposed by PSDDA is the highest AET among four (mortality of amphipod species, abnormality of oyster larvae, microtox, and field benthic community degradation). This is the concentration where all biological indicators (lab and field) have become consistently, significantly adverse. The screening level was set at 10 percent of the maximum level, as long as this value was at or above reference area values (for relatively clean and undisturbed areas) and below the lowest AET (where one biological indicator has become consistently adverse).

Figure 9 shows the relation of the PSDDA values to reference area values and interim criteria for several chemicals of concern in Puget Sound. An important feature of this system is that the dredging project's "pass" (acceptable for unconfined disposal) and "fail" (unacceptable for unconfined disposal) values have been separated. In contrast, the interim criteria have combined pass/fail volumes. In most cases, the screening level is below Fourmile Rock values, while the maximum level is above the Fourmile Rock values.

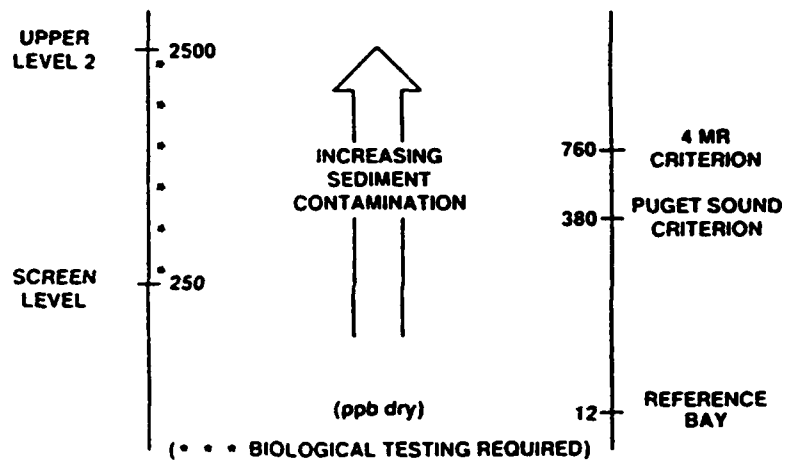
Biological testing is not needed when all chemicals are below screening levels, or when any two chemicals are above maximum levels (or one greatly above). Biological testing is conducted for all other cases. Interpretation of acute tests is shown in Table 5. Single species response guidelines are changed from 25 percent in the interim criteria to 50 percent in the proposed PSDDA approach. In addition, if any two of the PSDDA test species show significant toxicity (even if not 50 percent), then unconfined, open-water disposal is not recommended.

Past interpretation of bioaccumulation tests has relied on statistically significant chemical increases in tissues of test species relative to reference species (often from a relatively clean area). With the sophisticated analytical equipment available today, the finding of statistically significant increases of organic chemicals in test species tissues is increasingly common.

MERCURY



PCB



HPAH

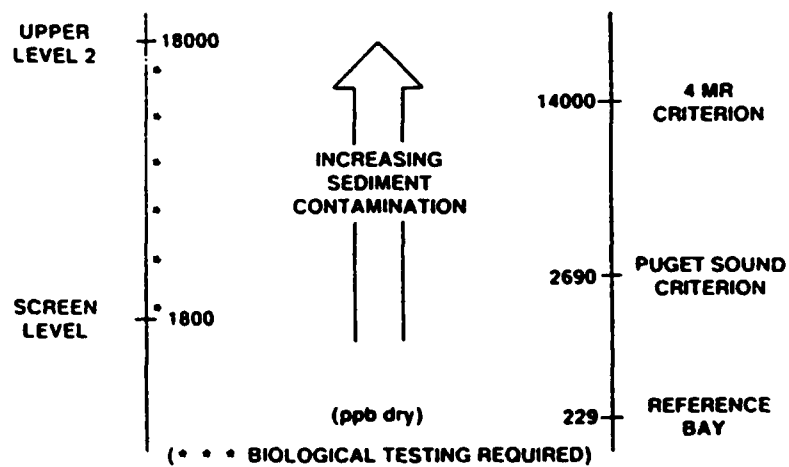


Figure 9. Comparative chemistry

TABLE 5. INTERPRETATION OF BIOASSAY TESTS

	Response Relative To		
	Control*		Reference**
Amphipod (10-day mortality)	< +50%†	<u>and</u>	+ SS††
Oyster larvae (4-day abnormality)	< +50%†	<u>and</u>	+ SS
Bacterial bioluminescence ("microtox") (15-min EC50)	NA‡		> -50%† <u>and</u> - SS

* Control sediment is collected from the source of the test species, typically sand. Control toxicity should not exceed 10 percent.

** Reference sediment is collected from relatively undisturbed areas, a match to dredged material grain size. Reference toxicity should not exceed 20 percent.

† +50% means less than 50 percent higher test response relative to control response. -50% means less than 50 percent lower test response relative to reference response. Values shown are those where a single test result (one of three) will indicate that dredged material is not acceptable for unconfined, open-water disposal. If any two of the three tests are significantly different from both control and reference, unconfined, open-water disposal will not be recommended, even without a 50-percent result.

†† SS = statistically significant.

‡ Controls for microtox testing do not use sediment and are not included in decisionmaking.

This is especially true when the reference area does not contain the chemical of interest. The finding then raises the issue of effects--what does the presence of chemicals in the tissues of test species mean?

In addressing this issue, PSDDA has opted to relate bioaccumulation test results primarily to human health concerns; i.e., when those chemicals of human health concern are elevated in the sediments, the 30-day bioaccumulation test is used to determine bioavailability of these chemicals. The chemicals used are those for which cancer potency (CP) values and reference risk dose (RFD) values (previously known as acceptable daily intake values) exist. Applying exposure analysis and CP and RFD models to the disposal sites, PSDDA has developed target tissue concentrations for interpreting bioaccumulation test results relative to human health concerns. Dredged material that produces lab tissues that exceed these values would not be recommended for unconfined disposal.

For material that is not acceptable for unconfined, open-water disposal, PSDDA is developing testing and design guidelines to assist in decisions on the disposal of these materials. Working through the Waterways Experiment Station, PSDDA has developed a framework for selecting needed design controls and treatment for contaminated sediments requiring confined disposal. The framework, known as the Dredged Material Alternative Selection Strategy (DMASS) (Cullinane et al. 1986) is undergoing refinement by EPWG to define acceptability guidelines for specifying the needed design features.

DISPOSAL SITE MANAGEMENT

The Management Plan Work Group is developing a management plan for each of the open-water disposal sites. The plan will define the roles of local, state, and Federal agencies. Issues include: permit application reviews and processing, permit compliance inspections, treatment of permit violations, disposal site environmental impact monitoring, responding to unforeseen site disposal effects, plan updating, and data management.

The following are preliminary findings and tentative conclusions for some of the issues addressed by this work group:

- a. Permit compliance. Periodic permit compliance inspections will help ensure that assigned disposal sites are properly used. The cost of these inspections will be borne by the project sponsors. Options considered range in cost and certainty from voluntary compliance to independent verification of all facets of site use. Self-monitoring by disposers with spot-checks by regulatory agencies is felt to be the most feasible option. A navigational positioning study concluded that a 600-ft radius target area should be used to ensure that all disposals occur within a 900-ft radius disposal zone (see Figure 6). The smaller target would compensate for some positioning uncertainty at the perimeter of the target area.
- b. Navigation positioning at the disposal site. A navigational positioning study suggests that a combination of LORAN and variable range radar would be a relatively inexpensive and accurate positioning method in areas that fall outside the scope of the Coast Guard's Vessel Traffic Service (VTS). VTS is currently being used to verify positioning before disposal at the Fourmile Rock disposal site. A buoy is proposed for Port Gardner, where LORAN signal inference problems have been experienced.
- c. Local government coordination. A survey conducted of local shoreline control programs revealed significant differences among cities and counties as to the handling of dredging and dredged material disposal. PSDDA is attempting, through coordination, to achieve uniformity and consistency in local government controls.
- d. Regional disposal sites. PSDDA is considering regional open-water disposal sites that will serve the needs of all cities and counties located within each region. One shoreline jurisdiction will be responsible for meeting the needs of all jurisdictions within their region. By participating in site selection and establishment of dredged material evaluation procedures, local governments will be assured that disposal will be environmentally safe, wherever it occurs, while meeting the economic development needs of their communities.
- e. Environmental monitoring. Each of the three Phase I area disposal sites will be monitored for physical, chemical, and biological effects resulting from site use in accordance with the PSDDA guidelines. The environmental monitoring is intended to ensure dredged material disposal accountability by field verification of predicted site conditions following disposal site use. Monitoring, as now

proposed, would involve before and after disposal surveys including sediment characterization onsite and offsite. Benthic infaunal tissue analysis would be performed at offsite stations to determine if any unexpected adverse biological impacts have occurred. The monitoring would be paid for by the State of Washington and by dredgers themselves through disposal site user fees.

PSDDA STATUS AND RELATION TO OTHER PUGET SOUND ACTIVITIES

As mentioned earlier, the cleanup of urban bays is a major activity of the Puget Sound Estuary Program (PSEP), which is also evaluating application of sediment quality values based on the AET concept. In their application, exceedance of sediment quality values would initially be used to identify areas where contaminant sources need to be controlled. They are additionally being considered for use in identifying sediment that is sufficiently hazardous in place to warrant removal or in situ confinement (remedial action). As part of their study, PSEP is responsible for evaluating unconfined, open-water disposal sites previously used for dredged material. PSDDA has coordinated closely with this other program, especially in the areas of sediment quality value and standard methods protocol development.

Another program in Puget Sound that is addressing sediment contamination issues is the comprehensive plan of the Puget Sound Water Quality Authority (PSWQA). The broad charter of PSWQA requires this state agency to act in a review mode for the products of PSDDA. The PSWQA draft comprehensive plan calls for the development of sediment quality standards for the Sound and for identification of multiuser, confined disposal sites for contaminated sediments that are unacceptable for unconfined disposal. While close coordination continues to be needed in areas where the two programs (PSDDA and PSWQA) overlap, the initiative for confined disposal sites is a welcome, and needed, complement to the PSDDA effort.

The release of the PSDDA Phase I draft Summary Report and EIS is scheduled for spring 1987. Several technical reports on the siting studies and development of sediment quality values will be published prior to that time. After completion of necessary reviews and securing shoreline permits from local government, the new PSDDA-developed sites and procedures should be in operation in Puget Sound by November 1987. Efforts for the rest of the Sound (Phase II) will be completed in 1988.

Prior to implementation of any of the work group recommendations, PSDDA is conducting an analysis of the cost implications of the proposals. Final recommendations will need to consider both economic and environmental consequences, and must also await public review and input scheduled for next year. Early response to the proposed changes has been favorable, as the new disposal sites, management plans, and evaluation procedures provide an objective, comprehensive, and long-term plan for dredged material management in the Sound.

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REMOVAL WORK OF ORGANIC DEPOSIT BY OOZER PUMP DREDGE

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BACKGROUND

Through drains from homes and factories, rivers that run adjacent to large cities are becoming more contaminated, worsening the living environment of residents along with development of the cities. The increase in contamination load is caused by the solution and elution of eutrophic salts from the contaminated bottom deposits.

The quality of water in rivers and lakes, which had been progressively deteriorating along with economic development, is today being gradually improved through elimination of contamination by strengthening the regulations for drainage standards and development of sewage works, etc. However, of the rivers that are designated "First Class," environmental standards are met in only about 75 percent of the cases; the remaining cases are unsatisfactory.

Dredging is the purification method of the bottom deposit most often selected. This paper outlines the work of dredging, dehydration, and treatment of surplus water of the subsoil by means of the transportable modular Oozer pump dredge.

PURIFICATION OF RIVERS ADJACENT TO CITIES

Controlling Pollution Sources

Controlling the pollution sources is the most basic and essential method. It includes the perfection of sewage works and the regulation of drainage, both of which should be planned and executed by the municipalities concerned.

Treatment of Bottom Sludge

Bottom sludge increases the pollutant load of the water by the elution of eutrophic elements, which greatly affects water quality. Furthermore, surface turbulence and suspension during flooding is also of great concern. To treat the bottom sludge, capping, on-the-spot solidification, or dredging is usually conducted. Table 1 shows typical bottom sludge treatment methods of rivers adjacent to cities.

Introduction of Purification Water

Though it has some effect by diluting the water quality or preventing the stagnation of flow, the overall pollution load is not changed, and there is a possibility of spreading contamination to a wider area. It is only recommended as a supplementary method to other measures. Also, this method is sometimes accompanied by economic problems such as construction of the estuary weir and complicated problems concerning water utilization and river improvement.



TABLE 1. BOTTOM SLUDGE TREATMENT SYSTEMS OF RIVERS ADJACENT TO CITIES

Main system	Subsystem	Flow	Remarks
On the spot disposal and treatment	Covering with soil/containment	I-1 Temporary closing --- Draining --- Covering/containment	Applicable to part of the river width.
	Solidification/containment	I-2 Temporary closing --- Draining --- Original position solidification (dry method) --- Covering/containment	
		I-3 Temporary closing --- Original position solidification (wet method) --- Covering/containment	
	Solidification/removal	I-4 Temporary closing --- Draining --- Original position solidification (dry method) --- Removal --- Transfer --- Disposal	Many examples
		I-5 Temporary closing --- Removal --- Original position solidification (wet method) --- Transfer --- Disposal	
Dredging disposal and treatment	Dredging/condensation/dehydration	II-1 Pump dredging --- Transfer of sludge --- Sediment condensation/residual water treatment --- Engineering dehydration --- Containment (At final disposal site)	Many examples
		II-2 Pump dredging --- Transfer of sludge --- Sediment condensation/residual water treatment --- Engineering dehydration (At treatment post) --- Transportation --- Disposal	
	Dredging/dehydration	II-3 Grab dredging --- Containment (At final disposal site)	
		II-4 Grab dredging --- Transfer of sludge --- Engineering dehydration or mechanical dehydration (At treatment site) --- Transportation --- Disposal	
	Dredging/condensation/solidification	II-5 Pump dredging --- Transfer of sludge --- Sediment condensation/residual water treatment --- Solidification --- Containment (At final disposal site)	
		II-6 Pump dredging --- Transfer of sludge --- Sediment condensation/residual water treatment --- Solidification (At treatment site) --- Transportation --- Disposal	
	Dredging/solidification	II-7 Grab dredging --- Transfer of sludge --- Solidification --- Containment (At final disposal site)	
		II-8 Grab dredging --- Transfer of sludge --- Solidification (At treatment site) --- Transportation	
	Dredging/special treatment	II-9 Dredging --- Transfer of sludge --- Burning or sintering --- Transportation --- Effective use	

DREDGING EXAMPLE

Of all the purification methods for rivers, dredging is the one most often used for disposal and treatment of bottom sludge. The following paragraphs introduce a project for the removal of organic sludge at Oxbow Lake (which was short cut by a new water-controlling work and became an old river at the winding river mouth).

Disposal and Treatment of Flow

Disposal and flow treatment flowcharts are shown in Figure 1.

Dredging

Dredge Requirements

Since this work has as its objective the removal of the deposited bottom sludge in the old river as part of the Pollution Control Project, the dredging should not cause environmental pollution. The dredging should satisfy the following conditions:

- a. Dredging is possible without fear of stirring, diffusion, etc., of sludge.
- b. Dredging is possible with high concentration so as not to agitate the sludge.
- c. Facilities are available to collect and dispose of the gas generated from the sludge by the dredging.
- d. Dredging is effective and economical.
- e. The equipment can be transported overland.

Dredge Characteristics

Table 2 shows the adaptability of different dredges which meet the requirements of selection listed above.

Selection of Dredge

In consideration of the requirements and the characteristics of available dredges, #1 Oozer pump (transportable, modular), which can be transported overland, was selected. The specifications, operation, arrangement, assembly, and suction inlet are detailed in Table 3 and Figures 2-5.

Transportation/Assembly

The #1 Oozer dredge was constructed in a compact form to remove sludge on small- and medium-sized rivers, canals, and lakes. The hull consists of eight divisional floats, and the driving parts are made of units for each operation. Therefore, it can be transported overland and assembled and disassembled in

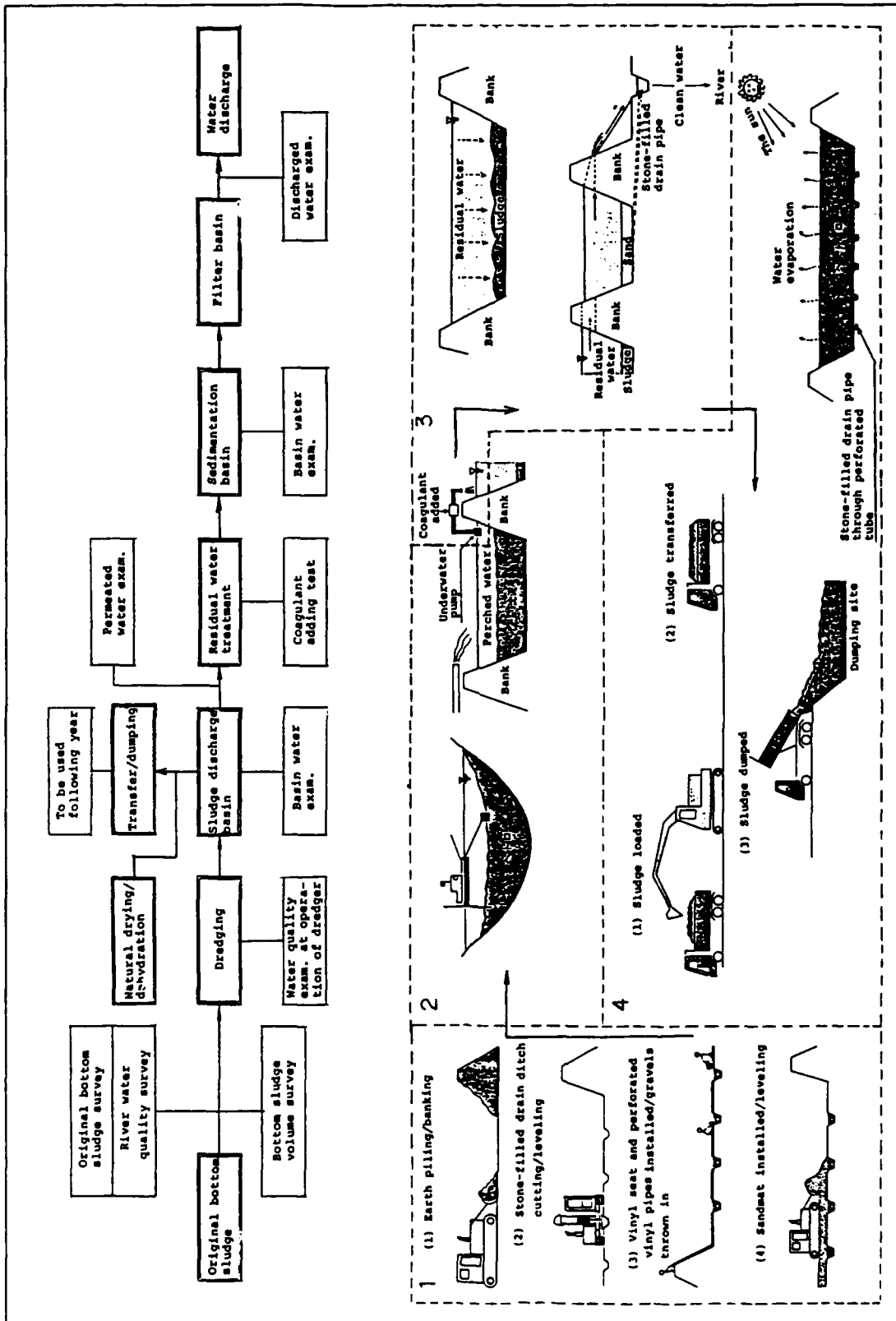


Figure 1. Disposal and treatment of flow

TABLE 2. DREDGE CHARACTERISTICS

Pump Type	Model	Delivery Distance	Outline	Limitations	Adaptability*
Pneumatic pump	Oozer pump	2,000 m	An improved type of Cirrus pump fixed with a pneumatic pump to be able to dredge the high concentration in shallow water. Possible to suck up to about 100% water content. Effective for treatment of residual water and lifted soil	Not suitable for rough and large obstacles	A
	Cirrus pump	1,200 m	Sucking sludge by pressure difference between atmosphere and water; it delivers sludge by compressor. Centrifugal pump is used for longer range booster. Water range of over ~10 m is required	Weak against shallow water and obstacles. Suction limit is about 200-300% water content generally and not good for high concentration	C
	Piston pump	300 m	Forced piston suction (like a water pistol) makes the high concentration dredging possible	Weak against obstacles and delivery range is short. Booster is required for a long range delivery	B
Improved type pump	Low position centrifugal pump	1,500 m	A small centrifugal pump fixed to the tip of an ordinary centrifugal pump for suction purpose; only dredges high sludge concentration. The head is short	Weak against change of water content and soil quality. Treatment of residual water and lifted soil are over its capacity compared with pneumatic pumps. Weak against obstacles and hard ground	B
	Closed grab	--	A type of conventional grab made closed in. Sludge concentration can be high because of direct dredging. Apt to stir original sludge and leave undredged bottom. Powerful against obstacles	Unsuitable for ultrahigh water content disposal and recovering of generated gas. Low dredging precision. Needs a barge	C

* A = Possible.
 B = Possible but inferior.
 C = Not suitable.

TABLE 3. SPECIFICATION OF A SLUDGE DREDGE

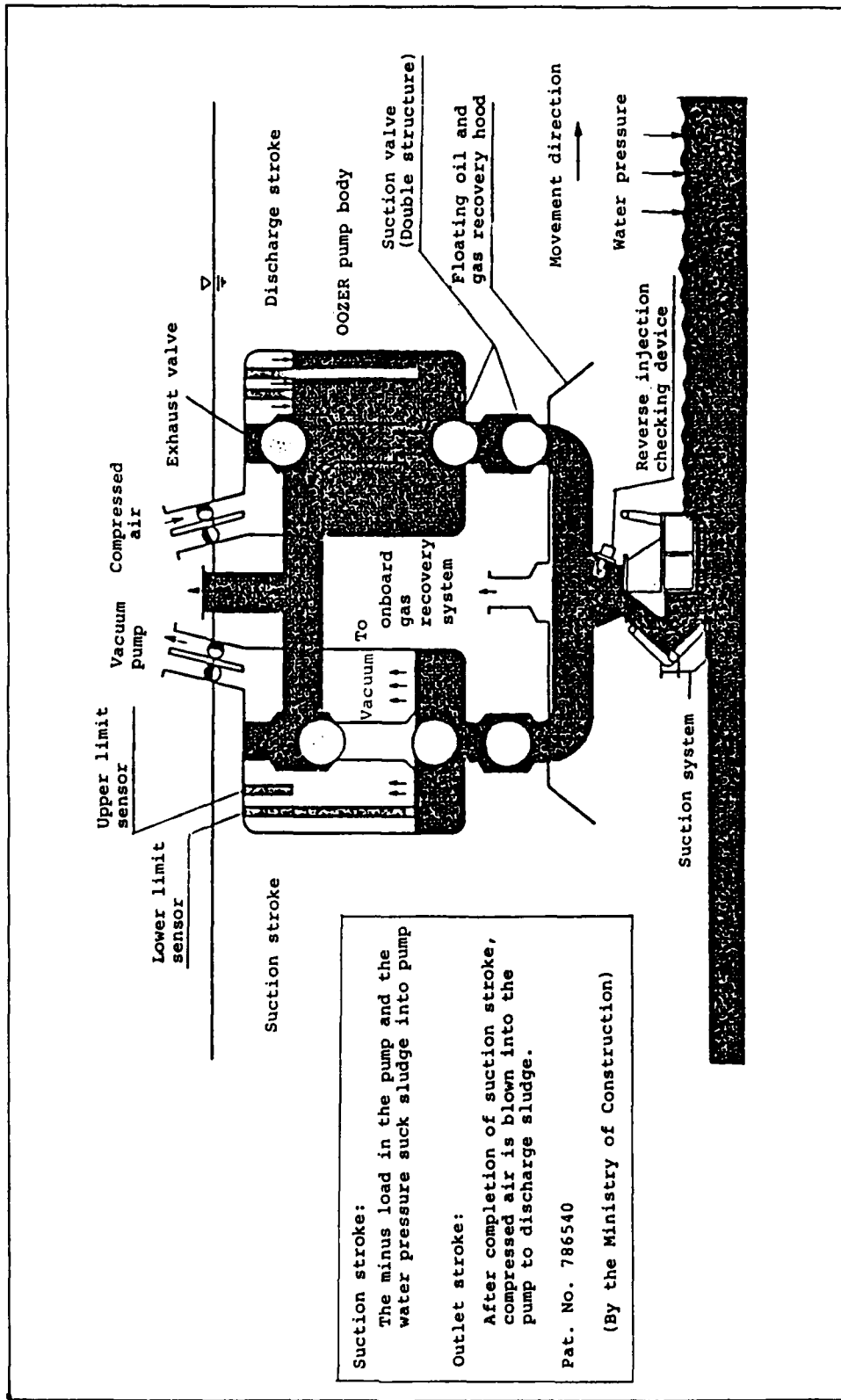
Ship's name	#1 Oozer pump
Vessel type	Eight divisional hull type
Vessel shape	25.0 m (L) by 10/0 m (W) by 1.5 m (D) by 1.0 m (draft)
Displacement	142 tons
Main generator	175 KVA (210 PS), 1 unit
Subsidiary generator	5 KVA (6 PS)
Pump type	Oozer pump energy-saving type
Pump output	
Suction	-500 mm HG by 10.0 m ³ /min vacuum pump, 1 unit
Outlet	17 m ³ /min by 170 PS compressor, 3 units
Dredging depth	-12.0 m
Quantity of solid dredged permit time	30-60 m ³ /hr (mixture: 100-300 m ³ /hr)
Suction mouth shape	Drag type B:1.4 m by H:0.4 m
Delivery tube diameter	Φ250 mm

places where waterway transport is impossible. Also, the ladder shears and spuds can be tilted to pass under bridges and overhead crossing as low as 3 m above the water.

Vehicles needed for overland transportation include:

<u>a.</u> 11-ton truck	2 units
<u>b.</u> Pole trailer	1
<u>c.</u> Low-deck trailer (9.0- by 3.2-m deck)	2
<u>d.</u> High-deck trailer (12.0- by 2.8-m deck)	7
<u>e.</u> Truck crane (100-ton haul)	1 (for assembly/ disassembly)
<u>f.</u> Truck crane (35-ton haul)	1 (for assembly/ disassembly)

Hull divisions are shown in Figure 6. The assembly process is shown in Figure 7.



Suction stroke:
 The minus load in the pump and the water pressure suck sludge into pump

Outlet stroke:
 After completion of suction stroke, compressed air is blown into the pump to discharge sludge.

Pat. No. 786540
 (By the Ministry of Construction)

Figure 2. Oozer pump operation diagram

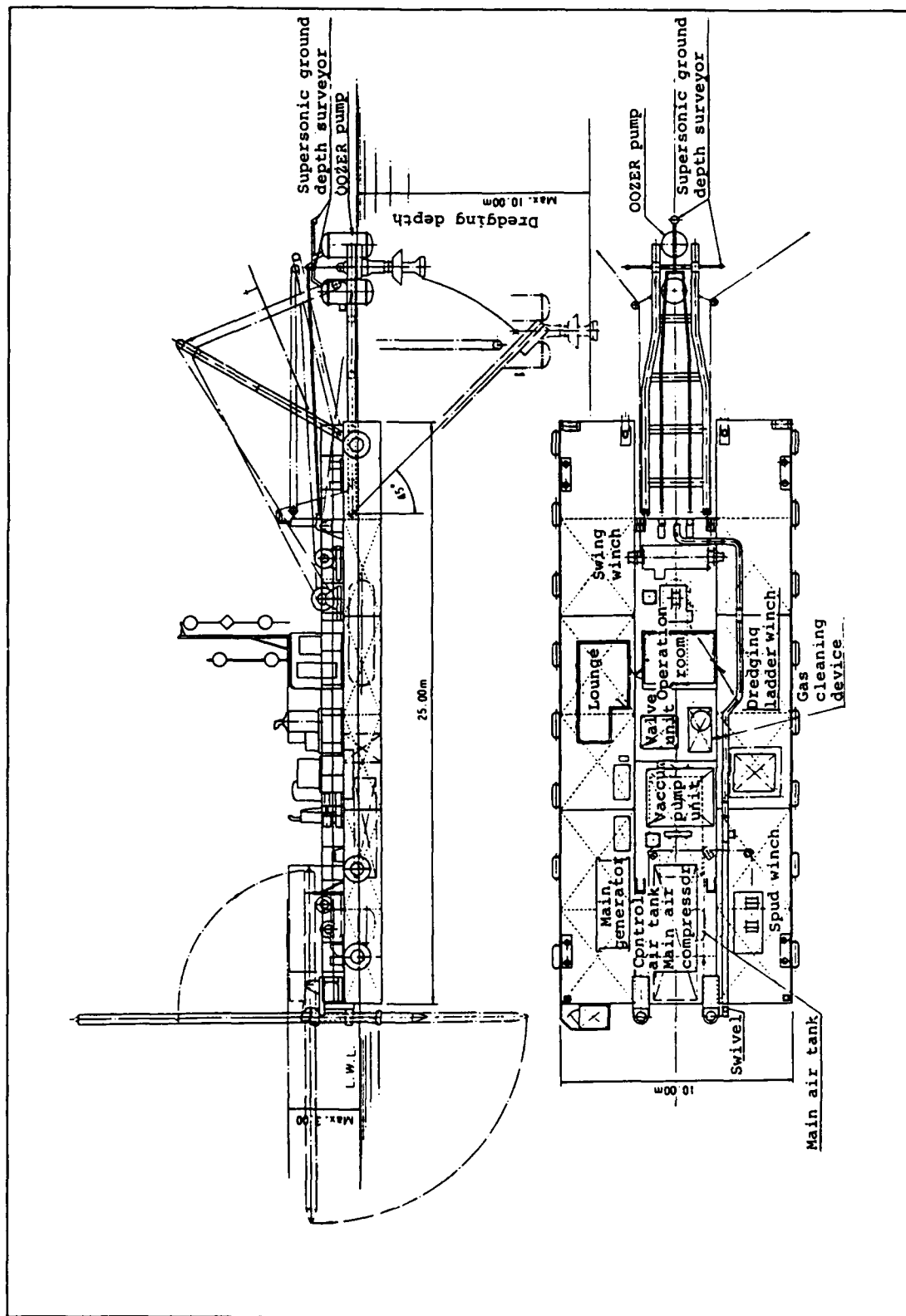


Figure 3. Arrangement plan

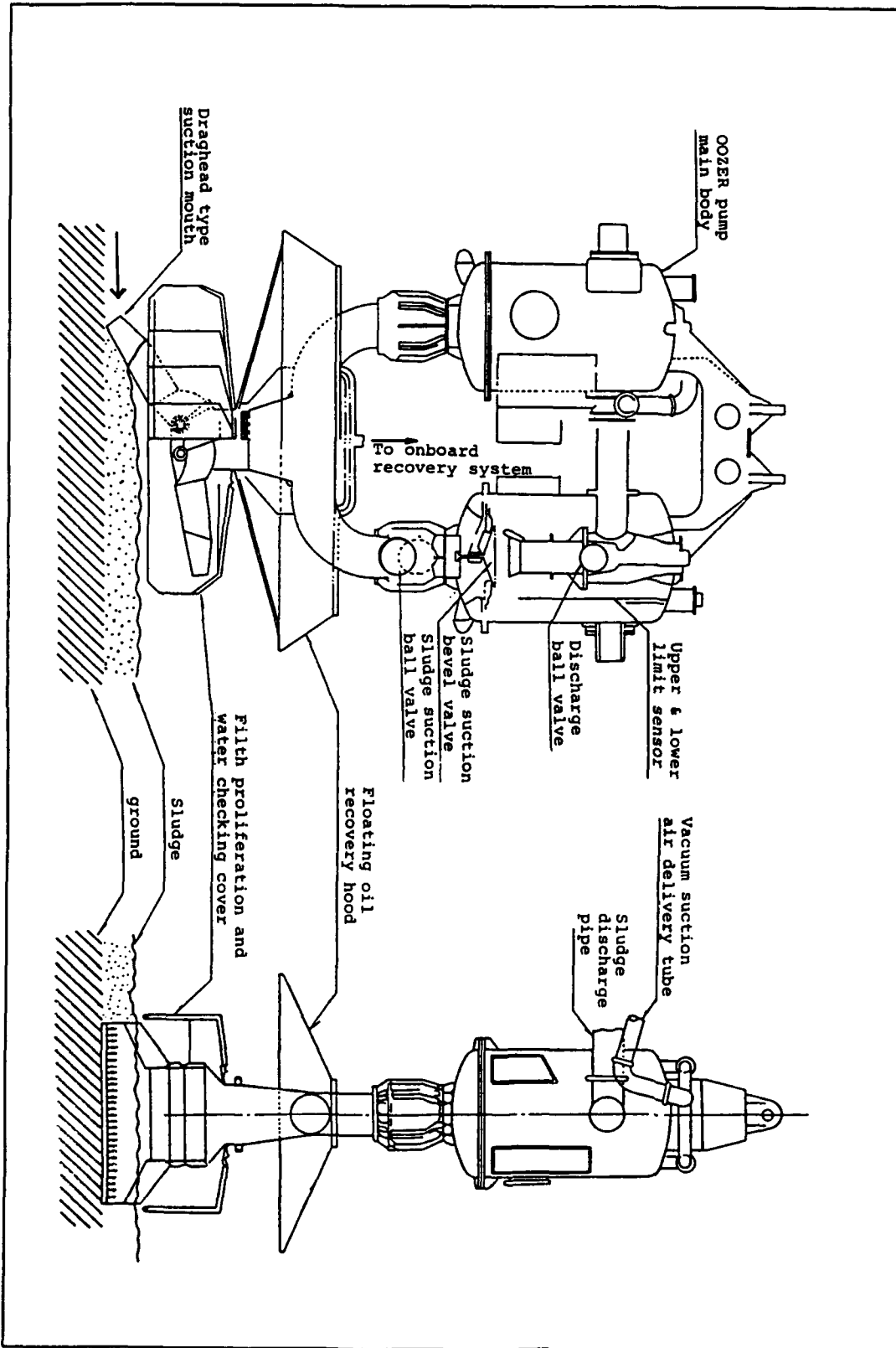


Figure 4. Oozer pump assembly drawing

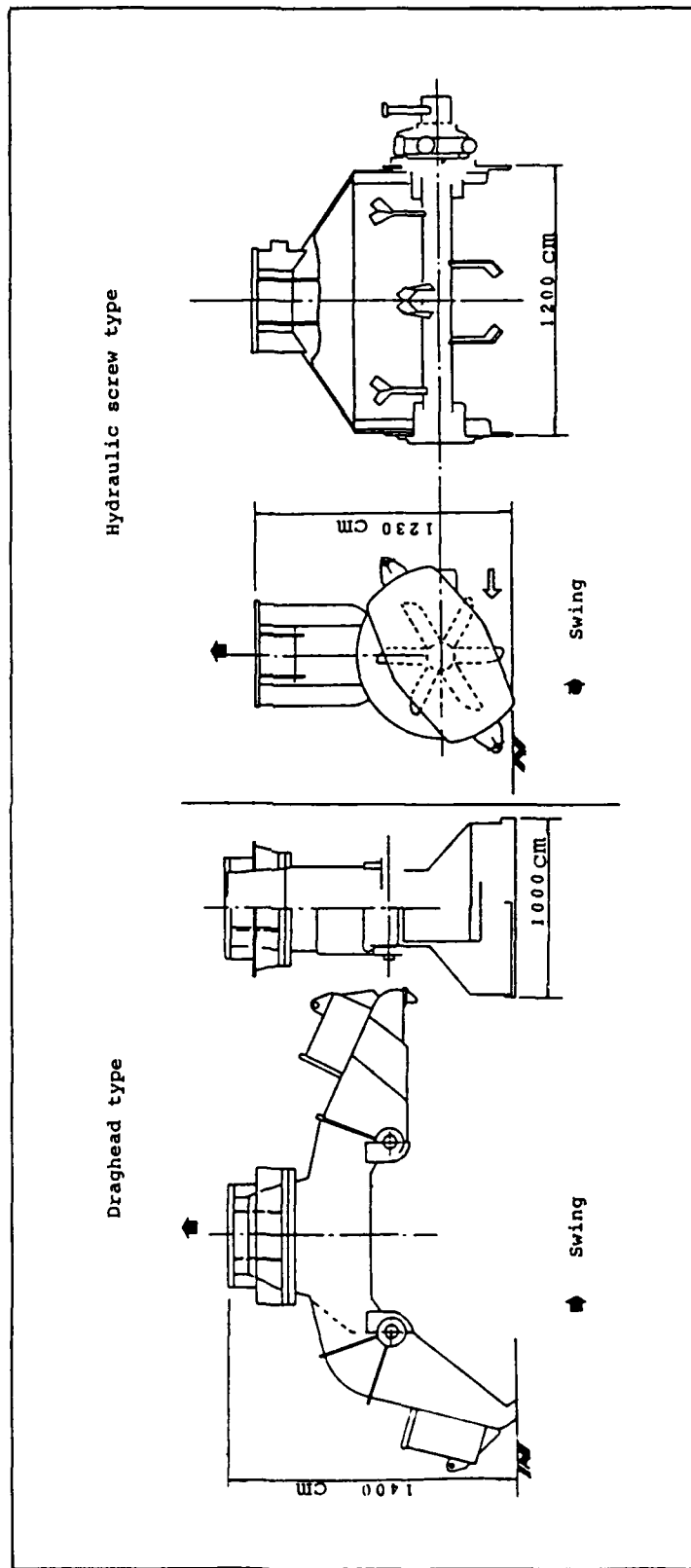


Figure 5. Detailed drawing of suction mouth

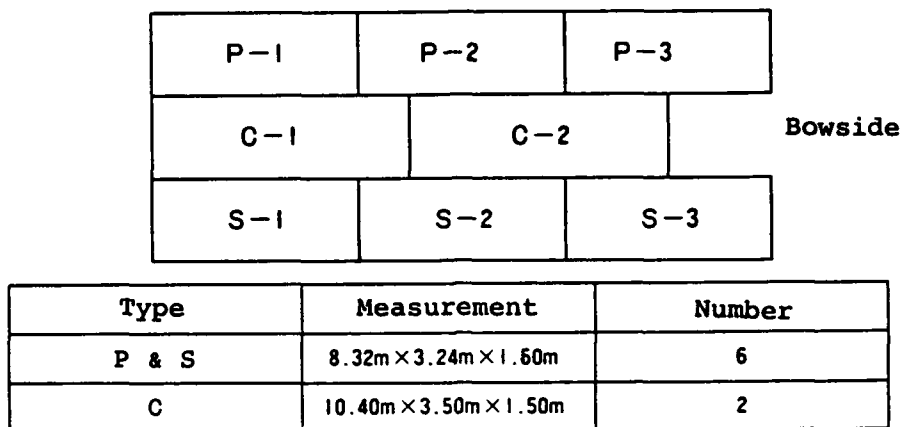


Figure 6. Hull division drawing

Discharge of Sludge

Sludge Discharge Basin

In order to speed dehydration from the bottom in addition to surface sun drying, perforated vinyl pipes with clog preventives (Table 4) were buried under the sandmat to discharge outside the bank (Figure 8).

Sludge Discharge Pipe

The sludge discharge pipe is detailed in Figures 9-12.

Residual Water Treatment

The water quality standard for the discharge water is:

SS density	25 ppm (25 mg/l)
pH	6.5-8.5

The residual water treatment process is illustrated in Figures 13-17.

OPERATION RESULTS

Soil Property

The properties of the bottom soil are shown in Table 5.

Dredging

Dredging Capacity

The overall average for 8 years was as follows:

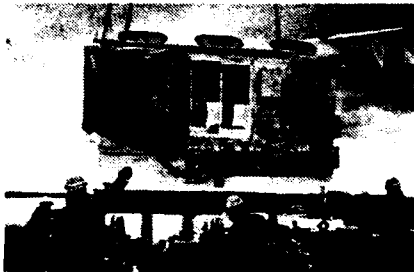
- a. Operation rate: 76.6% (per day of operation).
- b. Operation hours per day: 18 hr 11 min.



a. Transportation of components to the assembly site



b. Assembly of the hull parts



c. Fixing of driving compartments



d. Fixing of ladders and Oozer pump



e. Assembly completed

Figure 7. Assembly process

TABLE 4. CLOG PREVENTIVE DEVICE (TOUGHNER MAT u-60 PROPERTY VALUE)

Item	Property Value
Weight, g/m ³	300 ± 30
Thickness, mm	sby. 3
Tensile strength, kg/5 cm	Lengthwise over 50, breadthwise over 20
Ductility, %	Lengthwise over 80, breadthwise over 90
Tear strength, kg	Lengthwise over 15, breadthwise over 25
Coefficient of permeability, cm/sec	Horizontal: over 1×10^{-1} ; perpendicular: over 1×10^2

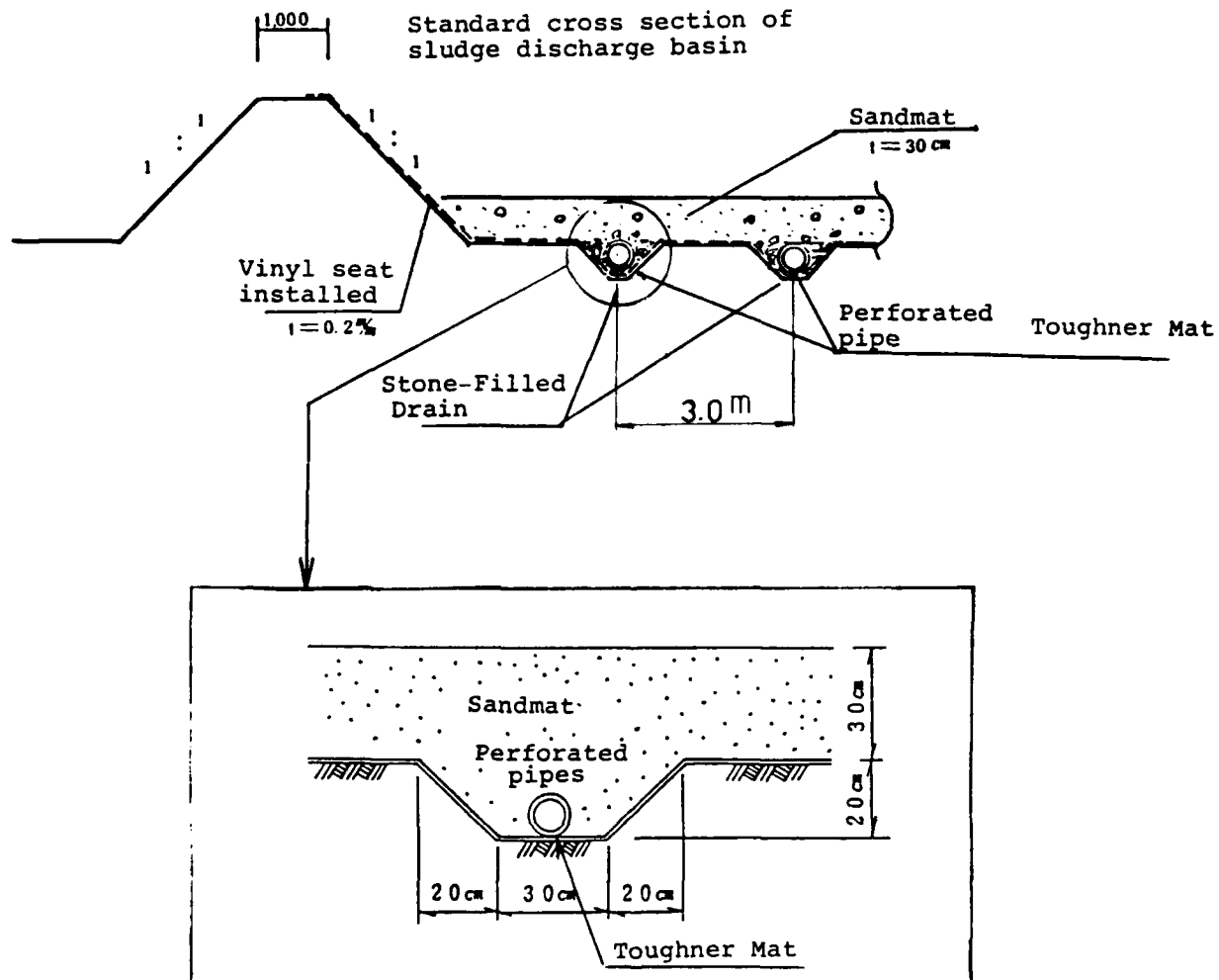


Figure 8. Stone-filled drain plan

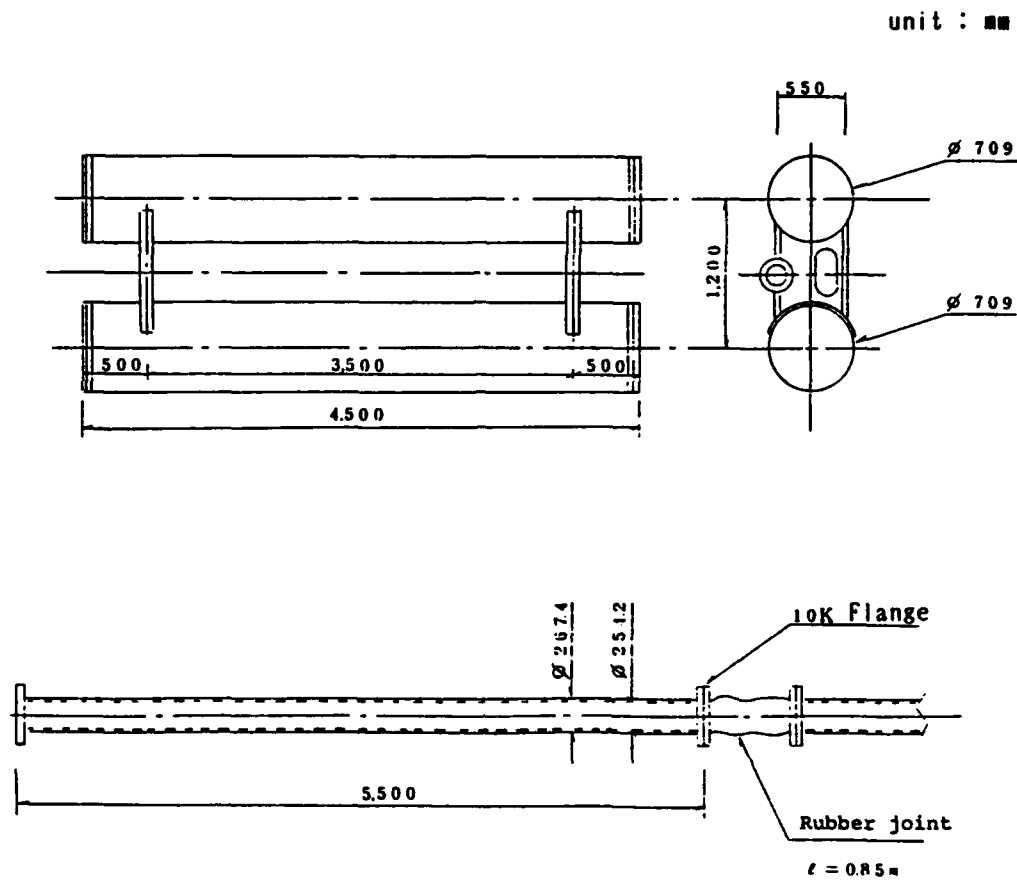


Figure 9. Structure of floating pipeline

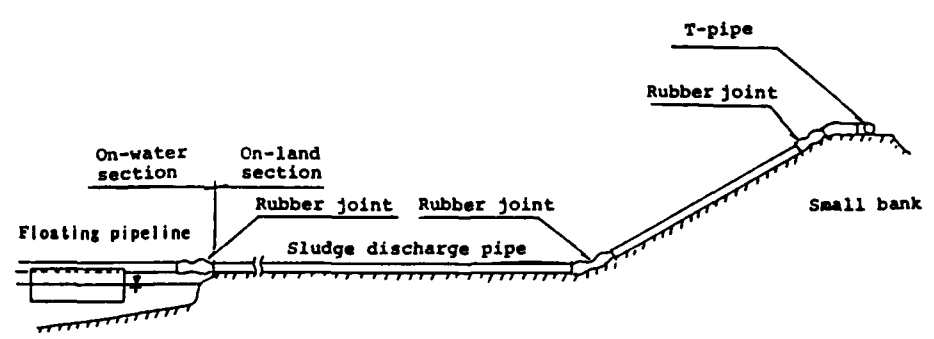


Figure 10. Starting point of land pipe

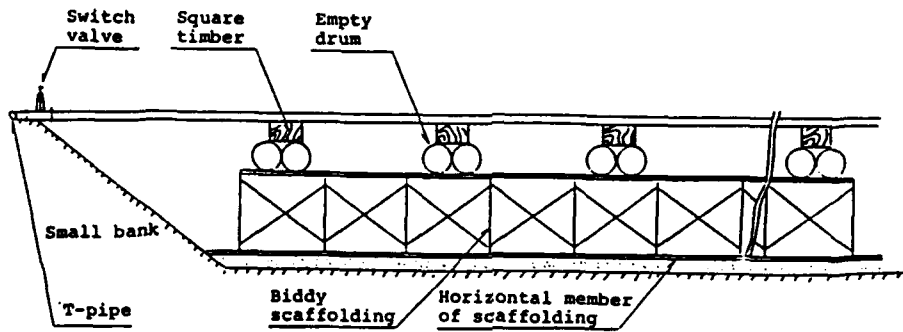


Figure 11. Sludge discharge basin receiving piping plan

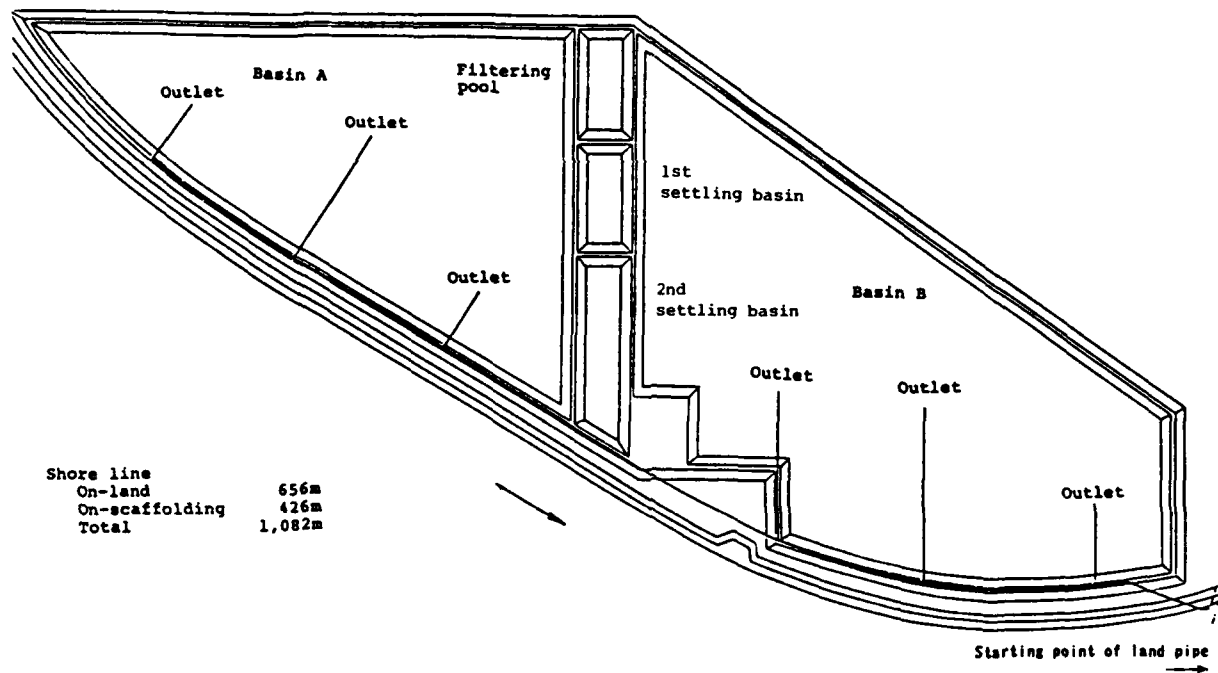


Figure 12. Sludge discharge basin on-land arrangement plan

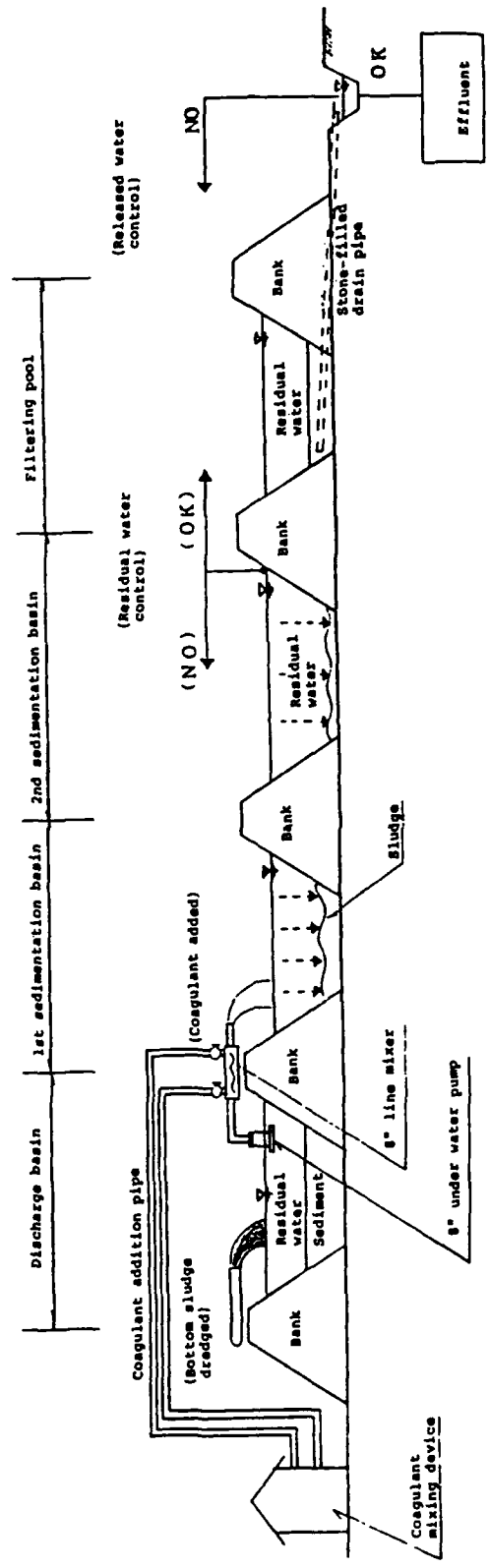
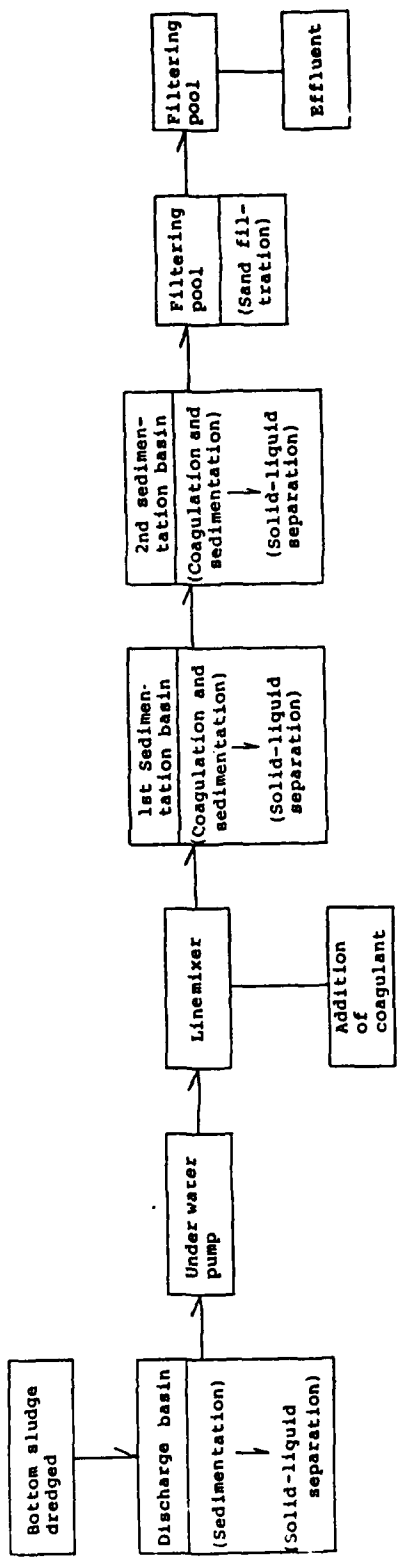


Figure 13. Residual water treatment process

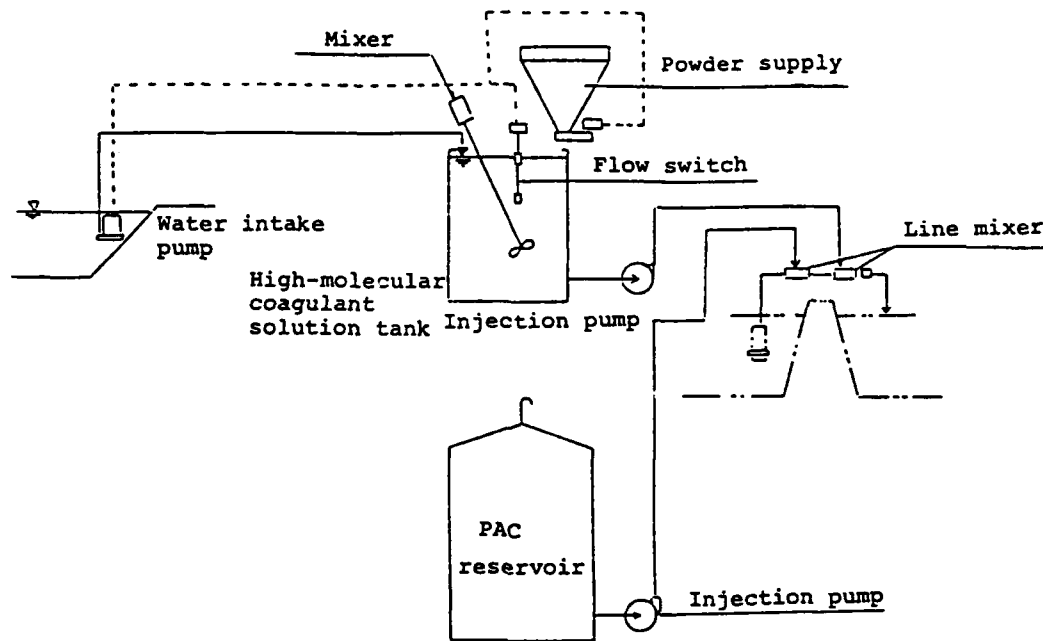


Figure 14. Treatment processing

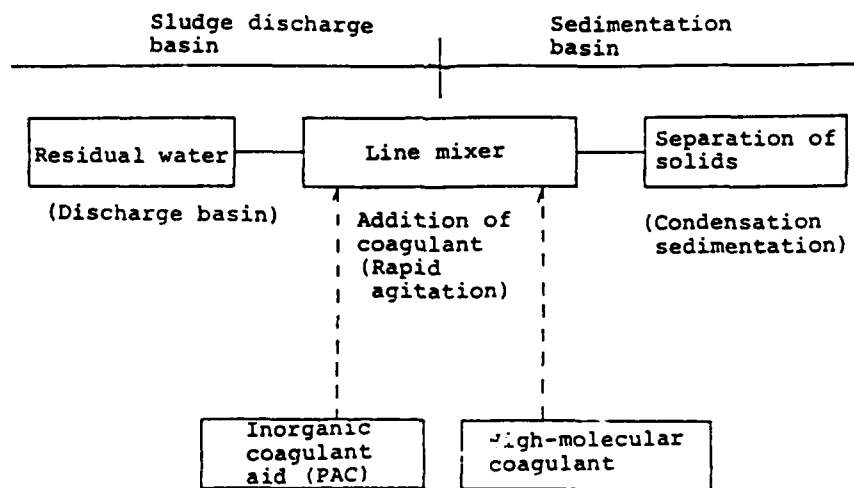


Figure 15. Coagulant addition

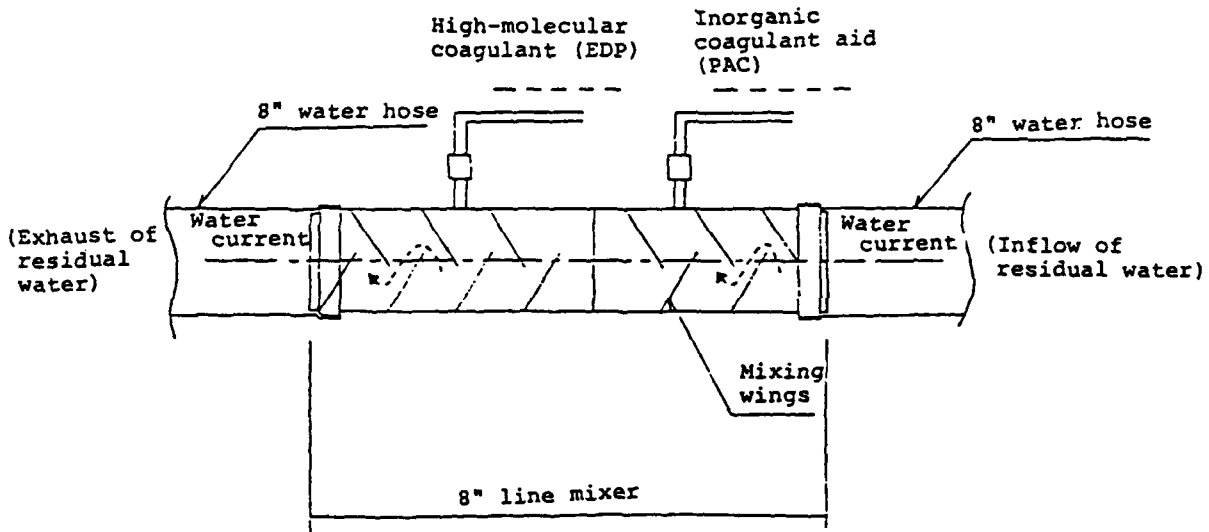


Figure 16. Line mixer

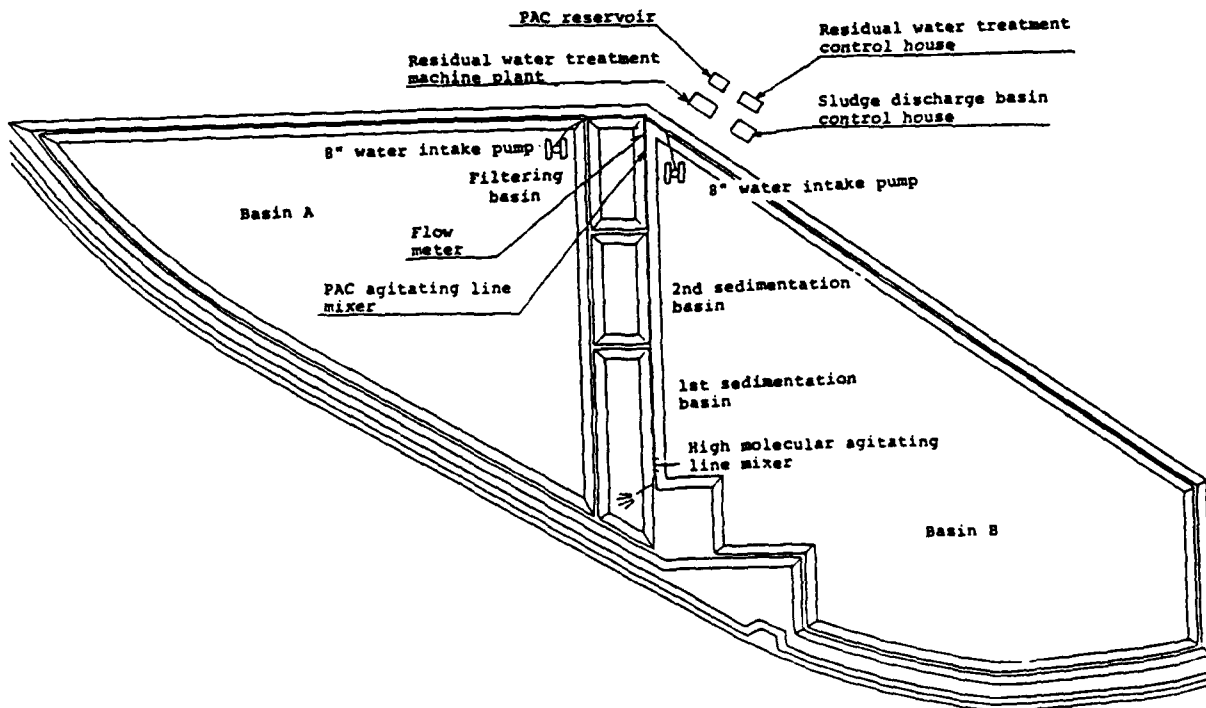


Figure 17. Sludge discharge basin

TABLE 5. WATER CONTENT/SPECIFIC GRAVITY OF SOIL

Year	Downstream		Upstream		Year	Downstream		Upstream	
	Water Content (%)	Specific Gravity	Water Content (%)	Specific Gravity		Water Content (%)	Specific Gravity	Water Content (%)	Specific Gravity
3rd	337	2,381	454	2,330	6th	178	2,355	161	2,452
	265	2,449	221	2,384		165	2,407	141	2,442
	383	2,454	165	2,392		153	2,377	177	2,364
4th	323	2,352	233	2,407	7th	90	2,532	105	2,555
	325	2,365	139	2,370		84	2,518	83	2,567
	213	2,417	147	2,389		81	2,461	74	2,515
5th	194	2,266	201	2,044	8th	394	2,413	403	2,405
	179	2,297	182	2,045		235	2,476	324	2,446
	140	2,272	133	2,185		227	2,511	248	2,522

Upper fig.: upper strata
 Middle fig.: middle strata
 Bottom fig.: lower strata

c. Quantity of solids dredged per unit time (production): 51.5 m³/hr.

d. Apparent suspended solids concentration: 22.9%.

Yearly details are provided in Table 6.

Dredging Precision

Figure 18 shows examples of the dredging precision. The actual result was 0 cm on the minus side and a maximum of 80 cm on the plus side against the required depth of dredging. The average extra dredging was 20 cm.

Pollution

As shown in Table 7, SS values sometimes showed an increase of about 10 mg/l and in other times a decrease; overall, however, pollution proved to be minor when compared with the ordinary pump dredge or the grab dredge.

Residual Water Quality

Table 8 lists the residual water quality values at each treatment processing. The pH value of the permeated water of 6.0 (which is more acidic than standard) is thought to be the result of the iron ions contained in the bottom sludge. However, since there is only a small amount of water involved, it seems to have no influence on the overall project.

Change in Soil Quality After Sludge Discharge

Figure 19 shows the chronological changes in the water content of the bottom sludge discharged into the sludge discharge basin.

SUMMARY

The removal of sludge in designated old rivers is part of the Environment Pollution Control Project of rivers adjacent to cities. This removal was carried out by dredging using the Oozer pump dredge, with dehydration through bottom drainage in the sludge discharge basin along with sun drying. The residual water was then treated.

Dredging Capacity

The capacity of dredging by #1 Oozer dredge turned out to be about 50 m/hr (its nominal capacity is 60 m/hr). Even though the hard bottom layers with low water content near the riverbed, and rocks, boulders, and other obstacles affected the capacity, an improvement in the capacity must be made (Table 6).

Dredging Precision

The average dredging depth was 20 cm over the target depths, which is a satisfactory result (Figure 18).

TABLE 6. SUMMARY OF OPERATION DATA

	Execution Year							Average
	1st	2nd	3rd	4th	5th	6th	7th	
Volume of dredging soil, m ²	18,000	67,170	105,400	102,000	83,475	89,850	65,933	63,589
No. of days of operation	28	63	98	91	76	107	72	64
No. of days of suspension	--	4	2	10	8	7	3	3
Total number of hours of operation	394 hr 05 min	1248 hr 10 min	1885 hr 45 min	1836 hr 55 min	1489 hr 00 min	1876 hr 30 min	1349 hr 10 min	1324 hr 50 min
Total number of hours of suspension	126 hr 40 min	359 hr 50 min	514 hr 15 min	587 hr 05 min	527 hr 00 min	675 hr 30 min	450 hr 50 min	283 hr 10 min
Operation rate	0.757	0.776	0.786	0.758	0.739	0.735	0.750	0.766
Operation hours per day	14 hr 06 min	18 hr 38 min	18 hr 51 min	18 hr 11 min	17 hr 44 min	16 hr 28 min	18 hr 44 min	20 hr 42 min
Volume of dredging soil, m ³ /day	643	1003	1054	1010	994	788	916	994
Volume of dredging soil, m ³ /hr	45.7	54	56	55.5	56.1	48	48.9	48
Apparent sludge concentration, %	35.9	35.8	20.3	18.6	22.9	18.0	16.1	15.6

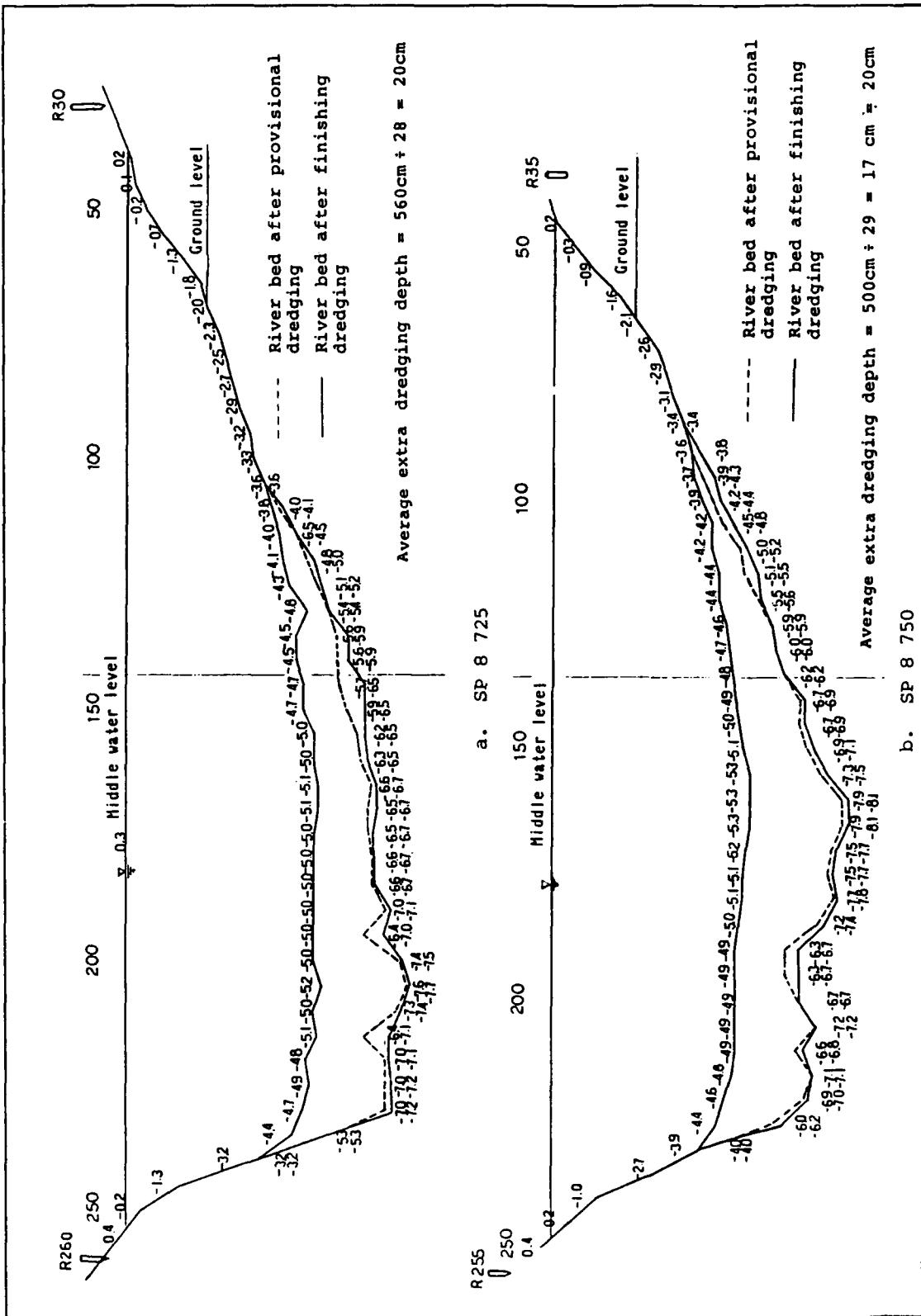


Figure 18. Cross section of dredged sites SP 8 725 and 750

TABLE 7. CONTAMINATION BY DREDGE

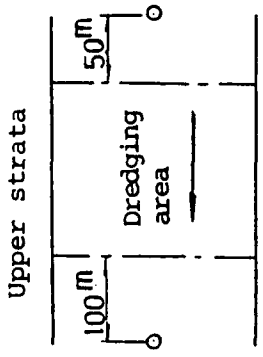
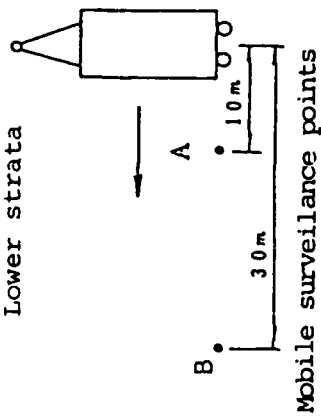
Point of survey	Point	Item	Execution Year				
			4th	5th	6th	7th	8th
Upper strata 	Back-ground	SS mg/l	25	30	19	21	14
		pH	7.5	7.8	7.6	7.9	7.4
Lower strata 	A	SS mg/l	26	26	22	18	15
	B	SS mg/l	31	33	33	23	18
			24	23	22	17	15
			28	27	30	17	17

TABLE 8. RESIDUAL WATER QUALITY AFTER TREATMENT
(FOR THE LAST 3 YEARS)

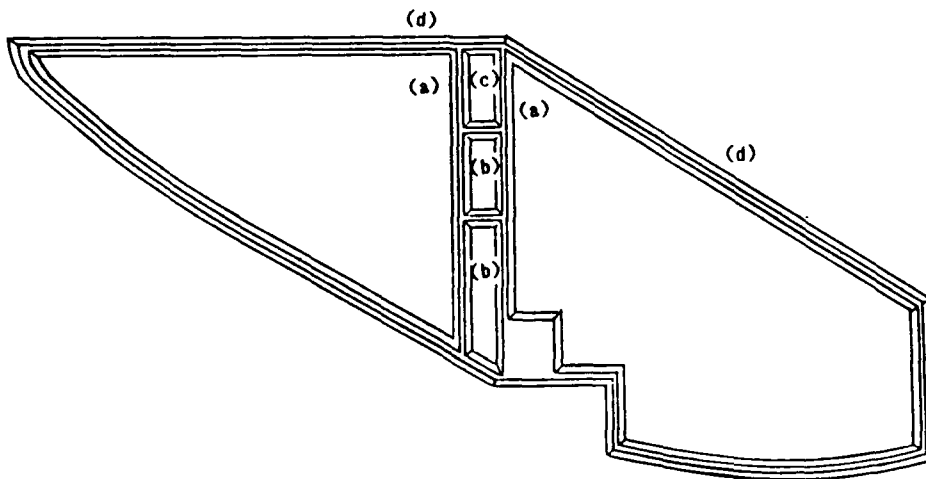
Measurement Point	Item	Execution Year		
		6th	7th	8th
Discharge basin	pH	7.2	7.4	7.0
	SS mg/l	1,135	2,930	1,696
1st sedimentation basin	pH	6.7	7.2	6.6
	SS mg/l	49	18	38
2nd sedimentation basin	pH	6.9	7.0	6.7
	SS mg/l	24	13	16
Effluent water	pH	6.9	6.8	6.8
	SS mg/l	20	10	15
Wrung water	pH	6.0	6.7	6.9
	SS mg/l	4	8	16

Standards for
release water

Allowable values: pH 6.5-8.5
SS below 25 mg/l

Measurement point

- (a) Overflow in discharge basin
- (b) Overflow in sedimentation basin
- (c) Effluent water
- (d) Wrung water



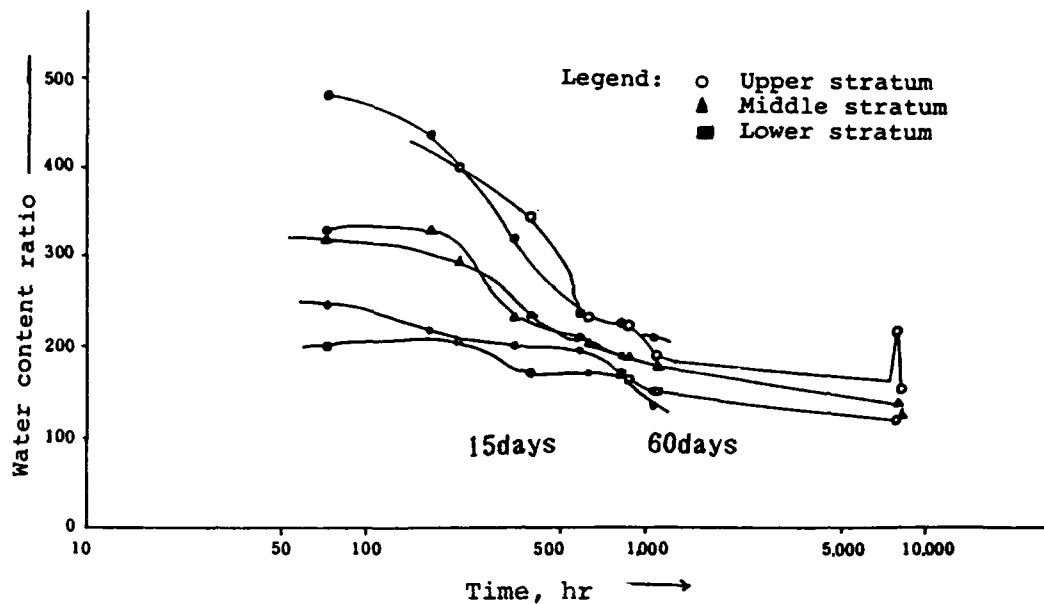


Figure 19. Chronological changes of water content per hour

Pollution

Pollution from the dredging did not show any noticeable differences in places and does not greatly influence the work (Table 7).

Residual Water Quality

The residual water quality conforms to the allowable standards for re-release water (pH of 6.5-8.5; SS below 25 mg/l (Table 8).

Change in Soil Quality After Sludge Discharge

In this project, sludge was dredged, discharged into a basin, dehydrated via sunshine and the stone-filled drain pipe, and dumped at a site on the river about 10 km away. The water content after discharge dropped to about half in 1 year (Figure 19) and caused no problems in transfer or disposal by the dumptruck.

The removal of the organic deposit in the river by the Oozer pump dredge ended with satisfactory results in precision, pollution, quality of the residual water, and dehydration in the discharge basin, although dredging capacity had room for future improvement. Therefore, this process is effective as a purification measure of rivers adjacent to cities.

CONCLUSIONS

There is an increased contamination load in rivers adjacent to large cities due to drains from homes and factories, and water pollution caused by eutrophication due to solution and elution from sludge of bottom deposits is a social problem. To solve this problem, various disposition and treatment

methods, including removal of sludge, have been used. This paper outlined one method--dredging, disposal, and treatment. Dredging was conducted using the transportable modular Oozer pump dredge, and the removal work is still ongoing.


Rivers adjacent to cities are expected to provide sites where citizens can relax and enjoy the site's natural beauty and harmony. We request that the authorities concerned with rivers adjacent to large cities ensure that the rivers be cleaned by the rapid development of sewerage works in tributaries, the removal of bottom deposits, and the self-purification of the river.

SEDIMENT CONTROL ON LAKE EUTROPHICATION
USING AN ECOLOGICAL MODEL

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AD-P006 904

INTRODUCTION

Eutrophication of closed water bodies is still one of the major environmental concerns in Japan. The percentage of lakes and reservoirs whose water cannot satisfy the Environmental Quality Criteria is the lowest when compared with those of rivers or sea areas (Environment Agency of Japan 1985). For the conservation and restoration of closed fresh waters, the Environment Agency enacted a new law, "Law Concerning Special Measures for Conservation of Lake Water Quality" on 13 May 1983. Local governments are required to make all possible efforts for eutrophication control, such as removal of polluted bottom sediments, artificial aeration/circulation, and reduction of nutrient loads.

Ecological models are widely used for assessing the effectiveness of various means for eutrophication control (Nyholm 1978, Orlob 1983, Matsuoka 1984). However, most conventional models presuppose that release rates of nutrients from bottom sediments are constant or are given by a forcing function (Larsen, Mercier, and Malueg 1973; Imboden and Gachter 1978; Gulliver and Stefan 1982). Sediment submodels, therefore, are rarely incorporated into the ecological models (Jorgensen 1976). Strictly speaking, it is almost impossible for these models to predict the effects of sediment dredging and/or reduction of external loadings on the profiles of nutrient concentrations along the depth of sediments and, consequently, interaction (i.e. release and sedimentation) between bottom sediment and overlying water.

The purpose of this study is to develop an ecological model of a lake, Lake Yunoko, combined with a dynamic model of bottom sediments and to assess various policies for eutrophication control with special emphasis on management of the bottom sediment.

STUDY AREA

Lake Yunoko is well known for its trout fishing and its hot spring resort in Nikko National Park. However, wastewater from lakeside hotels has been

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accelerating lake eutrophication. Lake Yunoko (Figure 1) is a dimictic lake (mean depth = 7.4 m, surface area = 0.35 km²) with a hydraulic retention time of ca. 40 days.

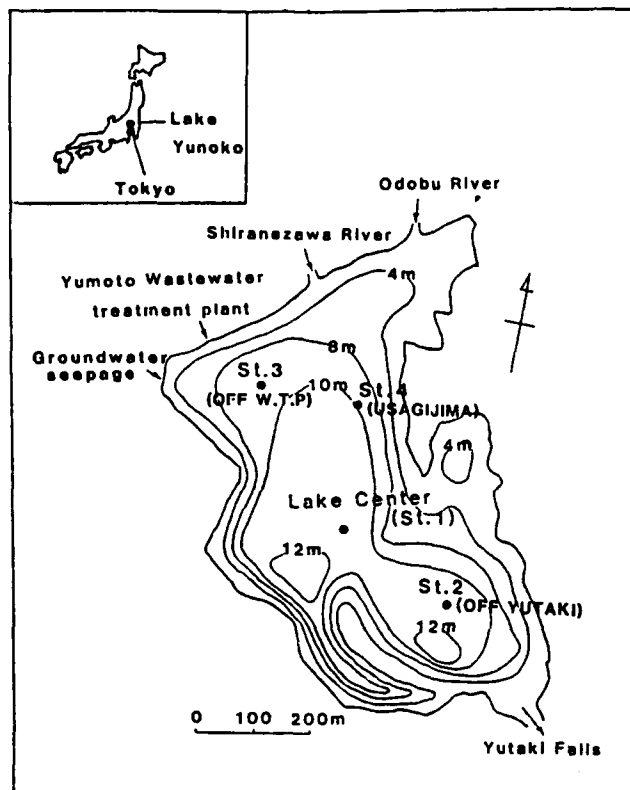


Figure 1. Lake Yunoko

Inflows include the effluent from Yunoko Wastewater Treatment Plant, Odobu River, and groundwater seepage. The only outflow is Yutaki Falls.

Intensive limnological surveys were carried out to obtain basic information both for construction and verification of the ecological model and to assess the response of the lake to advanced wastewater treatment for phosphorus removal, a phosphorus detergent ban, and dredging of the lake bottom sediment (Tochigi Prefectural Government 1978; Hosomi et al. 1984; Hosomi, Okada, and Sudo 1986).

These investigations revealed Lake Yunoko to be eutrophic. Chlorophyll-a concentrations often exceed 20 µg/l. Total nitrogen and total phosphorus concentrations in the epilimnion ranged from 0.2 to 0.8 mg N/l, and from 0.02 to 0.10 mg P/l, respectively. Bottom sediments were found to accumulate large amounts of organic materials (18-21% I.L., 1.6-2.7 mg P/g, 6-7 mg N/g) and sulfide. At present, the local government is planning to dredge all the contaminated bottom sediments.

MODEL STRUCTURE

The model proposed in this study is an ecological model combined with the water temperature model. This is a logical extension of the one-dimensional stratified deep reservoir model of the Water Resources Engineers, Inc. (WRE 1969) or the Massachusetts Institute of Technology (MIT) (Huber, Harleman, and Ryan 1972) type in order to include vertical transport of dissolved and

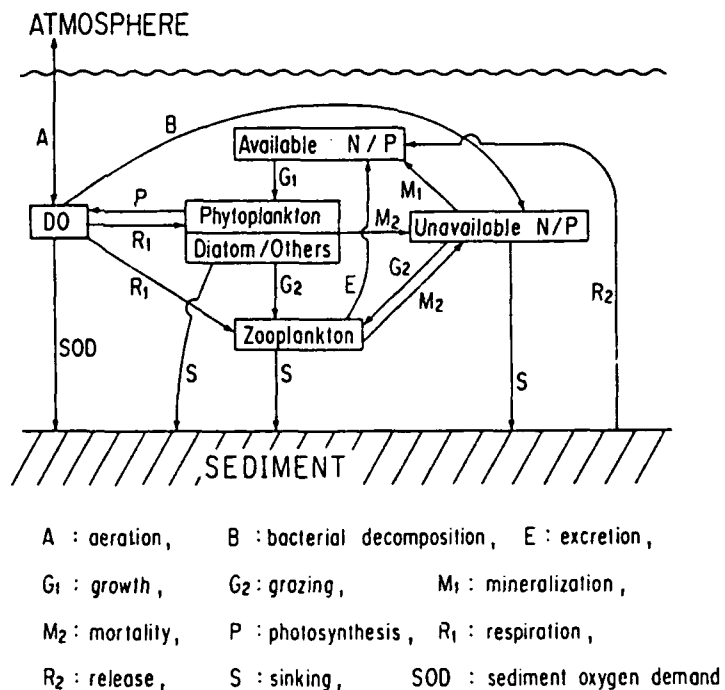
particulate materials. The model, therefore, should be called a one-dimensional, water temperature-water quality ecological model and is basically similar to LAKECO developed by Chen and Orlob (1975), which is embodied in the package WQRRS (Water Quality River-Reservoir Simulation) of the Hydrologic Engineering Center of the US Army Corps of Engineers (HEC 1978).

This model incorporated a detailed sediment submodel to simulate the nutrient cycle between water and sediment and to predict concentrations of dissolved and particulate nutrient in bottom sediments.

As Lake Yunoko is ice covered in winter (Mori and Yamamoto 1975), a one-dimensional water temperature model such as WRE and MIT cannot simulate the phenomenon of ice covering and melting. The Energy Balance model (Greene 1981) was introduced into the MIT model to simulate ice covering and melting and profiles of water and ice temperature during ice-covered periods.

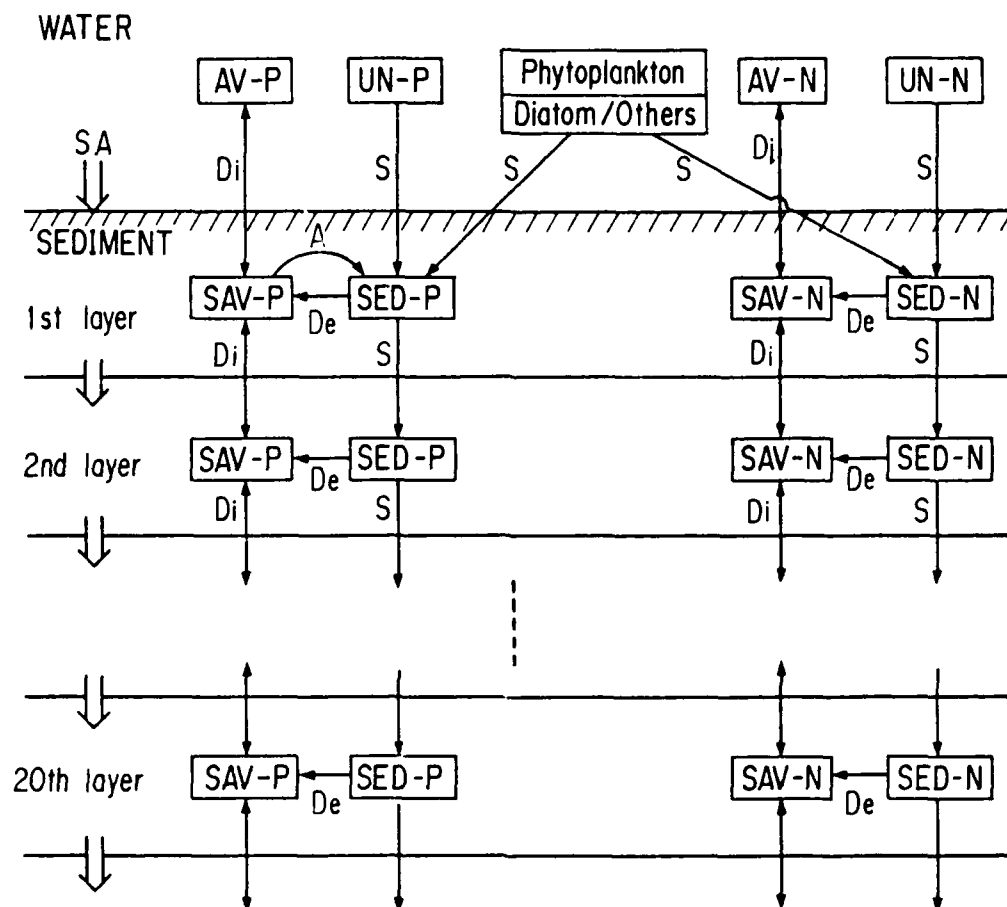
Figure 2 shows the relationships among the various constituents of the aquatic ecosystem. Available nutrients consist of AV-P (dissolved inorganic phosphorus) and AV-N (dissolved inorganic nitrogen). Unavailable nutrients correspond to detritus phosphorus and nitrogen because dissolved organic phosphorus and nitrogen concentration is low in Lake Yunoko (Hosomi et al. 1984).

Figure 2. Schematic diagram of the structure of the ecological model proposed in this study



Dominant species of phytoplankton in Lake Yunoko are diatom such as *Asterionella formosa*, *Synedra acus*, and *Fragilaria crotonensis* and flagellatae such as *Uroglena americana* and *Cryptomonas* sp. (Hosomi et al. 1984). It was found that sinking velocity of diatom such as *A. formosa* is much higher than that of flagellatae (Smayda 1964, Titman and Kilham 1976, Hosomi et al. 1984). Also, large diatoms are considered not to be grazed easily by *Keratella cochlearis* and *Bosmina longirostris*, which are dominant species of zooplankton in Lake Yunoko. Accordingly, in this model, phytoplankton were divided into two types: diatoms and other phytoplankton such as Cryptophyceae and Chrysophyceae.

Figure 3 is a schematic diagram of the structure of the sediment sub-model. The basic concept of this submodel is equivalent to the nitrogen release model developed by Hosomi, Okada, and Sudo (1985) and includes the transformation process of phosphorus and/or nitrogen compounds and the sedimentation process.



SA : sediment accumulation and compaction,
 AV-P : available phosphorus, AV-N : available nitrogen,
 UN-P : unavailable phosphorus, UN-N : unavailable nitrogen,
 SAV-P : dissolved phosphorus in pore water,
 SAV-N : dissolved nitrogen in pore water,
 SED-P : particulate phosphorus in sediment,
 SED-N : particulate nitrogen in sediment,
 Di : diffusion, S : sedimentation, De : decomposition, A : adsorption

Figure 3. A schematic diagram of the structure of the sediment submodel proposed in this study

Sediment accumulation by continuous deposition of particulate matter (allochthonous and autochthonous) at the surface layer of the bottom sediment shifts the sediment/water interface upward in reference to a fixed datum. In this model, this velocity term was defined as sedimentation rate.

Sedimentation processes include sedimentation of particulate phosphorus and nitrogen in the sediment and diffusion of available phosphorus and nitrogen in the pore water along with concentration gradients. Transformation processes of phosphorus and/or nitrogen compounds in the sediment-water system are as follows:

- a. Deposition of unavailable nutrients and phytoplankton in the overlying water to the surface of the bottom sediment.
- b. Reaction (solubilization) from particulate species to dissolved species by biological degradation and desorption.
- c. Adsorption of available phosphorus in the pore water onto the solid sediment phase.

The ecological model proposed in this study is a 12-layer model in which Lake Yunoko is divided into 12 horizontal slices along the depth of water at 1-m intervals. There are two sediment submodels, i.e., sediment A represents the bottom sediment of the littoral zone, from 0 to 4 m, and sediment B corresponds to the profundal zone, from 4 to 12 m. Each submodel is divided into 20 layers from the surface with intervals of 1 cm.

MODEL FORMULATION AND CALCULATION

Figure 4 shows a flow diagram for the ecological model. The system of differential equations and adjunct equations (not shown here) to define this model containing ecological processes and processes of interactions between sediment and water is constructed based on the concept shown in Figure 4.

Field data from Lake Yunoko and laboratory data using sediments of Lake Yunoko were used extensively to evaluate parameters in the model (Hosomi et al. 1984). When sufficient information was not available, the literature (Nyholm 1978, Jorgensen 1979, Scavia and Robertson 1979, Matsucka 1984) was also consulted to provide a range of values for the parameter.

A numerical procedure by the Runge-Kutta-Gill method was employed to solve the system of differential equations (HITAC M-280H, Computer Center, National Institute for Environmental Studies). The period of numerical calculation is from 1 May 1979 to 1 December 1982. Forcing functions are weather (Nikko Meteorological Station, near Lake Yunoko), volume and temperature of inflows, and inflow concentrations of nutrient and dissolved oxygen (daily data obtained by interpolation of observed data (Hosomi et al. 1984)). Observed values on 1 May 1979 were used as initial values of state variables in the model.

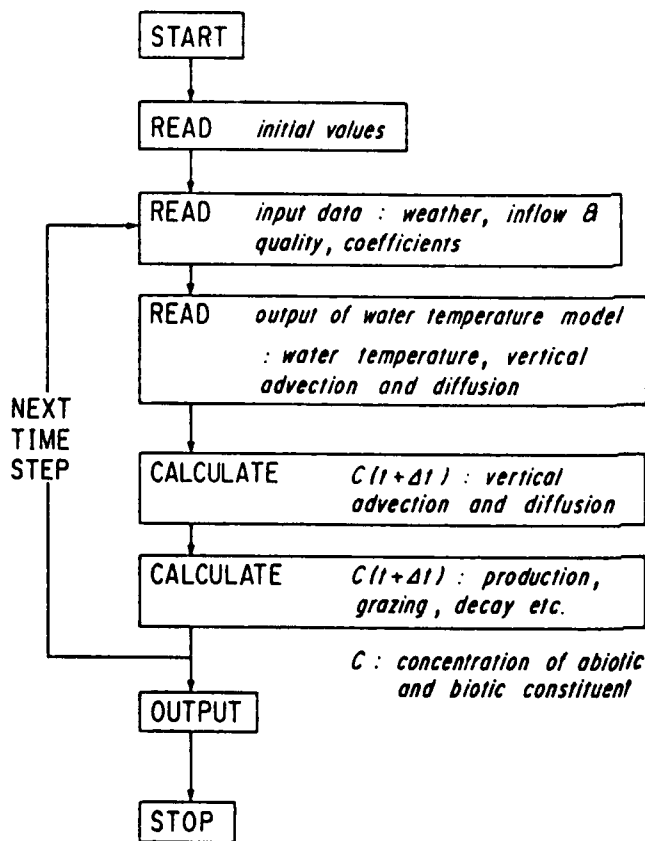


Figure 4. Flow diagram for the ecological model

MODEL CALIBRATION

The outputs of the water temperature model are shown in Figure 5. Good agreement exists between the model calculations and the measured profiles of water temperature. The water temperature model seems to simulate the phenomenon of ice covering and melting very well.

Figure 6 shows the results of simulation runs by the ecological model. Most of the calculated values are in reasonable agreement with the observed data. However, the ecological model could not simulate observed chlorophyll-a concentrations, which scattered widely during short terms in the summer stagnation period. Sinking velocities estimated from observed chlorophyll-a concentration in water and sedimentation fluxes of chlorophyll-a obtained by using sediment traps showed extensive variations during 1 week in summer. In this model, the mean values of these sinking velocities were employed, so the errors in averaging processes may significantly affect the accuracy of calculated curves. Further studies on the sinking processes of phytoplankton and other particulate materials (detritus) would be necessary.

Hosomi, Okada, and Sudo (1985) reported that the nitrogen dynamics of the upper layer (0 to 1 cm) of the sediment regulate the nitrogen transportation between sediment and water. So, we compared calculated with observed particulate phosphorus and nitrogen in the surface sediment.

The calculated values of particulate phosphorus and nitrogen in the surface layer of the sediment ranged from 1.2 to 2.2 mg P/g and from 5.7 to

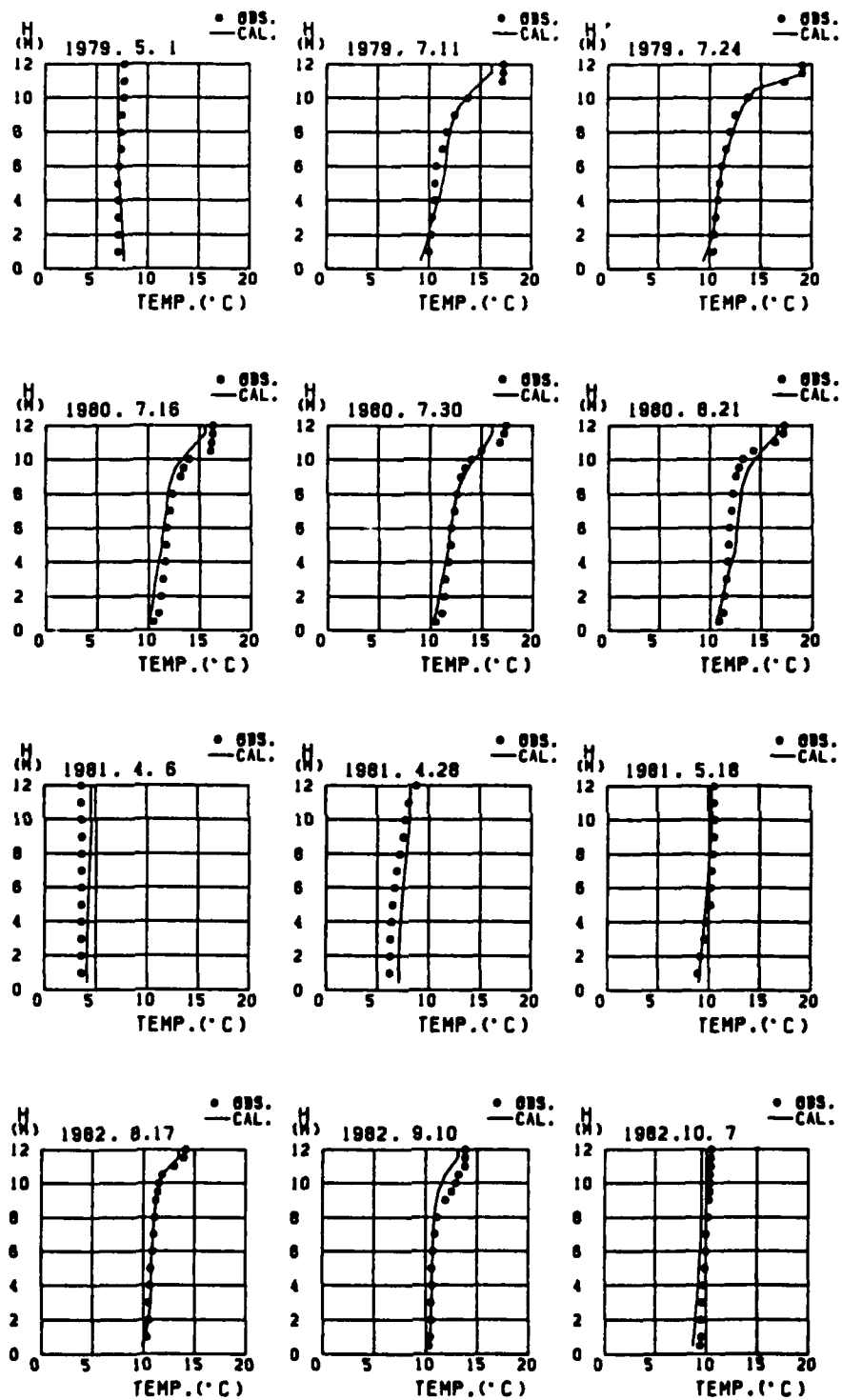
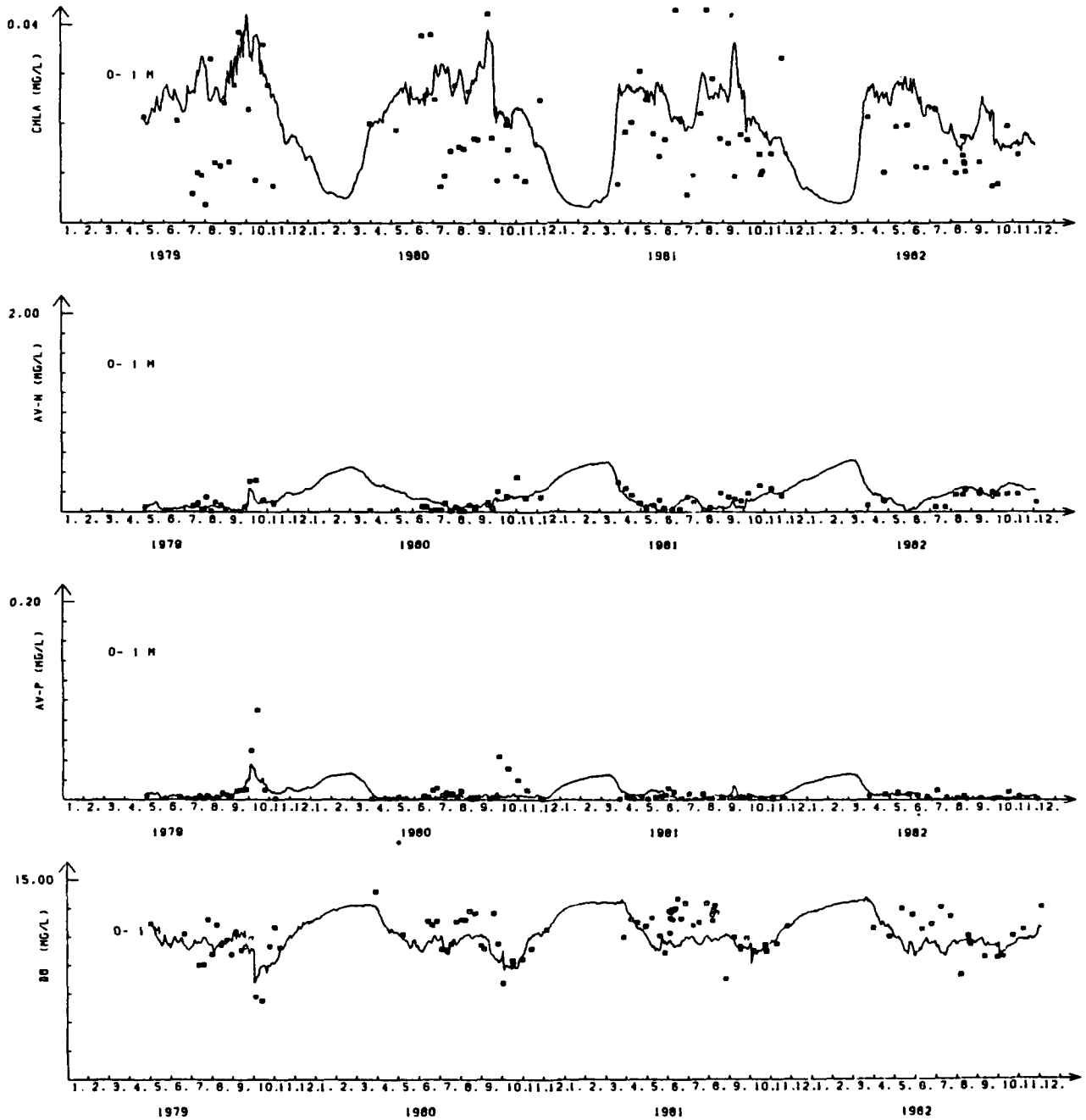
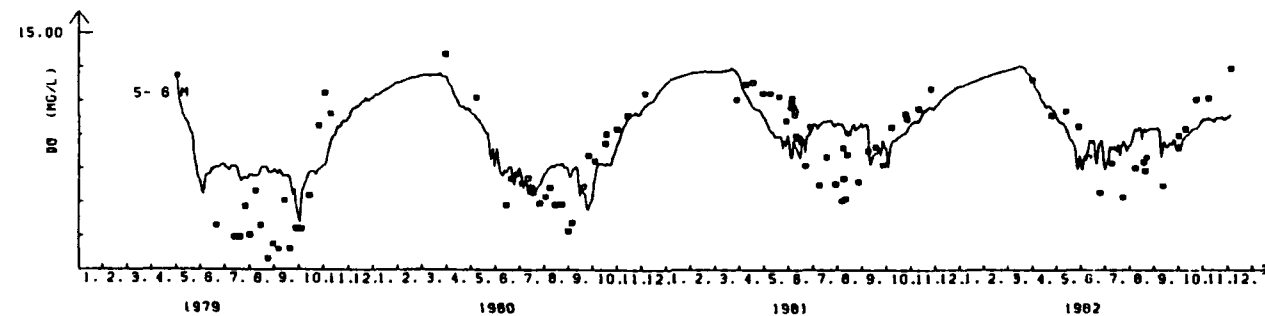
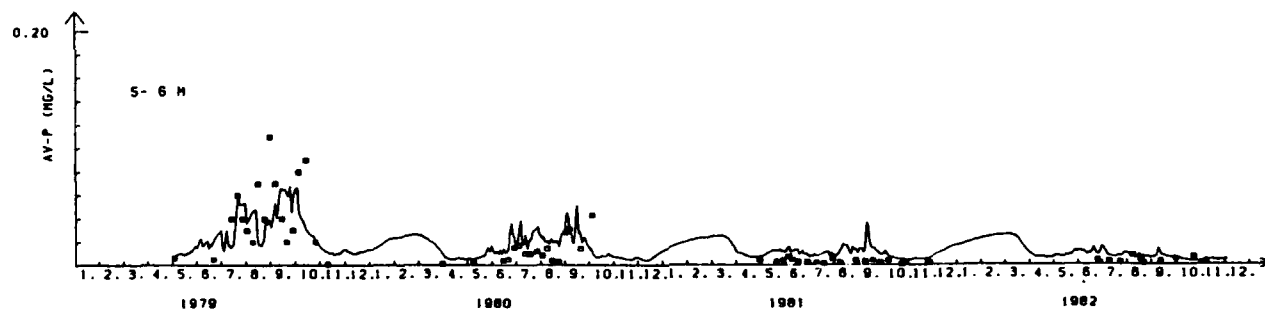
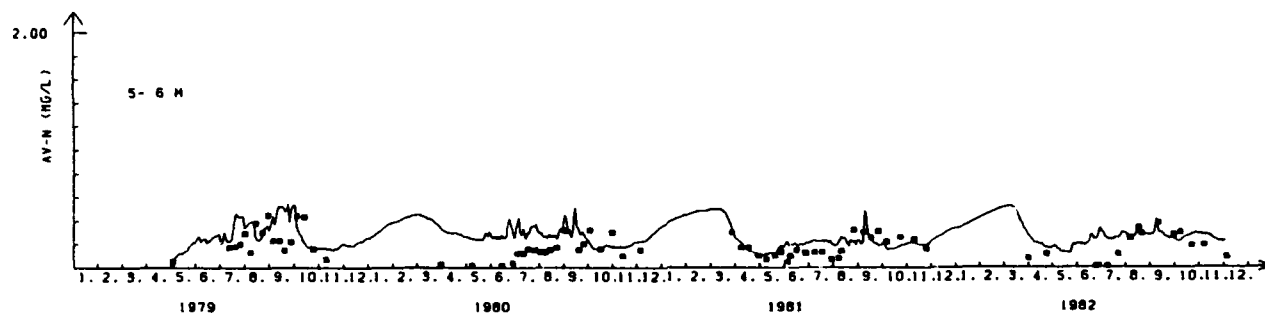
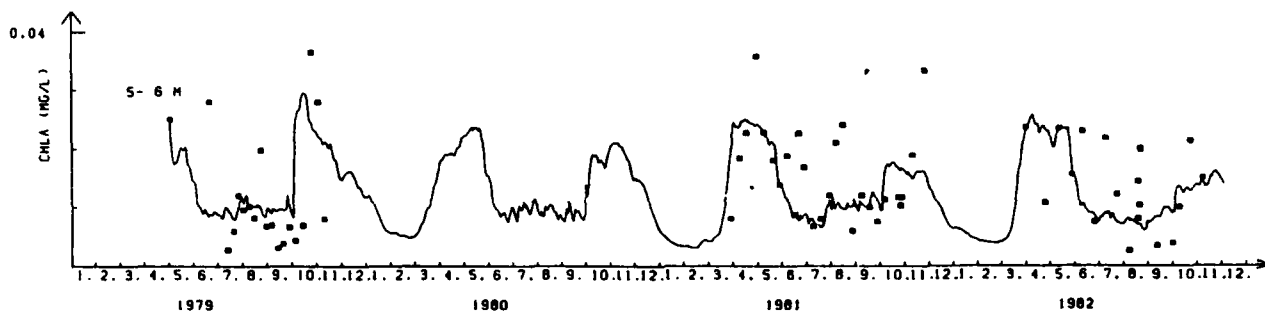


Figure 5. Profiles of water temperature in Lake Yunoko; comparison of calculated (---) and observed data (●)



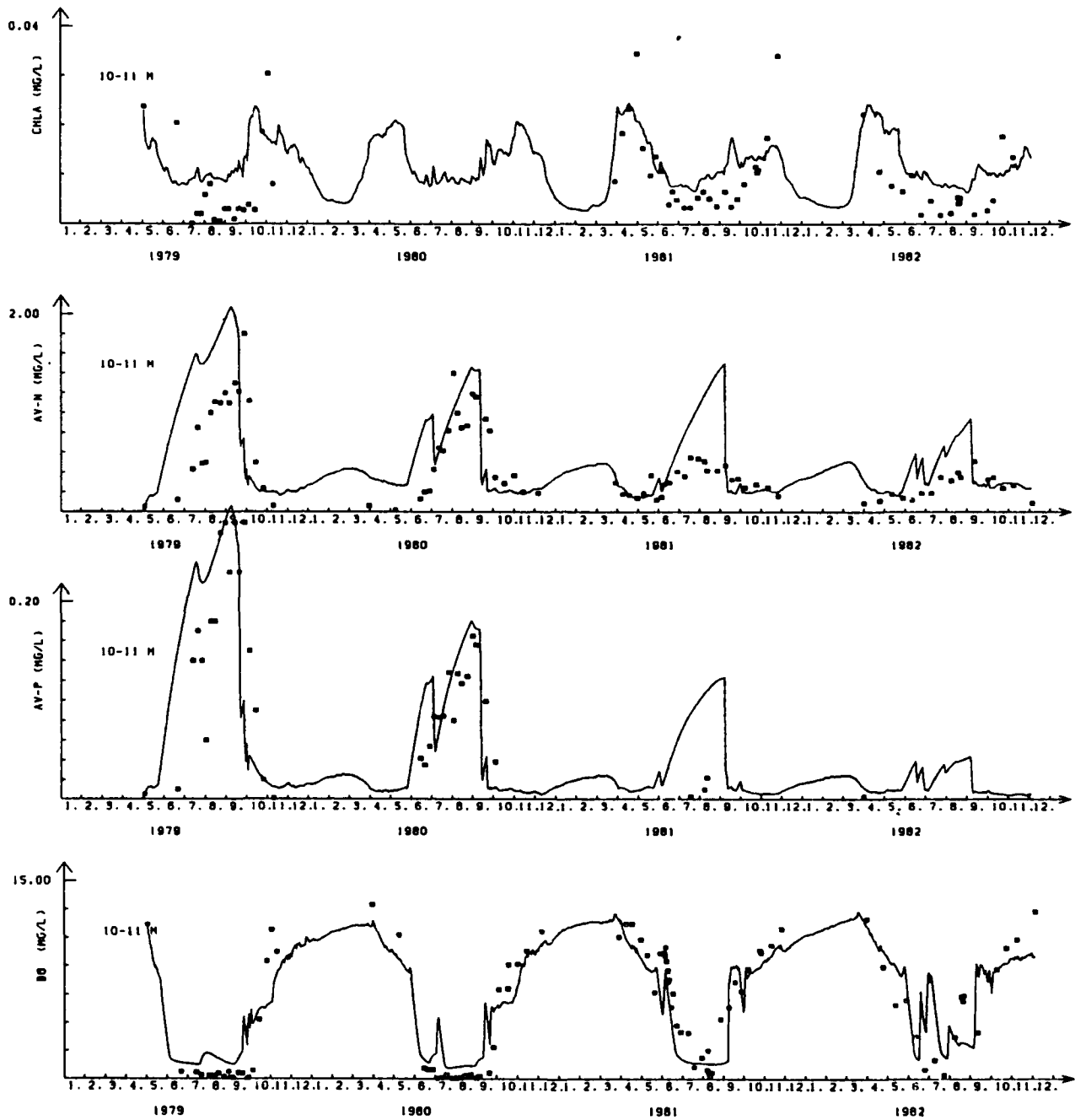
a. 0-1 m

Figure 6. Chlorophyll-a, available nitrogen, available phosphorus, and dissolved oxygen concentrations in Lake Yunoko; comparison of model calculations (---) and observed data (\square) (Sheet 1 of 3)



b. 5-6 m

Figure 6. (Sheet 2 of 3)



c. 10-11 m

Figure 6. (Sheet 3 of 3)

9.0 mg N/g, respectively. These ranges were within observed values of particulate phosphorus and nitrogen in the surface sediment (0 to 2 cm or 0 to 3 cm layer), i.e., 1.7 to 2.2 mg P/g and 6.0 to 7.2 mg N/g, respectively.

The calculated particulate nutrients in the first layer were inclined to decrease from 1979 to 1982. But, observed data in the surface sediments were approximately constant within the range of sampling and analytical errors. The inconsistency may be derived from the errors in calculating sedimentation flux of phytoplankton in model calculation.

Table 1 compares model calculations and observed data on phosphorus and nitrogen release flux from bottom sediments during summer stagnation periods in Lake Yunoko. The agreement between the model calculations and the measured data was good.

TABLE 1. RELEASE FLUXES OF PHOSPHORUS AND NITROGEN FROM BOTTOM SEDIMENTS DURING SUMMER STAGNATION PERIODS IN LAKE YUNOKO

	Values Estimated from Nutrient Release Experiments		Calculated Values Average
	Min.	Max.	
<u>Phosphorus Release Flux, mg P/m²/day</u>			
1979	4.0	6.8	6.6
1980	3.6	6.5	4.8
1981	0.4	2.4	2.3
1982	0.2	1.9	1.4
<u>Nitrogen Release Flux, mg N/m²/day</u>			
1979	56	60	43
1980		45	32
1981	28	71	27
1982	20	44	27

Phosphorus removal of wastewater began in 1981 (only summer and autumn season). The total phosphorus concentration in the euphotic zone decreased from 0.052 mg P/l (1979) to 0.036 mg P/l (1982). However, chlorophyll-a concentration in the euphotic zone remained constant at about 20 µg/l. Hosomi, Okada, and Sudo (1986) considered that this inconsistency was derived from the fact that the limiting nutrient for phytoplankton growth changed from nitrogen (before treatment) to phosphorus (after treatment) as a result of reduction of phosphorus loading.

The simulation demonstrated that the available phosphorus concentration in the epilimnion decreased to below the detection limit after phosphorus removal (Figure 6) without any remarkable reduction of chlorophyll-a concentration. Thus, the results of simulation seem to be consistent with those of the observation by Hosomi, Okado, and Sudo (1986).

ASSESSMENT OF EUTROPHICATION CONTROL PROGRAMS

The effectiveness of eutrophication control programs on chlorophyll-a and dissolved oxygen concentrations was evaluated by using the ecological model developed in this study. Sediment dredging, artificial circulation, and reduction of external loading as eutrophication control programs were evaluated taking forcing functions in 1980.

Initial values were assumed to be zero in sediment dredging simulation; i.e., there were no available nutrients and biodegradable particulate nutrients in sediment after dredging. Sediment oxygen demand was reduced to half by dredging. The same values as those in the control simulation were taken for other parameters.

In the case of artificial circulation, eddy diffusion coefficients in the water temperature model were increased from $0.005 \text{ cm}^2/\text{sec}$ in the control simulation to $1.0 \text{ cm}^2/\text{sec}$. Other parameters and conditions were the same as in the control simulation.

In the reduction of external phosphorus loading, phosphorus loading from the wastewater treatment plant was reduced to half or zero.

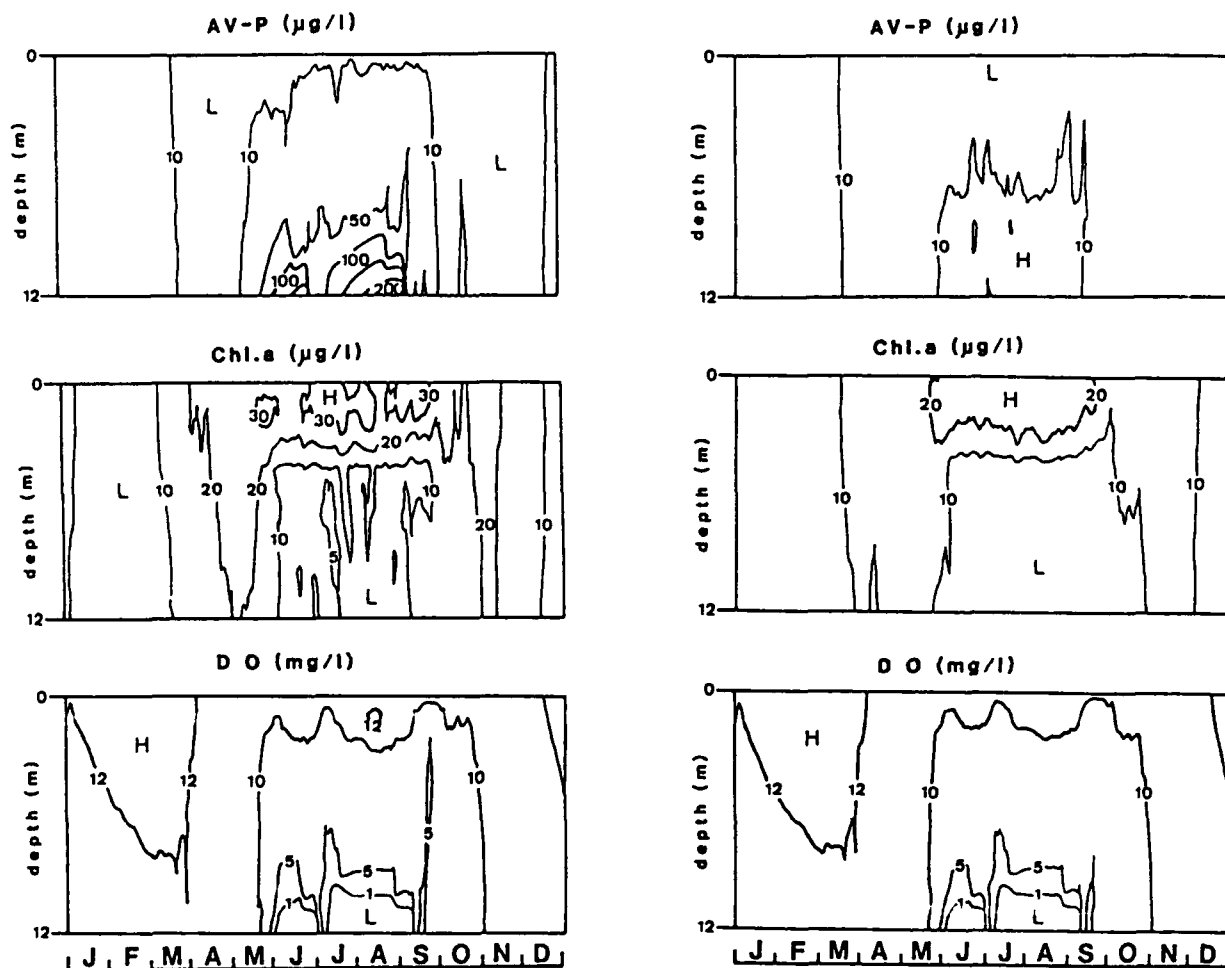
Figure 7 shows variations with depth and time of water quality constituents in the control simulation, sediment dredging simulation, artificial circulation simulation, and reduction of external phosphorus loading simulation.

In the sediment dredging simulation, annual average chlorophyll-a concentration in the surface water (0 to 4 m) decreased to 72 percent of that of the control simulation. The steep increase of available phosphorus toward bottom sediment diminished because nutrient release flux from bottom sediment became small. As a result, a peak in chlorophyll-a concentration in the surface water during the autumn circulation period did not appear. However, dissolved oxygen in the bottom water (10 to 12 m) during the summer stagnation period was scarcely restored by dredging.

A dredging case simulation was conducted with the sediment oxygen demand reduced to zero; for this case, dredging elicited the augmentation of only $1 \text{ mg}/\ell$ dissolved oxygen concentration (from $2.3 \text{ mg}/\ell$ to $3.3 \text{ mg}/\ell$). Accordingly, oxygen demand by sediment may have little effect on oxygen depletion in the bottom water during the summer stagnation period.

Artificial circulation changed the vertical profile of water temperature; i.e., uniform temperature was noted from the surface to the bottom, along with an increase in bottom water temperature of about 5° C . Dissolved oxygen concentration in the bottom water during the summer stagnation period increased from $1.9 \text{ mg}/\ell$ (control simulation) to $7.2 \text{ mg}/\ell$. However, chlorophyll-a concentration in the surface water scarcely decreased when compared with the control simulation, and chlorophyll-a in the water column increased.

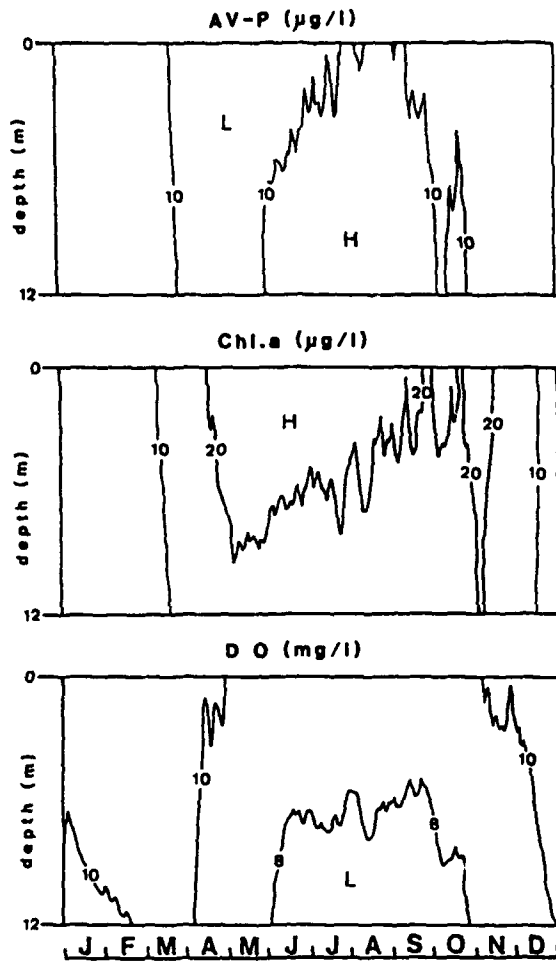
A 50-percent reduction in external phosphorus loading decreased chlorophyll-a concentration in the surface water in the summer to 82 percent of the control simulation. During the circulation in early autumn, the



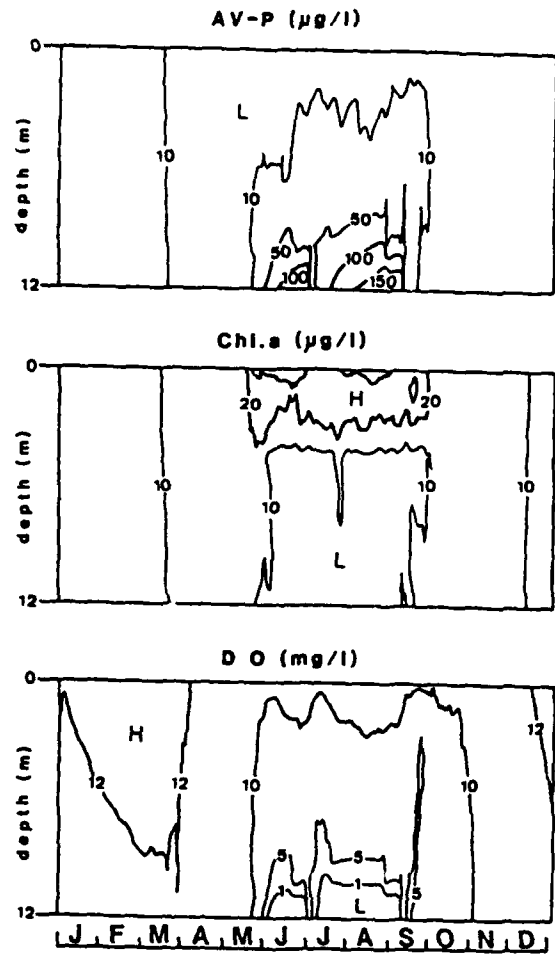
a. Control

b. Sediment dredging

Figure 7. Variations with depth and time of water quality constituents in the control simulation, sediment dredging simulation, artificial circulation simulation, and simulation of reduction of external phosphorus loading (Continued)



c. Artificial circulation



d. Reduction of external phosphorus loading

Figure 7. (Concluded)

calculated chlorophyll-a values were the same as those in the control simulation. This result was attributed to the transport of nutrients released from bottom sediments toward the epilimnion by autumn circulation.

Phosphorus loading from wastewater was reduced to zero, and annual average chlorophyll-a in the surface water decreased to 52 percent of that of the control simulation. From the chlorophyll-a concentrations in the surface water it was noted that the effectiveness of phosphorus reduction was the same as that of sediment dredging.

CONCLUSIONS

An ecological model of Lake Yunoko, a dimictic lake, was developed to assess various programs for eutrophication control with special emphasis on management of the bottom sediment. A multicomponent dynamic model for nutrients in the bottom sediment-water system was incorporated into the ecological model.

Good agreement was noted between the model calculations and observed nutrients, chlorophyll-a, and dissolved oxygen concentrations in the water and nutrients concentrations in the bottom sediment. The agreement between the model calculations and the measured nutrient flux from bottom sediment during the summer stagnation period was good.

The effectiveness of eutrophication control programs on chlorophyll-a concentration in the surface water and dissolved oxygen concentration in the bottom water was assessed using the calibrated model. Sediment dredging decreased nutrient flux from the bottom sediment. As a result, the peak of chlorophyll-a concentration in the surface water at autumn circulation period did not appear. However, the depletion of dissolved oxygen in the bottom water during the summer stagnation period was scarcely restored by dredging only.

Artificial circulation increased dissolved oxygen in the bottom water more than 7.2 mg/l throughout the year, whereas chlorophyll-a concentration in the surface water scarcely decreased when compared with control simulation.

A 50-percent reduction of external phosphorus loading could not decrease the peak of chlorophyll-a concentration during early autumn circulation. It did decrease chlorophyll-a concentration in the surface water during the summer.

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NEW TECHNOLOGIES APPLIED AT MEINOHAMA LANDFILL

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ABSTRACT

The landfill work which reclaimed an area of 74.0 ha at the Meinohama coastal area in Hakata Harbor was conducted in 1982-1985 by the Hakata Harbor Development Company. Several new technologies were applied, the most noticeable being the application of a new landfilling method characterized by placing dredged material in a pond with two to three staged intervals between placements. In this manner the dredged material is allowed to stabilize through self-weight consolidation during the intervals between placements.

New technologies were also adopted for spillwater treatment and dredged material dewatering.

This paper deals with the results of these technologies.

LANDFILL SITE

The landfill site (Figure 1), Meinohama, is on the western side of Hakata Harbor and has a total area of 74.0 ha. The reclaimed land will be used mostly for residences and other uses such as parks, roads, and port facilities.

PROPERTIES OF FILL MATERIAL

The properties of the fill material are represented by the triangular diagram (Figure 2) and the Casagrande plasticity chart (Figure 3). The fill material consists of sandy loam and clay. In the plasticity chart, the liquid limits lie between 30 and 60, and the plasticity indexes between 10 and 30. Since these points lie above the A-line, as a general rule the material is normally considered to be inorganic and is therefore suitable as fill material.

DREDGING

The dredged area is located about 500 m from the reclaimed land and has an area of 370,000 m² (Figure 4). The silty material in the upper layers of the sea bottom was dredged mechanically and was placed onto the pond bottom. The sandy material in the deeper layers was dredged hydraulically and deposited upon the previously deposited silty material (Figure 5). The boring profiles from the dredged area are shown in Figure 6.

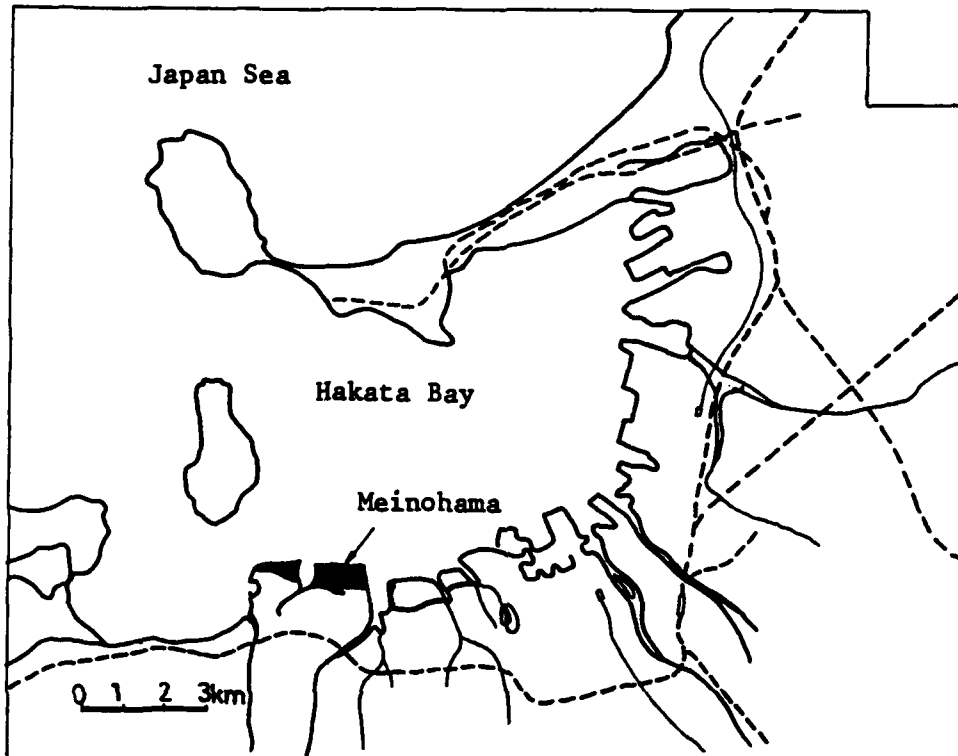


Figure 1. Landfill site

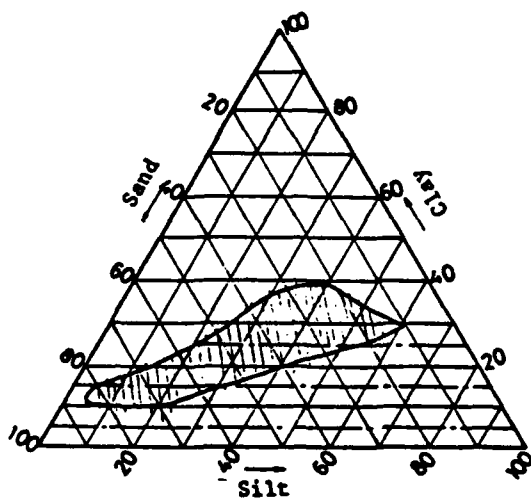


Figure 2. Triangular diagram

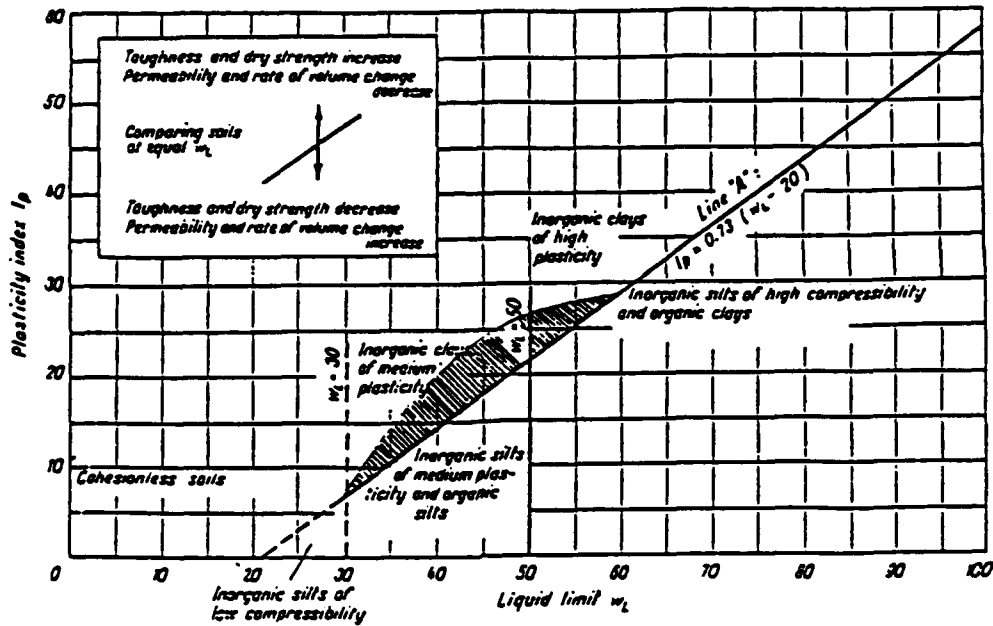


Figure 3. Casagrande plasticity chart

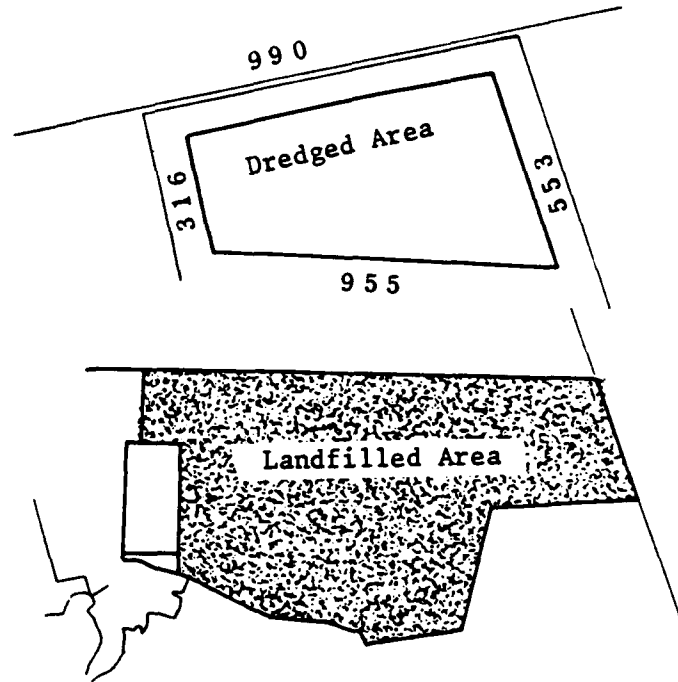


Figure 4. Dredged and landfilled areas

SELF-WEIGHT CONSOLIDATION OF DREDGED MATERIAL DISPOSED IN THE WATER

The disposed material in the pond is subjected to consolidation by its own weight (Figure 7) and water weight. In this case the thickness of the material to be consolidated is changing. Therefore, the normal equation of consolidation (Terzaghi) cannot be applied.

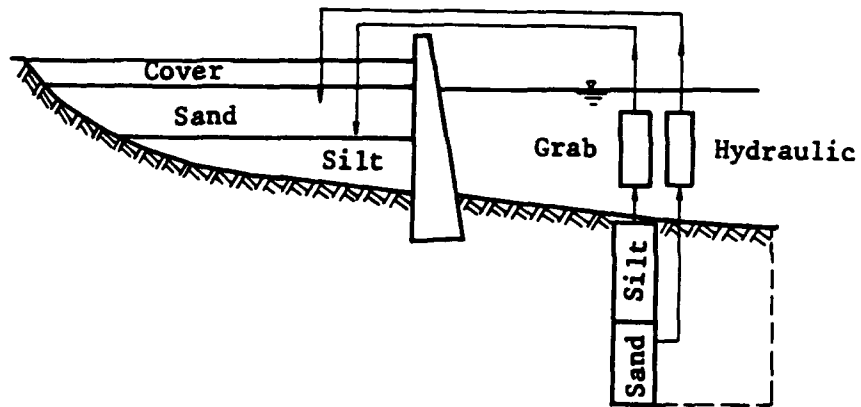


Figure 5. Dredging methods

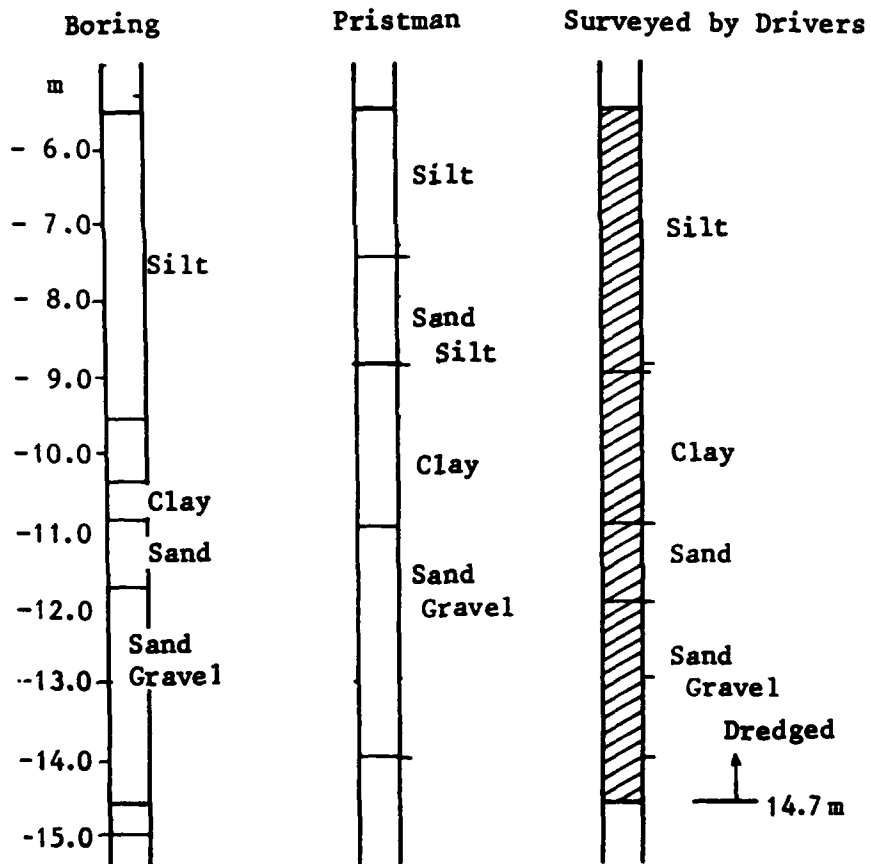


Figure 6. Boring profiles

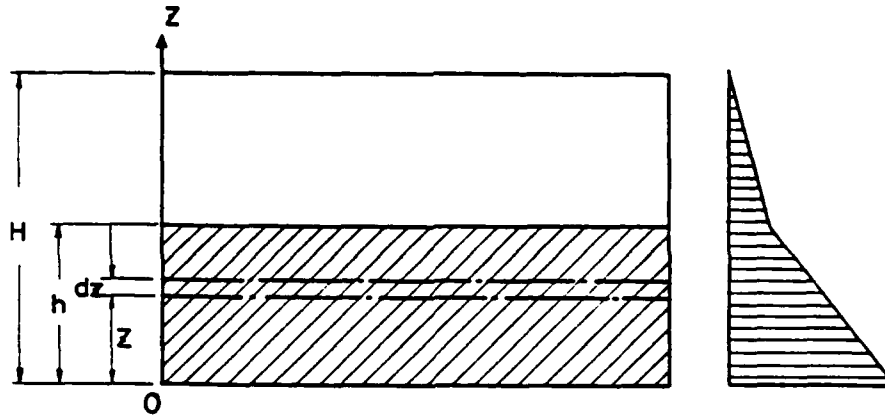


Figure 7. Self-weight consolidation

The total load acting on the slice layer ΔP at the height of Z in Figure 7 is represented as follows:

$$\Delta P = (H - h)\gamma_w + (h - z)\gamma'_s \quad (1)$$

where

γ_w = unit weight of water

γ'_s = unit weight of wet dredged material

The settlement of the slice layer ΔS becomes

$$\Delta S = U m_v \Delta P dz \quad (2)$$

where

U = degree of consolidation

m_v = coefficient of volume compressibility

Then the settlement for the disposed height h is

$$\Delta S = \int_0^h U m_v \Delta P dz \quad (3)$$

The coefficient of volume compressibility m_v can be approximately represented by the following hyperbolic function

$$m_v = A \Delta P^n \quad (4)$$

where A and n are curve fitting constants. Then we have

$$\begin{aligned}
S &= \int_0^h U A \Delta P^n \Delta P \, dz \\
&= U A \int_0^h \Delta P^{n+1} \, dz \\
&= U A \int_0^h \left[(H-h)\gamma_w + (h-z)\gamma_s \right] dz \\
&= U \frac{\gamma_s^{n+1} H^{n+2}}{n+2} \left\{ \left[\left(1 - \frac{h}{H}\right) \frac{\gamma_w}{\gamma_s} + \frac{h}{H} \right]^{n+2} - \left[\left(1 - \frac{h}{H}\right) \frac{\gamma_w}{\gamma_s} \right]^{n+2} \right\} \quad (5)
\end{aligned}$$

The m_v curve is shown in Figures 8 and 9.

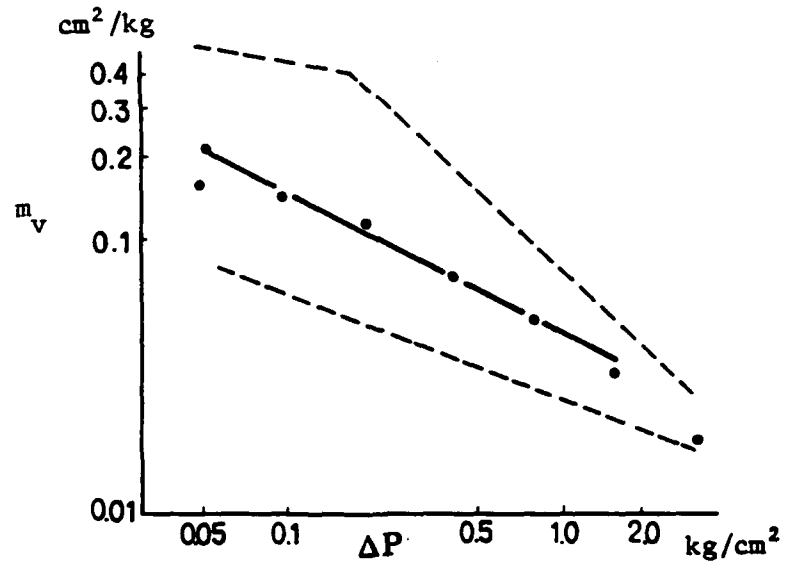


Figure 8. m_v versus ΔP (October 1984)

If we put

$$f\left(\frac{h}{H}\right) = \left[\left(1 - \frac{h}{H}\right) \frac{\gamma_w}{\gamma_s} + \frac{h}{H} \right]^{n+2} - \left[\left(1 - \frac{h}{H}\right) \frac{\gamma_w}{\gamma_s} \right]^{n+2} \quad (6)$$

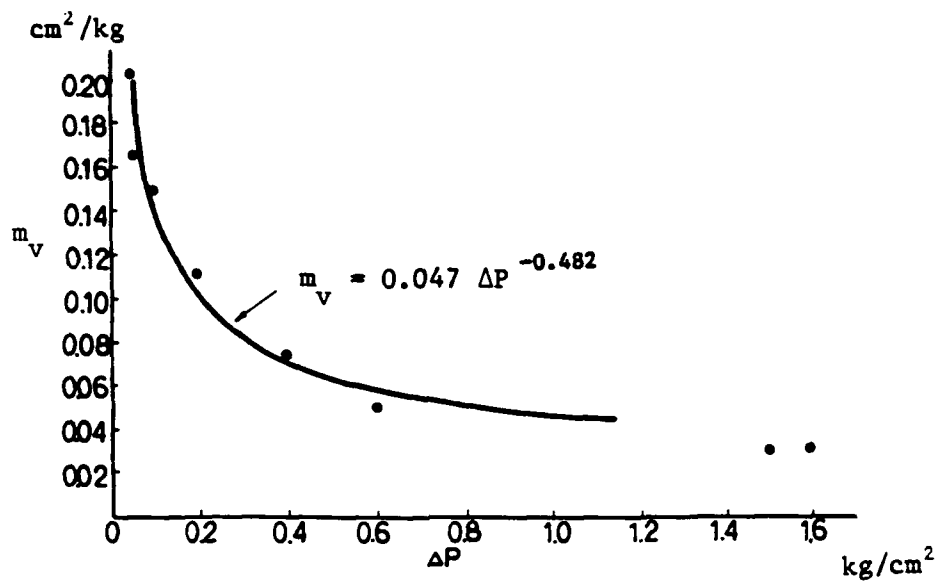


Figure 9. Approximated curve of m_v

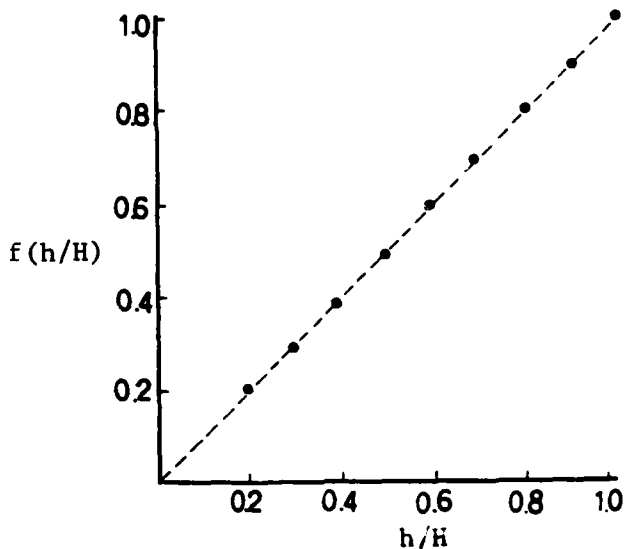
Equation 5 becomes

$$S_h = U A \frac{\gamma_s^{n+2} H^{n+2}}{n+2} f \frac{h}{H} \quad (7)$$

As Figure 10 shows, the function of (h/H) can be approximated as follows:

$$f \frac{h}{H} = \frac{h}{H} \quad (8)$$

Figure 10. Plot of $f(h/H)$ versus h/H



Finally, we have

$$S_h = \alpha \frac{A \gamma_s^{n+2} H^{n+2}}{n+2} \frac{h}{H} \quad (9)$$

This is a settlement equation of dredged material disposed in water.

DEGREE OF CONSOLIDATION

The coefficient of consolidation c_v can be approximated by the following hyperbolic function:

$$c_v = B\Delta P^m \quad (10)$$

The curve of c_v can be seen in Figures 11 and 12.

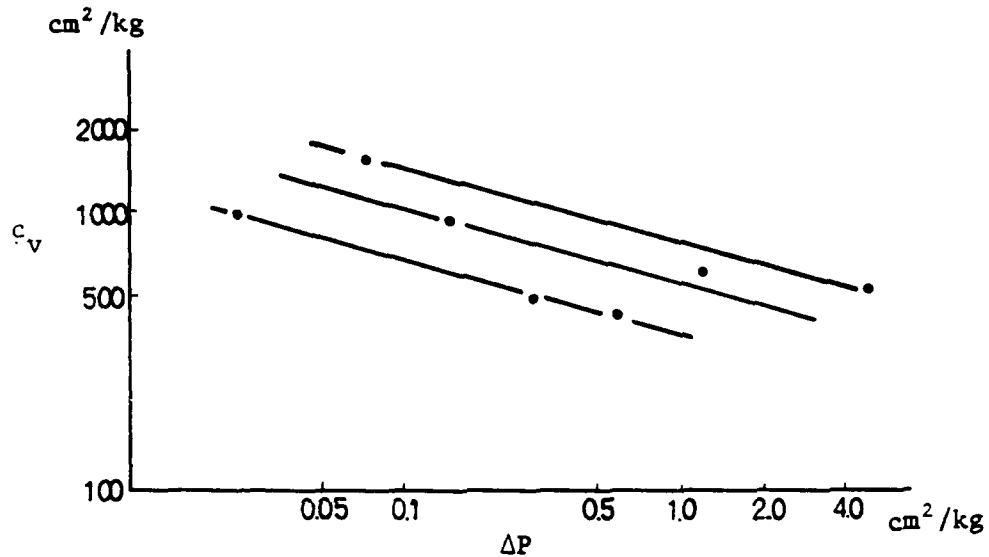


Figure 11. c_v versus ΔP (1984)

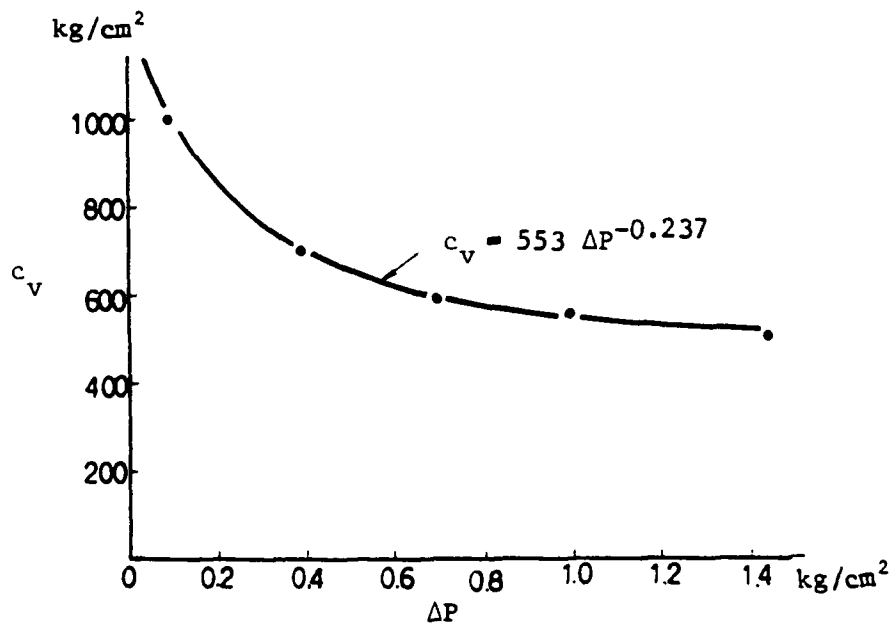


Figure 12. Approximated curve of c_v

The mean value of c_v for dredged material disposed in the water is calculated as follows:

$$\begin{aligned}
 c_v &= \frac{1}{h} \int_0^h B \Delta P(z)^m dz \\
 &= \frac{1}{h} \int_0^h B \left[(H - h) \gamma_w + (h - z) \gamma_s' \right]^m dz \\
 &= \frac{BH^{m+1} \gamma_s'^{m+1}}{(m+1) \gamma_s' h} \left[\left(\left(1 - \frac{h}{H}\right) \frac{\gamma_w}{\gamma_s'} + \frac{h}{H} \right)^{m+1} - \left(\left(1 - \frac{h}{H}\right) \frac{\gamma_w}{\gamma_s'} \right)^{m+1} \right] \quad (11)
 \end{aligned}$$

If we put,

$$g\left(\frac{h}{H}\right) = \left(1 - \frac{h}{H}\right) \frac{\gamma_w}{\gamma_s'} + \frac{h}{H} \quad \quad \quad - \left(1 - \frac{h}{H}\right) \frac{\gamma_w}{\gamma_s'} \quad \quad \quad (12)$$

we have

$$c_v = \frac{B \gamma_s'^m H^{m+1}}{(m+1) h} g\left(\frac{h}{H}\right) \quad (13)$$

The function of $g(h/H)$ (Figure 13) also can be approximated as follows:

$$g\left(\frac{h}{H}\right) = \frac{h}{H} \quad (14)$$

Then Equation 11 becomes

$$c_v = \frac{B \gamma_s'^m H^{m+1}}{(m+1) h} \times \frac{h}{H} = \frac{B (\gamma_s' H)^m}{m+1} \quad (15)$$

If we represent the filling rate as a (Figure 14), the fill height h becomes:

$$h = at \quad (16)$$

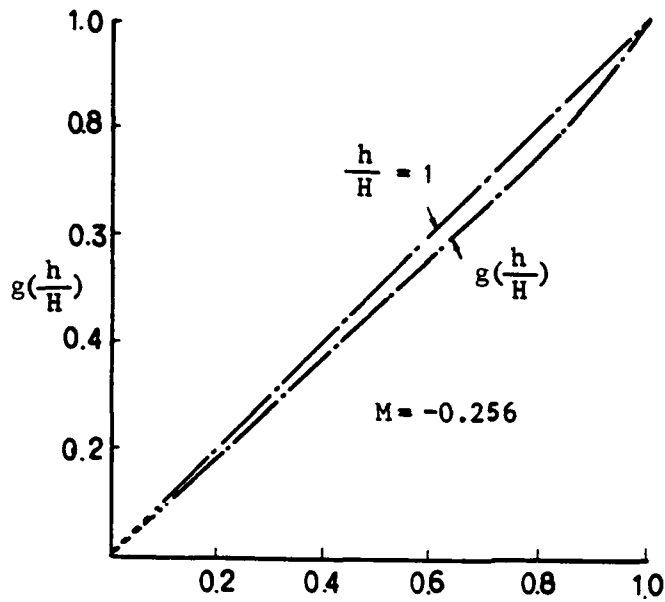


Figure 13. Plot of $g(h/H)$ versus h/H

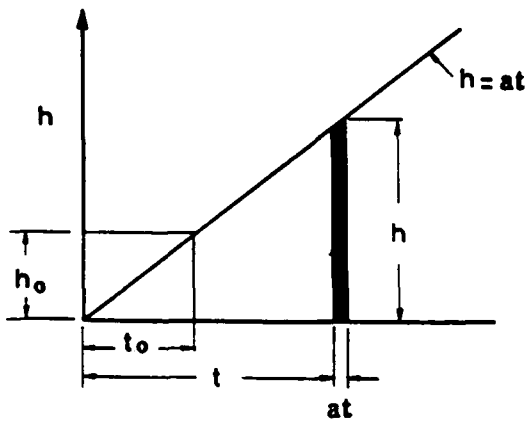


Figure 14. Filling rate, a

At the point of time when the fill height reaches h , the consolidation acts on the fill material during dt . Then we have the following time factor:

$$dT = \frac{c_v dt}{(at)^2} \quad (17)$$

where T is the time factor. Then

$$\begin{aligned} T &= \int_0^t \frac{c_v}{(at)^2} dt \\ &= \frac{c_v}{a^2} \left(\frac{1}{t_0} - \frac{1}{t} \right) \end{aligned} \quad (18)$$

When the fill height is small, it is presumed that no consolidation occurs. The time when the consolidation begins is assumed to be t_0 .

PLACEMENT OF DREDGED MATERIAL

The landfilled area was divided into 28 sections as Figure 15 shows. Each section was filled in such a way that the dredged material was placed in two to three stages with inactive intervals between placements. For example, Figure 16 shows the two-staged placement conducted at C-section. The first placement height was 2.275 m and the disposal rate was 9.89 cm/day, for 23 days. After placement, an inactive interval of 42 days was provided. After day 65 the second placement commenced with a rate of 2.66 cm/day. After 111 days, the height of the deposit reached the appointed level of 3.5 m. Figure 17 shows the three-staged filling diagram for the P-section. The objective of the inactive intervals was to allow self-weight consolidation of the dredged material to occur. Therefore, the multistage disposal method is presumed to promote the stabilization of the material.

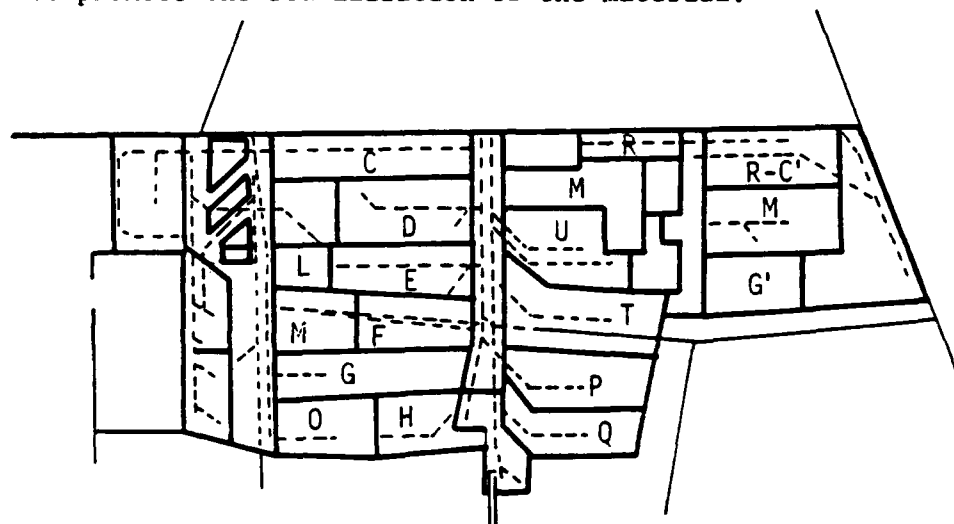


Figure 15. Division of pond area

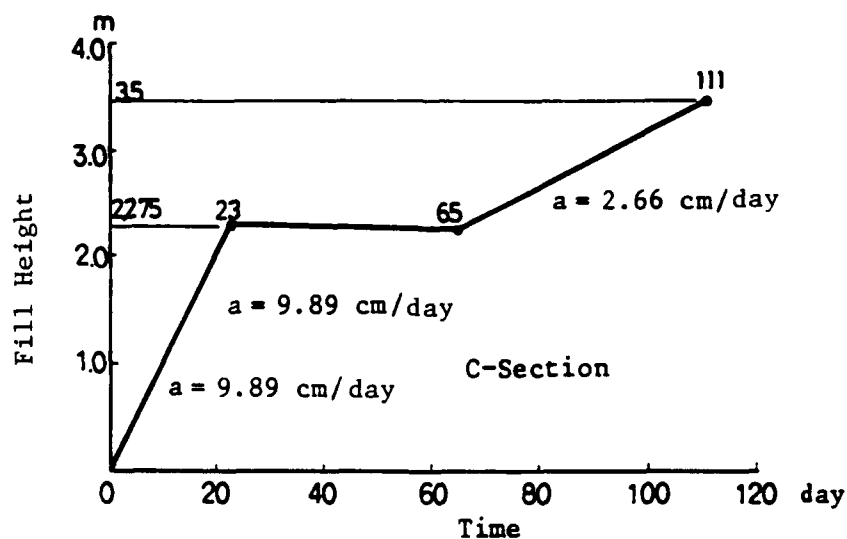


Figure 16. Fill diagram of C-section

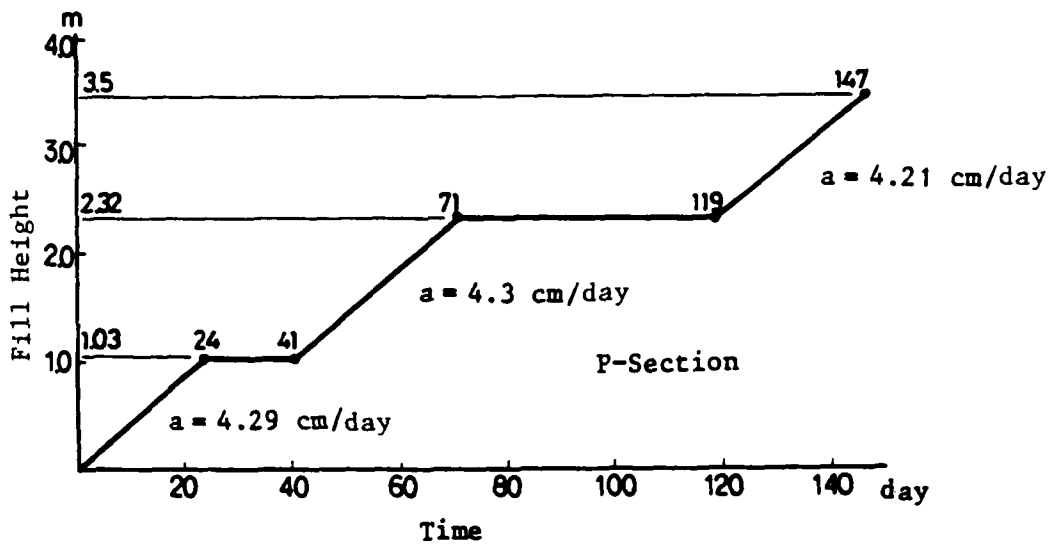


Figure 17. Fill diagram of P-section

Figure 18 shows the settlement curve calculated by Yoshida's equation described on page __. Figure 19 indicates the change in degree of consolidation during disposal of the dredged material. From Figure 19 it can be seen that the consolidation advances considerably in spite of the presence of water, primarily due to the slow disposal rates a of this method. It is presumed that the degree of consolidation reaches 80 percent at the completion of filling.

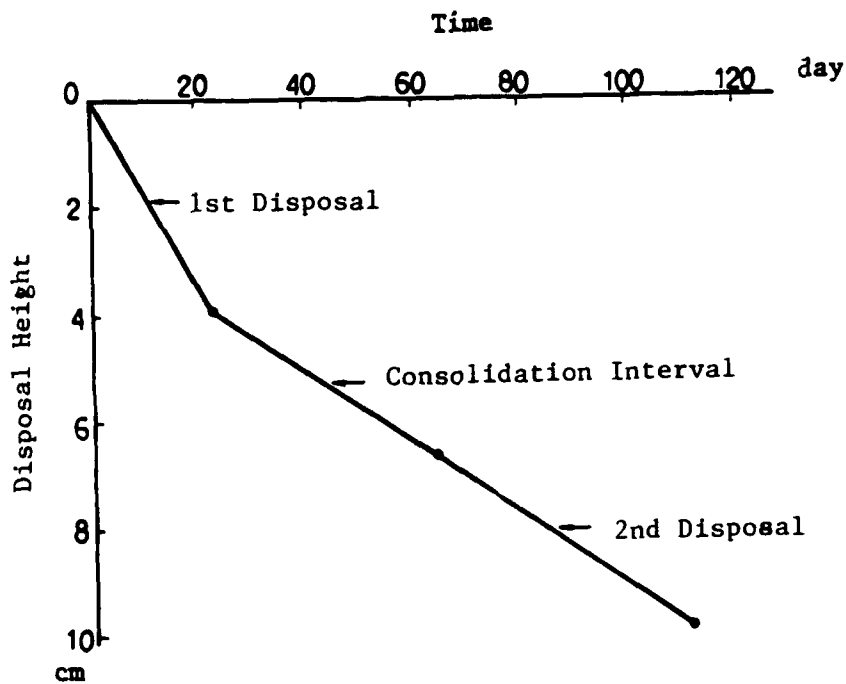


Figure 18. Settlement of disposed material

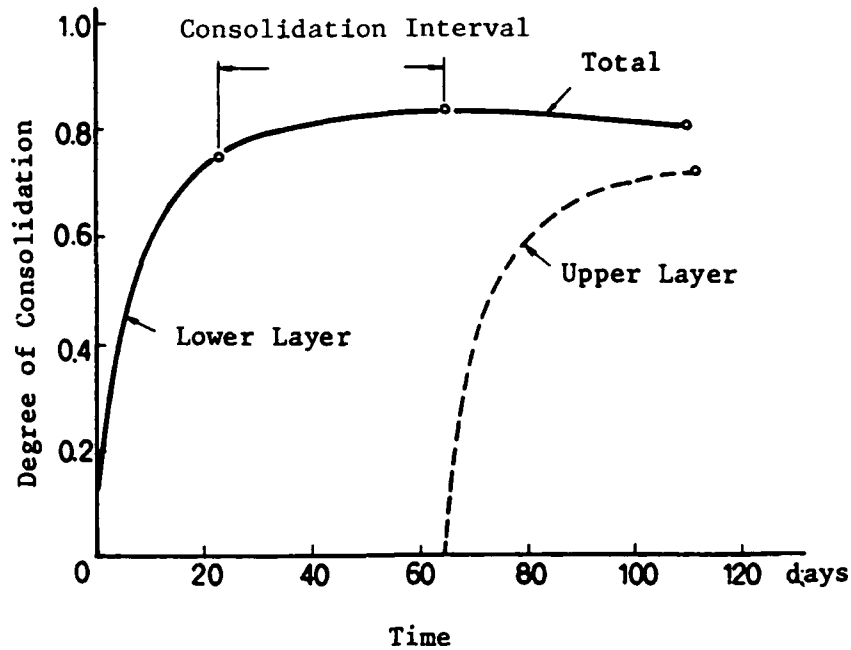


Figure 19. Degree of consolidation versus time

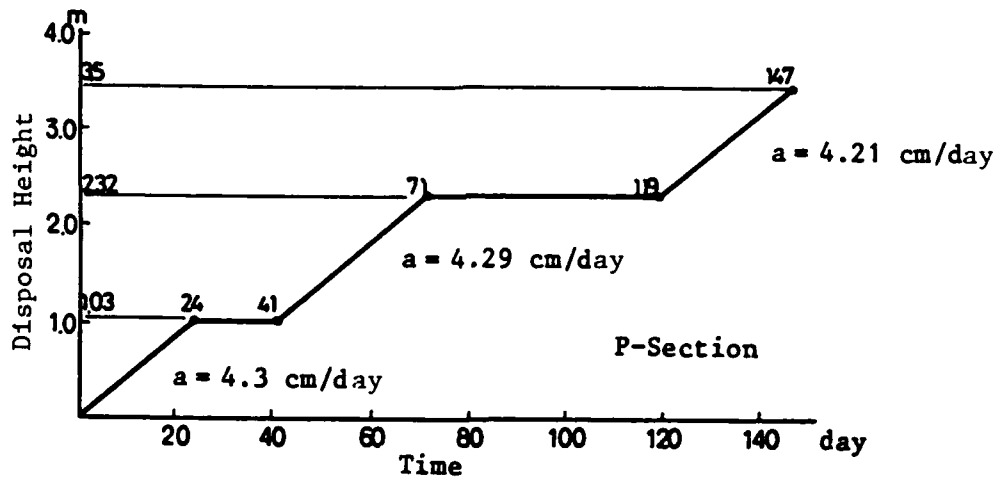


Figure 20. Three-stage disposal

Figure 20 shows the three-stage disposal conducted in P-section. In this case the disposal rate was about 4.3 cm/day with inactive intervals of 17 and 48 days. Figure 21 shows the calculated settlements in P-section. Figures 22 and 23 show the degrees of consolidation.

In this case, it is presumed that the degree of consolidation reaches 0.94 at the completion of filling.

The consolidation states of the material placed by the various filling methods are compared in Figure 24. It is evident that the consolidation state of fill material differs for the different filling methods. The settlement resulting from placement of material in three stages is the largest, and that resulting from single-staged placement of material is the smallest. There is

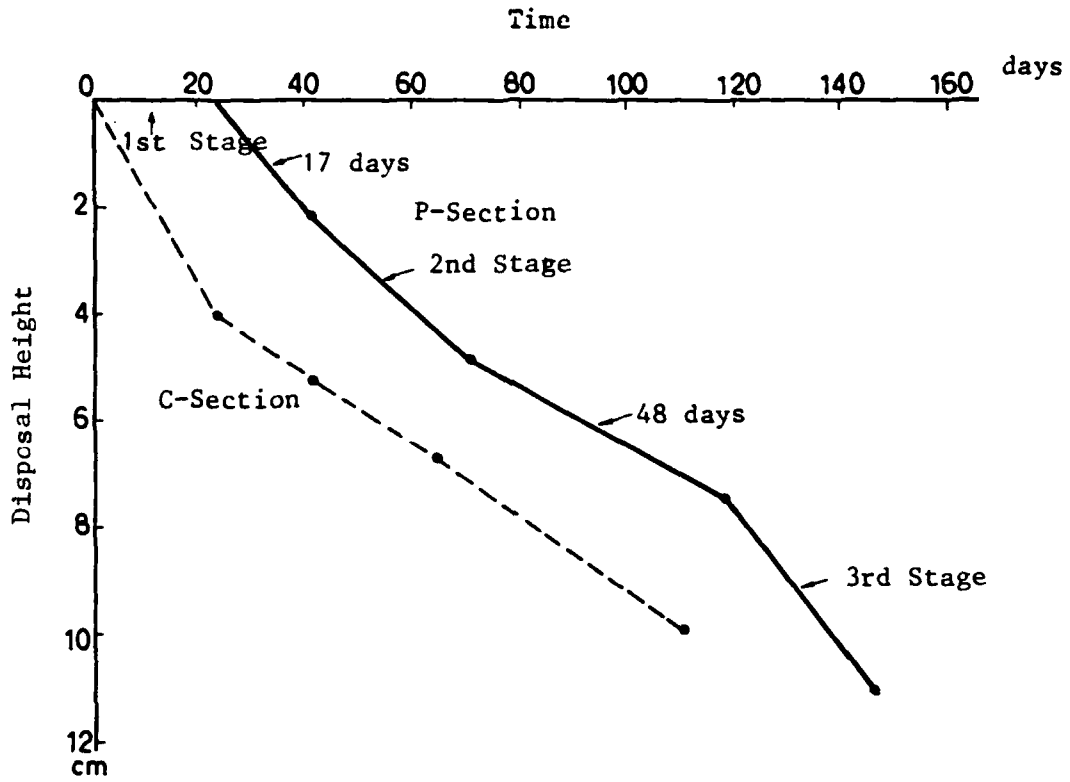


Figure 21. Settlements of three-stage disposal

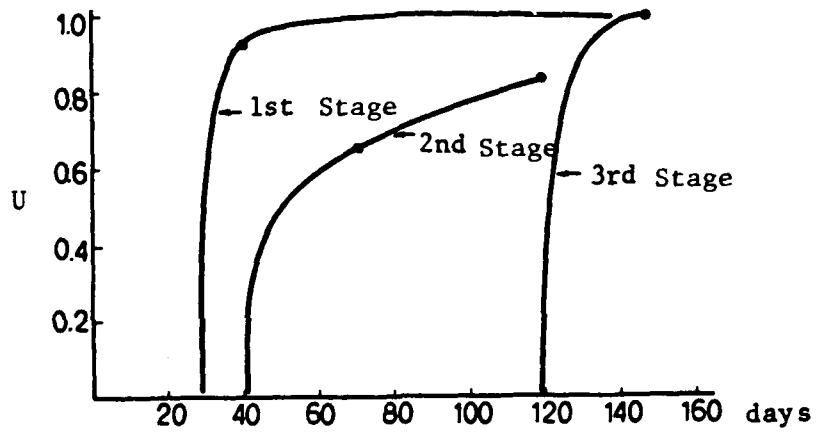


Figure 22. Degree of consolidation of each stage

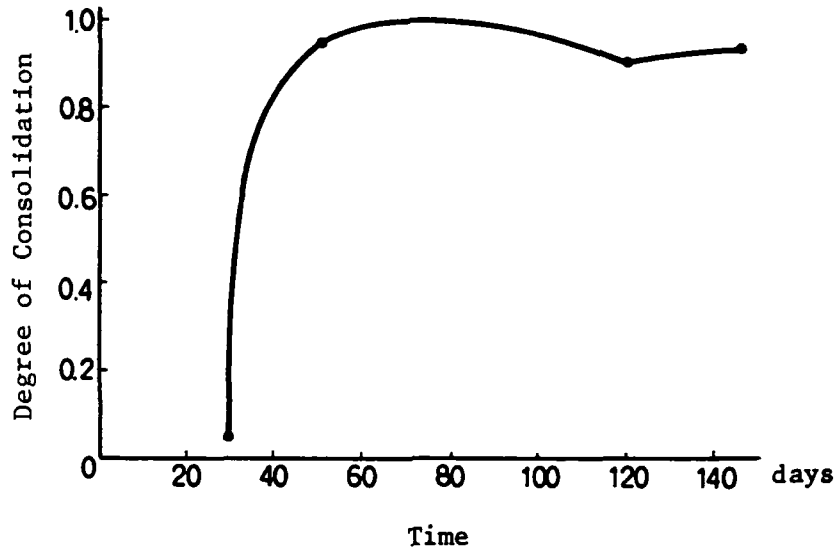


Figure 23. Degree of consolidation of the entire deposit

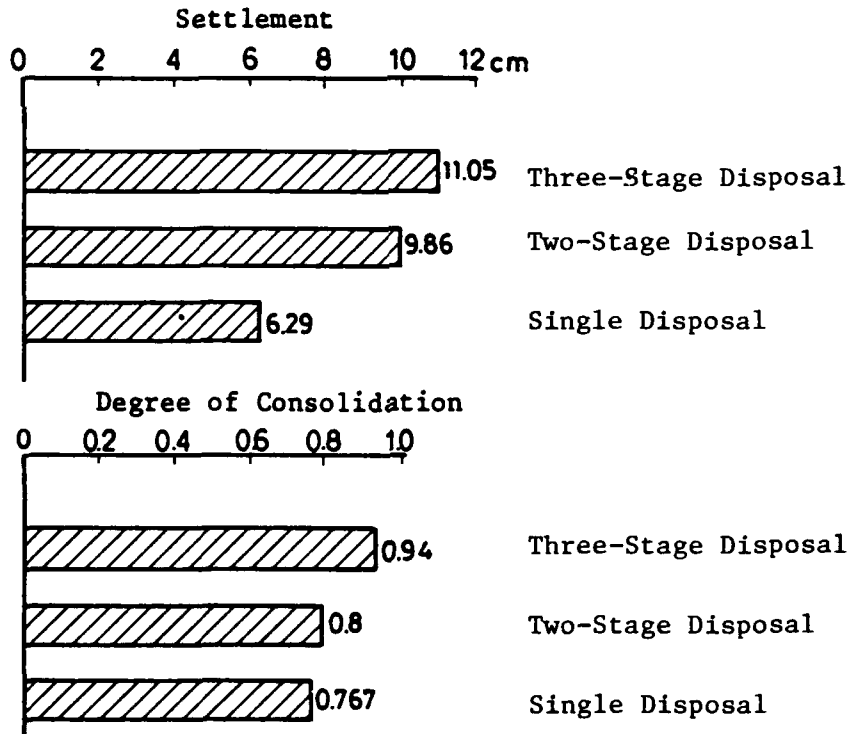


Figure 24. Comparison of various disposals

a difference of 4.76 cm between them. For the single-stage method to attain the same settlement as the three-stage method, it is necessary to apply larger cover loading than that of the three-stage method (Figure 25).

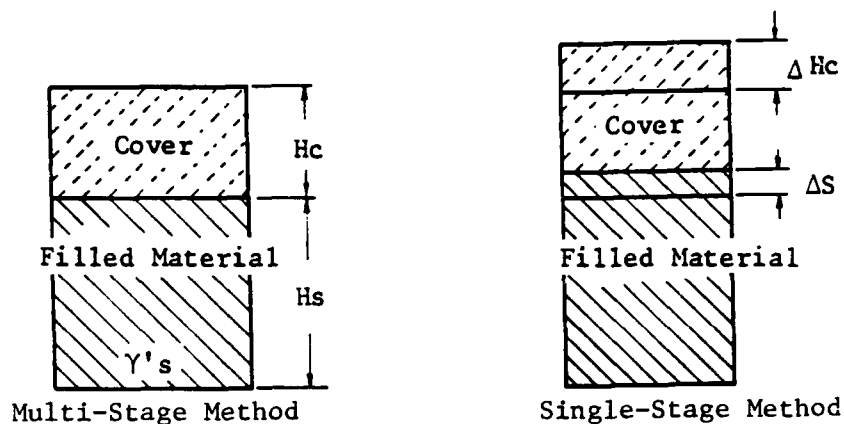


Figure 25. Difference in cover thickness according to methods

If the cover height necessary to yield settlement ΔS of filled material is represented as ΔHc , we have the following equation:

$$\Delta Hc = \frac{1}{\gamma'_s} \left(\frac{\Delta S}{\alpha A H_s} \right)^{\frac{1}{n+1}} \quad (19)$$

If we put the following numerical values in Equation 19

$$\begin{aligned} A &= 0.0578 \text{ cm}^2/\text{kg} \\ n &= -0.31 \\ \Delta S &= 4.21 \text{ cm (average difference)} \\ H_s &= 350 \text{ cm} \\ \gamma'_s &= 0.00165 \text{ kg/cm}^3 \end{aligned}$$

we get

$$\Delta Hc = \frac{1}{0.00165} \left(\frac{4.21}{1 \times 0.0578 \times 350} \right)^{\frac{1}{-0.31+1}} = 58.86 \text{ cm}$$

Therefore, the increment of cover volume becomes

$$\Delta V = 0.5886 \text{ m} \times 716,000 \text{ m}^2 = 421,438 \text{ m}^3$$

If we assume a covering cost of \$17/m³, the total cost of \$7,164,446 (\$17/m³ × 421,438 m³) would be saved by using the new method. Consequently, we can say that the merits of the new method are great.

SPILLWATER TREATMENT

The settling pond was divided into four sections with three interior spillways and one final spillway (Figure 26). These spillways were arranged so that spill water would flow as long a distance as possible, thereby increasing retention time of the slurry. The longer the passage, the stronger the sedimentation. When the dredged material was poured into A-section, the SS concentrations of spill water at SR₃ were about 2 to 4 mg/l due to a long distance of more than 500 m. However, when the pouring point moved to B-section and then to C-section, the SS concentrations were 11-15 mg/l and about 100 mg/l, respectively. In this study, the permissible SS concentration of spill water at the final spillway, SR₄, was less than 50 mg/l. In order to attain this safely, it was necessary to treat spill water at the final spillway.

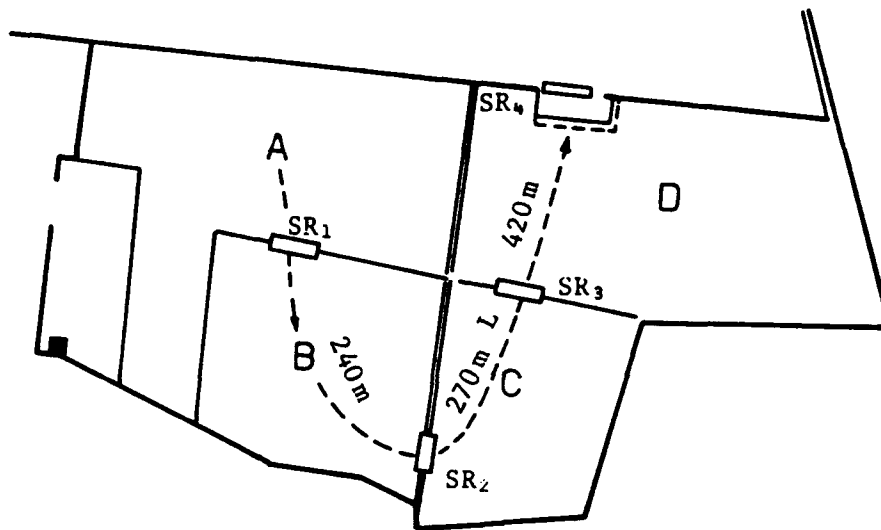


Figure 26. Spillways

Figures 27 and 28 show the construction of the final spillway. After operation began, it was discovered that a normal settling pond was insufficient to reduce the concentrations of spill water to the desired level of 50 mg/l. Therefore, two fibrous sheets were added as an improvement. The main sheet was an unwoven textile for filtration, and the supplementary sheet was a permeable textile to prevent access of solid material settled on the pond bottom. These geotextiles gave good results.

DEWATERING TESTS OF DREDGED MATERIAL

When dredged material is placed into a pond from a delivery pipe (hydraulic pipeline), the coarse-grained material settles out near the point of discharge while the fine-grained material settles at some distance from the pipe (Figure 29). This fine-grained material often remains very wet. In order to overcome this problem, dewatering tests were performed using drainpipes.

At the bottom of a pond having an area of $52 \times 48 = 2,496 \text{ m}^2$, 14 drain pipes (150 ϕ) were placed. The water which drained from the overlying dredged material was gathered in two colgates and pumped up to the ground (Figure 30).

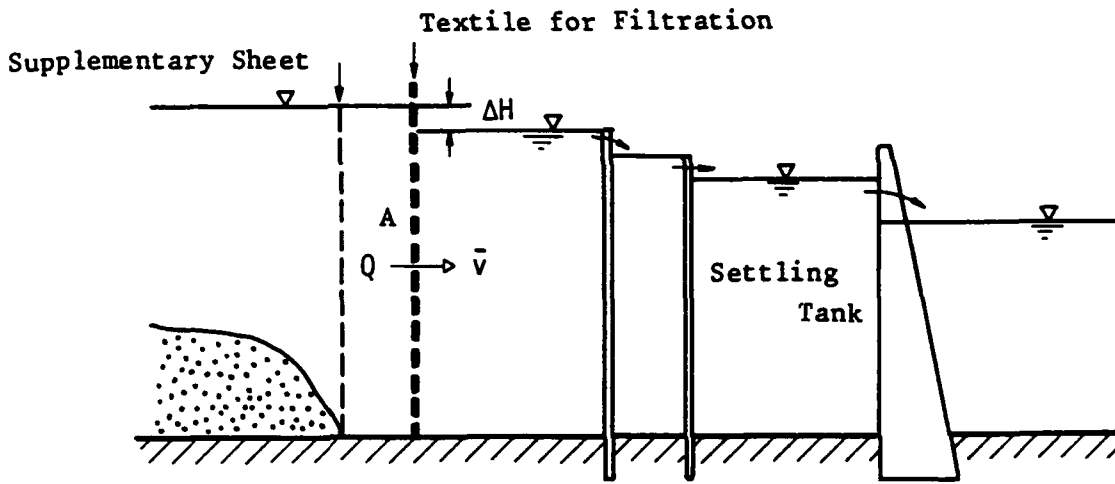


Figure 27. Vertical section of final spillway

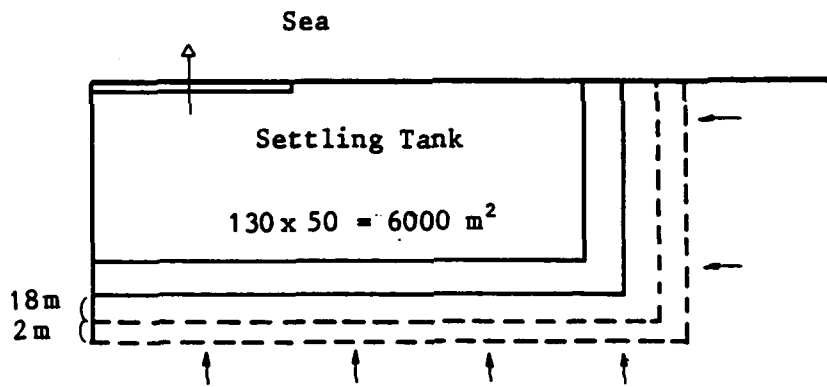


Figure 28. Plane figure of final spillway

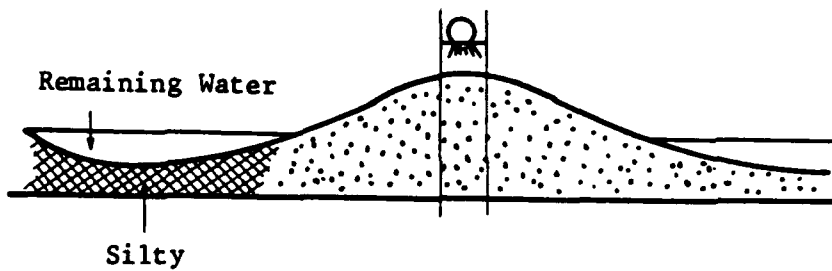
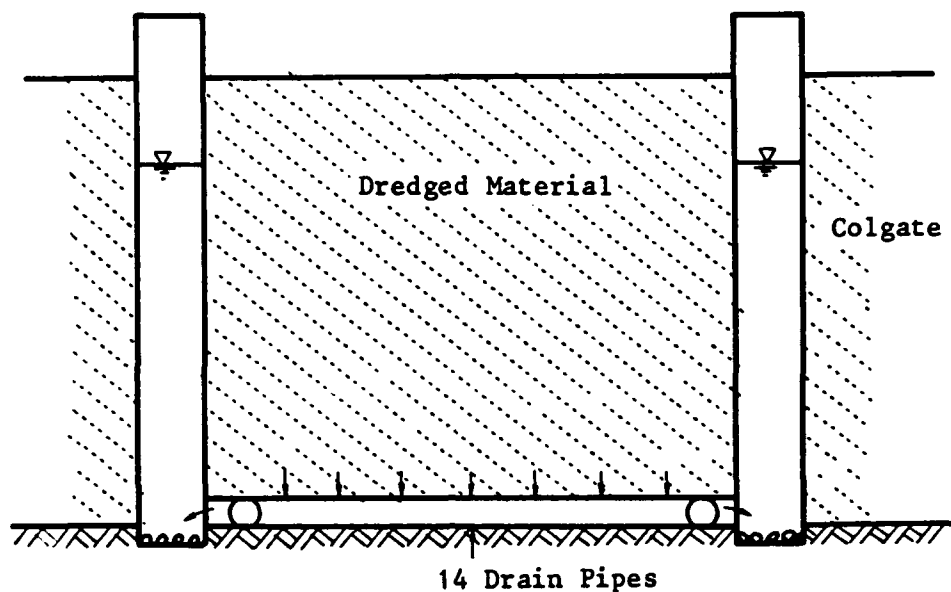


Figure 29. Undewatered silty material



$$A = 52 \times 48 = 2,496 \text{ m}^2$$

Figure 30. Drain pipes for dewatering

The water content of the dredged material at the test area was 62.7 percent, which corresponds to a total quantity pore water of $3,895 \text{ m}^3$. This system drained 972 m^3 of water. Therefore, the dewatering efficiency became

$$Y = \frac{972}{3,895} \doteq 0.25$$

This means that the initial water content of 62.7 percent was reduced to 47 percent.

CONCLUSIONS

Several new landfilling technologies were introduced and applied at Meinohama. The new filling method was the first attempt in our country. It was found that they were beneficial from both a technical and economical viewpoint. It is worthwhile to note that these technologies have contributed to the improvement of reclamation techniques.

The total load acting on the slice layer ΔP at the height of Z is represented as follows:

$$\Delta P = (H - h)\gamma_w + (h - z)\gamma_s$$

where

$$\gamma_s = \text{saturated unit weight of soil mass}$$

or

$$\Delta P = (H - h)\gamma_w + (h - z)\gamma'_s$$

where

$$\begin{aligned}\gamma'_s &= \text{effective unit weight of the soil mass} \\ &= \gamma_s - \gamma_w\end{aligned}$$

The effective load is

$$\Delta P' (\text{or } \sigma') = (h - z)\gamma'_s$$

This effective load is the one and only load which causes consolidation. The load caused by the water does not cause consolidation.

According to our sources (References attached), degree of consolidation is identified as a U . In addition, settlement ΔS is calculated by the following equation

$$\Delta S = m_v \Delta P dz$$

As you will notice, degree of consolidation is not used in the settlement calculation. It should also be noted that ΔP in the above equation should be the effective pressure, not the total pressure. Therefore ΔP used in the above equation should be calculated as follows:

$$\Delta P = (h - z)\gamma'_s$$

It should not be calculated as shown

$$\Delta P = (H - h)\gamma_w + (h - z)\gamma'_s$$

References for settlement equation:

Terzaghi, K. 1943. Theoretical Soil Mechanics, John Wiley and Sons, Inc., New York, Article 102, p 281.

Terzaghi, K., and Peck, R. B. 1967. Soil Mechanics in Engineering Practice, John Wiley and Sons, Inc., New York, Article 41, p 278.

THE ROLE OF MICROORGANISMS IN MOBILIZATION
OF CONTAMINANTS FROM SEDIMENT

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ABSTRACT

This paper describes the results of a literature survey on microorganisms and their ability to enhance the movement of contaminants from sediment into the overlying water. The sediments included in this discussion are from both subaqueous and wetland environments. The survey is specifically concerned with the role of microorganisms in several processes occurring in flooded soils and sediments; these processes include: formation of dissolved organic matter, both directly from nutrients in flooded soils and sediments and from living and dead plant and animal tissues; incorporation and/or conversion of contaminants into dissolved organic form; and transformation of contaminants that enter flooded soil or sediment systems from an external source, such as tides, floodwaters, and overland runoff. This information is used to assess the importance of dissolved organic matter as a vehicle for transport of total organic matter and organic contaminants away from flooded soils and sediments. Special attention is given to the importance of microorganisms in the formation of dissolved organic matter and the transformation of contaminants into soluble form.

INTRODUCTION

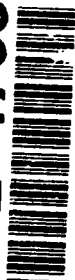
This paper is based on a literature evaluation conducted for the US Army Engineer Waterways Experiment Station by Blum and Mills (1986). Other information has been used, where appropriate, to provide clarity.

Sediment and its microbial inhabitants collectively provide many of the biological, chemical, and physical properties necessary for the processes of

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mineral cycling and the degradation of man-made chemicals. Sediment is a multifaceted interface--between solid and liquid, between aerobic and anaerobic, and between sorptive and desorptive. Depending upon hydrodynamic conditions in the overlying water column, sediment may serve as a source or sink for contaminants in the ambient environment (Gunnison, Engler, and Patrick 1985). In addition, microbial activities in sediment may, in large part, determine what chemical constituents are released into or removed from the overlying water column (Brannon, Chen, and Gunnison 1986; Gunnison, Engler, and Patrick 1985). Thus, activities of microorganisms can often serve as major determinants of water quality.

Wetlands are special sediment environments, serving as transitional interfaces between aquatic and terrestrial habitats and often sharing commonalities with both systems. In the past these intermediate zones were considered to be useless wastelands that offered benefits only to sportsmen, and the principal use of these regions was achieved only by drainage to provide available land for agriculture, urban, or industrial development. While loss of wetlands to such activities is still occurring, the rate of this loss is decreasing, largely in recognition that these areas are of value in ways not previously recognized.

Increasing interest in interactions between aquatic sediment microorganisms and their environment is reflected by the number of recent papers dealing with various aspects of this subject (Brannon, Chen, and Gunnison 1986; Collins 1977; Gunnison, Engler, and Patrick 1985; Marshall 1980). The study reported herein is concerned with the specific role of microorganisms in the formation (or removal) of dissolved organic materials that complex with contaminants and alter their mobility, either increasing or decreasing release of the materials from sediment. Many compounds form complexes with dissolved organic materials, and the resulting complex may also be soluble, increasing the mobility of the original compound of interest. In sediment, as in many other aquatic environments, the activities of microorganisms can exert a strong influence on the chemical status of the habitat. This influence may involve direct action on the contaminant itself, or indirect action on material that may subsequently react abiologically with the contaminants.

For purposes of this paper, contaminants are defined as any chemical substance in an aqueous or wetland sediment that is undesirable with respect to man's intended use of the sediment. Dissolved organic matter (DOM) refers to those organic materials in solution which are small enough to pass a 0.1- μ m filter.

NATURE OF SEDIMENTS AND FLOODED SOILS

Ponnamperuma (1972) has characterized sediments as substrates having dark color; low redox potential; no dissolved oxygen, at least within the strata underlying the first few millimetres or centimetres of its surface; reduced forms of N, S, Mn, and C and refractory organic matter; and often, a thin, light-colored oxidized layer at the surface. The composition of sediment varies according to the source of the material and the hydrodynamic processes occurring in the area in which the sediment settles. Those areas of the world that have heavy industrial activity and/or large levels of harbor traffic are

likely to have sediments heavily polluted with organic and metallic contaminants. Many of these areas also require frequent dredging to keep the channels open to ship traffic.

Flooded soils differ from sediments in that they are flooded intermittently rather than permanently. A large variety of habitats contain flooded soils or some type of wetland. These include the field flooded only briefly during periods of heavy accumulation of precipitation, temporary pools and riverbanks that are flooded for a portion of the year, tidal marshes that are inundated on a daily basis, and rice paddies that may be flooded for months at a time. As a habitat, the flooded soil may undergo comparatively pronounced extremes in temperature and moisture during periods when the soil is not flooded. When water is present over a soil, the depth is often fairly shallow. As a consequence, the water is easily mixed by the wind, and dissolved oxygen levels are high. Shallowness also contributes to the ability of sunlight to penetrate the water column to the soil surface, supporting active growth of algae and cyanobacteria. The level of organic matter can also be high owing to the presence of rooted plants. These factors encourage the formation of a system with intensely reducing conditions in the underlying layers and an oxidized surface layer. There is ample opportunity for chemical and microbiological processes favored by these conditions to occur. Examples of such processes include nitrification-denitrification, release of iron and manganese in the reduced form, and sulfate reduction to sulfide (Gunnison, Engler, and Patrick 1985). If the reduced forms of these materials are released into the water column, they may then be reoxidized; some are then reprecipitated on the sediment surface.

Flooded soil undergoing an accumulation of material and gradually rising above the water level may evolve into a high marsh and then into meadowland. The source materials for this process are often obtained from both aquatic sediment and an admixture of living and decomposed organic material derived largely from aquatic and terrestrial plants. In the case of newly filled reservoirs, a formerly dry or intermittently flooded soil surface undergoes a transition associated with transformation of this substrate into an aquatic sediment.

NATURE OF DOM PRODUCTION

Aquatic ecosystems typically contain large amounts of DOM relative to the concentrations of particulate organic material (Vallentyne 1957; Wangersky 1965). In both marine and freshwater systems, the level of DOM is an order of magnitude greater than the concentration of particulate organic matter (POM) (Fenchel and Blackburn 1979; Parsons and Strickland 1962; Wetzel et al. 1972). Freshwater concentrations of dissolved organic carbon under unpolluted conditions, for example, range between 1 and 30 mg C/l (Wetzel 1983). DOM is derived mainly from two major sources: allochthonous sources through direct input of organic materials such as autumn shed leaves (Kaushik and Hynes 1971), and indirect inputs of soil organic matter from overland flow and shallow ground-water inputs; and autochthonous sources such as cellular debris and extracellular excretion products of aquatic organisms (Fogg 1958). Allochthonous DOM dominates most freshwater systems. In coastal regions, the carbon levels vary from 0.1 to 100 mg C/l (Menzel and Vaccaro 1964; Wilson 1961), while in open ocean systems, the range is from 0.4 to 1.0 mg C/l (Barber 1968; Kerr and Quinn 1980). The relationship between DOM and POM is relatively

constant, with the POM concentration being an order of magnitude less than the DOM concentration. The relationship of DOM to ecosystem productivity is unclear, due in part to the inability to reliably measure DOM turnover rates and in part to the chemical complexity of DOM and the resultant difficulty in distinguishing between competing biological, chemical, and physical reactions, such as POM formation. It is important to understand DOM before we can examine the interactions of this material with contaminants and understand the consequences of processes occurring in aqueous and wetland sediments.

In most of the areas not dominated by man's influence, macrophytes and some phytoplankton are responsible for the production of most of the dissolved organic matter. The amount of organic material that is excreted or exuded is dependent on light, temperature, ionic composition of the medium, presence of epiphytes, and conditions of the plant or type of plant. Generally, DOM release reaches a maximum during periods of intense photosynthetic activity. Plant excretion can occur at the roots or from the foliage. DOM can also be leached from foliage of emergent and terrestrial plants by rainfall and enter water and, later, sediment. Submerged macrophytes exude DOM directly into the water column, and from their roots and rhizomes into the sediment. Phytoplankton can also contribute significant amounts of DOM to wetlands. Large amounts of DOM are also released by leaching and autolysis of plant tissues. Information concerning the relative importance of DOM from macrophytes and phytoplankton in wetlands is lacking.

While there are many origins of DOM, it appears that release of extracellular dissolved organic carbon by healthy primary producers is a major source of dissolved organic carbon. Secretion of dissolved organic compounds occurs in both freshwater and marine plants. Below-ground biomass of marsh plants constitutes a significant portion of total organic carbon in wetland ecosystems. Estimates of the relative contribution of roots and rhizomes vary widely. There appears to be a lack of suitable methods to determine live and dead root and rhizome biomass. If wetland plants behave in a manner similar to terrestrial plants, then the rhizosphere effect may be very important in influencing the fate of contaminants in the root zone. The rhizosphere of wetland plants can include an oxidized zone in otherwise reduced sediments. The Eh, pH, and biological and abiological reactions characteristic of sediments would not be expected to occur in the rhizosphere. While it is known that microbial biomass is elevated and activity is stimulated in the rhizosphere of terrestrial plants, there appear to be few reports of microbial activity in the rhizosphere of wetland and aquatic plants. Extensive investigations of microbial processes in the rhizosphere of wetland and aquatic plants are required to determine what the reactions of importance are in that zone and what contributions they make to the release of organic matter in the sediment.

Wetland plants, emergent and submergent aquatic macrophytes, and phytoplankton are responsible for production of much of the highly labile DOM found in sediments. Once DOM is produced, it may condense to form POM through abiotic processes. Alternatively, the DOM may be assimilated by microbes and incorporated into cell biomass, or it may be mineralized by microorganisms. The proportion of DOM available for microbial assimilation is unknown and is likely to be a function of the DOM source and the physical and chemical characteristics of the environment into which the DOM is released. Lush and Hynes (1973) found that a large percentage of the leachate from autumn maple leaves

was converted directly to POM without microbial assimilation. This was dependent on pH and the calcium ion concentration. DOM that does not condense may be available to microorganisms and may be quickly metabolized by microbial assemblages found in association with plants and detritus.

The role of microorganisms in conversion of DOM to POM is just beginning to be investigated as is the role of microbes in cycling of nutrients in wetlands through mineralization of DOM. The role of microorganisms is important in production of refractory DOM and the alteration of the physical and chemical environment as a result of microbial metabolism of DOM. Studies have shown that the amount of soluble microbiologically available organic material present in marine sediments decreases with increased sediment depth (Seki, Skelding, and Parsons 1968). Total organic carbon was observed to increase with depth as a consequence of compaction and the accumulation of organic carbon resistant to microbial attack. The level of DOM in the pore water was observed to decrease sharply with depths below 2 cm. Results of other studies suggest that turnover of fatty acids (Balba and Nedwell 1982) and mineralization of organic nitrogen (Abd. Aziz and Nedwell 1979) are detectable only within the top 5 cm of the sediment profile. Even though the total organic carbon content of the sediment was found to increase with depth, the proportion available for microbial metabolism decreased from about 1 percent at the surface to less than 0.1 percent at 20-cm depth; at least part of this difference results from a decrease in pore water due to sediment compaction.

MICROORGANISMS AND CONTAMINANTS: DIRECT EFFECTS

Natural products, materials formed by plants, animals, and microorganisms, often comprise insoluble organic polymers, including aromatic macromolecules, which are unique in that only microorganisms have the required enzyme complements for decomposition (Evans 1977). Although microorganisms have evolved numerous enzymes to accomplish the degradation of natural products, tannins and lignins seem to decompose very slowly (Crawford 1981, Dickinson and Pugh 1974). The degradation products of these two materials, along with aromatic compounds, amino acids, proteins, and other materials that result from microbial activities, are combined in a complex series of processes to form humus (Evans 1977). Humus is a persistent compound, but is subject to continuous modification in soils. Aquatic humus is composed of both water-soluble and particulate-sorbed components of terrestrial origin and a component that may be formed in situ from degradation products of plankton (Gjessing 1976). Little appears to be known about fate and effects of humus in aquatic sediments.

Contaminants that are man-made xenobiotic compounds are often biotoxic, readily bioaccumulated, and frequently move through food webs. Many xenobiotics, along with several of the aromatic petroleum hydrocarbons and petroleum derivatives, are difficult to degrade. Degradation often depends upon the environmental circumstances. Degradation of aromatic compounds is frequently accomplished by microorganisms utilizing one or more alternative pathways. Aerobic and anaerobic catabolism of aromatic compounds has been reviewed by Evans (1977), who gave special consideration to anaerobic dissimilation. Among the pathways considered were anaerobic photometabolism of benzoate, anaerobic metabolism of benzoate through nitrate respiration, and methanogenic fermentation of benzoate and other aromatics by a microbial consortium. Xenobiotic compounds, especially the pesticides, often possess chemical

structures that interfere with metabolic processes. The basic chemical structure of many xenobiotics is similar to key biochemical intermediates; as such, they are theoretically susceptible to degradation by some microorganisms. However, addition (by man) of certain substituents, for example halogens, to various positions on an aromatic structure can serve to increase toxicity, while simultaneously decreasing the degradability of the molecule. The latter phenomenon occurs because the substituents shield the molecule itself from enzymatic attack. As a consequence, the substituent group(s) must be removed before biodegradation of many synthetic compounds can occur. Two other factors that inhibit biodegradation include competitive inhibition between microorganisms for growth factors and nutrients (Haller and Finn 1978) and diauxy, the preferential metabolism of easily degradable compounds over other, more refractory substrates (Atlas 1981, Chou and Bohonos 1979).

Microorganisms have an array of strategies for dealing with refractory compounds. Certain strategies require the presence of several different members of a microbial community, rather than individual microorganisms. Evans (1977) discussed several of these strategies, including cometabolism (Horvath 1972), enzyme induction, transfer of metabolic plasmids, a mutation that results in production of the requisite enzymes, and the formation of microbial consortia. Other investigators have provided evidence supporting the possibility that biodegradation of refractory molecules may be stimulated through application of a readily available carbon source to encourage development of a decomposer population (Chou and Bohonos 1979; Clark, Chian, and Griffin 1979), to facilitate cometabolism (Horvath 1972), or to encourage induction of appropriate enzyme systems (Atlas 1981, Chou and Bohonos 1979).

There are a number of ways in which microorganisms can interact with contaminants. These have been discussed by a number of authors, most recently by Gunnison (1986). Perhaps the most familiar aspect of microorganism-contaminant interactions deals with direct transformation of chemicals through biodegradation. Biodegradation can have a broad range of effects on the biological and chemical activities of xenobiotics. Alexander (1977) delineated the various degradational processes and their consequences as follows:

- a. Detoxication. Conversion of a molecule from a substance that is inhibitory in the concentration normally applied to the environment into a product that is nontoxic.
- b. Degradation. Transformation of a complex molecule into simple products.
- c. Conjugation, complex formation, or addition reactions. Processes involving either an increase in the complexity of a substrate or a combination of the substrate with microbial cellular metabolites.
- d. Activation. Conversion of a nontoxic substrate or potential pesticide into a toxic substrate.
- e. Alteration of the pattern (spectrum) of activity. Processes wherein the metabolism of a pesticide or other biotoxic compound changes the range of the compound from the species originally intended as the target to another, dissimilar species.

The environment has a bearing on which of the above pathways are able to function in the degradation of aromatic compounds. Environmental factors involved in promoting, inhibiting, or preventing microbial degradation have been considered by several investigators (Alexander 1977, Evans 1977, Matsumura and Bousch 1971, Woodcock 1971, Yoshida 1975). Some of these factors are particularly important in microbial metabolism in aquatic sediment, including, for example, the presence of aerobic or anaerobic conditions, the level of nutrients, pH levels, salinity, and temperature (Evans 1977). Some authors have even discussed the possibility of altering the soil environment to favor those microorganisms capable of degrading contaminants that are refractory in the present soil system (TNO 1984). It is possible that similar processes can also be applied to the sediment environment to promote degradation of contaminants.

An additional environmental factor of critical importance in soil and aquatic sediment is the nature and availability of sorptive surfaces. Sorptive capacity is of direct concern with regard to the mobility of chemical contaminants, particularly any microbial processes that alter either the sorptive capacity of the sediment or the ability of the contaminant itself to be sorbed. Sorptive capacity varies from one sediment to another, depending on textural composition and organic matter content (Buckman and Brady 1969, Dale 1974, Marshall 1980) and on the grain size of the sediment (Dale 1974). The nature and solubilities of contaminants already sorbed by the sediment are important determinants of mobility. Microbial modification of these substances that results in a change in solubility can have both negative and positive aspects. Iron-oxidizing thiobacilli assist in the oxidation of iron pyrites, possibly increasing the severity of an acid mine drainage problem. Alternatively, similar processes occur when metal sulfide ores are exposed during mining activities. Here the formation of an acid mine drainage from these deposits can be an efficient means of extracting ore and may even be encouraged by the mining company (Ehrlich 1981). In a similar vein, microbial modification of organic chemical contaminants in a sanitary landfill or a confined disposal facility may be important, if the modification increases the solubility or lowers the sorptive capacity of the soil or sediment. The concern here is that the more mobile chemical species may be able to move out of the landfill/disposal facility and into the ambient environment or down into the ground water. Similar concerns have been expressed with respect to undesirable water quality impacts resulting from environmental alterations of the sediment environment during resuspension and deposition of contaminant-laden dredged material in dredging and disposal operations.

Extensive literature is available on biodegradation, but very little is known of the effect biodegradation has on the mobility of contaminants or their degradation products. Theoretically, some degradation intermediates behave very differently from the parent molecule. For example, high molecular weight organic compounds are generally highly insoluble. If the molecular weight of polymeric compounds is reduced by microbial activity, the solubility and, consequently, the mobility may increase (Blum and Mills 1986). Other kinds of degradation reactions may be important. The potential for dehalogenation is important because the total number of halogens attached to an aromatic ring can greatly influence the environmental activity of the molecule. The contribution of groups to activity can be expressed as a function of the partition coefficient (Fujita and Nishizawa 1982). Each functional group is assigned a PI value based on empirical determinations, and this can then be

used to compute a partition coefficient. The relationship may be an approximation, but it demonstrates that for many compounds, removal of halogens may alter the partition coefficient; this, in turn, is related to the solubility of the compound. With a decrease in the partition coefficient, the solubility may increase to some extent, and the tendency for the compound to segregate into organic complexes as a result of hydrophobic interactions decreases. Most of the contaminants of public concern are not high molecular weight compounds, and depolymerization of these compounds during degradation is not likely to alter their mobility.

DOM is important in these reactions because dehalogenation is not coupled to biosynthetic pathways in microorganisms. As a result, an energy source other than the contaminant is required, at least for this part of the degradation. Higher concentrations of available DOM would result in higher levels of indigenous microbial species, possibly expediting the degradation process. This process can be inhibited by a situation in which contaminants are complexed with DOM, shielding both the contaminant and the DOM from degradation.

Ring cleavage also has the potential to enhance the mobility of contaminants. The meta-cleavage pathway normally results in formation of dicarboxylic acids. As a result of the presence of easily dissociating protons on the carboxyl groups, these compounds are often highly polar, readily solvated, and very mobile in the environment. These compounds are quite different from the original contaminant and may have important properties as described above. Very little is known about the mobility of contaminant degradation products, and future investigations may demonstrate the potential for breakdown intermediates of interest.

An additional area of direct interaction of microbes with contaminants includes uptake, either actively or passively, and/or surface sorption of the contaminant to the cell. These reactions include both organic and inorganic substances. Once sorbed, the contaminant would move with the bacteria in suspension. If the ionic or chemical status of the environment changes, the contaminant could desorb from the bacteria. Alternatively, cell death could also release the contaminant into the sediment or water.

Concentration of organic contaminants in sediment and water can influence the manner in which these chemicals are metabolized and the rate at which they disappear. For example, Wang, Subbarao, and Alexander (1984) have shown that a chemical may be metabolized at one concentration and cometabolized at another. The presence of substrates in addition to the concentration of an organic contaminant is an important factor in determining the behavior of contaminants in the environment. The rate of the cometabolism is highly dependent on the concentration of microbially available DOM in sediment and water. One theory holds that heterotrophic bacteria require a minimum concentration of organic carbon to meet requirements for maintenance; below this level, bacterial growth is unable to occur (Powell 1967). Microorganisms responsible for degradation of contaminants may be included in this group. As a result, it is anticipated that levels of contaminants below the threshold necessary to support microorganisms carrying out degradation will result in very slow disappearance of the contaminant.

Sorption of Organics

Sorption reactions are important with respect to interactions between contaminants and DOM for two reasons: (a) the contaminant may be directly sorbed onto the surface of a particle, with the result that the contaminant and the particle it is sorbed to may settle onto the sediment surface and become buried, and (b) DOM is often sorbed to organics and may contribute to the sorption processes occurring in association with the particle. This role of DOM in the reaction of contaminants and inorganic particles requires further investigation.

Particulate and DOM Interactions

DOM can influence many aspects of the physical state and environmental fate of organic contaminants. Humic substances, for example, can alter the solubility of nonpolar compounds (Boehm and Quinn 1973; Hassett and Anderson 1979; Landrum et al. 1984; Mathur and Morely 1978; Matsuda and Schnitzer 1971; Ognier and Schnitzer 1970; Wershaw, Burcar, and Goldberg 1969). Humic substances can also bind organic compounds by one of several different mechanisms, hydrolyze pesticides (Perdue 1983), photosensitize contaminants (Zepp, Baughman, and Schlotzhauer 1981a, b), alter the bioavailability of contaminants to aquatic organisms (Leversee et al. 1983), and control the rate and extent of biodegradation of organic compounds at low concentrations (Holm et al. 1980, Schmidt and Alexander 1985).

DOM in general increases the aqueous solubility of hydrophobic organic compounds, including pesticides (Marshall 1980; Wershaw, Burcar, and Goldberg 1969), n-alkanes, aromatic hydrocarbons, and dibutyl phthalate (Boehm and Quinn 1973), and cholesterol and 2,2',5,5'-tetrachlorobiphenyl (TCB) (Hassett and Anderson 1979, 1982). DOM has been observed to influence hydrolysis of organic contaminants (Li and Felbeck 1972a, b; Schmidt and Alexander 1985), and a model has been formulated that includes the effect of DOM on hydrolysis (Perdue 1983). Other studies have demonstrated that some chemicals photoreact more rapidly in water when DOM is included (Mills et al. 1982; Powell 1967; Ross and Crosby 1975; Zepp et al. 1975; Zepp, Baughman, and Schlotzhauer 1981a, b).

REACTIONS WITH METALS

Direct Reactions

Microorganisms exhibit a variety of activities toward many metals that can have major influences on the mobility of these elements. Processes carried out by microorganisms may alter the Eh-pH of a system, and microorganisms can also participate in a series of redox reactions that affect the mobility of metals. For example, reduced metals are often more soluble and mobile than their oxidized forms. Several metals are thought to function as redox species; however, with the exception of iron, there is little evidence to suggest that the microorganism obtains energy from the oxidation process or by the oxidized metal serving as the electron acceptor. Metals thought to undergo microbial oxidation and reduction include As, Zn, Cu, Hg, U, and perhaps Cd and Cr. In many cases, there is little clear information to indicate whether there is a direct interaction between the metal and the microorganisms or if the interaction occurs secondarily as a result of microbial modification of

the environment. For mercury there is good evidence for production of a mercury reductase system in several bacteria (Wood 1974); the product of mercury reduction is elemental mercury, which is very mobile due to its volatility.

DOM would not be expected to influence oxidation of metals, providing these reactions occur directly. It is expected that these oxidations are carried out by autotrophs. However, reduction reactions can be strongly influenced by the quantity and quality of DOM; in this case, metals most likely serve as terminal electron acceptors for heterotrophs. There is little information to indicate that metals other than iron and mercury undergo significant direct oxidation or reduction. Some metals, including Hg, Sn, As, and Se, undergo methylation in anaerobic sediments with consequent formation of volatile methyl complexes such as mono and dimethyl mercury. These substances diffuse rapidly through water and, under suitable conditions, into the atmosphere. Such methylation/volatilization reactions are thought to be a part of a microbial strategy to convert toxic material into a form that moves away from the area of the cell. Methylations occur under anaerobic conditions, and all microorganisms carrying out such reactions are heterotrophs.

Indirect Reactions

Metals generally sorb to particles or complex with organic matter through a mechanism related to ion exchange, although many organometallic complexes are bound together with covalent bonds.

Clays and other inorganic particles often have a net surface charge resulting from the arrangement of O and OH groups on the crystal surface as well as from the internal composition of the crystal. The charge associated with unshared oxygens or hydroxyl groups results from acid dissociations. At low pH, the negative charge will be saturated with protons, and there will be no charge to attract other cations. At higher pH, the supply of protons decreases and the resulting negative charge is fulfilled by other cations, e.g., metals. At pH ranges from 4 to 10, the pH of water in contact with sediment particles may play a major role in releasing the metals from the surface of the clay. Thus, in some low pH wetlands (acid bogs, for example) metals may be more mobile as a result of the high proton concentration in the sediment pore water and the water column displacing metals from clay surfaces. In higher pH environments (marshes), cations such as metals are held more firmly due to the lack of protons for displacement.

Davis (1982) has shown that a large percentage of the surface area on most inorganic particles is coated with organics derived from dissolved material. The coating can interact in several ways, depending on pH, relative surface area, amount of sorbable organic matter in the system, nature of the solid substrate, and inorganic electrolytic composition. Dissolved organics can act through various mechanisms to increase the sorption of cations through increasing the effective number of negatively charged sites associated with a particle.

Ionic reactions are largely responsible for sorption of metals to organic particles. Carboxylic acids and phenols, the primary active sites on organic particles (Schnitzer and Kahn 1972), are also important in interactions with DOM. As was the case with clay, such reactions are pH dependent with the specific effective pH range for effectiveness being dependent on the pKa of

the specific functional group. Because microorganisms mineralize organic matter in the sediment, they can play a critical role in determining the mobility of metals bound to particles by organic compounds. In spite of this, there has been no direct examination of this type turnover, nor any examination of the fate of materials adsorbed to the particles.

There are several different types of reactions that may result in complexation of metals to dissolved compounds. Presence of carboxyl groups of phenolic hydroxyls will create an ion-binding condition identical to that presented for organic and inorganic particles. Chelation is typical of this type of reaction and is very important in movement of iron-oxide organic colloid complexes in fresh waters as well as in liberating metals from surrounding soils and in liberating metals from soils in the area and allowing movement of metals into wetlands (Zyczynska-Bakoniak and Szpakowska 1985).

While many plants are able to remove metals from wetland sediments, the major environmental concern is with the release of metals upon death and decomposition of the plant. This obvious area of research has never been addressed in a meaningful way. Many scientists consider particulate detritus as an effective extractor of metals from water and an assistant in moving metals to the sediment. The potential for metal release from decomposing saltmarsh cordgrass was considered by Breteler et al. (1981), but their data showed an enrichment in metal concentration in detritus over time. This suggested retention of plant-bound metals and perhaps sorption of metal from the water column. Specific studies of metal exchange have not been performed; however, present evidence suggests that net flux would be zero or into the organic matter (Giblin et al. 1983).

Changes in the Physical Environment

A potentially important contribution of DOM to the mobilization of contaminants in aquatic sediment and wetland soil is a consequence of the alterations in pH and Eh that occur in anoxic areas. Little is known about this process. Much of the metal binding occurs on negatively charged sites associated with deprotonations associated with DOM and POM, and the pH of the system greatly influences the number of sites available. Release of protons by microorganisms during the decomposition of organic matter could decrease the pH, if it were not for the fact that the protons released are usually accompanied by an anion to balance the charge. The question of how the decrease in pH occurs is not always clear. In addition, in most sediments and flooded soils, oxygen is absent and reduced products accumulate. The decrease of Eh often has a marked influence on mobility of materials capable of undergoing oxidation-reduction reactions, especially metals.

The literature does not present any information on the movements of most contaminants between sediments and the water column when the sediment is reduced and oxidizing conditions are present in the water column. Several laboratory studies have examined the solubility of different redox species for a variety of compounds. Integrated studies of the role of organic decomposition on maintenance of low pH-Eh conditions in conjunction with mobilization processes are lacking. This represents a potentially important means of releasing contaminants from sediments, and it is important to develop a better understanding of this area.

CONCLUSIONS

The evaluation of existing literature on the role of microorganisms in production of DOM and mobilization of contaminants makes it possible to say something about the metabolism of contaminants in natural ecosystems, with particular reference to aquatic sediment. A large variety of chemical compounds exists in sediment. These materials are of both natural and anthropogenic origin. Often, members of the native sediment microbial community are able to multiply and utilize these substances. As indicated previously, a microbial species having the enzymes necessary to degrade these materials may not be present. However, there is always the possibility that microorganisms having the capability may appear through one or more mechanisms; these include invasion and genetic modification of native or invading species. Alexander (1977) has noted that not all pollutants are organic, and a large number of the organic contaminants are destroyed either principally or only by way of microbial activities. Several organisms are capable of complete degradation, partial conversion, or cometabolism of contaminants (Bourgquin and Pritchard 1979, Leisinger et al. 1981). Nonetheless, while some contaminants are easily degraded, others are not and may be quite persistent. Metabolism of pesticides has been used by Alexander (1977) as an example of the ways in which microorganisms act or fail to act upon pollutants.

Problems often arise as a result of the relationship of contaminants present in a sediment to the use or intended use of the aquatic ecosystem. Problems may arise with the intended use of other environments, if the latter are potential locations for placement of a contaminated sediment. The problem becomes a concern from a legal standpoint when and if the accumulation of materials in the sediment is high enough to cause a conflict with legislative mandates. For example, open-water disposal of contaminated dredged sediment requires consideration of several legislative items, including the Fish and Wildlife Coordination Act, the National Environmental Protection Act, the Marine Protection Act, the Research and Sanctuaries Act, the Federal Water Pollution Control Act, the National Ocean Pollution Planning Act, and the London Dumping Convention. In contrast, freshwater concerns with aquatic sediment relate to many of the same, plus several additional pieces of legislation. The latter include the Clean Water Act, National Drinking Water Standards, and state and local water quality criteria.

Interactions between microorganisms and DOM and contaminants have many ramifications with respect to submerged sediments and wetlands. In addition to the possible failure of microorganisms to degrade some contaminants or the ability of other microorganisms to form undesirable products, consideration must also be given to the possible changes that may occur if a sediment is moved. Movement of a sediment from one location to another in an aquatic ecosystem may not necessarily cause a change in the sediment as a microbial habitat. However, placement of a sediment into another environment may result in some severe changes, and these may then alter or influence the activity of the sediment microflora. Finally, alteration of existing environments, such as wetlands, may cause difficulties if contaminants are present or if the wetland is important in removing contaminants from inflowing waters.

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TESTS ON SEDIMENT GAS GENERATION

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AD-P006 907

ABSTRACT

In the past, many clean water projects have been implemented because of complaints from residents about sediment odor. Thus, sediment odor is an important environmental effect of sediment pollution. On the other hand, research on its behavior has not advanced satisfactorily. The second Port and Harbor Construction Bureau of the Ministry of Transport is now studying the fundamental behaviors of sediment gases and pollution control regarding sediment odor. For this purpose, laboratory tests on sediment gas generation were performed in 1984 and in 1985 by the Sediments Management Association. This paper deals with those test results.

SAMPLING SITES AND SEDIMENT PROPERTIES

Sediment samples were taken from the Shibaura Canal near Haneda Airport in Tokyo Bay (Figure 1). The sediment there is highly polluted; therefore, it was dredged in 1978 and 1980. The samples were taken from five points in 1984 and from three points in 1985.

TEST APPARATUS

The jar test column, made of acryl, had a diameter of 190 mm and three columns with three height stages: 80, 130, and 180 mm (Figure 2). The sediment sample was placed at the bottom of the column to a height of 300 mm. After the air above the sediment and water was removed by N_2 gas, testing began. Figure 3 indicates the relationship between ignition loss and COD concentrations. The samples were divided into two groups, high and low pollution. The water depth was 2.7 m and the sampled sediment depth was 15 cm. Properties of the sediment samples (Figure 4) are listed in Table 1. The heights of the overlying water were 40, 90, and 140 cm. The gas concentrations dissolved in the water in 1984 were as follows:

<u>Gas</u>	<u>Concentration After 20 days</u>
Methane (CH_4)	0.578 mg/l
Hydrogen sulfide (H_2S)	26.6
Methyl mercaptan	<0.0001
Methyl sulfide	<0.0001
Methyl disulfide	<0.0001



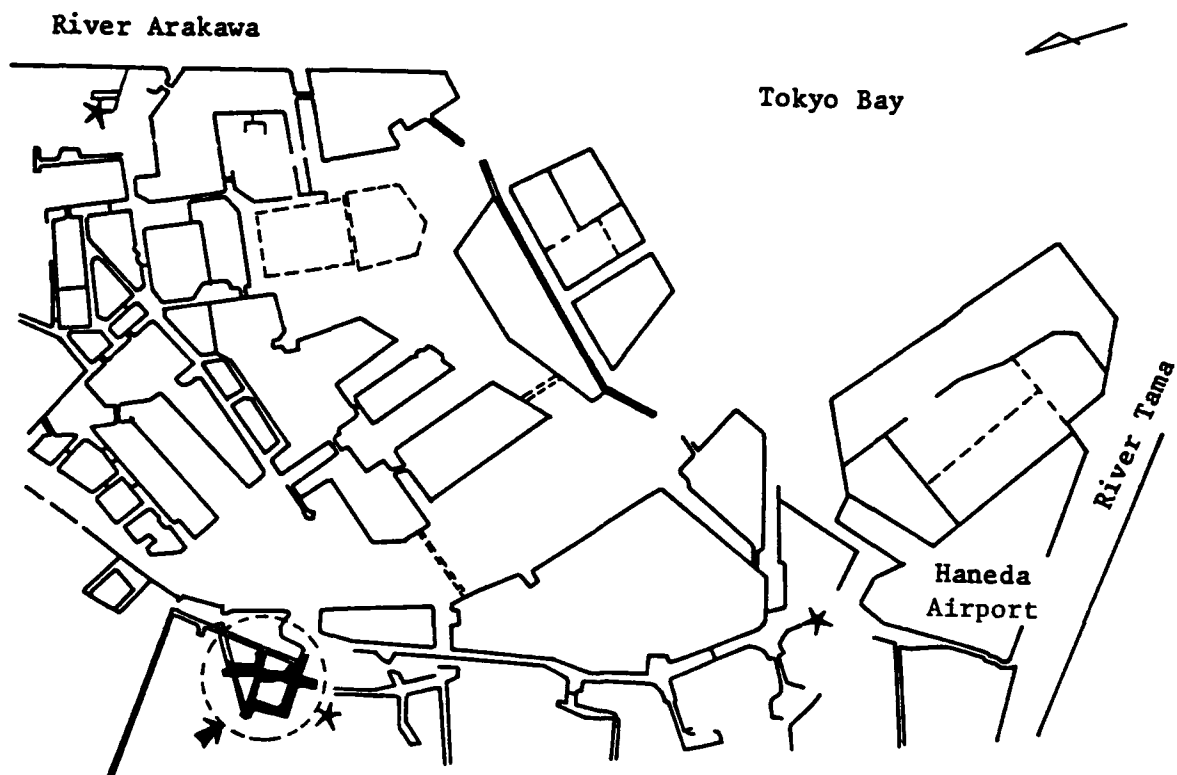


Figure 1. Sampling sites

From this, it was found that the generated gas was composed predominately of CH_4 and H_2S . Therefore, gas analysis in 1985 was conducted only for CH_4 and H_2S .

OVERALL GAS TRANSFER COEFFICIENT

Gas transfer is depicted in Figure 5. The overall gas transfer coefficient K_{La} is defined by the following equation.

$$\frac{m}{Vt} = K_{La} (C_g - C_w) \quad (1)$$

- m = gas concentrations dissolved in water, mg/l
- V = volume of water
- t = time
- K_{La} = gas transfer coefficient
- C_g = concentration of gas
- C_w = concentration of water

The values of overall gas transfer coefficient for H_2S calculated from the test results are shown in Table 2 and Figure 6. It is noted that the coefficient K_{La} depends upon the value of m (Figure 6).

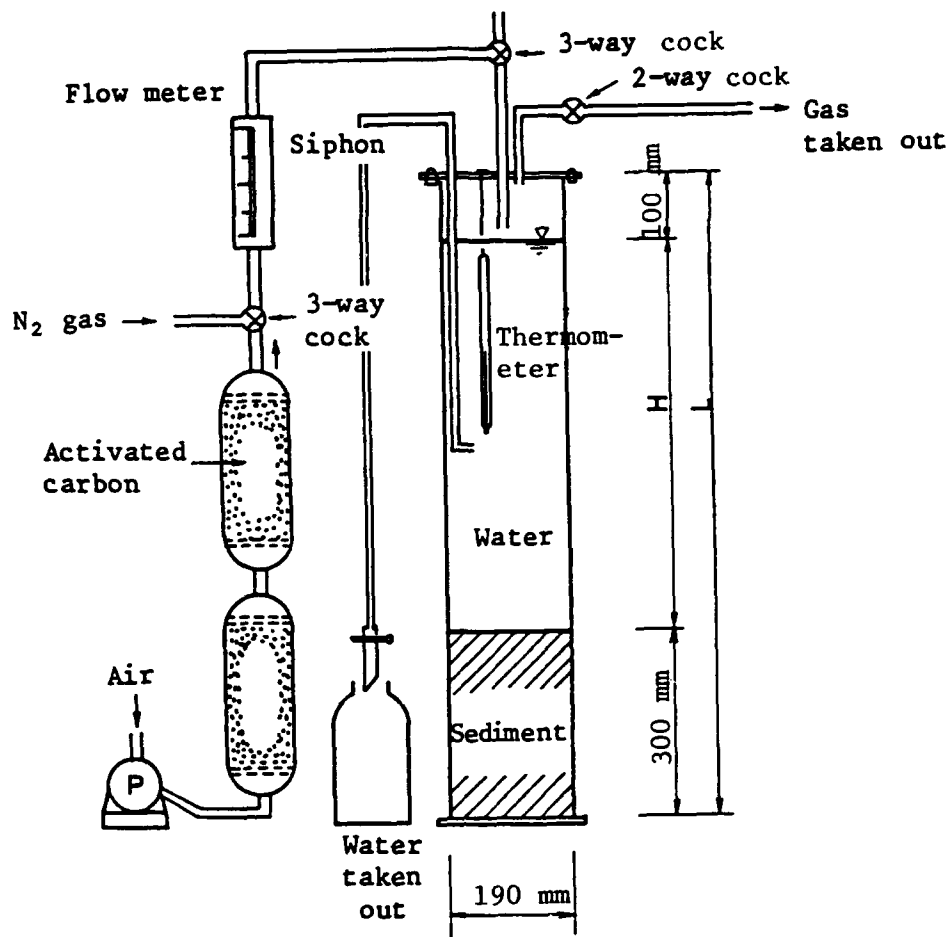


Figure 2. Test apparatus

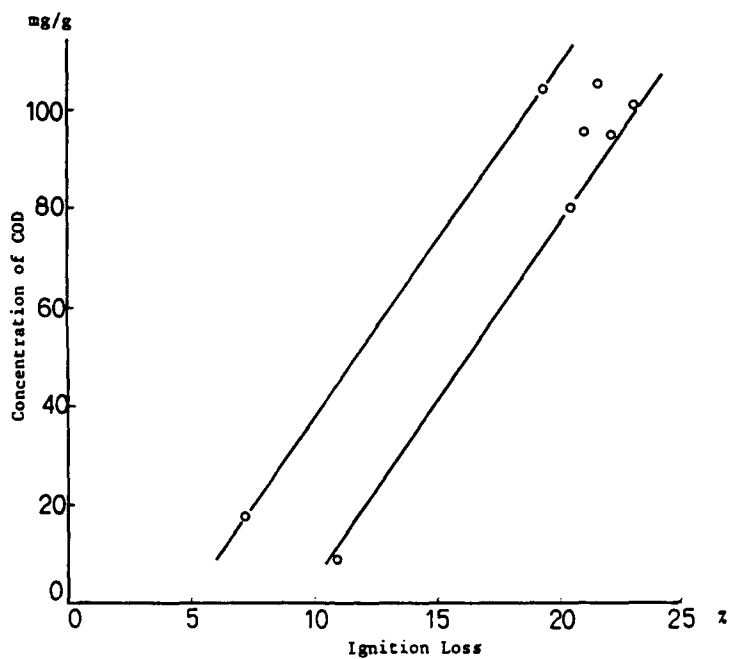


Figure 3. Relationship between ignition loss and COD

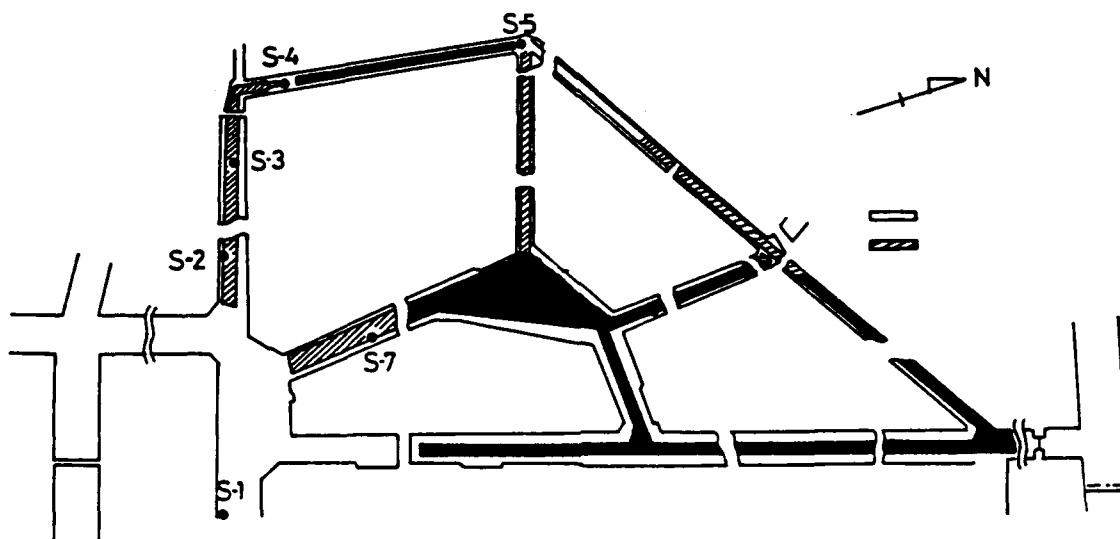


Figure 4. Sampling points

TABLE 1. PROPERTIES OF THE SEDIMENT SAMPLES

Year	Sampling Point	Ignition loss %	COD mg/g	Sulfide mg/g
1985	S-1	7.2	17.9	0.73
	S-3	19.3	104	7.35
	S-7	20.4	105	10.2
1984	S-1	20.5	79.4	5.71
	S-2	21.0	95.9	8.04
	S-3	23.0	101	7.91
	S-4	22.1	94.8	7.30
	S-5	11.0	8.8	0.21

TEMPERATURE EFFECTS UPON GAS GENERATION

Figure 7 shows the relationship between the dissolved gas concentrations m and water temperatures for methane. This correlation can be represented by the following equation:

$$m_T = m_{20} 1.14^{T-20} \quad (2)$$

m_T = dissolved gas concentrations at temperature $T^\circ \text{C}$

m_{20} = dissolved gas concentrations at 20°C

For H_2S gas (Figure 8), the same equation can be obtained.

Figure 5. Movement of gas

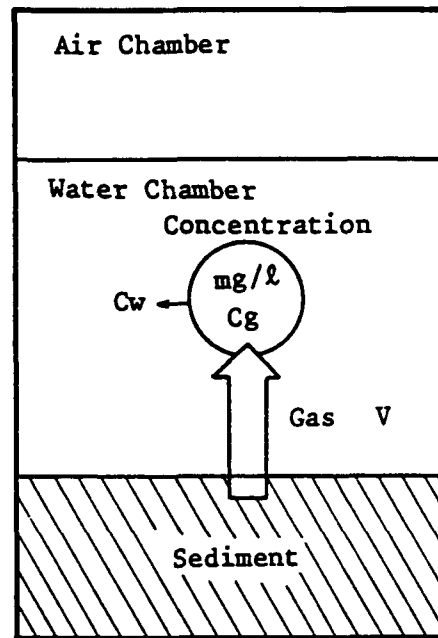


TABLE 2. K_{La} OF H_2S

Sampling Point	Height of Water			Average
	140 cm	90 cm	40 cm	
S-1	5.48×10^{-5}	9.28×10^{-5}	4.14×10^{-5}	6.30×10^{-5}
S-2	7.28×10^{-4}	5.11×10^{-4}	4.48×10^{-4}	5.62×10^{-4}
S-3	8.78×10^{-4}	5.41×10^{-5}	9.63×10^{-5}	5.05×10^{-5}

RELATIONSHIP BETWEEN DISSOLVED GAS CONCENTRATIONS AND SEDIMENT POLLUTION

Figure 9 shows the relationships between sulfide concentrations in sediment and dissolved gas in overlying water. The figure clearly indicates the strong relationship between sediment pollution and gas generation: the greater the sulfide content in sediment, the larger the dissolved gas concentrations in the overlying water, and the higher the liquid height, the larger the gas concentrations in the overlying water. Figure 10 shows the relationship between TOC and dissolved gas concentrations. From the figure it can be seen that CH_4 concentrations dissolved in the water increase rapidly if the concentration of TOC exceeds 70 mg/g.

GAS GENERATION RATES

The gas generation rates (Figure 11) from sediment, represented by G ($mg/m^2/day$), can be obtained by the following equations:

$$GAt = m + G'At \quad (3)$$

$$G = \frac{m}{At} + G' \quad (4)$$

G = gas generation rates from sediment
 A = cross-sectional area of the test column
 t = time
 m = dissolved gas concentrations in the overlying water
 G' = gas generation rates at the water surface

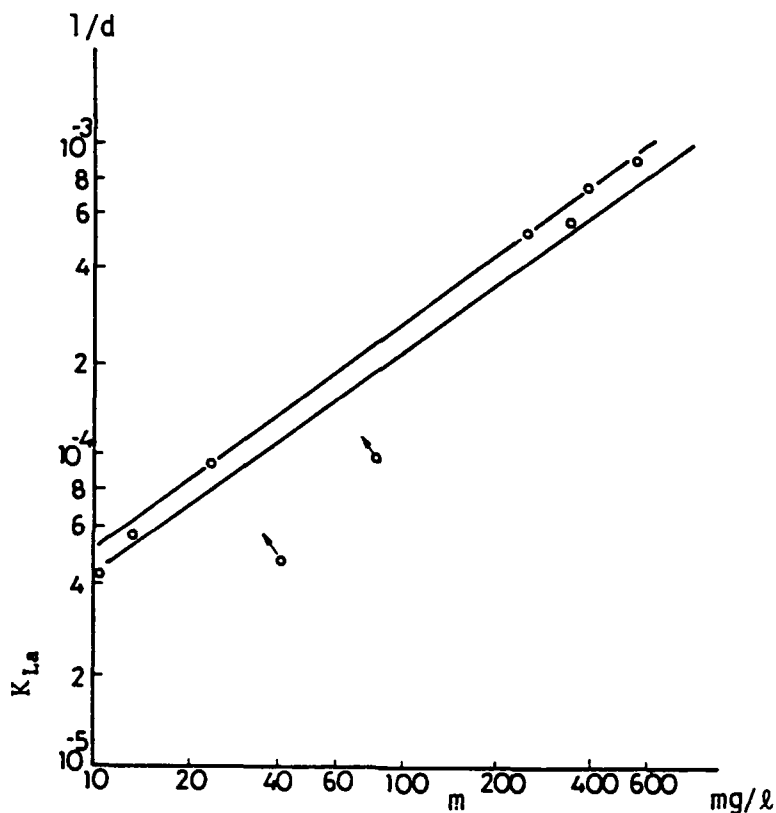


Figure 6. Relationship between m and K_{La}

Since the values of m and G' are measured, the values of G can be calculated from Equation 4 (see Table 3). Figure 12 shows the relationships between sulfide concentrations in sediment and H_2S generation rates. There is clearly a linear correlation. There is also a strong correlation between TOC and CH_4 gas generation rates (Figure 13). In this case the correlation curve is not linear and rapidly increases according to the increase of TOC.

GAS CONCENTRATIONS IN THE OVERLYING WATER

The dissolved gas quantities in the water are represented below as mentioned before:

$$\frac{m}{Vt} = K_{La} (C_g - C_w) \quad (1, bis)$$

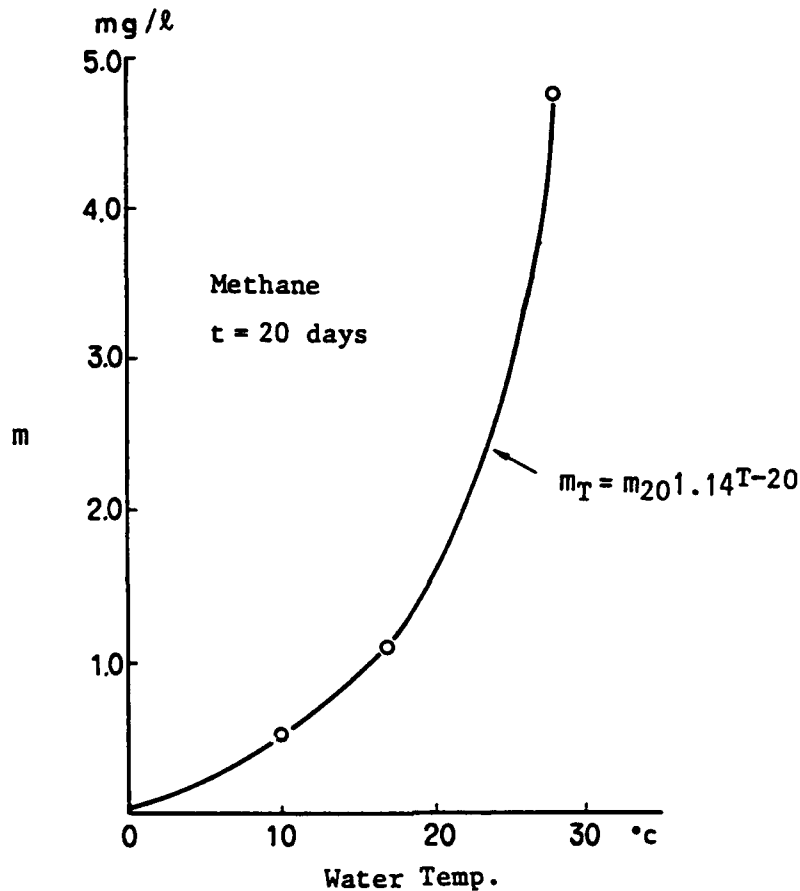


Figure 7. Temperature effects for CH_4

This can be rewritten as follows:

$$\frac{m}{At} = HK_{La} (C_g - C_w)$$

The left term of the above equation is the dissolution rate of gas, which is equal to the product of HK_{La} and the difference of gas and water concentrations. In Figure 14, taking a thin layer dz at the height of Z , we can consider the balance of inward- and outward-flowing gas material as:

$$\frac{\partial c}{\partial t} = K \frac{\partial^2 c}{\partial z^2} - K_{La} H \frac{\partial c}{\partial z} \quad (5)$$

where

- c = concentration in the water
- K = diffusion coefficient of gas molecules in the vertical direction
- z = height in liquid column
- K_{La} = overall gas transfer coefficient
- H = liquid height (total)

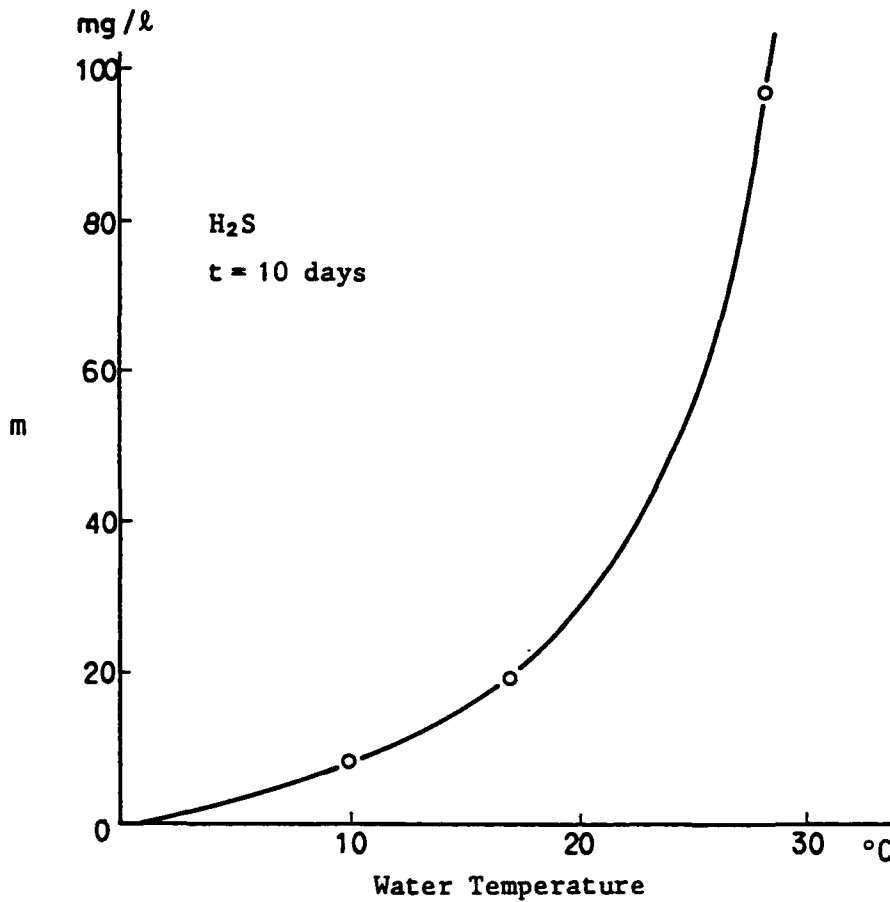


Figure 8. Temperature effects for H₂S

The general solution of Equation 5 is represented as follows:

$$C = Ae^{at-bz} \quad (6)$$

By putting this into Equation 5

$$a = Kb^2 - K_{La}Hb$$

we have then

$$C = Ae^{(b^2K - bHK_{La})t - bz} \quad (7)$$

This equation represents the vertical distribution of water concentrations. The boundary condition (Figure 15) at the sediment surface is as follows:

$$-K \left. \frac{\partial c}{\partial z} \right|_{z=0} = G \quad (8)$$

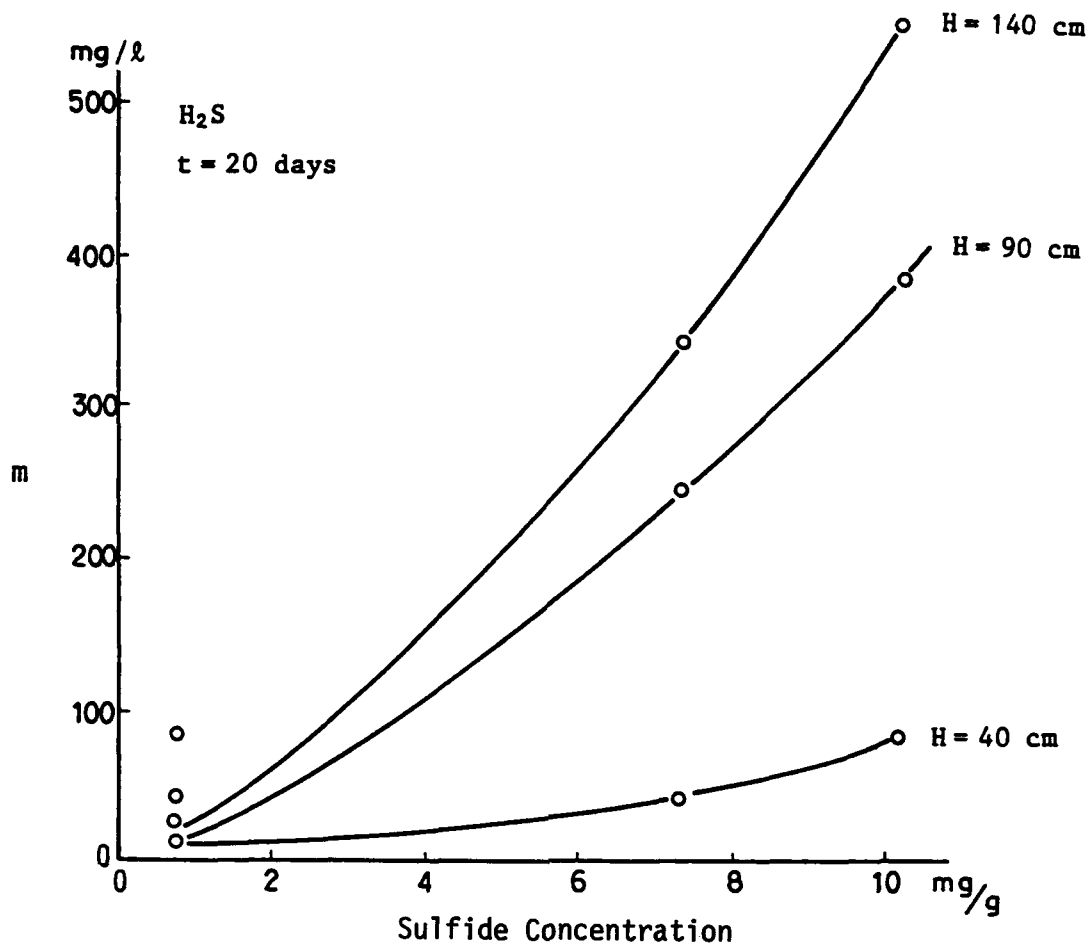


Figure 9. Relationships between sulfide concentration in sediment and the water

Then we get

$$A = \frac{G}{bK} \quad (9)$$

The boundary condition at the liquid surface is

$$-K \left. \frac{\partial c}{\partial z} \right|_{z=H} = G'$$

Therefore, we have finally

$$G' = G e^{\left(b^2 K - bHK_{La} \right) t - bH} \quad (10)$$

From this equation, the value of K can be obtained. For example, from the following data:

$$G = 699.1 \text{ mg/m}^2/\text{day}$$

$$G' = 0.0055 \text{ mg/m}^2/\text{day} \text{ (} t = 10 \text{ days)}$$

$$G' = 0.0085 \text{ mg/m}^2/\text{day} \text{ (} t = 20 \text{ days)}$$

we get

$$b = 8.7 \text{ l/m}$$

$$K = 7.16 \times 10^{-4} \text{ m}^2/\text{day} = 8.29 \times 10^5 \text{ cm}^2/\text{sec}$$

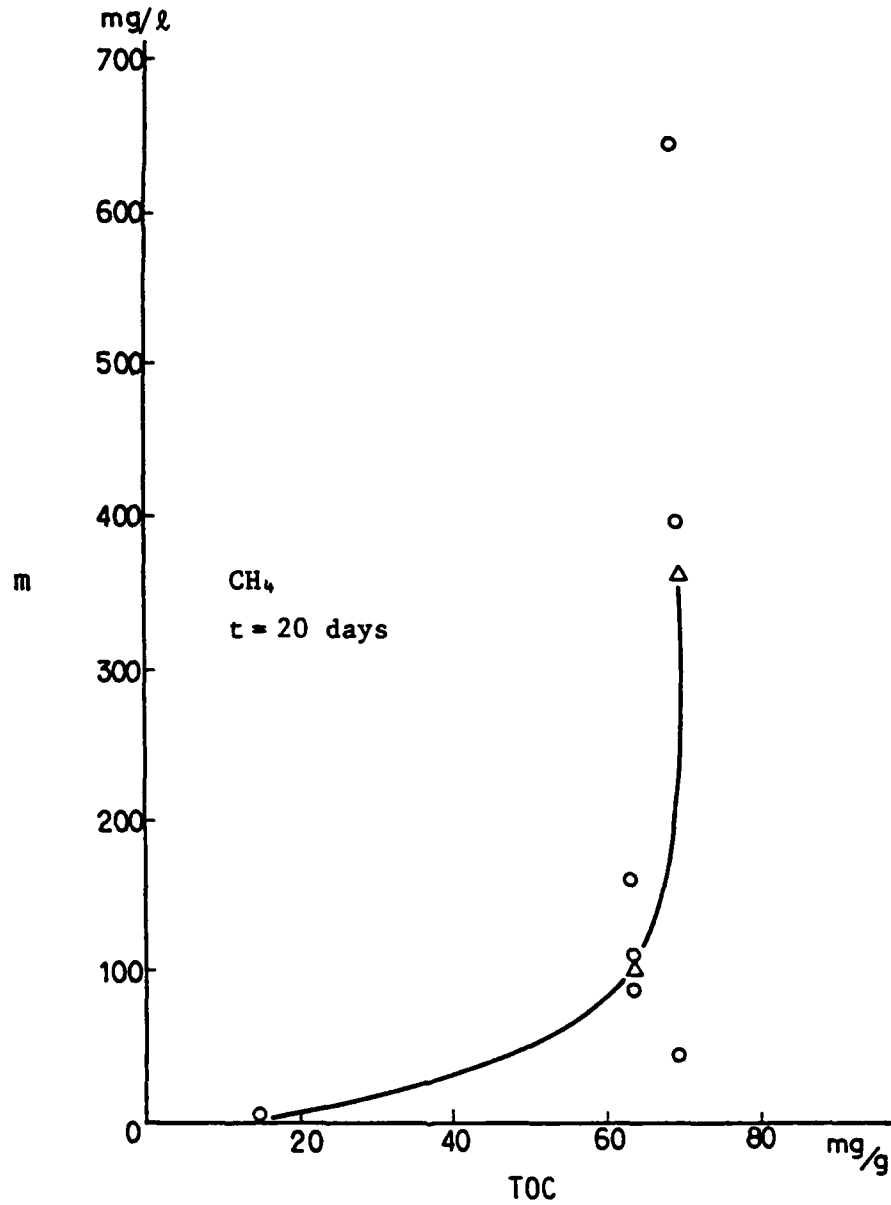


Figure 10. Relationship between TOC and m

Figure 11. Gas generation rates

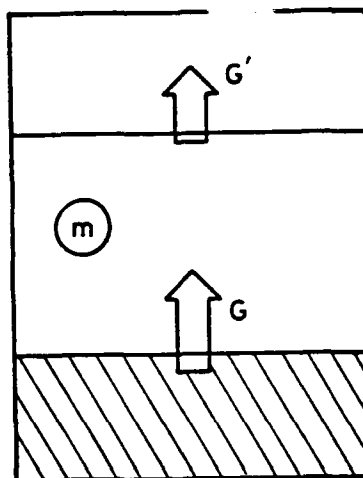


TABLE 3. GAS GENERATION RATE VALUES*

Parameter	Height of Water		
	140 cm	90 cm	40 cm
G' , mg/m ² /day	0.0085	0.0296	0.0205
m , mg/m ² /day	555	342	82.6
m/At , mg/m ² /day	699.1	670.1	364.2
G , mg/m ² /day	699.1	670.1	364.2

* S-7, $t = 20$ days.

Since the vertical distribution of water concentrations is now known, we can calculate the value of m :

$$m = \frac{V}{H} \int_0^H \frac{G}{bK} e^{\left(\frac{b^2 K - bHK}{dz} \right)_{La}} t - bz \quad (11)$$

$$= 507 \text{ mg}$$

There is a fairly good agreement between the calculated value of 507 mg and the measured value of 555 mg (Table 3).

GAS EMISSION INTO THE AIR

This section will consider the odor problem at a sampling site. The important matter is gas emission into the open air (Figure 16). As mentioned before, gas generation rates in the overlying air are represented as follows:

$$G' = Ge \left(\frac{b^2 K - bHK}{La} \right) t - bH \quad (10, \text{bis})$$

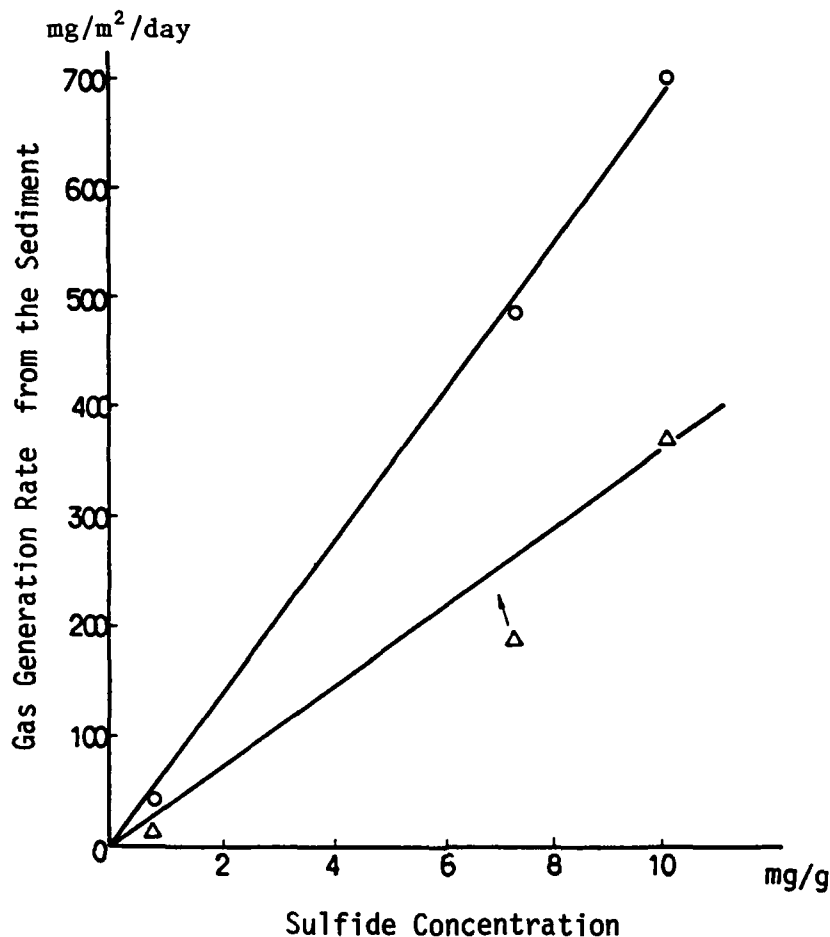


Figure 12. Relationship between sediment sulfide concentration and G

From this it is known that gas generation rates at the liquid surface are proportional to e^{-bH} . This means that the lower the water depth, the larger the gas emission into the air. For example, such a correlation is indicated in Figure 17 regarding the S-3 sample. That is, if we compare the gas emission at water depths of 0.75 m and 1.4 m, the gas quantities at the former are 100 times larger than the one of the latter. Therefore, we must pay particular attention to odor emission at sites in shallow water.

The gas concentration at the water surface is represented as follows:

$$C_A = C_o \frac{G'}{G} = C_o e^{(b^2 K - bHK_{La})t - bH} \quad (12)$$

Then we have at the initial condition ($t=0$)

$$C_A = C_o e^{-bH} \quad (13)$$

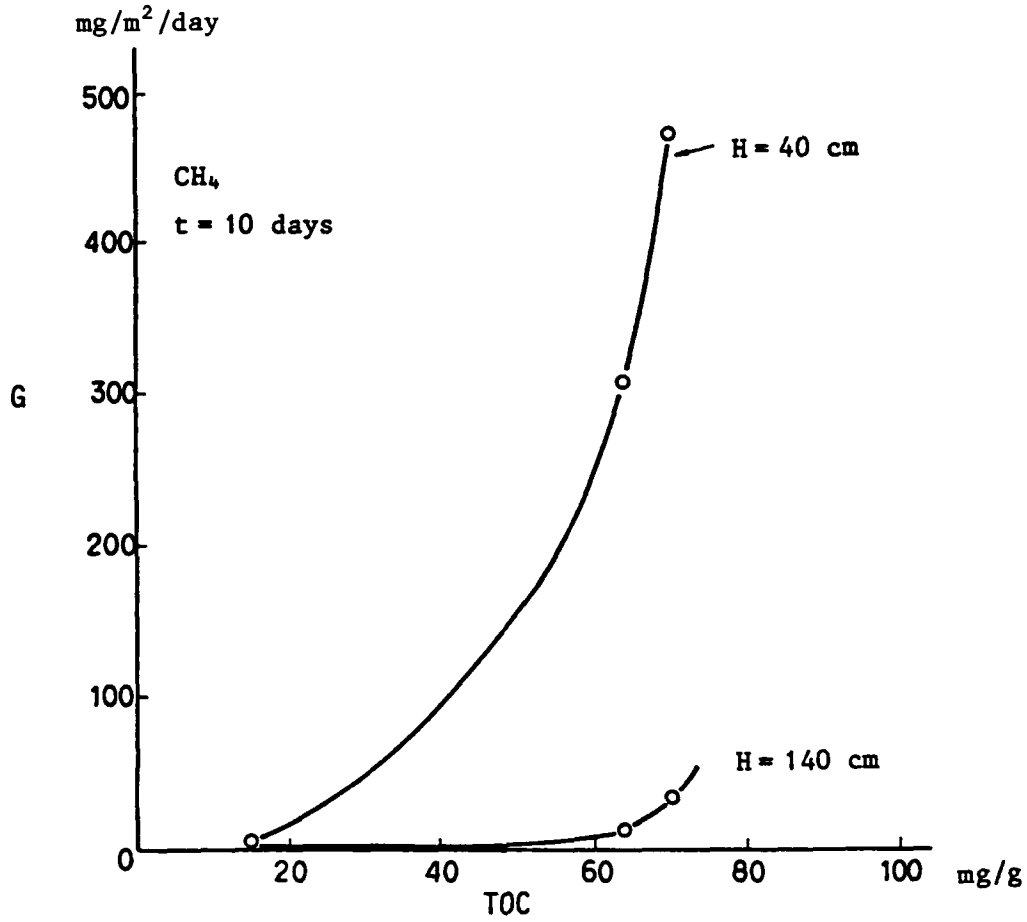
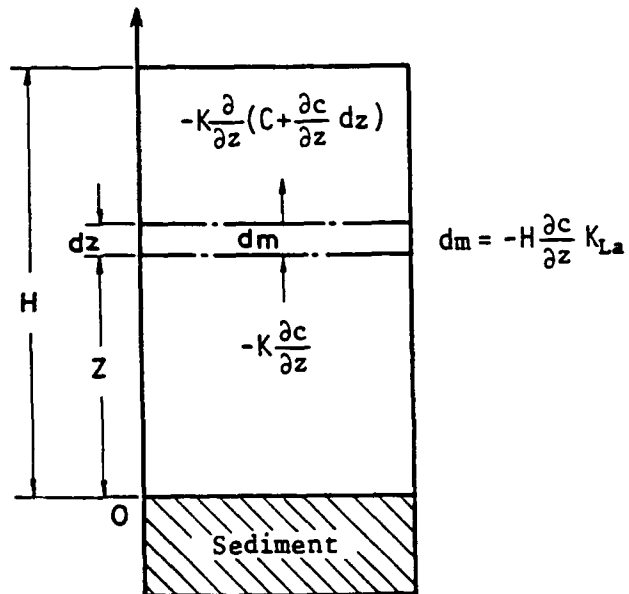


Figure 13. Relationship between TOC and G for methane

Figure 14. Movement of gas in the water



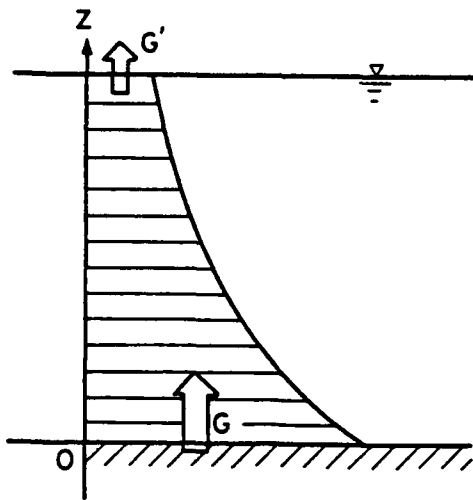


Figure 15. Boundary conditions

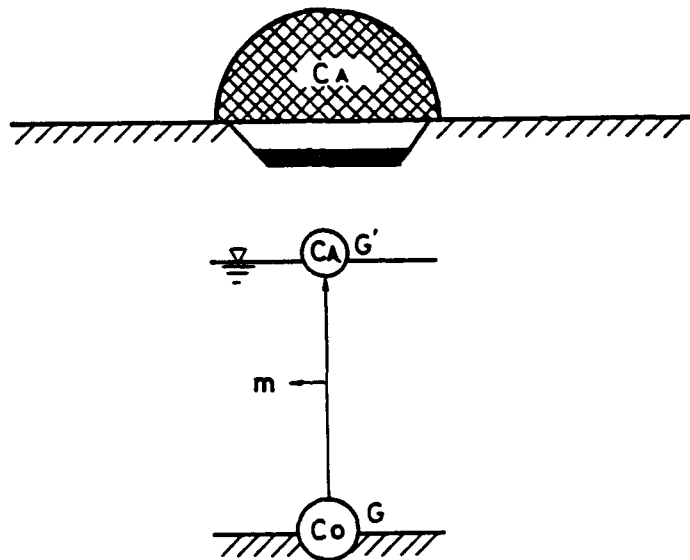


Figure 16. Gas emission into the air

From this it is found that the larger the value of b , the lower the gas concentration at the water surface. Therefore, in order to investigate odor problems, it is necessary to know the value of b .

POLLUTION CONTROL WITH REGARD TO SEDIMENT ODOR

As mentioned before, the distribution coefficient b plays an important role in gas emission into the air. If the relationship between the values of b and sediment pollution is known, it is possible to obtain the gas concentrations in the overlying air resulting from sediment pollution.

Figure 18 shows the relationship between the values of b and sulfide concentrations in sediment. The higher the sulfide concentration, the smaller the value of b . Figure 19 shows the calculated gas concentrations C_A at

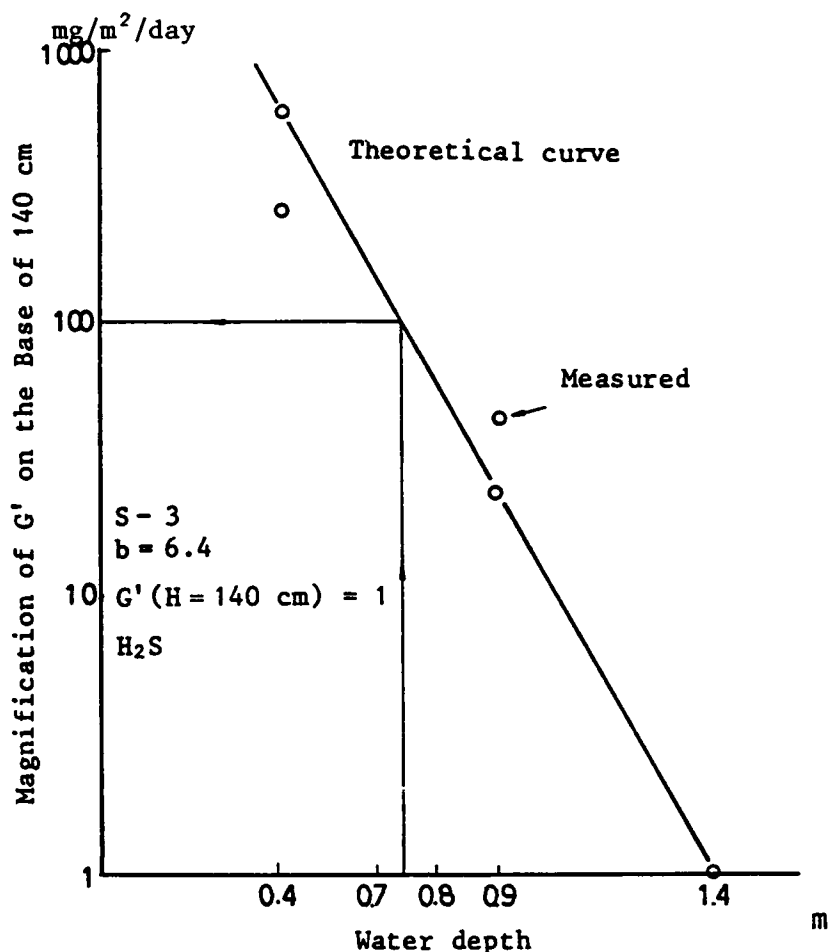


Figure 17. Gas emission according to water depth

the liquid surface with a parameter of b . Thus, the relationship between sulfide and gas concentrations in the overlying air can be obtained as shown in Figure 20. The toxicity of gases guideline is indicated in Table 4. If the correlation curve is intersected by an appointed permissible concentration, the sediment removal limit can be obtained. For example, if the permissible concentration of H_2S is set at 0.2 ppm, the sediment removal limit becomes 6.5 mg/g of sulfide in sediment. That is, sediment having concentrations greater than 6.5 mg/g should be removed to prevent sediment odor.

CONCLUSIONS

These tests indicated that warm water temperature and shallow water depth have adverse effects on odorous gas emission into the air. Therefore, a serious odor problem may occur in shallow rivers or canals and in summer. Although the relationship between sediment pollution and odorous gas generation has been obtained, this relationship is not enough for practical application because in laboratory tests the effects of water flow are not taken into account.

The concentration change equation in an authentic canal or river is as follows:

$$\frac{\partial c}{\partial t} = K \frac{\partial^2 c}{\partial z^2} - K_{La} H \frac{\partial c}{\partial z} - U \frac{\partial c}{\partial z}$$

where U is the water flow velocity. Needless to say, it is necessary to perform in situ tests to verify the effects of water flow.

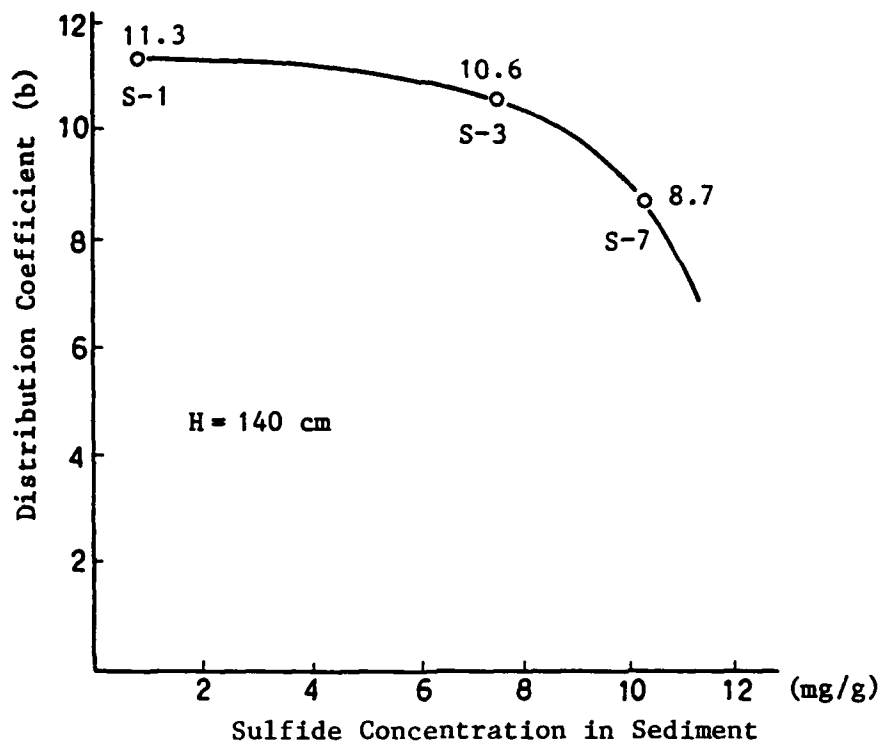


Figure 18. Relationship between sulfide concentration and b

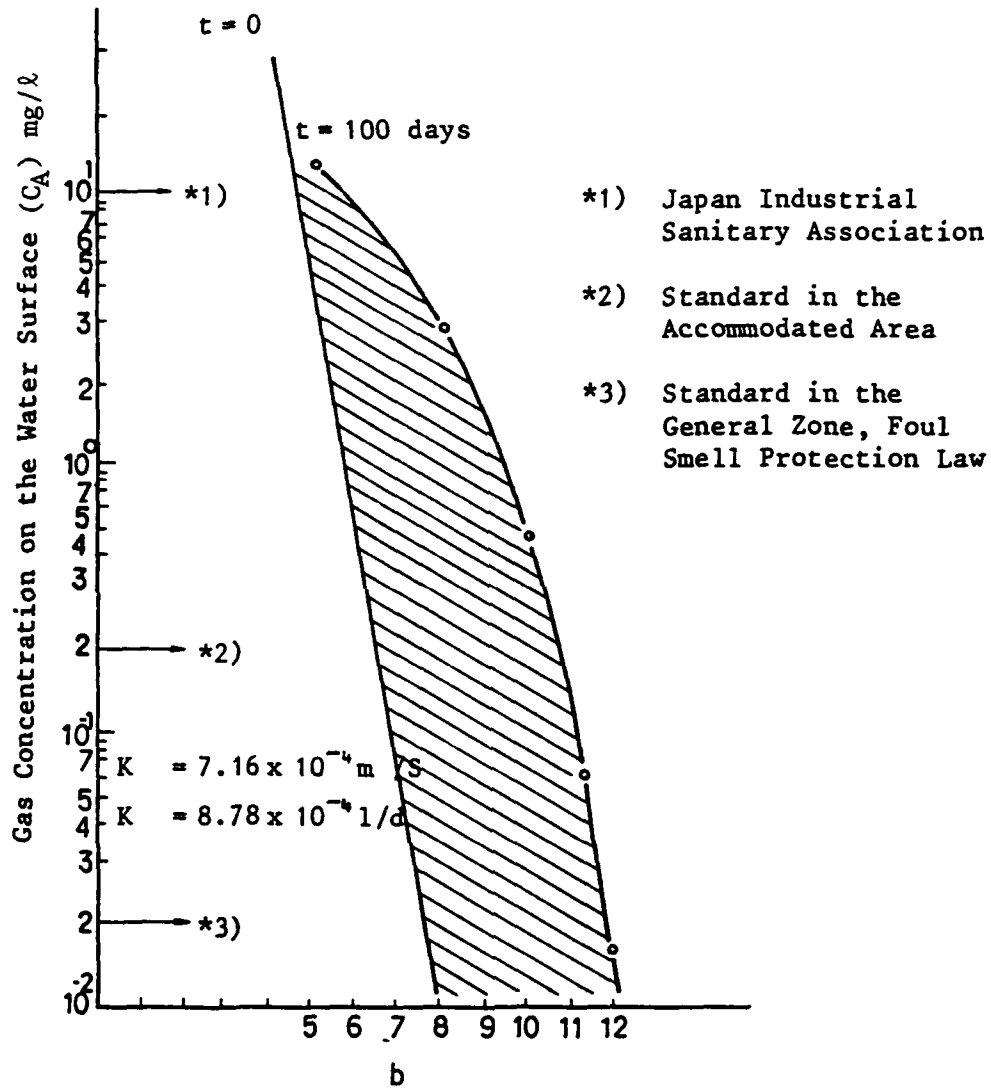


Figure 19. Calculated gas concentrations in the air

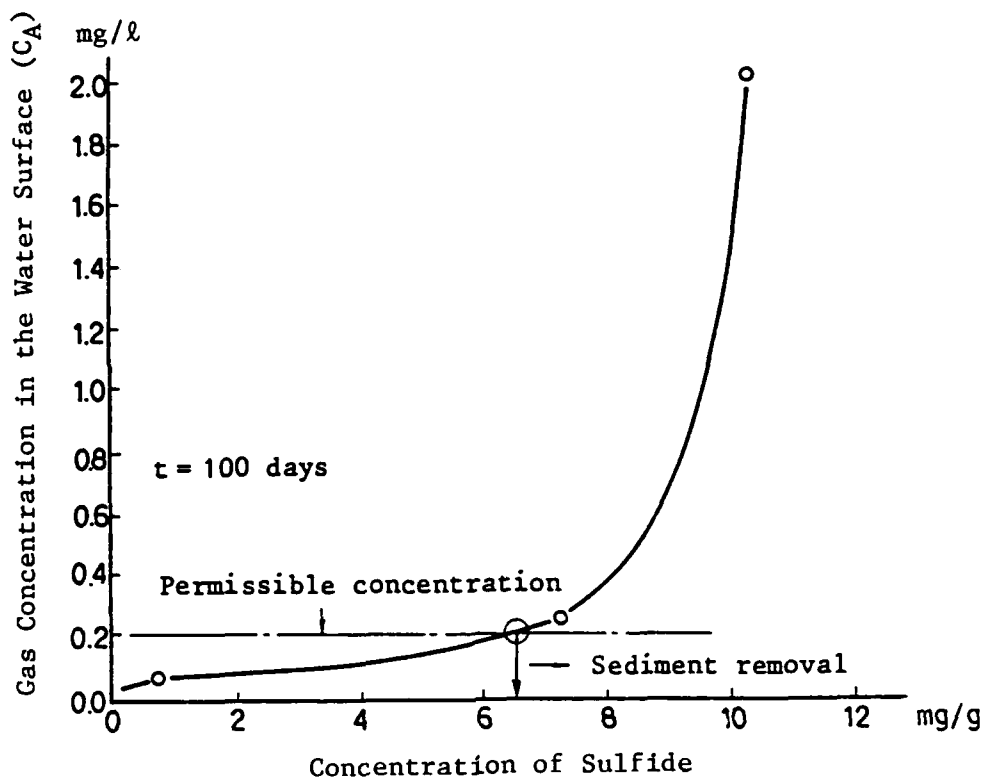


Figure 20. Relationship between sulfide concentrations and C_A

TABLE 4. TOXICITY OF H_2S

Concentration		Effects to Human Body
ppm	mg/l	
0.02	0.00028	Standard in the general zone, Foul Smell Protection Law
0.2	0.0028	Standard in the accommodated area
10	0.014	Japan Industrial Sanitary Association
70-100	0.1-0.15	Symptomatic after long hours
100-150	0.14-0.21	Symptomatic after several hours
170-260	0.24-0.36	
360-500	0.4-0.7	Absorption during 0.5-1.0 hr is dangerous
420-600	0.6-0.87	Sudden death after 0.5-1.0 hr
850-1000	1.2-2.8	Instantaneous death
1000-3000	1.4-4.2	Instantaneous death

DEALING WITH TOXIC POLLUTED SEDIMENTS IN THE GREAT LAKES BASIN

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AD-P006 908

ABSTRACT

The United States and Canada signed the Great Lakes Water Quality Agreement in 1972. In 1978, the Agreement was expanded to cover pollution from toxic chemicals. Since 1973 the Great Lakes Water Quality Board has identified specific areas such as harbors, river mouths, and connecting channels as geographic areas of concern. Now the major problem in the majority of these areas is in-place polluted sediments. The Water Quality Board instituted a new ranking system for these areas in 1985. The purpose of the new system is to encourage progress and to allow the Board to report the status of investigatory and remedial activities accurately. The United States and Canada have banned production and use of certain toxic compounds such as DDT, dieldrin, and PCB's. The concentrations of these chemicals in fish have decreased over the last several years. Now the concentrations seem to be leveling off, probably due to the continuing reservoir in the sediments. The highest concentrations of PCB's, up to 520,000 ppm, are found in Waukegan Harbor. The design of the proposed remedial action, under the United States Superfund authority, has been completed for Waukegan.

INTRODUCTION

This paper examines the binational approach of the United States and Canada of dealing with the problem of in-place polluted sediments. It will articulate the history of the designation of some 42 areas of concern (AOC's) in the Great Lakes Basin (Figure 1) and outline the scheme that was developed by the Great Lakes Water Quality Board to track progress and provide incentives to the states and the Province of Ontario to deal with their polluted sediment problems. The paper will also discuss the environmental condition of Lake Michigan as an example of the impact of in-place polluted sediment on the Great Lakes Basin. As an example, the paper will review the in-place polluted sediment problem found in Waukegan Harbor. Waukegan Harbor has the highest sediment concentrations of polychlorinated biphenyls (PCB's) in the Basin



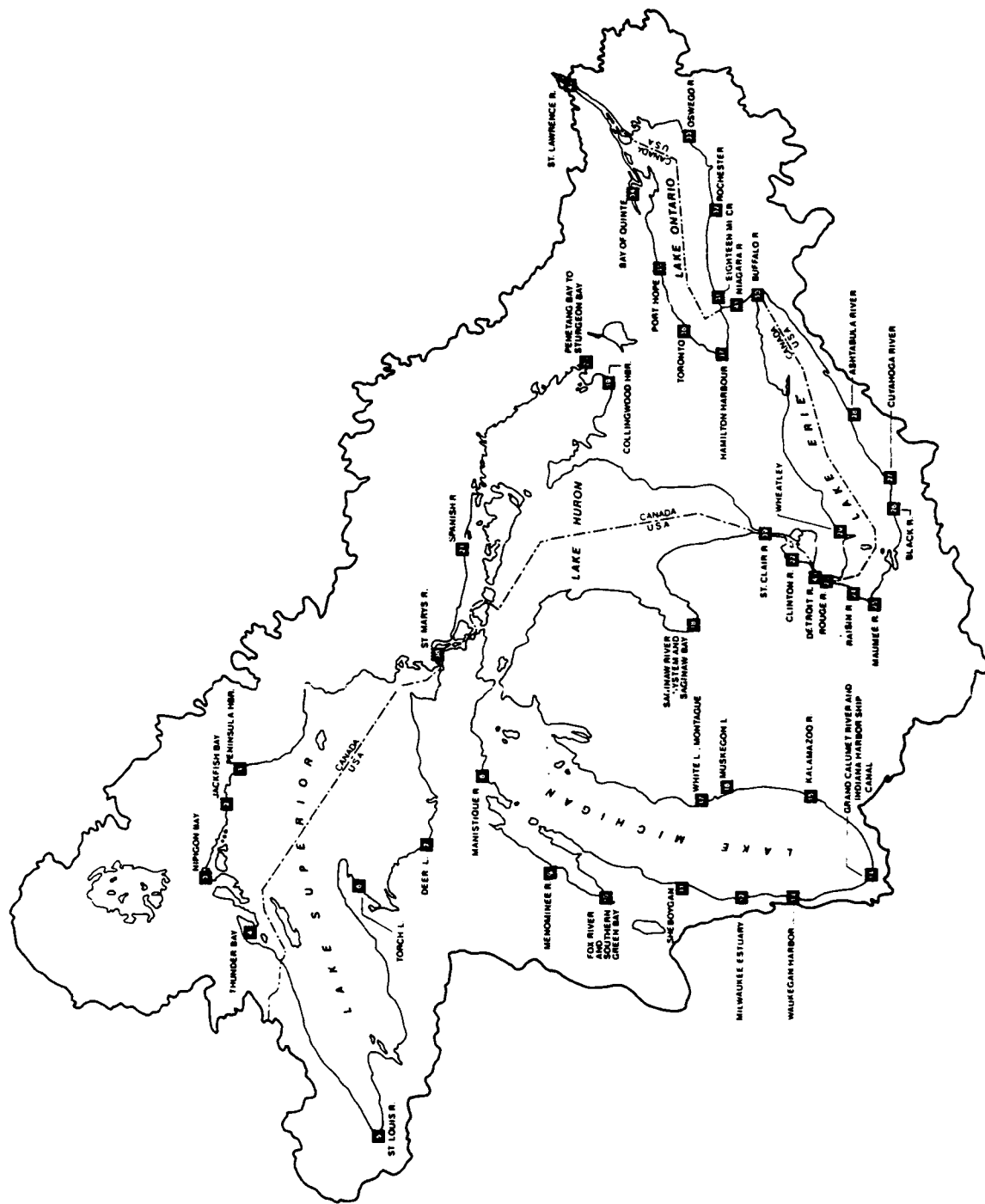


Figure 1. Areas of concern in the Great Lakes Basin

with concentrations of up to 520,000 ppm. The paper will outline the alternatives that have been considered and selected for remediation of the problem in Waukegan Harbor.

In 1909 the Boundary Waters Treaty was signed by the United States and Canada. Although concerned with water levels management, the Treaty provided that neither party could pollute the waters of the other. The International Joint Commission (IJC) was created by the Treaty to foster international cooperation in research, planning, and resource protection and to resolve disputes arising under the Treaty.

Since 1973 the IJC's Great Lakes Water Quality Board has identified specific areas such as harbors, river mouths, and connecting channels which have serious water pollution problems. In 1974, the Board identified 23 such areas in the upper lakes and 46 in the lower lakes. From 1976 to 1977, the Board continued to report on the problems of these AOC's. The 1978 Board report suggested that these areas be classified into three categories: (1) water quality objectives not achieved because remedial programs were not yet complete; (2) remedial programs completed but delayed before conditions in the lake showed improvement; and (3) further remedial programs may be required.

A total of 48 problem areas were identified with these new criteria. Of these areas, 26 were classified in Category 1, 6 in Category 2, and 16 in Category 3. In the intervening years between 1978 and 1985, the Water Quality Board (WQB) struggled with varying ways to report on progress in the AOC's. However, in preparing for the 1985 Board report, the Board was distressed because these areas had been AOC's for over a decade; yet, in almost every case, little or no measurable progress had been made toward the restoration of beneficial uses.

In order to overcome these difficulties and to provide a more complete and coordinated assessment of the data and information provided by the jurisdictions, the Board developed a new procedure for assessment of AOC's. In the past, the Board had not always been clear on how to track and measure progress in AOC's or how to remove the site from the AOC list. To alleviate these problems, the Board adopted a new system of categories which represents a logical sequence for problem solving and resolution. The categories identify the status of the information base, programs which are under way to fill an information gap, and status of remedial efforts. The heart of the concept is the completion of remedial action plans. These plans are being developed to address specific problems causing use impairments. Resolution of the problems takes place when evidence can be presented that the full complement of uses has been restored and the site can be removed from the AOC list.

Specifically, the appropriate jurisdiction will now describe each of the AOC's in relation to the following six-category sequence:

1. Causative factors are unknown and there is no investigative program under way to identify causes.
2. Causative factors are unknown and an investigative program is under way to identify cause.

3. Causative factors are known, but remedial action plan has not yet been developed and remedial measures are not fully implemented.
4. Causative factors are known, remedial action plan has been developed, but remedial measures are not implemented.
5. Causative factors are known, remedial plan has been developed, and all remedial measures identified in the plan have been implemented.
6. Confirmation that uses have been restored and deletion as an AOC.

In Category 5, the effectiveness of remedial measures should be evaluated within a reasonable period of time, say 3 years, to determine whether or not the measures were sufficient to restore beneficial uses in the AOC. If not, it will be necessary for the responsible state or province to either determine additional remedial measures that will need to be implemented or indicate the level of environmental quality expected in the AOC should further remedial actions be adopted.

This new scoring system of the AOC's serves two purposes. First, it allows the WQB to report on progress in AOC's. Secondly, and of equal importance, it provides an incentive for the states and the Province of Ontario to move forward with the completion of remedial action plans and the implementation of programs to restore uses.

The WQB asked each of the states and the province to evaluate their AOC's related to the six categories. The WQB asked these jurisdictions to then designate each of the AOC's, the category the area was currently in, and the time frames required to complete the remedial action plans and submit them to the WQB.

In order to continue to provide incentives to the states and the province, the WQB did two additional things. First, they developed a generic outline that articulates the components of remedial action plans, and, secondly, they developed a review procedure by which the Board would review and comment on the plans once they were submitted.

In all but four of the AOC's, surveillance and monitoring have found concentrations of one or more toxic substances which exceed guidelines or objectives for the protection of aquatic life, human health, or even open-water disposal of dredged material. In some cases the reported levels only exceed objectives and guidelines on a periodic rather than chronic basis; this suggests improving conditions. Fish advisories related to consumption for the protection of human health are in effect in 31 of the 42 AOC's.

The most common source of continuing water quality problems in AOC's is in-place pollutants, in many cases attributable to past waste discharges or spills. According to EPA Great Lakes sediments guidelines, sediments in 38 of the 42 AOC's are moderately to heavily contaminated, mainly with toxic substances. In some cases, the contaminated sediments have led to the destruction of local aquatic communities and are contributing to the high contaminant levels observed in other environmental media. Where dredging for navigation is required, it is often necessary to place restrictions on dredging-related activities. Where such situations arise, special precautions with disposal

must be taken. In some cases, the sediments are so contaminated that they must be treated as toxic or hazardous waste. In many of the AOC's, the sediments the Corps of Engineers dredges must be disposed of in confined disposal facilities. The siting of confined disposal facilities has become more difficult due to increased public and Agency concerns over the suitability of these facilities for sediments contaminated with toxics.

As EPA requires increasingly effective physical and biological isolation of the pollutants in the sediments from the Great Lakes aquatic system, the cost differential of confined disposal facilities vis-a-vis upland disposal will continue to diminish.

The Corps of Engineers is undertaking a number of special studies with EPA assistance to look at alternatives for disposing of heavily contaminated sediments and to identify and quantify bioaccumulation and transfer of pollutants to the aquatic and terrestrial food chains. Two of these studies are being undertaken in Indiana Harbor, an AOC, and in the Buffalo River area of Buffalo, New York.

One of the significant shortfalls in the US environmental legislative authority and funding support is direct assistance for the removal and treatment of in-place polluted sediments. Four of the 28 US AOC's are currently designated on the national priority list for Superfund action; they are Waukegan Harbor, Ashtabula River, Torch Lake, and Sheyboygan Harbor. In these cases there are Superfund sites eligible for Federal cleanup, not because of bioaccumulation into the food chain, which is the most significant problem caused by in-place polluted sediments, but rather because there are existing drinking water intakes located within a prescribed distance from polluted sediments. In addition, the Corps of Engineers undertakes routine navigational maintenance dredging. However, this dredging oftentimes does not deal with the vast quantities of contaminated sediments that lie outside the shipping channel, or in areas that are not commercially navigable.

CONTAMINANTS IN THE BIOTA

Using Lake Michigan as an example, the concentrations of some toxic organic chemicals in Great Lakes fish have declined over the last decade. Despite this decline, fish advisories exist for all or parts of each of the Great Lakes. These advisories warn against consumption of certain species and recommend limiting the consumption of others. The chemicals that are currently driving both the banning of commercial fishing and the advisories related to sports fishing are primarily pesticides such as DDT and dieldrin and industrial compounds such as PCB's. The reduction of these chemicals in Great Lakes fish can be directly attributed to the fact that both the United States and Canada have banned the production and use of these chemicals in the early 1970's. It is now hypothesized, however, that the concentrations of these chemicals may be approaching a new equilibrium in the biota. This equilibrium could be caused by the existence of continuing reservoirs of the chemicals in the sediments as well as atmospheric sources. If it is true that we are at a new equilibrium and that the fish concentrations will remain near existing levels, the fishery will not return to an edible state unless sources such as in-place polluted sediments are addressed.

The 1985 WQB report on Great Lakes water quality reported on trends of contaminants in the biota by lake. Lake Michigan will be used as an example here. There is now a consistent health advisory for all Lake Michigan fish for all states. This health advisory on Lake Michigan fish is based on 1984 data and was released in March 1985. In general, fish contaminant monitoring data for Lake Michigan show that levels of PCB's and banned pesticides contaminants are declining. The 1984 data show that the contaminants in coho salmon, rainbow trout, and lake trout less than 51 cm in length have decreased to the point where 90 percent or more of the fish tested do not exceed US Food and Drug Administration (FDA) Action Levels. However, contaminant levels in over 90 percent of the lake trout, carp, and brown trout remain high. It is recommended that no one eat these fish (Table 1).

Examining fish concentration data for bloater chubs, lake trout, and coho salmon, the US Fish and Wildlife Service and EPA have documented substantial declines in both DDT and PCB's from 1969 to 1982. Dieldrin has apparently not responded as rapidly to remedial actions, as evidenced by its apparent increase since 1969. In general, dieldrin concentrations in lake trout increased between 1970 and 1979 followed by a decrease between 1979 and 1982 (Figure 2). Coho salmon data confirm this trend between 1980 and 1984. It is now believed the dieldrin problem is related to deep depositional areas.

The Canadian Government has demonstrated that concentrations of DDE, DDT, HCB, MIREX, and total PCB in herring gull eggs have decreased dramatically in the last decade (Figure 3). As is the case with Lake Michigan fish, dieldrin concentrations in herring gull eggs from Lake Michigan remain relatively high and have not decreased rapidly as a result of remedial measures.

The majority of the fish addressed in Lake Michigan health advisories are species that are not typically residents of a particular harbor or river mouth. Salmon and trout tend to migrate freely throughout the entire basin. Therefore, little detailed evidence can be submitted that would indicate that the polluted sediments in these harbors are causing the continuance of levels of these chemicals in Great Lakes fish. Carp, however, remain on the advisory because they continue to have high levels. It is worth note that carp from areas of contaminated sediments have higher levels, for example, 90 to 100 ppm PCB's in Sheyboygan Harbor.

EPA, in conjunction with the states, is beginning a series of evaluations on localized species, such as spottail shiners. In addition, specific bioaccumulation tests are being conducted in a number of the AOC's to determine the bioavailability of the chemicals to the biota.

WAUKEGAN HARBOR REMEDIAL ACTION

The EPA has attempted to use as many of its authorities and resources as possible in addressing AOC's. Because of the administrative limitations on Superfund, the hazardous waste cleanup authority has been brought to bear in only four of the AOC's. Waukegan Harbor, perhaps the most notorious AOC in the Great Lakes, is one of the most highly PCB-contaminated sites in the United States. The figures are overwhelming; over 300,000 lb of PCB's are contaminating slip 3 in the upper harbor of Waukegan Harbor and over 750,000 lb of PCB's are contaminating portions of the soils of the nearby Outboard Marine Corporation property. Sample results from the harbor sediments

TABLE 1. HEALTH ADVISORY ON LAKE MICHIGAN SPORT FISH*

	Group I	Group II	Group III
Species of fish	Yellow perch Smelt Coho salmon Rainbow trout Lake trout less than 51 cm	Chinook salmon over 64 cm Lake trout 51-64 cm	Brown trout** Lake trout over 64 cm Carp
Contaminants of concern		PCB	PCB, dieldrin, chlordane
Level of contaminants	Lowest	Moderate	Highest
Advice	90% or more of the fish tested in this group met FDA Action Levels Eating these fishes poses the lowest risk of exposure to contaminants	One or more contaminants were found to be above the FDA Action Level in 50% or more of the fish tested in this group Pregnant women who expect to bear children, nursing mothers, and young children should not eat these fish Others should limit their consumption to no more than one meal per week and heed preparation and cooking recommendations	One or more contaminants were found to be above the FDA Action Level in 90% or more of the fish tested in this group No one should eat these fishes

* This advisory is based on the presence of lipid soluble contaminants such as PCB, DDT, dieldrin, chlordane, or toxaphene in edible portion samples. Heavy metal contaminant levels were not significant. Not enough samples of brook trout or chinook salmon less than 64 cm have been collected to adequately establish contaminant levels.

** Brown trout contained primarily PCB and showed a wide range in levels from location to location.

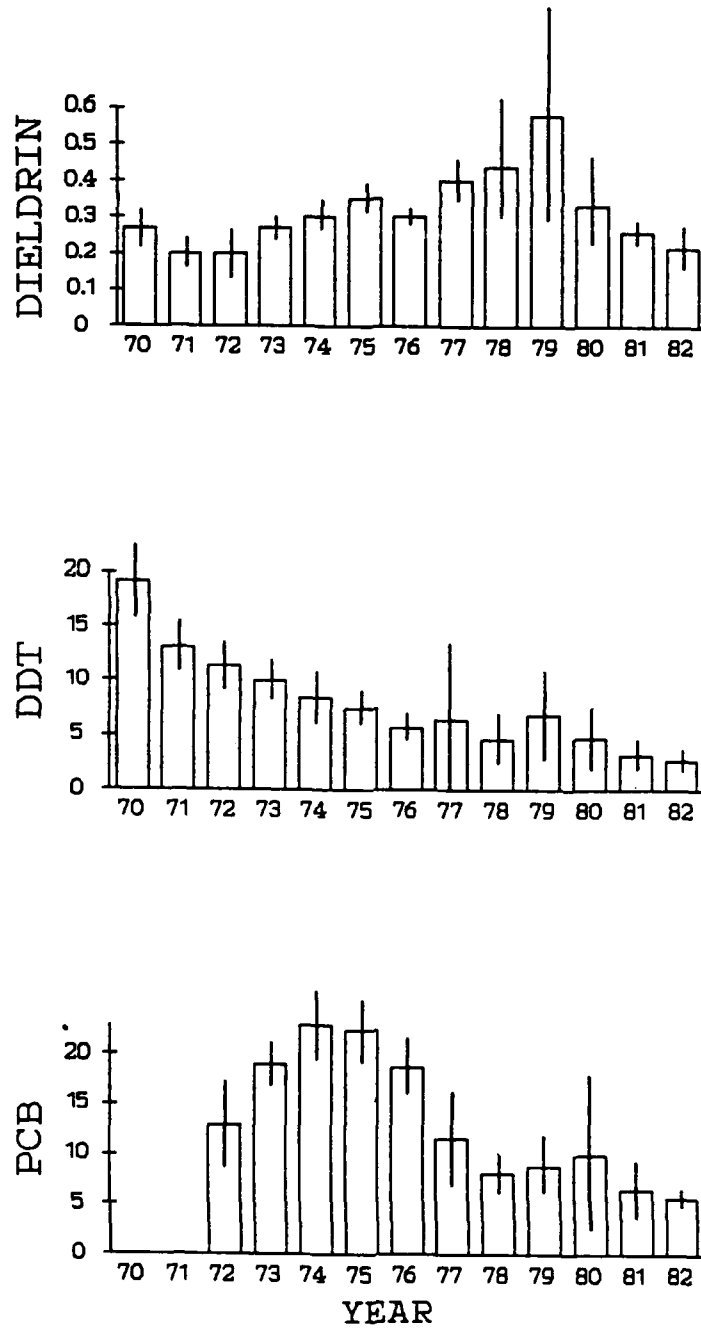


Figure 2. Mean annual dieldrin, total DDT, and PCB concentrations (mg/kg wet weight; with 95% confidence intervals) in whole fish samples of lake trout collected from eastern Lake Michigan. (Data from US Fish and Wildlife Service. From 1972-76 PCB was quantified using 1:1:1 Aroclor 1248, 1254, and 1260; beginning in 1977, only Aroclor 1254)

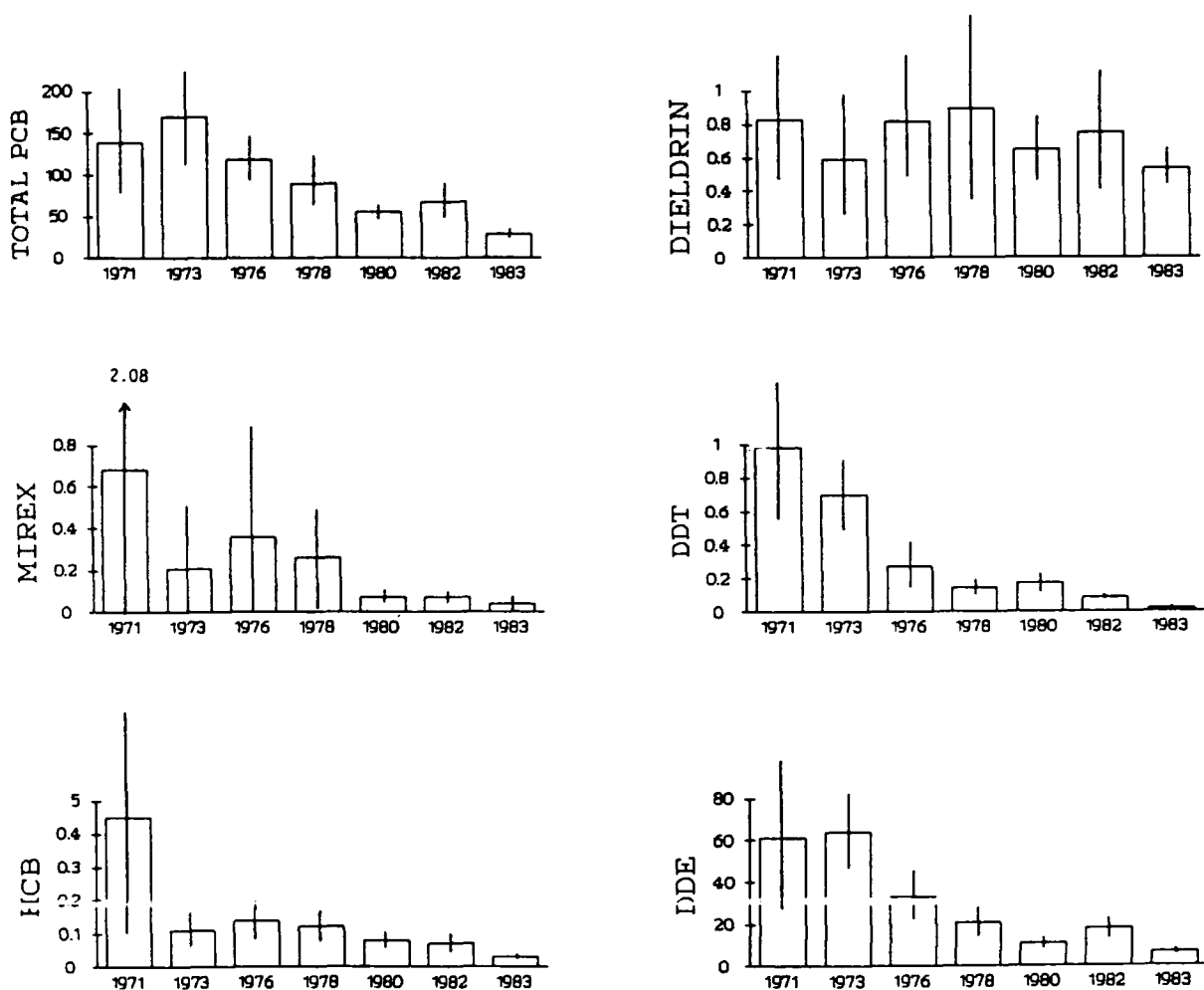


Figure 3. Mean organochlorine contaminant concentrations (mg/kg wet weight \pm S.D.) in herring gull eggs from Lake Michigan's Big Sister Island, 1971-1983. (Data from Canadian Wildlife Service)

have shown PCB levels and concentrations up to 520,000 ppm (52 percent). EPA has estimated that between 30 and 40 lb of PCB's are released to Lake Michigan by water movement annually, while some 12 to 40 lb are released into the atmosphere. There is a ground-water plume migrating toward Lake Michigan that in the future will contribute another 8 lb/year to Lake Michigan proper. The City of Waukegan has an emergency water intake in Waukegan Harbor, and while it has been used only six times since 1978, this intake is the basis for Superfund involvement. EPA fish studies in Waukegan Harbor indicate that levels of PCB's exceed the FDA Action Level of 2 ppm. Samples taken range from 2.7 to 187 ppm.

There is little dispute that this contamination resulted from years of discharge into Waukegan Harbor by the Outboard Marine Corporation (OMC). Between 1952 and 1971, OMC purchased approximately 9 million pounds of PCB for use in its die casting machinery and other equipment. The major portion of the PCB's was used as a cooling fluid in the die cast operation. It is estimated that at least 10 to 15 percent of this material leaked through floor drains into Waukegan Harbor and the North Ditch drainage system.

EPA has attacked this Waukegan Harbor problem from two perspectives, legal and technical. On the legal front, EPA and OMC differed greatly on whether the concentrations of PCB's present a danger and what to do about them. OMC repeatedly asserted that PCB's in and around Waukegan Harbor present no danger. Indeed, OMC is fond of asserting that EPA "admitted" as much when it responded in the course of litigation that it knew of no specific persons affected by PCB contamination from Waukegan Harbor. Fortunately, with the passage of the Clean Water Act, Congress provided EPA the authority to act on chemical exposure via the water route before the fact of demonstrated human injury. EPA must act to protect the public health. PCB's are not natural to the Great Lakes ecosystem. Therefore, evidence of toxicity in animals is sufficient for EPA to take action. The truth is less ingenuous.

Years of Agency experience with PCB's have demonstrated the chronic dangers which PCB's present. PCB's are known to have carcinogenic effects on rodents and to cause chloroacne and reproductive disorders in humans. They are also suspected of other disorders in both humans and animals.

In 1978, the Agency used the only options available to it by commencing judicial action against OMC. Many frustrating years of litigation followed. When the Superfund statute passed, the Agency believed it had a means for accomplishing its goal of a cleanup, which years in court had not achieved. Subsequent to the engineering design for the removal and disposal of the PCB's from Waukegan Harbor, OMC engaged in litigation against EPA, this time over property access issues. The remedial activity, which will be described later, is still in its design phase.

Early in the design process, the Agency requested access to OMC's property for the limited purpose of doing survey work and taking soil borings. OMC refused. Although EPA did not prevail in court on the access issue, now with pending Superfund reauthorization, it appears that this stumbling block will be removed. OMC and EPA have recently met to once again begin negotiations for the express purpose of reaching agreement on how and what should be done in terms of removal.

In 1984, EPA signed a decision document authorizing the expenditure of \$21 million for the design and cleanup of the entire Waukegan site. Although many alternatives were examined for the cleanup, the recommended alternative consists of: (1) dredging of the upper harbor down to 50 ppm; (2) dredging and deep excavation of slip 3; (3) construction of a bypass sewer line through the north ditch; (4) construction of a slurry wall around and a clay cap over both the oval lagoon and the crescent ditch after hot spot (over 10,000 ppm) removal; (5) containment of parking lot contamination by slurry wall; (6) construction of an aboveground containment cell on the parking lot area to hold contaminated materials removed from the upper harbor in slip 3; and (7) fixation and offsite disposal of materials excavated from slip 3, the Crescent Ditch, and the oval lagoon (hot spots). (This material would be disposed of offsite in certified hazardous waste landfills.) The design work began in December 1984 but was put on hold in March 1985 due to the ongoing litigation.

There are two remaining issues related to this example of a Great Lakes polluted harbor that need to be mentioned. The new Superfund reauthorization now pending in Congress makes a strong statement about the ultimate destruction of toxic materials rather than the removal and disposal of toxic

materials. There is a belief that the removal of the toxic material from Superfund sites and the transport by rail or road to disposal facilities increase the risk of human exposure and set up the situation where the disposal facility ultimately will have to be dealt with sometime in the future. If Waukegan Harbor is not resolved quickly, it may come under the auspices of the new Superfund legislation. It is likely that the destruction of the dewatered sediment by an incineration-type method will be carefully considered and perhaps recommended. If this technology is, in fact, chosen, the cost--which in 1984 was \$21 million--will undoubtedly double and likely triple. The second remaining issue relates to the level of cleanup. EPA has estimated that if sediments are removed down to a 50-ppm level, that would virtually eliminate movement between the harbor and the lake. Certainly the vast majority of the exposure would have been removed. It is likely, however, that once the cleanup is completed, tests will have to be run as to the bioavailability of the remaining PCB's in the harbor, and additional steps by either the restriction of movement of the biota or additional cleanup may be required.

CONCLUSIONS

In conclusion, the sediments contaminated with toxic materials in the Great Lakes Basin remain one of the most significant issues in the basin. The WQB has recognized this and has attempted, by establishing a new process of ranking AOC's, to provide the incentive to the Great Lakes States and the Province of Ontario for action. Although scientifically not a certainty, it is projected that until Canada and the United States address in-place sediment problems in harbors and river mouths, it is reasonable to believe that consumption advisories on sport fish and limits on commercial fishing will remain. This would certainly be true in some localized areas.

EPA and the Corps of Engineers have limited authority to deal with this problem. To date our Superfund program has been able to deal with only 4 of the 28 US AOC's and, as demonstrated by Waukegan Harbor, this is a time-consuming and costly process.

Both the US House of Representatives and the Senate have reauthorized the Clean Water Act. This bill recognizes the need to address in-place polluted sediments. They have authorized EPA to undertake a 5-year demonstration program for the removal, treatment, and disposal of in-place sediments in Great Lakes AOC's. If funds are appropriated for this activity, EPA will work with the US Army Corps of Engineers to continue to develop technical approaches for dealing with in-place polluted sediments. Ultimately, however, either the Federal Government or the States will have to bear the burden of the costs of the removal and destruction of polluted sediments.

EVALUATION OF CHEMICAL POLLUTANTS BY ANALYSIS OF OIL
COMPONENTS IN SEDIMENTS

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ABSTRACT

Partitioning of oil and chemical substances in the bottom sediments off the coast of Yokohama was investigated to study the behavior of chemical substances in the environment. Concentrations of lipophilic substances were proportional to concentrations of oil in the sediments. From these results we could characterize the sediment contamination of the area.

INTRODUCTION

As a direct consequence of the PCB environmental pollution issue, the "Law Concerning the Examination and Regulation of Manufacture, etc., of Chemical Substances" was enacted in October 1973 in Japan. As a part of the Chemical Assessment Project, the Environment Agency of Japan systematically examines the safety of chemical substances in the environment, and estimates contamination levels in the environment (Environment Agency 1982).

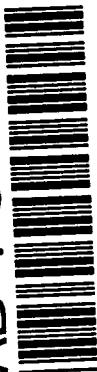
Lipophilic substances such as PCBs and chlordanes, which are nearly insoluble in water, tend to be accumulated in the fat of fish (Japanese Society of Public Health 1984). Therefore, both the concentrations of organochlorine compounds and the content of fat can be determined simultaneously in analyses of fish (Environment Agency 1982). Previous laboratory experiments revealed that about 90 percent of hexachlorobiphenyl in water was extracted into suspended oil and sediment after efficient mixing (Sayler and Colwell 1976) and that both particle-size distribution and total organic carbon in bottom sediments are important factors in determining the extent of PCB partitioning (Steen, Paris, and Baughman 1978). Similarly, lipophilic substances in the environment should mix with oil since oil has analogous physicochemical properties.

This paper will report on an investigation of the relationship between concentrations of oil and chemical substances in the bottom sediments off the coast of Yokohama (Figure 1), the western part of Tokyo Bay (Ninomiya et al. 1984).

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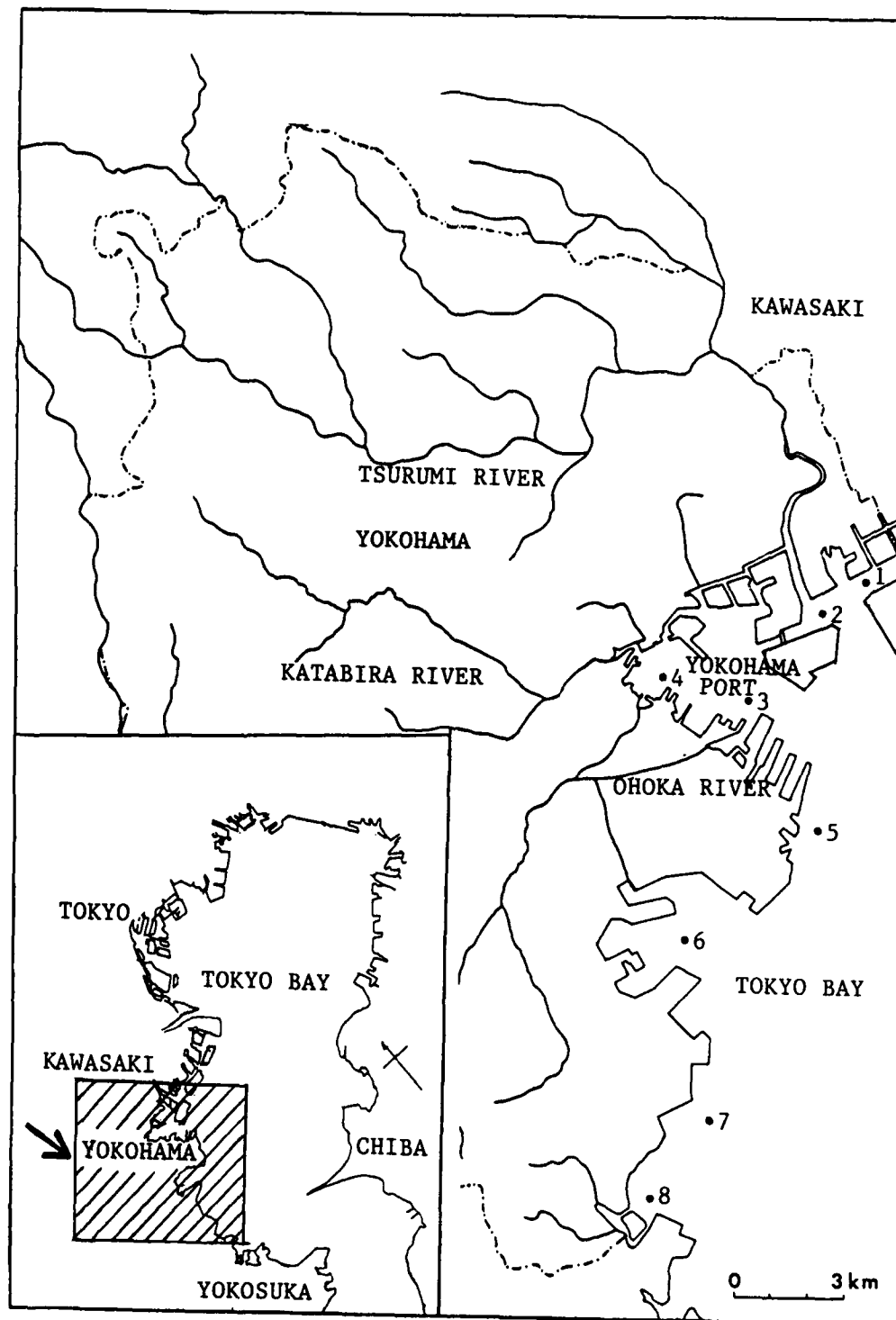


Figure 1. Sampling sites along the coast of Yokohama

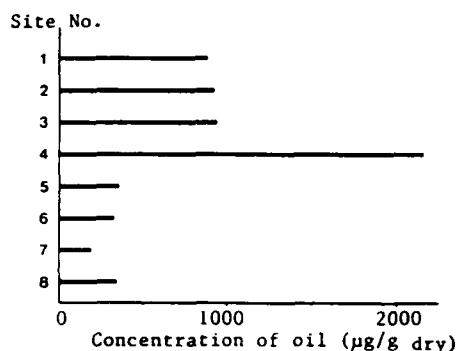
SEDIMENT CONTAMINATION IN YOKOHAMA WATERS

Concentration of Oil in the Bottom Sediments

Coast of Yokohama

After sediment samples were collected at eight sites along the coast of Yokohama (Figure 1) and dried at room temperature, oil was extracted using carbon tetrachloride. The concentrations of oil are shown in Figure 2.

Figure 2. Concentration of oil in sediments off the coast of Yokohama



The total concentration of oil was higher in the northern coast (sites 1-4) because of effluents from the factories located in Keihin Industrial Area, established in the 1930's. The concentration of oil was highest at site 4 (Yokohama Port), in which the current is stagnant and the discharge from many ships and tankers is difficult to disperse (Ninomiya 1982).

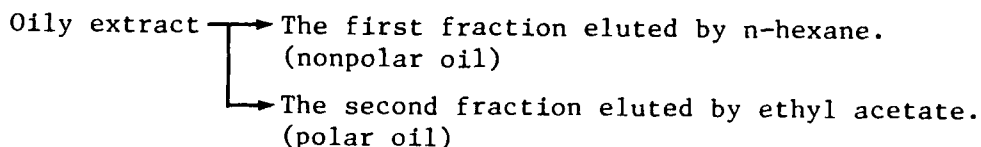
Tsurumi River

Tsurumi River, which runs through northern Yokohama, is about 42 km in length and flows into Tokyo Bay. Sediment samples were collected at 20 sites along Tsurumi River. The sites were located within 29 km of the mouth of the river (Figure 3). In this experiment, ethyl acetate was used as extractant. The concentrations of oil are shown in Figure 4.

The concentration of oil was higher at the sites located between the mouth of the river and about 6 km upstream. The maximum concentration was measured at site 5, about 4 km from the mouth of the river (Ninomiya 1983).

Nonpolar Oil as a Characterization of Oily Pollutants

By fractionating the oily extract of the sediment from Tsurumi River using chromatography on silica gel, oil was separated into two components, nonpolar and polar. The percentage of the nonpolar component in the total oil is shown in Figure 5.



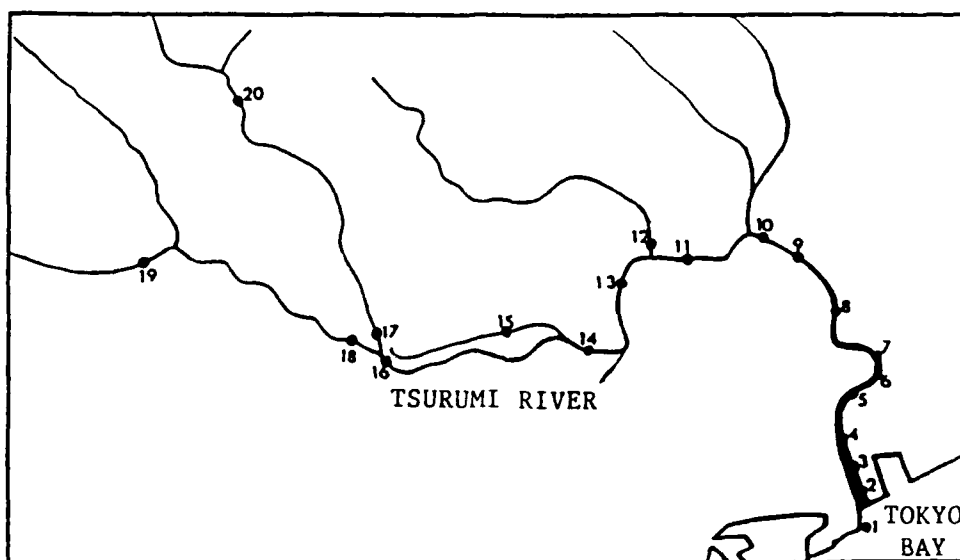


Figure 3. Sampling sites along Tsurumi River

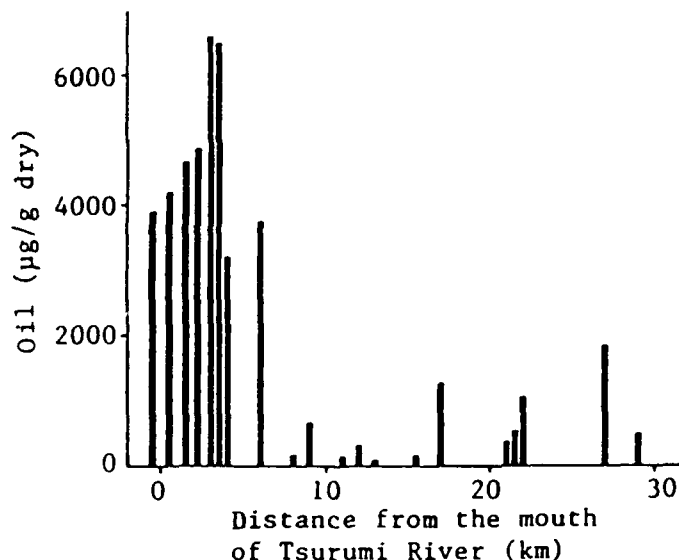


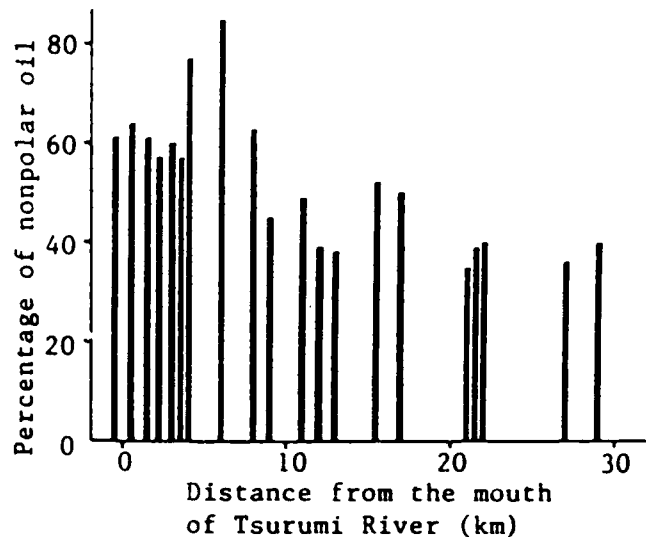
Figure 4. Concentration of oil in sediments in Tsurumi River

The higher percentage of polar oil at the middle and upstream sites is suspected to be caused by the inflow of domestic effluents and the benthic organisms, while the percentage of nonpolar oil at the downstream sites might come from the effluents containing petroleum from industrial and urban areas. The extraordinarily high percentage of nonpolar oil, as high as 85 percent, at site 8 should be explained by the local pollution of the river due to petroleum-containing effluent (Ninomiya 1983).

OUTLINE OF ENVIRONMENTAL SURVEY OF CHEMICAL SUBSTANCES

We have investigated the levels of contamination by chemical substances in water and bottom sediments since 1976 in cooperation with the program

Figure 5. Percentage of nonpolar oil in total oil sediments in Tsurumi River



planned by the Environment Agency. Samples were collected at three sites each in the mouth of Tsurumi River (T) and in Yokohama Port (Y) of Tokyo Bay, as shown in Figure 6.

The drainage basin of Tsurumi River in Yokohama is about 143 km², with a population of 800,000. Katabira River and Ohoka River, which run into the bay of Yokohama Port, drain a total area of 151 km². Approximately 1,060,000 people live in the drainage basin of these two rivers.

From 1976 to 1985, field surveys were conducted on 161 substances, and 44 of the 161 substances were detected. Reflecting the tendency of accumulation in the bottom sediments, a larger number of substances was detected in bottom sediment than in water, as shown in Table 1.

Figure 6. Sampling sites in the mouth of Tsurumi River and Yokohama Port

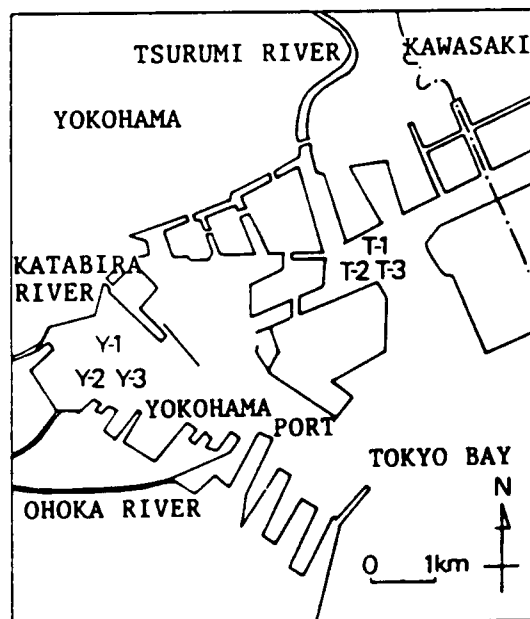


TABLE 1. SUBSTANCES DETECTED, DETECTION RANGE, AND DETECTION RATE IN YOKOHAMA, 1976-1985

Year	Substance	Sample	Detection Range*	Detection Rate
1976	Aniline	Water	0.00011-0.0011	20/20
	Aniline	Sediment	0.0016-0.50	20/20
	m-Chloroaniline	Water	0.00021-0.00034	5/20
	m-Chloroaniline	Sediment	0.0005-0.019	13/20
	p-Chloroaniline	Sediment	0.001-0.27	17/20
	o-Chloroaniline	Sediment	0.001-0.008	9/20
	3,4-Dichloroaniline	Water	0.00032-0.00042	3/20
	3,4-Dichloroaniline	Sediment	0.0077-0.49	14/20
	2,4-Dichloroaniline	Sediment	0.005-0.0058	2/20
	N-Ethylaniline	Sediment	0.0083-0.037	6/20
	p-Toluidine	Sediment	0.0038-0.046	9/20
	o-Toluidine	Sediment	0.011-0.046	8/20
	m-Toluidine	Sediment	0.0023-0.038	12/20
	p-Anisidine	Sediment	0.0017-0.0055	4/20
	o-Anisidine	Sediment	0.0083-0.057	6/20
	2,3-Xylidine	Sediment	0.028-0.09	4/20
	3,5-Xylidine	Sediment	0.0022	1/20
	PCN	Sediment	0.34-0.67	3/3
	PCT	Sediment	0.65	1/3
2-Naphthylamine	Sediment	0.027	1/20	
1977	Tributhyl phosphate	Water	0.00003-0.00005	6/6
	Tributhyl phosphate	Sediment	0.0026-0.0074	6/6
	Nitrobenzene	Water	0.0007-0.001	2/6
1978	PCT	Sediment	0.14-0.30	3/6
	HCB	Sediment	0.0016-0.0041	4/4
	Polyoxyethylene-alkylphenylether	Sediment	8.6-29	6/6
1979	1,2,4-Trichlorobenzene	Sediment	0.0092-0.03	5/6
	Ethylenediaminetetraacetic acid	Sediment	2.3-13	5/6
	Organosilicone compounds	Sediment	4.2-12.2	5/6
	Chlorinated paraffins	Sediment	1.7-7.4	6/6
1980	Organosilicone compounds	Sediment	6-22	6/6
	Chlorinated paraffins	Sediment	1.4-8.5	6/6
	N-Phenyl-1-naphthylamine	Sediment	0.03-0.04	5/6
	N-Phenyl-2-naphthylamine	Sediment	0.02-0.03	3/6
1981	N-Phenyl-2-naphthylamine	Sediment	0.006-0.016	4/6
	Hexachlorophene	Sediment	0.005-0.098	6/6

(Continued)

* Units: $\mu\text{g/ml}$ for water, $\mu\text{g/g}$ dry basis for sediment.

TABLE 1. (Concluded)

Year	Substance	Sample	Detection Range*	Detection Rate	
1982	trans-Chlordane	Sediment	0.0008-0.0064	5/6	
	cis-Chlordane	Sediment	0.0005-0.004	5/6	
	trans-Nonachlor	Sediment	0.0005-0.007	5/6	
	cis-Nonachlor	Sediment	0.001-0.0031	4/6	
	Heptachlor	Sediment	0.0012	1/6	
	γ-Chlordene	Sediment	0.0003-0.001	2/6	
	Hexachlorophene	Sediment	0.009-0.17	6/6	
	Di-n-butyl phthalate	Water	0.00006-0.0001	4/6	
	Di-n-butyl phthalate	Sediment	0.068	1/6	
	Di-n-heptyl phthalate	Sediment	0.071-0.24	5/6	
	Di(2-ethylhexyl)phthalate	Sediment	0.069-3.5	6/6	
	Polyoxyethylene-type nonionic surfactants	Sediment	0.16-2.0	6/6	
	Polyoxyethylene alkylether	Sediment	0.27-1.0	5/6	
	Polyoxyethylene alkylphenylether	Sediment	3.5-4.9	4/6	
	1983	Alkyldimethylbenzyl- ammonium chloride	Sediment	0.9-5.2	6/6
		Fluorene	Sediment	0.03-0.09	6/6
Diphenylmethane		Sediment	0.02-0.16	5/6	
1984	Acenaphthylene	Sediment	0.67	1/6	
1985	1,3-Dimethyl naphthalene	Sediment	0.062	1/6	

PARTITIONING OF OIL AND CHEMICAL SUBSTANCES

Thirteen substances were detected in bottom sediments in 1982 (Table 1). The detection rate (number of detected samples/total number of samples) was more than 5/6 for eight substances as shown in Table 1. They were trans-chlordane, cis-chlordane, trans-nonachlor, di(2-ethylhexyl)phthalate, di-n-heptyl phthalate, polyoxyethylene-type nonionic surfactants, polyoxyethylene alkylether, and hexachlorophene (Table 2).

This study was aimed at examining concentrations of chemical substances in the environment. Concentrations of the substances in the bottom sediments are presented in Table 3 (Ninomiya et al. 1984).

Check on Local Contamination of Oily Discharge

Generally, oil is composed primarily of fatty matter (polar oil) from animal and vegetable sources and of hydrocarbons (nonpolar oil) of petroleum origin. Nonpolar oil percentage increases as local contamination by oil discharge of petroleum origin increases. The percentage of nonpolar oil at all sites was compared as shown in Table 4. The average percentage of nonpolar oil was 59 percent for the samples from the mouth of Tsurumi River and 70 percent for the samples from Yokohama Port. The percentage for the port was higher than the percentage for the river.

Figure 7 shows the concentration of polar and nonpolar oil as a function of the concentration of total oil for Yokohama Port and Tsurumi River. The total oil versus polar oil plot can be approximated by a single straight line, while the total oil versus nonpolar oil plot consists of two lines of similar slopes but unequal intercepts corresponding to the two areas. The set of three points in Yokohama Port consists of a straight line that is parallel to but higher than the line from the set of three points in the mouth of Tsurumi River. The long-term accumulation of mineral oil from the ship discharges is believed to be the cause of a remarkably high percentage of nonpolar oil in the sediment of Yokohama Port. Accordingly, there was no sample that was locally contaminated by oily discharge in either area.

Relationship Between Oil and Chemical Substances

To examine the contamination of the two areas of Tokyo Bay, the concentrations of the chemical substances were plotted against the concentration of total oil (Figure 8). As shown in Figure 8, the total oil versus chemical substances plots consist of two lines of similar slopes but unequal intercepts corresponding to the two areas, except for polyoxyethylene alkylether and hexachlorophene.

Lipophilic chemical substances such as chlordanes dissolve into oil on their way of running off in drainage basin, because the chemical substances and oil are analogous in physicochemical properties (Taylor and Colwell 1976). And the oil containing the chemicals has accumulated in sediment at mouth of river and off coast. Therefore, the concentration of chemical substance in bottom sediments should be proportional to that of oil. Although there are only a few samples, the total oil versus chemical substances plots can be

TABLE 2. WATER SOLUBILITY, USE, AND AMOUNT OF PRODUCTION OR IMPORT OF THE EIGHT SUBSTANCES DETECTED IN 1982

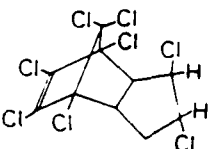
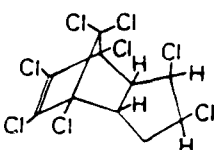
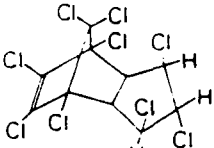
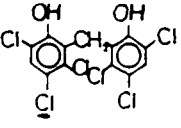
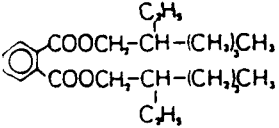
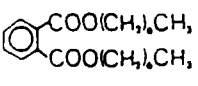
<p>trans-Chlordane</p> 	<p>Water solubility : 9.45×10^{-2} $\mu\text{g/ml}$</p> <p>Use : for termites, wood and plywood</p> <p>Amount of import : 1,320 tons as chlordanes</p>
<p>cis-Chlordane</p> 	<p>Water solubility : 6.60×10^{-2} $\mu\text{g/ml}$</p> <p>Use : for termites, wood and plywood</p> <p>Amount of import : 1,320 tons as chlordanes in 1981</p>
<p>trans-Nonachlor</p> 	<p>Water solubility : 5.66×10^{-2} $\mu\text{g/ml}$</p> <p>Use : for termites, wood and plywood</p> <p>Amount of import : 1,320 tons as chlordanes in 1981</p>
<p>Hexachlorophene</p> 	<p>Water solubility : Sparingly soluble</p> <p>Use : In past as bathroom agent and industrial antiseptic, now as disinfectant</p> <p>Amount of import : A ton in 1981</p>
<p>Di(2-ethylhexyl)phthalate</p> 	<p>Water solubility : 6-18 $\mu\text{g/ml}$</p> <p>Use : Plasticizing agent for vinyl chloride products</p> <p>Amount of production : 223,243 tons in 1981</p>
<p>Di-n-heptyl phthalate</p> 	<p>Water solubility : 20 $\mu\text{g/ml}$</p> <p>Use : Plasticizing agents for leather, hose, and paste</p> <p>Amount of production : 36,175 tons in 1981</p>
<p>Polyoxyethylene-type nonionic surfactants</p>	<p>Water solubility : Well soluble</p> <p>Use : Detergents and permeative agents, etc.</p> <p>Amount of production : 146,233 tons in 1981</p>
<p>Polyoxyethylene alkylether</p> <p>$\text{RO}(\text{C}_2\text{H}_4\text{O})_n\text{H}$</p>	<p>Water solubility : Well soluble</p> <p>Use : Detergents, etc.</p> <p>Amount of production : 146,233 tons as polyoxyethylene-type surfactants in 1981</p>

TABLE 3. BOTTOM SEDIMENT DATA FROM THE MOUTH OF TSURUMI RIVER AND YOKOHAMA PORT, 1982 ($\mu\text{g/g}$ DRY BASIS)

Site	Water Content	Ignition Loss	Total Oil	Nonpolar Oil	Polar Oil
T-1	43	5	414	198	147
T-2	69	14	3880	1790	1150
T-3	66	16	823	388	261
Y-1	81	13	5650	3700	1530
Y-2	73	16	5530	3310	1520
Y-3	73	12	4270	2710	1100

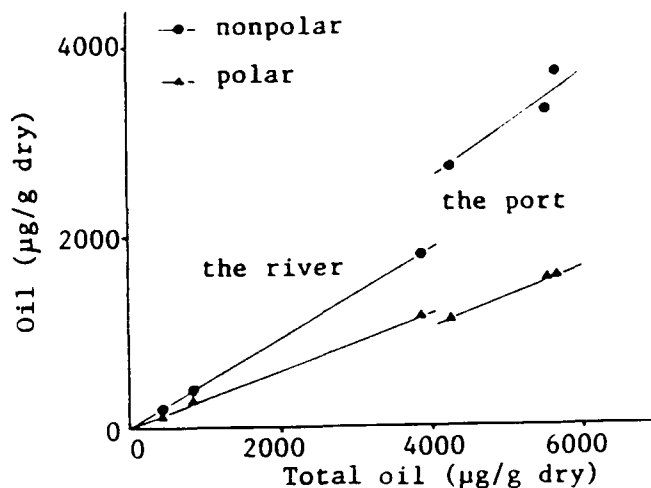
Site	Total Carbon	Total Nitrogen	trans-Chlordane	cis-Chlordane	trans-Nonachlor
T-1	0.58	0.058	0.0002	0.0002	0.0002
T-2	4.0	0.44	0.012	0.0067	0.0070
T-3	1.4	0.12	0.0008	0.0005	0.0005
Y-1	4.5	0.41	0.0064	0.0040	0.0054
Y-2	4.6	0.34	0.0060	0.0038	0.0061
Y-3	3.4	0.27	0.0022	0.0013	0.0010

Site	Di(2-ethylhexyl) phthalate	Di-n-heptyl phthalate	Polyoxy-ethylene nonionic surfactants	Polyoxy-ethylene alkylether	Hexachlorophene
T-1	0.069	0.012	0.16	0.2	0.009
T-2	3.5	0.23	1.7	1.0	0.050
T-3	0.36	0.071	0.54	0.5	0.016
Y-1	2.5	0.084	1.5	0.27	0.17
Y-2	3.2	0.24	2.0	0.35	0.16
Y-3	0.88	0.11	0.99	0.41	0.068

TABLE 4. PERCENTAGE OF NONPOLAR OIL IN TOTAL OIL

	Site					
	T-1	T-2	T-3	Y-1	Y-2	Y-3
Percentage	57	61	60	71	69	71

Figure 7. Plot of concentrations of nonpolar and polar oil versus concentration of total oil



approximated by two straight lines for each area as shown by the following formula:

$$Y = A(X - B)$$

where

Y = concentration of chemical substance

A = slope of line

X = concentration of oil

B = x-axis intercept of line

With respect to the substances except polyoxyethylene alkylether and hexachlorophene, A for the mouth of Tsurumi River was nearly equal to A for Yokohama Port. However, B for the mouth of the river was much closer to the origin of the plot than B for the port. Chemical substances such as chlordanes in sediment are mostly derived from urban runoff (Fukushima 1986). On the other hand, oily pollutants come from two different sources; urban runoff and ship discharge. Oil from ships accumulates in sediment off coast, soon after it discharges. Therefore, it contains far less amounts of chemical substances than oil from urban runoff. As a considerable portion of oil in the port comes from ship discharge, B for the port should be greater value than B for the mouth of the river.

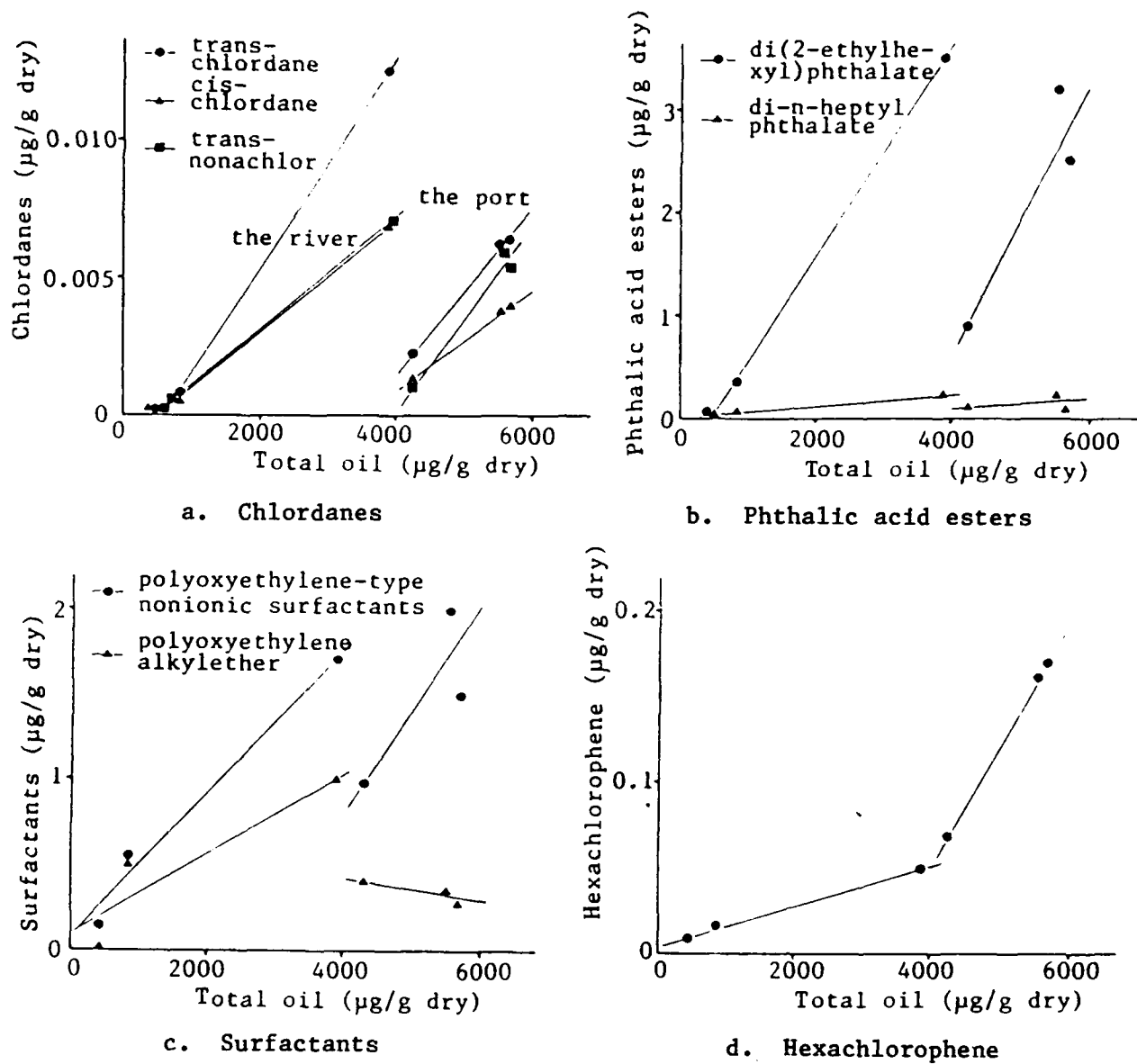


Figure 8. Plot of concentrations of chemical substances versus concentration of total oil

For the chemicals, B represents the concentration of oil not related to the chemicals, and A is equal to the average proportion of the chemicals in the oil containing the chemicals. Nearly equal A values for the two areas of the bay implies that the concentrations of the chemicals in the contaminated oil are similar in the two areas. This leads to the conclusion that the contamination by the chemicals and the oil containing the chemicals proceeds similarly in the two areas of the bay.

Evaluation of Sediment Contamination

Table 5 indicates the slope A and the intercept B in the two areas of the bay.

TABLE 5. SLOPE A AND X-AXIS INTERCEPT B

	Area	trans-Chlordane	cis-Chlordane	trans-Nonachlor	Hexachlorophene
A*	T	3.5×10^{-6}	2.0×10^{-6}	2.1×10^{-6}	12×10^{-6}
	Y	3.0×10^{-6}	2.0×10^{-6}	3.6×10^{-6}	73×10^{-6}
B**	T	500	500	500	0
	Y	3,500	3,600	4,000	3,300

	Area	Di(2-ethylhexyl) phthalate	Di-n-heptyl phthalate	Polyoxyethylene nonionic surfactants	Polyoxyethylene alkylether
A*	T	$1,000 \times 10^{-6}$	60×10^{-6}	420×10^{-6}	250×10^{-6}
	Y	$1,500 \times 10^{-6}$	70×10^{-6}	550×10^{-6}	-100×10^{-6}
B**	T	400	100	0	0
	Y	3,700	3,000	2,800	--

* Unit: $\mu\text{g}/\mu\text{g}$.

** Unit: $\mu\text{g}/\text{g}$.

As expected, A values for all the lines are positive except polyoxyethylene alkylether, which shows a slightly negative A. The negative A may arise from the fact that surfactants are more soluble in water and decompose faster via the microorganisms than the other substances. The hexachlorophene A value for Yokohama Port was about six times as large as A for the mouth of Tsurumi River, probably because the hexachlorophene load in the port is considerably larger than in the mouth of the river. Of the other substances, A values for Yokohama Port are slightly larger than A for the mouth of the river.

The values of A for the chemicals decreased in the following order: di(2-ethylhexyl)-phthalate, polyoxyethylene-type nonionic surfactants, polyoxyethylene alkylether, di-n-heptyl phthalate, hexachlorophene, and chlordanes. The values of A for the chemicals were compared with the amount of their productions or imports in Japan (Table 2) since A values for the chemicals should be correlated with the amount of the chemicals consumed in the drainage basin and since the amount of their consumptions is said to be proportional to the amount of productions or imports. In accord with this hypothesis, A values increase in the increasing order of the productions or imports of the chemicals except for hexachlorophene; this hypothesis was supported experimentally.

The values of B for chlordanes in the mouth of Tsurumi River are nearly 500 µg/g, but those in Yokohama Port are 3,500-4,000 µg/g. The values of B for di(2-ethylhexyl)phthalate is nearly equal to B for chlordanes. The values of B for the other chemicals in both areas are lower than B for chlordanes and di(2-ethylhexyl)phthalate.

The data show a definite empirical relationship between oil and chemical substances. The relationship is that the concentrations of chemicals in bottom sediments are proportional to the concentration of oil. If the relationship is confirmed through additional future efforts, it would be expected to provide a method for qualitative estimation of contamination by chemical substances: there are complications and difficulties in determining the concentrations of chemical substances in bottom sediment. We can consider a simplified method to estimate the concentrations of chemical substances, when we want to know the distribution of chemical substances. First, we derive a regression formula from the field survey at several sites in an area. Second, we can estimate the concentrations of chemicals from the concentration of oil at the other sites in the same area.

SUMMARY

This investigation studied the relationship between the concentrations of oil and the chemical pollutants in bottom sediments off the coast of Yokohama. The data show a possible relationship, although the amount of data are only three data points per contaminant per harbor. Additional data from other sampling locations in these and other harbor would be needed to develop an accurate predictive methodology to estimate the concentrations of chemicals from the concentration of oil.

ACKNOWLEDGMENTS

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GAS GENERATION IN CANALS OF TOKYO PORT

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ABSTRACT

Gas generation in canals of the Tokyo port has been studied to identify its mechanism. The major component of generated gas is hydrogen sulfide. Gas emissions from sediment into water are closely related to the depth of water and tide. Emissions per half days were 4 to 11 ℓ/m^2 for the 1-m depth of water or less and 1 ℓ/m^2 for water more than 2 m depth. Emission of gas into water began during the ebb tide, indicated maximum value about 1 hr before low tide, reduced during the flood tide, and became almost zero after high tide. According to seasonal variations of generated gas, emissions are greatest in August, being over 100 times more than those in October. Gas emissions are suppressed significantly by sediment dredging; gas emissions measured after dredging were reduced to 1/300 to 1/200 of those measured before dredging. The concentration of hydrogen sulfide was attenuated by water at a rate of 1/3 per meter. Generally, the relationship between sediment thickness and gas emissions was not clearly determined.

INTRODUCTION

The importance of water areas in maintaining the good condition of the environment of a city has been pointed out lately--primarily because the water areas are attractive as landscape and can be utilized as an element for recreation, and thus makes up part of the living environment.

The canals of the Tokyo Port have been used not only to transport cargo but also as living areas for Metropolitan citizens and for recreation. The grounds alongside the canals have been redeveloped, changed from sites for warehouses to areas for residences including apartment houses and parks.

But the canal area has problems with contamination of water and sediment and offensive odors. To solve these problems, the Tokyo Metropolitan Office has been engaged in dredging the canal since 1972. This dredging project will have removed 3 million cubic metres of sediment by the end of 1986, costing 10 billion yen.

92-17643



Along with this project, the Tokyo Metropolitan Government has performed various surveys, including quantitative examination of effects of sediment removal and investigation of its standard. This paper, following "Analytical Survey of Accumulations of Sediments in Canals of Tokyo Port" presented by Kimura, Naganuma, and Wano in the 11th Conference, identifies the mechanism of sediment contamination and gas generation.

Gas generation is studied by using gas samplers installed in the canal to determine (1) the relationship between tide and gas generation, (2) seasonal changes of gas generation and effects of dredging, (3) attenuation of hydrogen sulfide in water, and (4) relationship among gas emissions, water depth, and sediment thickness. The status quo of the organic pollution of sediment that significantly affects gas generation, and actual environments of the canal including seasonal changes of water contamination are presented in Shimura (1985).

STUDY SITES AND METHOD OF STUDY

Study Sites

The study sites are shown in Figure 1 as oblique lines in canals of the interior of Tokyo Bay. These canals are located in Kohtoh, Shibaura, and Keihin Districts. The canals of the Shibaura District are considerably polluted in terms of water quality and sediment (Shimura 1985). In order to determine some of the characteristics of gas generation, the present study was carried out in the Shibaura District where the worst sediment contamination is found. Figure 2 shows the stations of study.

Method of Study

A funnel-like gas sampler (Figure 3) was used to collect generated gas. Five hours before low tide, a set of gas samplers was fixed on a post, and the collector bottles filled with seawater up to the zero mark. Water levels in the bottle were recorded every 30 min from 4 hr before the beginning of low tide to 1 hr after the end of low tide. At the same time the range from the collector bottle to the water surface, and the water depth were measured.

Gas volume data taken every 30 min were corrected with respect to water pressure. Gas was also collected on upper, intermediate, and bottom layers in order to investigate attenuation of the components of generated gas in water. Gas generation was surveyed in the ebb tide periods during the spring tide in summer (August) because this month corresponds to the worst pollution period of water and sediment. Gas generation was also surveyed in March, October, and December to examine seasonal variations.

RESULTS AND DISCUSSION

Gas Generation

Gas Emission Volume

Figure 4 shows the maximum gas emissions and variations in the concentration of hydrogen sulfide in stations. This figure shows that more gas is emitted and the concentration of hydrogen sulfide is higher at stations in the

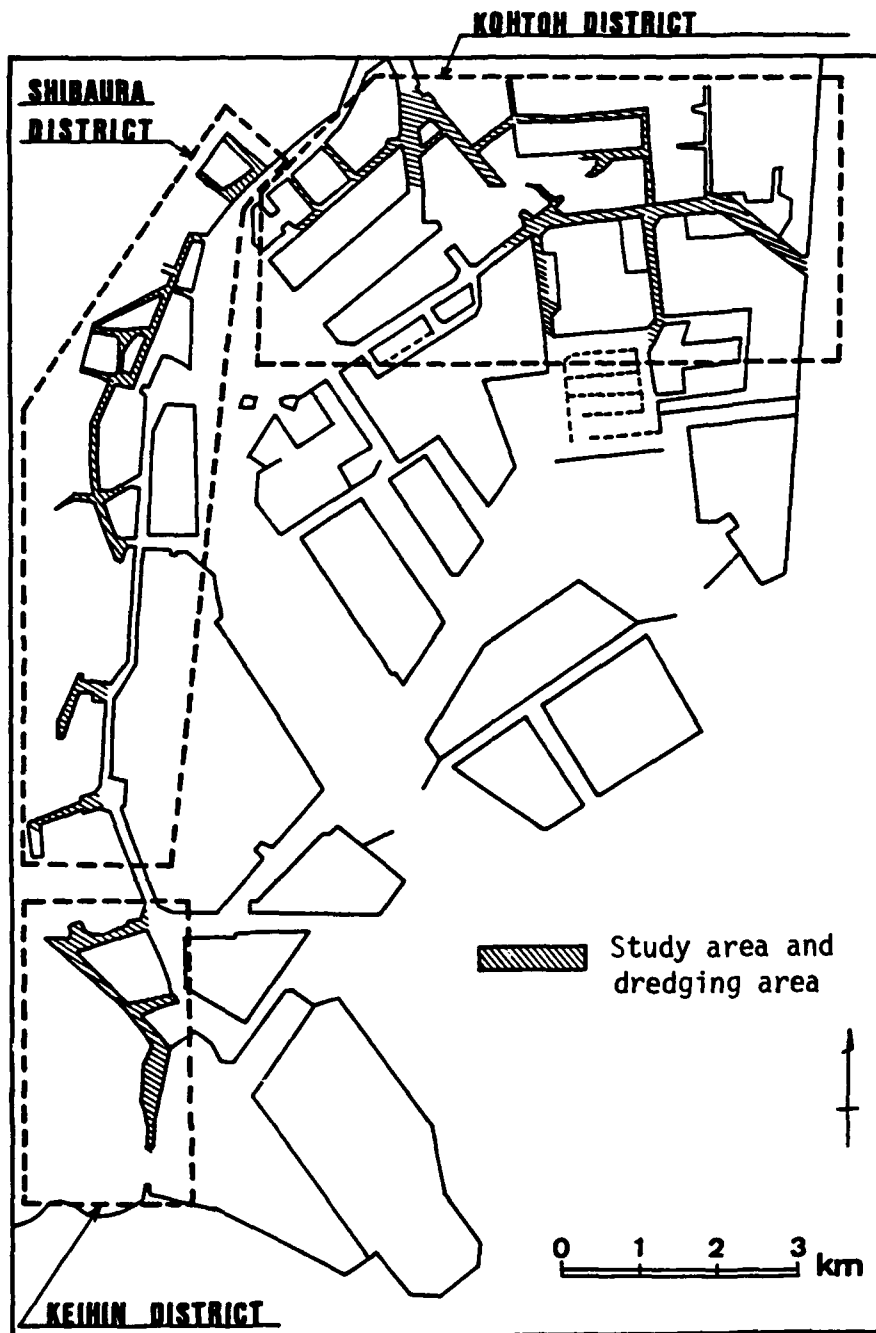


Figure 1. General map of Tokyo Port

interior of the Shibaura District (S-4, S-5, S-6, S-7, S-8, S-13) and at stations belonging to the west Shibaura canal (S-1, S-2, S-3). This coincides with the report of sites worse in water and sediment contamination identified by Shimura (1985). This water area, adjacent to the Shibaura sewage disposal plant, contains more organic matters due to sewage discharges.

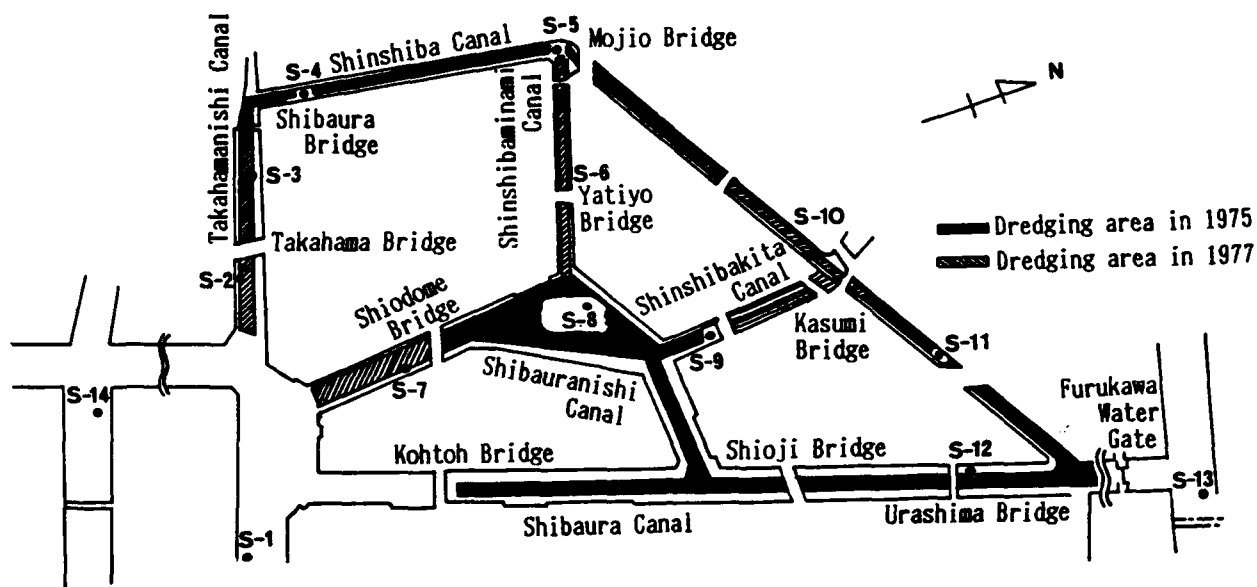
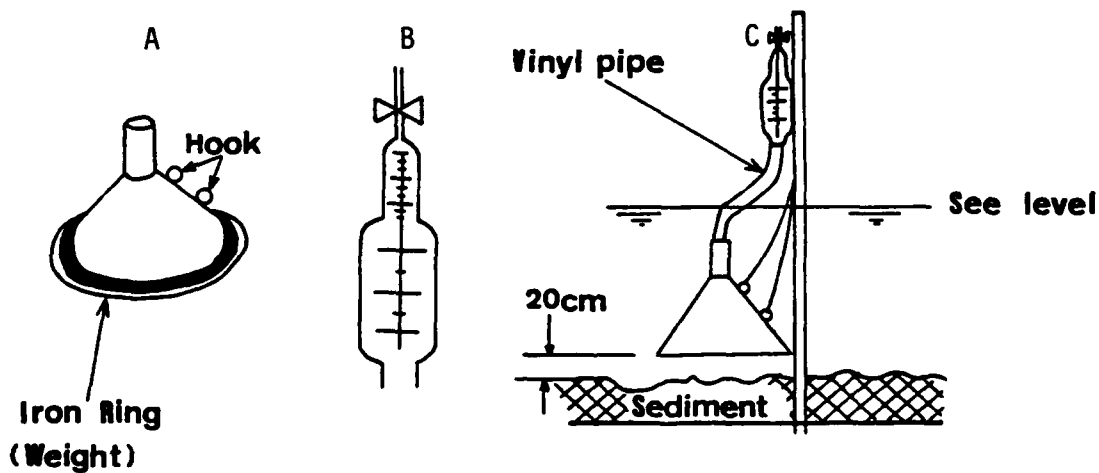


Figure 2. Typical area of sediment contamination (Shibaura District).
Stations measured: S-1-S-14



- A: Gas Collector
- B: Glass Bottle for Gas Measurement (H_2S, CH_3SH)
- C: Installation of Sampler

Figure 3. Gas sampler

Offensive Odor Gas Components

This study analyzed two components: hydrogen sulfide and methyl mercaptan. Figure 4 indicates that the stations where the concentrations of hydrogen sulfide are $\geq 6,000$ mg/l correspond to S-2, S-3, S-4, S-5, S-6, S-7, S-8, and S-13 belonging to the interior area and the west Shibaura canal water area. At these stations, more gas is generated. For methyl mercaptan, few values available for analysis were detected.

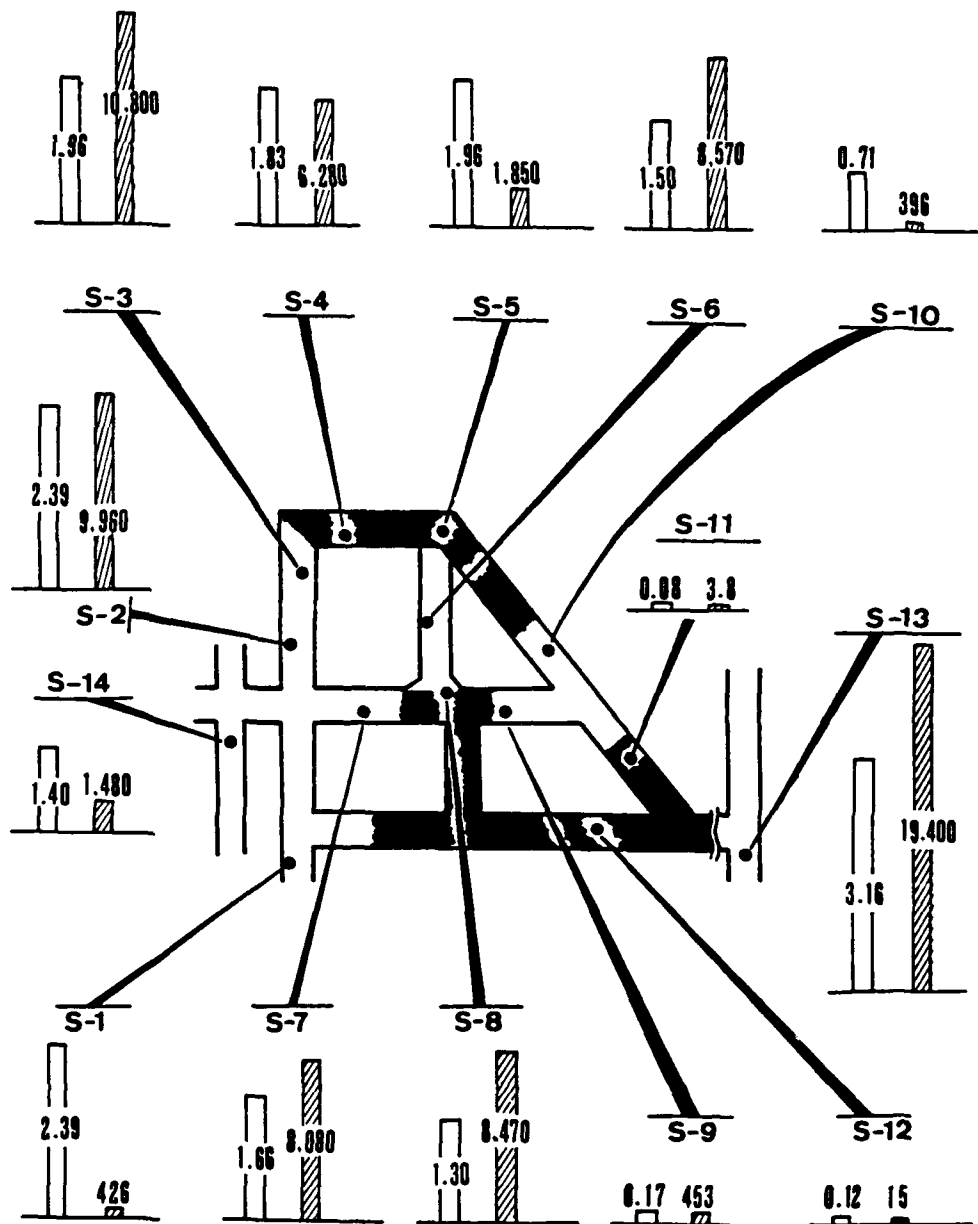


Figure 4. Maximum value of the gas emission and concentration of hydrogen sulfide (H₂S). Open bar: maximum value of generated gas volume (l/m²/hr); hatched bar: H₂S concentration (ppm)

Tide and Gas Generation

The maximum tidal range in the Tokyo Port is 2.10 m. As gas generation may be affected significantly by variations in water pressure, the relationships among water depth, tidal changes, and gas emissions have been investigated.

Gas Emissions and Water Depth

Figure 5 shows the relationship between gas emissions and water depth. At four stations, S-2, S-3, S-8, and S-13, where the water depth is below 1 m during low tide, generated gas volumes ranged from 4.1 to 10.9 ℓ/m^2 /half day. At the stations where the water depth is above 2 m during low tide, generated gas volume is 1.0 ℓ/m^2 /half day. Thus, gas emissions are significantly affected by water depth during low tide. Stations S-1 and S-7, however, show high generated gas values of 4.0 to 7.0 ℓ/m^2 /half day, although the water depth during the low tide was about 2.5 m. This would result because gas emissions at these stations were affected by water depth, as well as geographical conditions and accumulated contaminated sediment.

Gas Emissions and Tidal Changes

Figure 5 shows the relationship between gas emissions and temporal tide changes. Gas emission is highest from an hour before low tide to low tide except for S-13, where a high gas emission began 3 hr before low tide and the maximum emission occurred an hour before low tide. Thus, for the relationship between tide and gas emissions:

- a. Gas emissions vary depending on tidal change.
- b. While the gas emission pattern is different quantitatively depending on the accumulated contaminated sediment and geographical condition, gas emissions begin during the ebb tide, peak about 1 hr before low tide, are reduced during flood tide, and close after high tide.

The above facts lead to an understanding that anerobic generated gas accumulated in sediment is suppressed by water pressure during high tide and is not discharged into water, and gas is emitted during low tide when the water pressure is reduced. Hence, it is obvious that water depth is closely related to gas emission.

Seasonal Variations of Gas Emissions and Dredging Effects on Gas Emissions

At five stations, S-1, S-3, S-4, S-6, and S-10, seasonal variation on gas emissions and dredging effects on gas emission were investigated. Figure 6 shows the maximum gas emissions and seasonal variations in the concentration of hydrogen sulfide at these stations, and during years when dredging was performed. For seasonal variations of gas emissions and effects of dredging on gas emissions, the following facts have been identified:

- a. Seasonal variations in gas emissions.
 - (1) Gas emissions at station S-1 reach a maximum value in August and are reduced from October and subsequently become nearly zero. They then begin to increase again in March. Through 1 year, gas emissions during August are at a maximum and are over 100 times more than the emissions from October on.

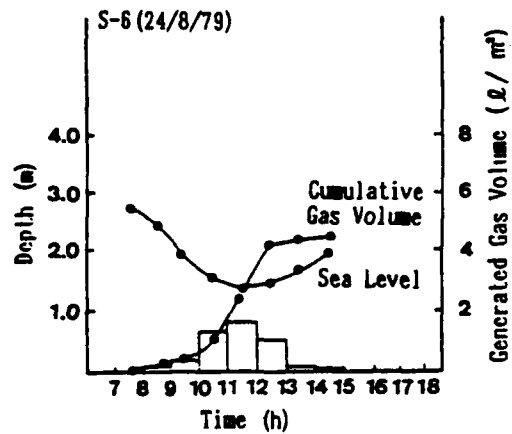
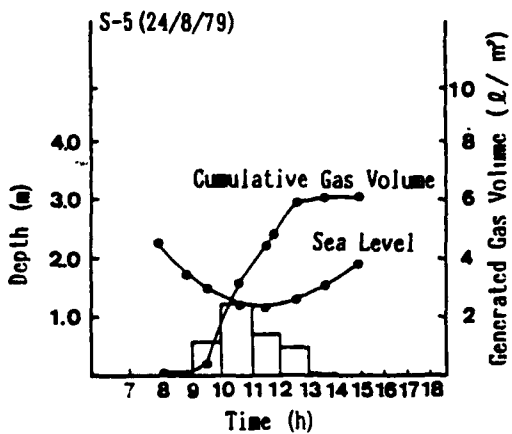
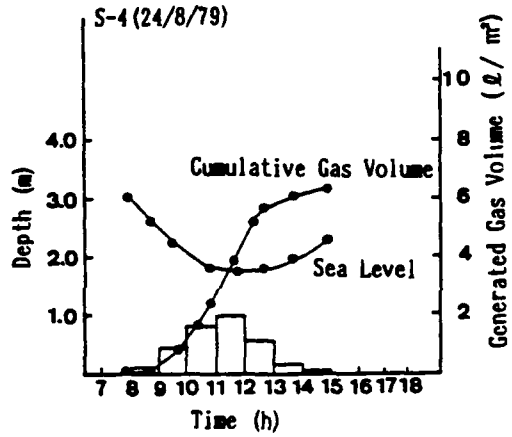
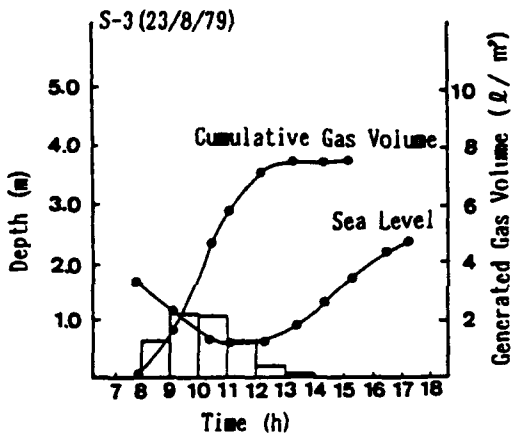
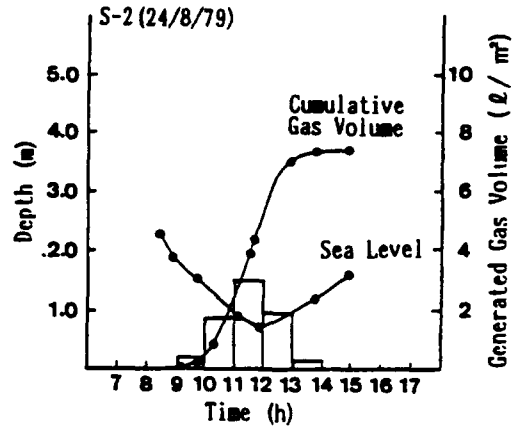
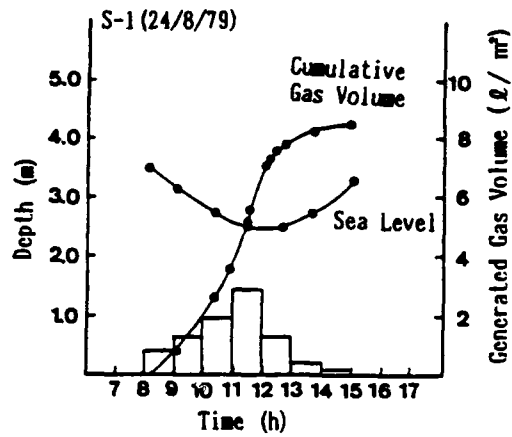


Figure 5. Time series of gas emission volume and sea level (Continued)

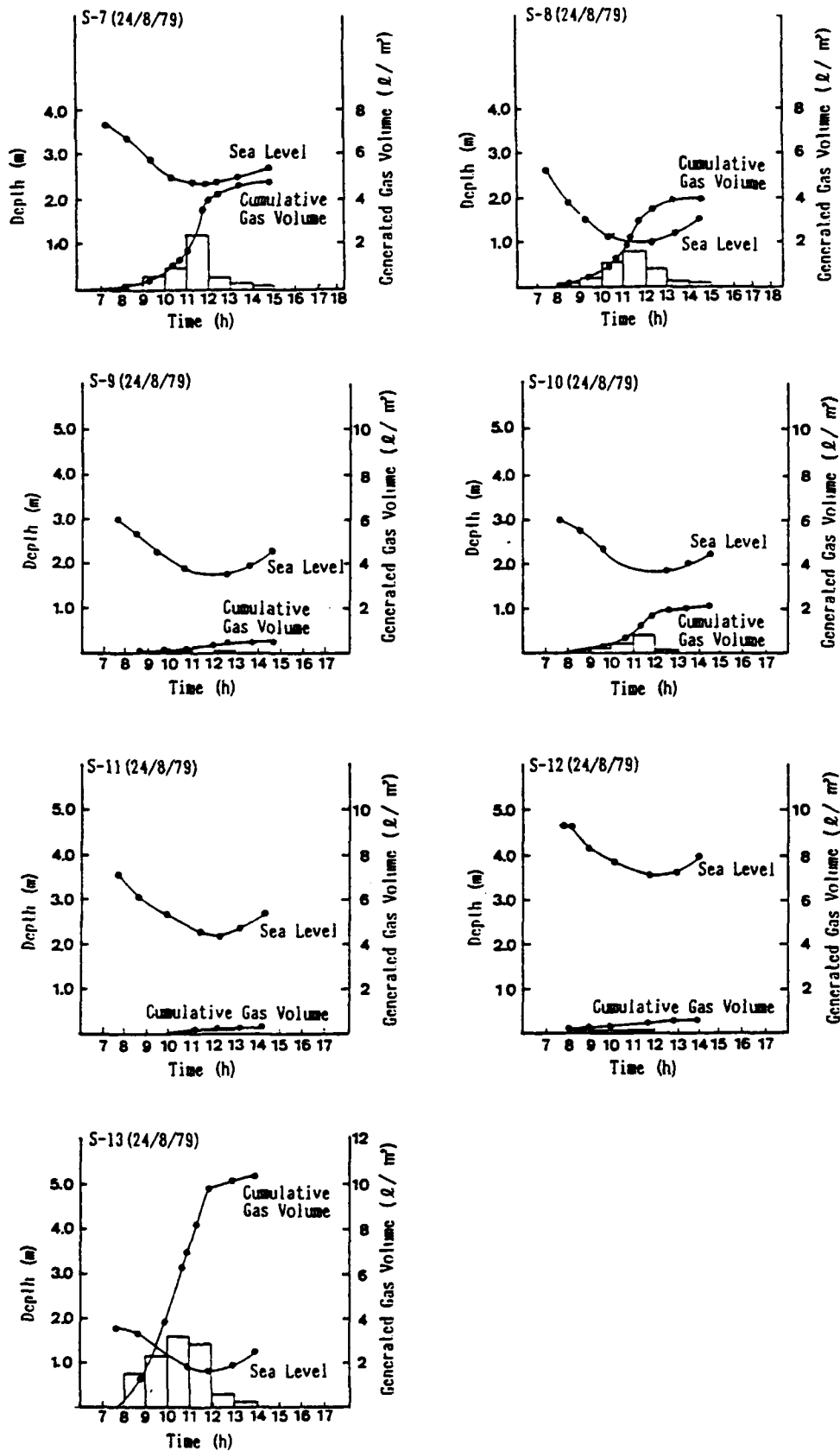


Figure 5. (Concluded)

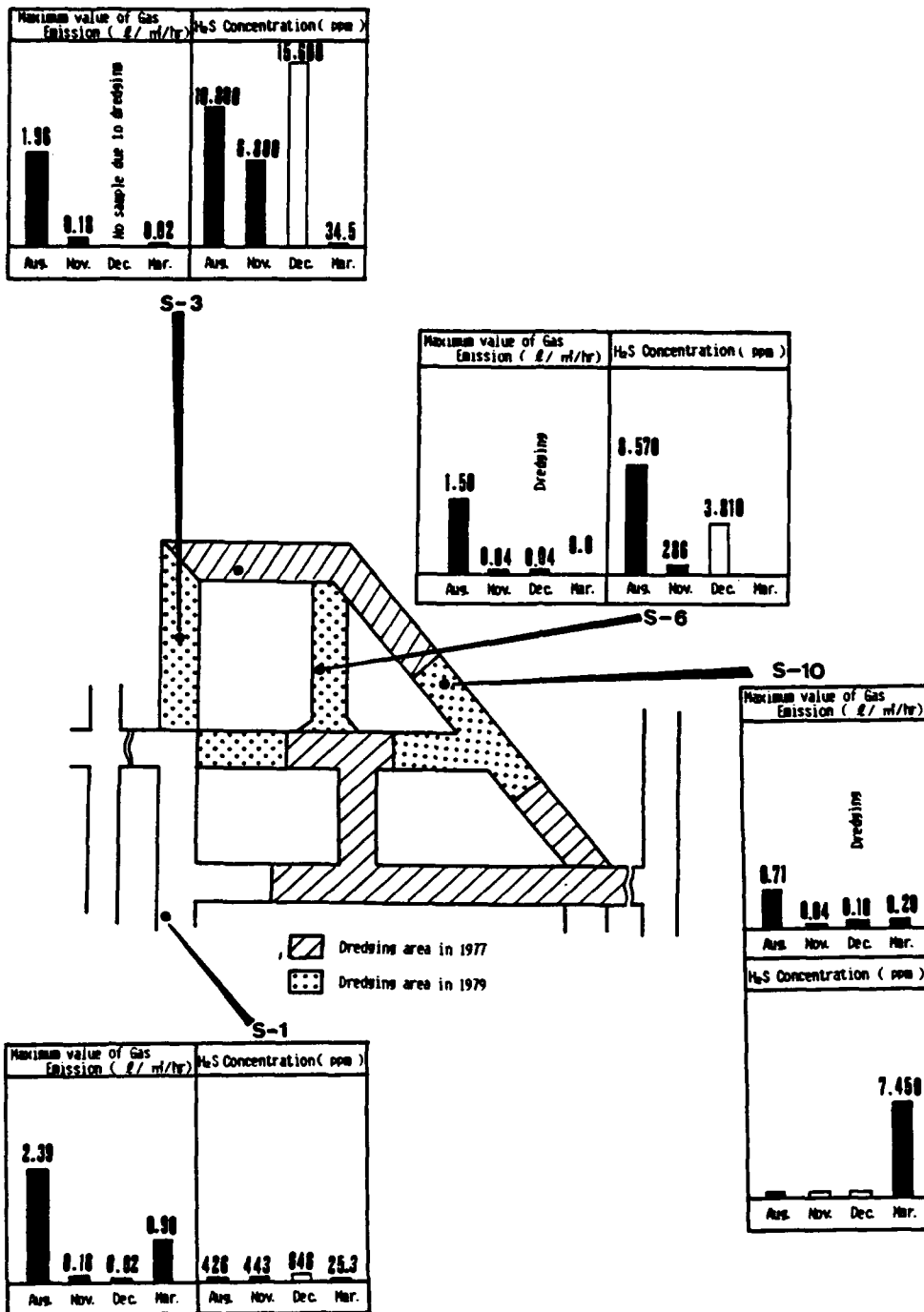


Figure 6. Seasonal variation of gas emission. Open bar: value by forced ejection

- (2) Several hydrogen sulfides were sampled by forced ejection in the study after October; therefore, the concentration of hydrogen sulfide was not measured under the same standard through all periods of the study. But it may be reduced from October on. Thus, August has the worst conditions for seasonal variations of

gas emissions and gas components because the sediment temperature is higher, thereby causing higher activity of anaerobic bacteria.

- b. Effects of dredging. The effects of suppressing gas emissions by dredging have been estimated by comparing the gas emissions and concentration of hydrogen sulfide at a station without dredging (S-1) with dredging stations (S-3 and S-6). At station S-1, the concentration of hydrogen sulfide in March is only somewhat lower than that in August. At S-6, however, no gas is sampled in spite of an attempt at forced ejection. The concentration of hydrogen sulfide at S-3 after dredging is 34.5 ppm, only 1/300 to 1/200 of the value before dredging. This signifies that gas emission is considerably suppressed by dredging.

Attenuation of Hydrogen Sulfide in Water

Anaerobic gas generated and accumulated in the sediment is emitted into water as the sea level decreases. Part of the anaerobic gas is dissolved into water while the rest is emitted into the air. Emitted gas can be sampled at several depths, and the concentrations of hydrogen sulfide contained in the sample gas measured at these depths can be used to examine attenuation of hydrogen sulfide in water. Table 1 shows the relationship between the water depth and the residual rate of concentration of hydrogen sulfide at the depth of sampling (residual rate: concentration of hydrogen sulfide at sampling depth/concentration of hydrogen sulfide on sediment). Figure 7 shows these relationships.

The residual rate of the concentration of hydrogen sulfide (including some deviated values) indicates nearly a linear relation for the passing depth. The regression equation is:

$$y = -0.3234x + 1.0132$$

where

y = residual rate

x = passing depth

Thus, it has become obvious that the concentration of hydrogen sulfide is attenuated in water by about 1/3 per meter. Hence the maintenance of sufficient water depth will significantly reduce the amount of hydrogen sulfide gas emitted to the air.

Relationships Among Depth of Water, Sediment Thickness, and Gas Emissions

As presented earlier, water depth has a negative correlation with gas emissions. As the sediment is related to gas generation and accumulation, the thickness of sediment will be positively correlated to the gas emissions. To identify the relationship of water depth and sediment thickness to gas emissions, new stations are arranged transversely near station S-3 (Figure 8). Figure 9 shows the maximum value of generated gas ($l/m^2/hr$), cumulative gas volume (per half days), water depth, and sediment thickness.

TABLE 1. RESIDUAL RATE OF H₂S IN WATER

Station	Sampling Time	Passing Depth of H ₂ S, m	Residual Rate of H ₂ S
S-3	Before dredging ↓ Oct. 1979 ↓ Dec. 1979	0.30	0.868
S-3		0.10	1.009
S-3*		0.70	0.079
S-6		1.05	0.450
S-6		1.05	0.840
S-11*		2.30	1.000
S-11		1.90	0.919
S-1		2.81	0.093
S-1		2.73	0.228
S-1		2.56	0.205
S-3		0.83	0.950
S-6	2.36	0.203	
S-6	1.84	0.119	
S-6	1.70	0.950	
S-10*	2.97	0.506	
S-1	Dec. 1979	2.40	0.192
S-6	Dec. 1979	1.30	0.560
S-10	Dec. 1979	1.50	0.246
S-4*	After dredging	1.36	0.030
S-4	After dredging	1.38	0.637

* Shows the neglected data.

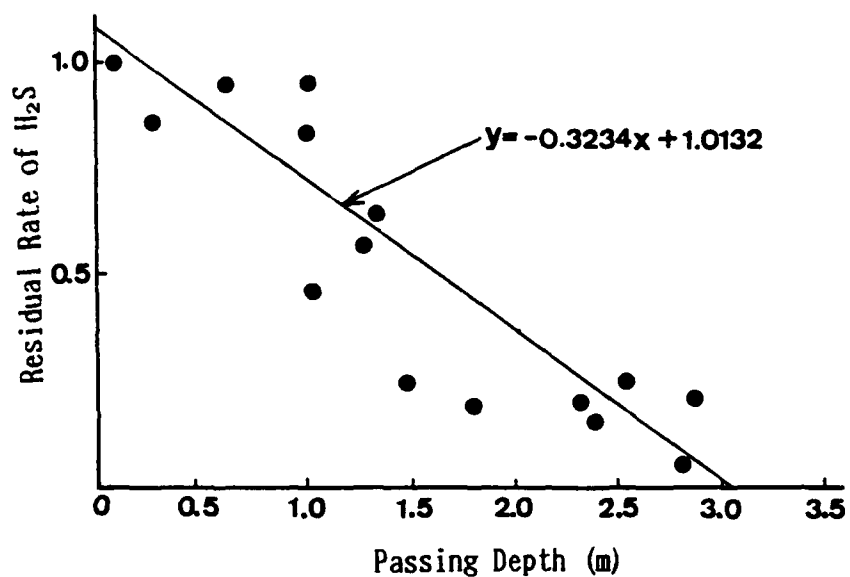


Figure 7. Relationship between the residual rate of H₂S in the water and the passing depth

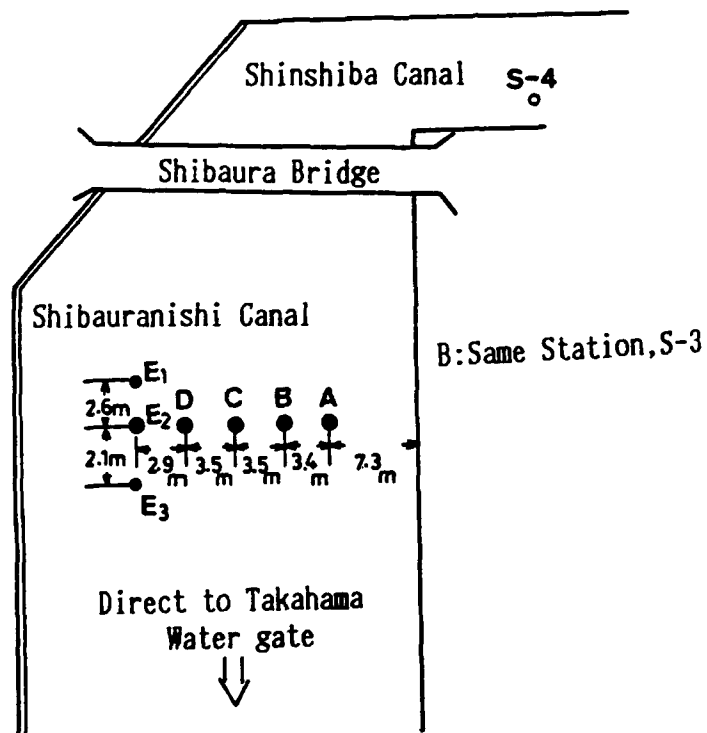


Figure 8. Distribution of new stations to analyze the relationship among water depth, gas emission, and sediment thickness (near Shibaura Bridge)

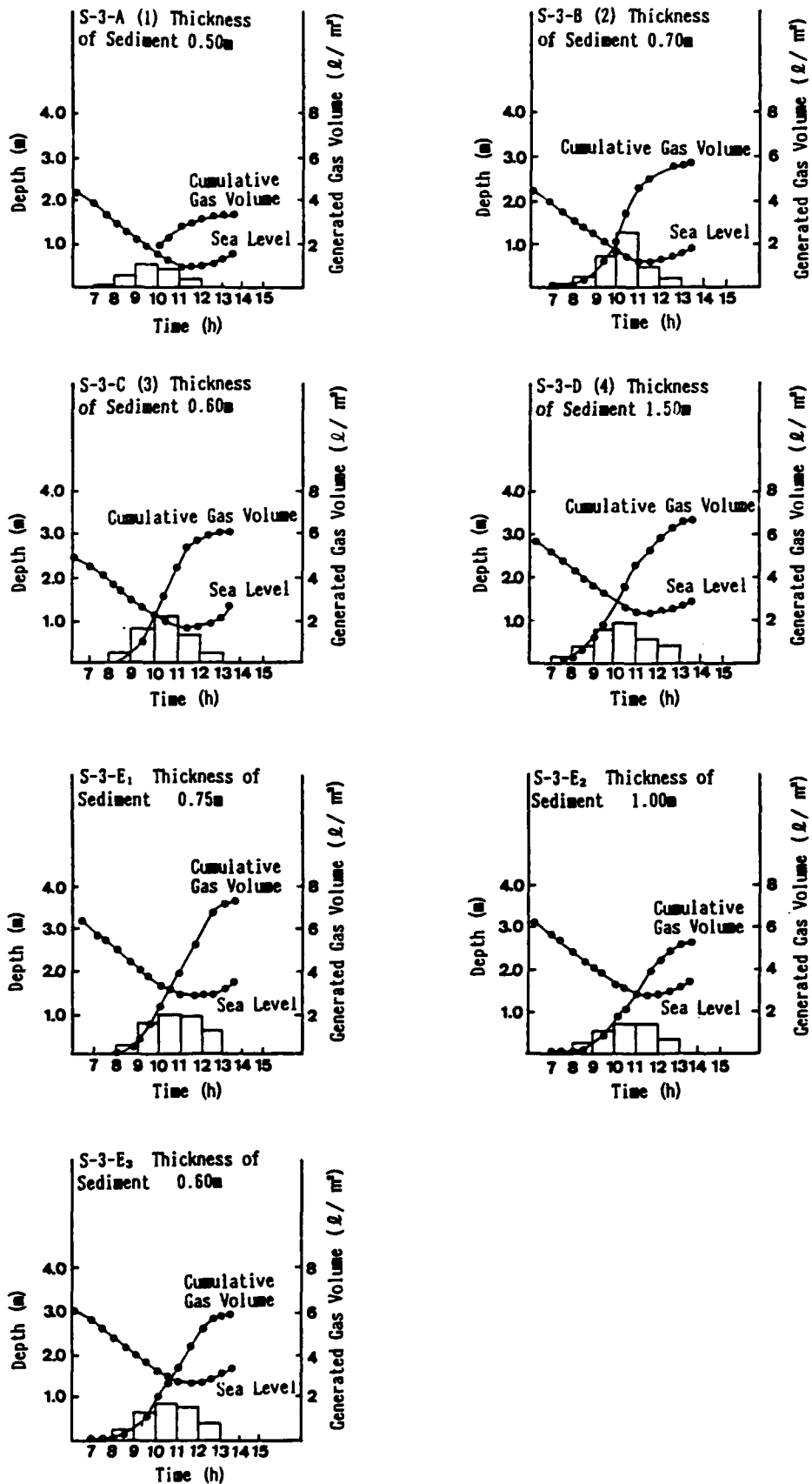


Figure 9. Time series of gas emission and sea level

Gas emissions at the new stations showed the same pattern as the old stations, peaking about an hour before low tide. The maximum gas emissions ($\ell/m^2/hr$) are shown under multiple regression analysis by using the water depth and sediment thickness. Figure 10 shows the results. The maximum gas emission is negatively correlated with the water depth except at station A. It is difficult to find any relationship between sediment thickness and the maximum gas emission due to considerable deviations.

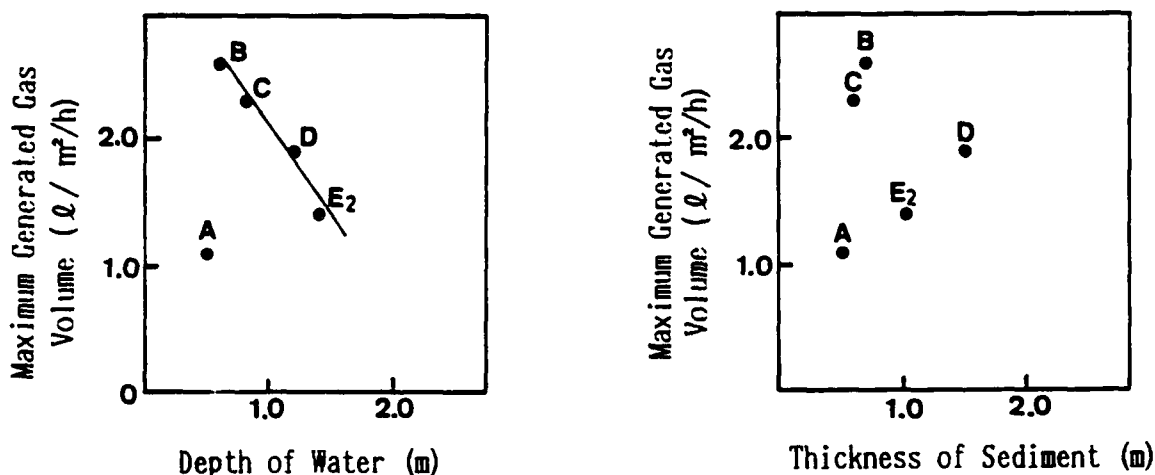


Figure 10. Relationship between gas emission volume and water depth or sediment thickness (stations as shown in Figure 9)

Table 2 shows the results of the multiple regression analysis. Except for station A, the multiple regression coefficient is not less than 0.9, showing a high correlation. Two partial regression coefficients indicate that the maximum gas emission has a negative correlation with water depth and a positive correlation with sediment thickness. It can be seen that water depth affects gas emission more significantly than sediment thickness.

TABLE 2. MULTIPLE REGRESSION ANALYSIS

	Multiple Correlation Coefficient		Regression Equation	
	All Data	Data Except for A	X_1 :Depth of Water	X_2 :Thickness of Sediment
Maximum Gas Emission (Y)	0.295	0.999	$Y = 3.39 - 1.63 X_1 + 0.30 X_2$	

CONCLUSIONS

The results of this study can be summarized as follows:

- a. Gas emissions in canals are 4 to 11 g/m^2 /half day for the stations where water depth is below 1 m, and 1 g/m^2 /half day for the stations where the water depth is above 2 m; gas emission is therefore closely related to water depth.
- b. Gas emissions in the canal area are affected considerably by tidal variations. The variations begin during the ebb tide and peak about an hour before the lowest tide. Gas emissions are then reduced during flood tide and nearly cease after high tide.
- c. Through 1 year, the maximum gas emission occurs in August, being over 100 times more than the emissions produced from October on.
- d. After dredging, the concentration of hydrogen sulfide is reduced to 1/300 to 1/200 of that observed before dredging, indicating that the effect of dredging is significant.
- e. Hydrogen sulfide concentrations are attenuated by water at a rate of about 1/3 per meter.
- f. Gas emission are correlated negatively with water depth and positively with sediment thickness. Water depth affects gas emissions considerably. On the other hand the relationship between gas emissions and sediment thickness is not clearly determined.

FUTURE PROBLEMS

The mechanism of gas emission in canals of Tokyo Port has been almost understood through this study.

Canals have been dredged by the Tokyo Metropolitan Office to prevent offensive gas. It has become obvious that this dredging does suppress gas emission.

The water quality in Tokyo Port has been improved significantly compared with that in the last half of 1960's through reinforcement of drainage restrictions and extension of sewage systems. In recent year, however, the water quality has stopped improving; there are some problems including frequent generation of red tides due to eutrophication in Tokyo Bay. To solve these problems it is necessary that new ideas and operations for prevention of water and sediment contamination will be carried out in the near future.

The Bureau of Harbor Service now has a plan for investigating the re-sedimentation mechanism after dredging. Including this study, the new operation of sediment removal will be fulfilled soon. Details of these projects will be reported separately.

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DEVELOPMENT OF THE PORT OF LONG BEACH:
PAST, PRESENT, AND FUTURE PROJECTS

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ABSTRACT

The development and expansion of the Long Beach and Los Angeles Harbors in San Pedro Bay has been an ongoing effort involving dredging and associated landfill, construction of breakwaters and bulkheads, and other measures related to the conservation, development, and management of water and related land resources in the harbor area.

The southwestern and midwestern sections of the United States are in a period of extensive growth, creating a need for an expanded port; increased use of containerization in shipping also requires more land in order to guarantee an efficient operation. Through extensive studies of historical, current, and projected use of the Ports of Long Beach and Los Angeles, it has been determined that San Pedro Bay will need an additional 2,600 acres of land to help handle the anticipated 223,210,000 short tons of cargo passing through the Ports by year 2020.

This paper describes the development of the Long Beach and Los Angeles Harbors from the late 1800's, when the area was a large expanse of mudflats and shallow water, to the present: the largest harbor complex along the west coast of the United States. A discussion of future developments is also presented. Particular attention will be given to the subjects of dredging and landfill and their important role in the development of the Port of Long Beach, and to the high regard given to mitigation trade-offs.

INTRODUCTION

Over the past 75 years, the Port of Long Beach has grown to be one of the busiest ports on the west coast of the United States. This achievement is the result of hard work and dedication of many people as well as some fortunate natural circumstances.

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Geologic conditions, including natural land topography, sediment properties, and subsidence, have provided a physical setting conducive to port development.

Since the late 1800's, when the harbor was a muddy slough, land and water have been rearranged continuously. The Federal Government initiated the first improvement of San Pedro Bay in 1871 by constructing a 6,700-ft jetty connecting the west end of Rattlesnake Island with Deadman's Island, which provided better protection for ships and deepened the entrance channel from 2 ft to 10 ft at low tide (Smongesky 1983).

Dredging and landfill are still reshaping the harbor; however, over the years, this process has come to demand serious consideration. Each design plan must be studied not only for construction detail, but also for its effect on the environment.

These plans, their implementation, and the resulting mitigation projects are discussed in this paper.

EARLY HISTORY OF THE PORT OF LONG BEACH

The Port of Long Beach is located in San Pedro Bay, a half-moon body of water with a natural protection on the west by bluffs; however, on the southwest, south, and southeast, harbor protection is by man-made breakwater. The San Gabriel and Los Angeles Rivers emptied into the original tidal lagoon, and in 1825 a great flood increased this drainage basin from 700 square miles to approximately 1,700 square miles.

Great quantities of silt and debris were brought down the rivers over several centuries. The action of winds, waves, and ocean currents formed a narrow sand spit approximately 4 miles in length known as Rattlesnake Island (Figure 1).

North and west of Rattlesnake Island was a large expanse of mudflats and shallow water about 2 ft deep at low tide. The construction of a jetty connecting the westerly end of Rattlesnake Island with Deadman's Island provided better protection for ships and deepened the entrance channel by 8 ft.

From 1907 to 1909 much work was performed on the harbor. The San Gabriel River was deepened and straightened for 1 mile, the turning basin and certain navigable channels were dredged, and the barrier beach separating the ocean from the San Gabriel River channel was dredged out at high tide creating the ocean entrance to Long Beach Inner Harbor (Smongesky 1983).

This entrance channel was to be protected by rock jetties projecting into the Pacific Ocean on either side of the channel, each approximately 1.6 miles long with a middle breakwater 3.5 miles long.

Figure 2 shows development of the Port from 1905 through 1960. Figure 3 represents the Port of Long Beach as it exists today.

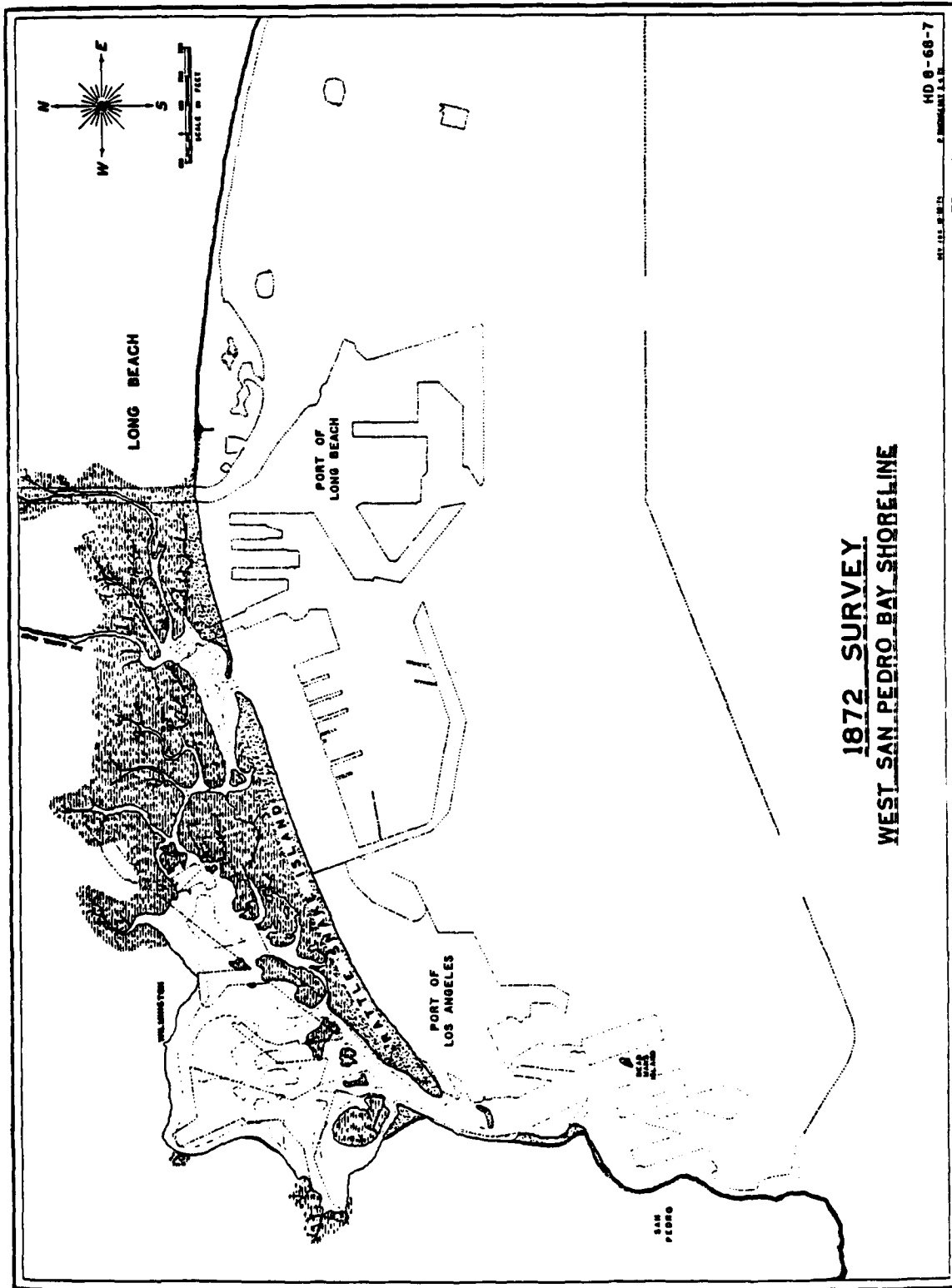


Figure 1. San Pedro Bay

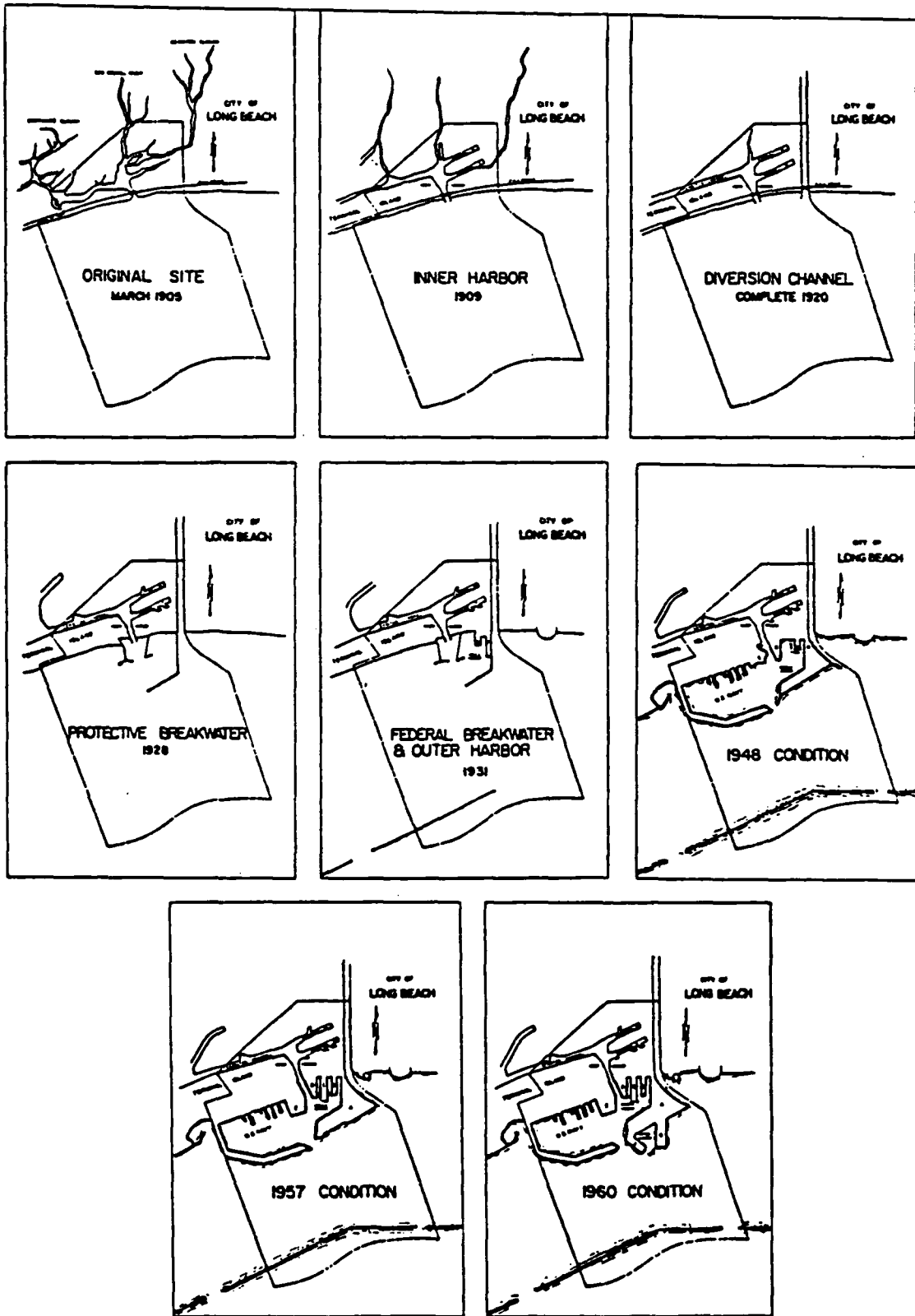


Figure 2. Port of Long Beach development

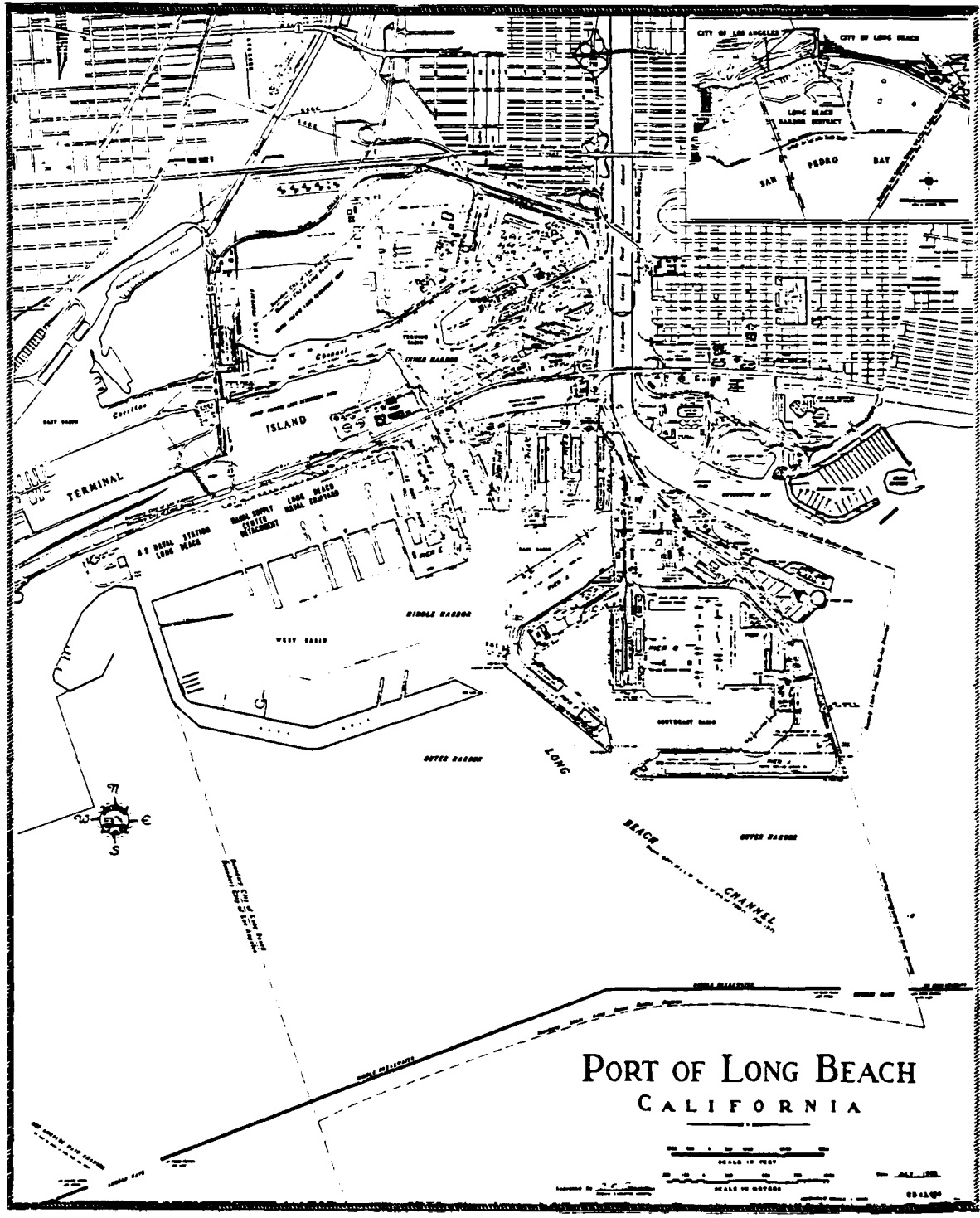


Figure 3. Port of Long Beach, 1986

NATURAL CONDITIONS FACILITATING PORT DEVELOPMENT

San Pedro Bay is located in the southeast region of the Los Angeles Basin, a gentle sloping basin approximately 50 miles long and 20 miles wide, trending northwest, containing thousands of feet of sediments overlying a crystalline basement rock (US Army Corps of Engineers 1984b).

In Long Beach Outer Harbor, the sediments are fairly uniform consisting of a veneer of loose to firm surface sediments overlying very dense older (Pleistocene) sediments. The veneer consists primarily of silts with isolated lenses of sand. The older stratum consists mostly of sands with some lenses of silt, clay, and shales to at least a depth of -150 ft mean lower low water (MLLW). The near-surface materials in the middle and inner harbors are similar to the outer harbor, except that the top 6 ft contain more organic residue (US Army Corps of Engineers 1984a).

Successful use of dredged material for landfill depends on the type of sediment dredged. The Los Angeles Basin, being a desert area, has dry sandy soil--an attractive advantage over moisture-holding sediments found in many other port areas of the United States. In addition, the harbor's distance from the mountains is such that the sediments settling out from the rivers are a desirable size and strength for use as landfill. The abundance of fine- to coarse-grained sand in the harbor has been a valuable asset for the Port. The diminishing supply of this resource as an ideal material for landfill use is an ever-growing concern among port planners and engineers.

Subsidence in the harbor has caused extensive damage to buildings, docks, pipelines, and other facilities. It has also created deeper channels, which are a welcomed by-product of this otherwise devastating occurrence. This subsidence was caused by extraction of oil from the Wilmington Oil Field. Between 1928, when pumping was initiated, and 1965, up to 29 ft of subsidence and substantial horizontal movement occurred. Since 1966, subsidence has been arrested by the injection of water into the oil zone, thus preventing the soils from compacting when oil is removed. Due to the injection, some rebound has occurred, reaching a maximum of 1.4 ft between the peak rebound years of 1961 and 1973. Currently, the maximum ground surface elevation fluctuation is less than 0.2 ft annually and generally not measurable (US Army Corps of Engineers 1984a) (Figures 4 and 5).

Another advantage of the geographic location of the Port is the minimal need for maintenance dredging due to the natural barrier created by the Palos Verdes Hills, located west of the harbor, and the redirection of river outlets away from Port channels by flood control facilities. The annual average maintenance dredging cost for the Port of Long Beach is \$72,000, which is substantially lower than that for other US ports (Long Beach Harbor Consultants 1976).

The Port of Long Beach has also been very fortunate in that it has not had to alter or deter dredge and fill projects because of the existence of toxic substances. The US Environmental Protection Agency (EPA) requires that chemical analysis be conducted on the material to be dredged to determine the levels of concentration of certain constituents which may be toxic to marine

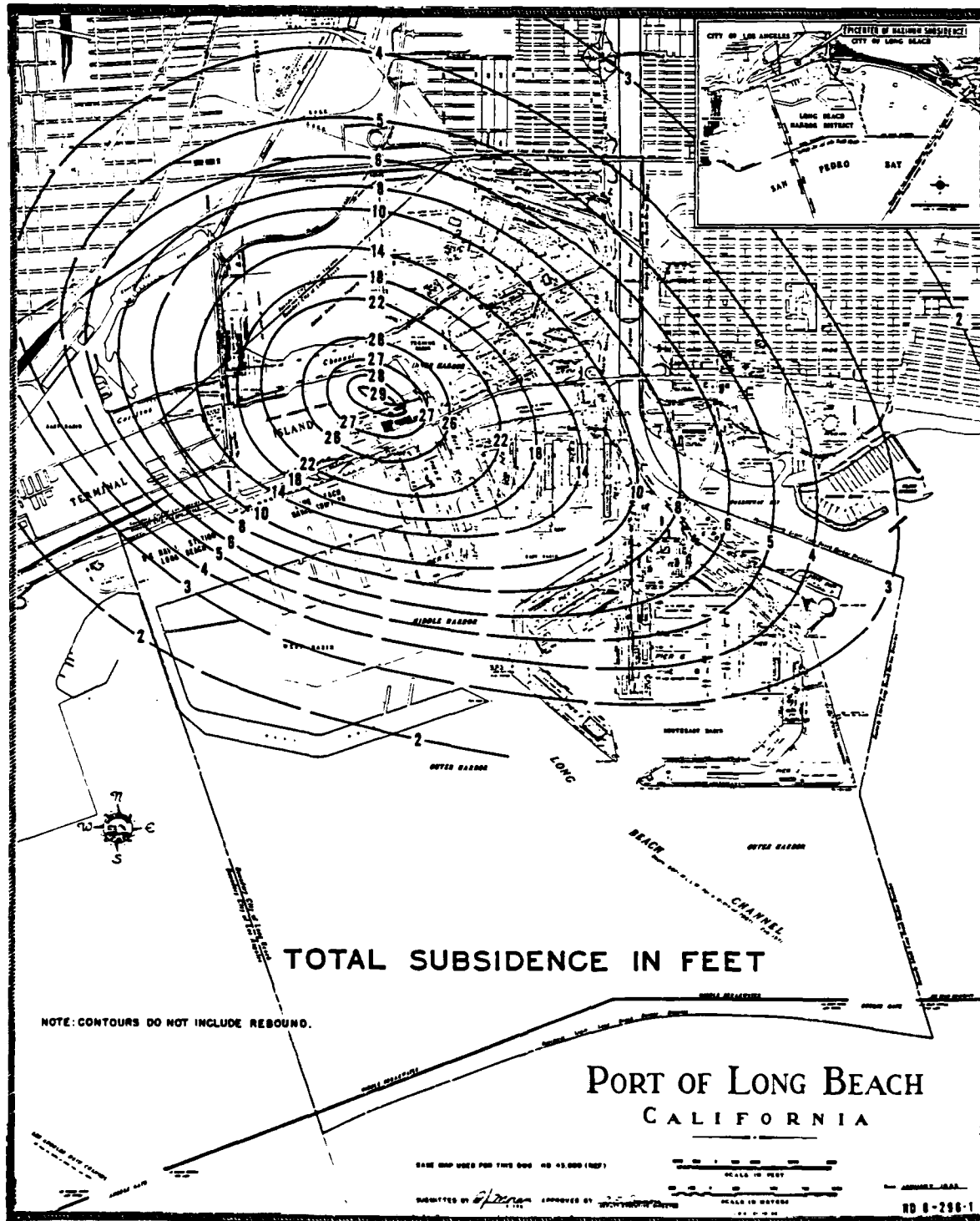


Figure 4. Total subsidence

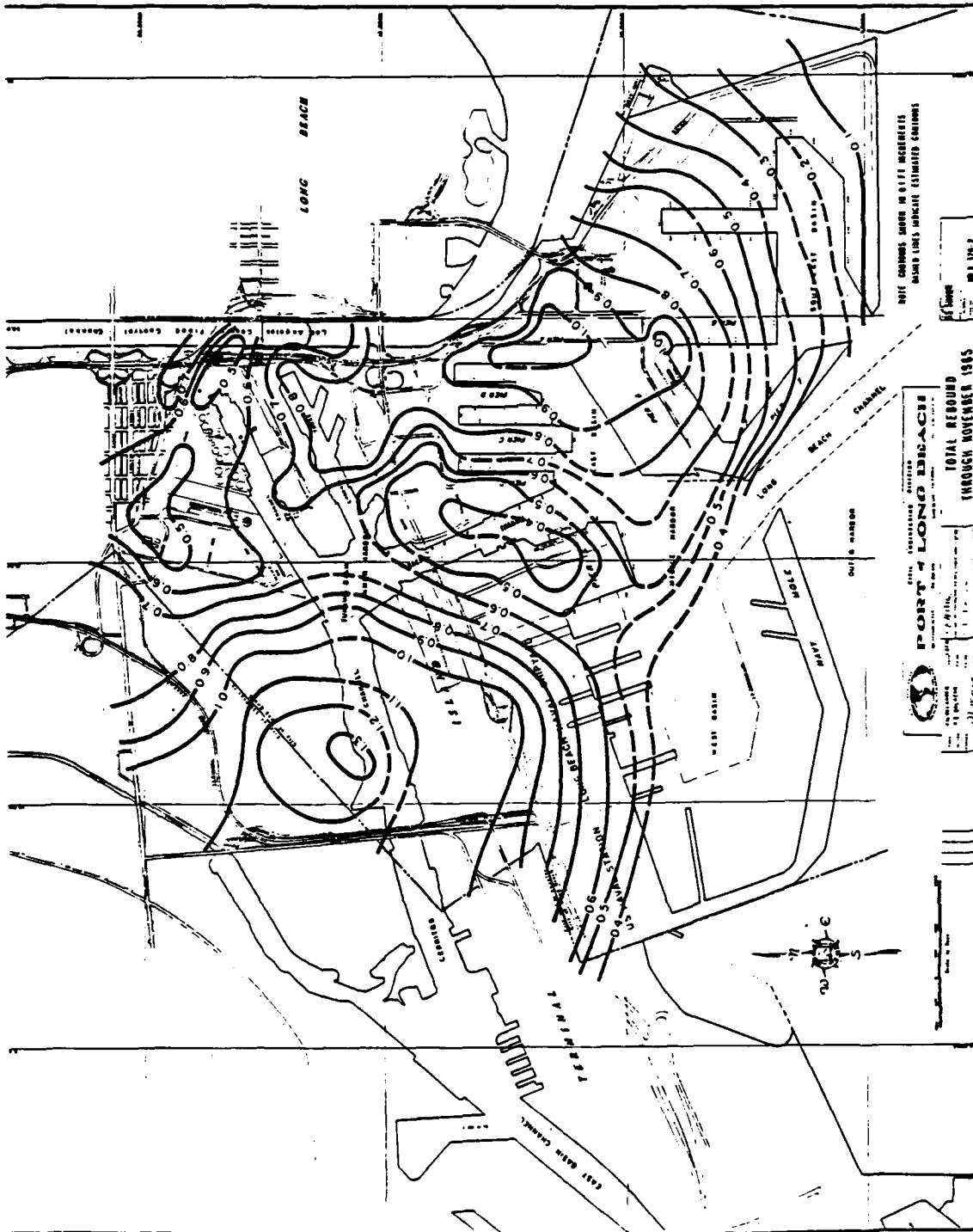


Figure 5. Total rebound

organisms. The analyses are required from the areas proposed to be dredged on samples obtained between the existing sediment surface and the proposed project dredge depth.

Prior to bioassay test requirements, maximum concentration levels were allowed by the EPA for certain constituents. These levels were as follows: mercury, 1.5 parts per million (ppm); cadmium, 3.0 ppm; lead, 180 ppm; zinc, 300 ppm; and oil and grease, 4,000 ppm. In a report prepared for the Port by Long Beach Harbor Consultants in 1976, 200 separate samples were collected and analyzed from over 40 stations throughout the harbor and outside the Federal breakwater (Long Beach Harbor Consultants 1976). Concentrations of each constituent tested were less than or equal to the maximum allowable limits at all stations except for three stations located in the inner harbor. However, the inner harbor has not been used as a source of landfill material since 1924. See Figures 6 and 7 which illustrate the dredge and fill history of the Port.

USE OF DREDGING AND LANDFILL IN PORT DEVELOPMENT

Hydraulic dredging and filling have been used extensively in developing the existing harbor complex; approximately 50 percent of the Port's land area in hydraulic fill (Johnson and Burke 1986).

Obtaining enough fill material for future development will require dredging the entire outer harbor and well beyond the existing breakwater. The Port has been extremely fortunate in the past to have quality material readily available for landfill; however, the supply of the best materials is nearly exhausted. The Port will then need to consider transporting landfill from offsite locations, or develop ways to make better use of the available dredged material. One such technique, involving a special compaction process used along dike perimeters, was tested and will be discussed later in this paper.

Water depths in the terminal approaches within harbor range from 50 ft in the middle harbor basin and channels two and three, to 60 ft in the main and back channels. Adjacent to berths, water depths range from 36 ft to 70 ft. The deep berths and channels in the harbor can presently accommodate vessels in the 50,000 to 165,000 DWT class with drafts not exceeding 58 ft. This is an insufficient depth to accommodate the larger petroleum tankers and dry bulk carriers. ARCO's terminal on Pier E, where depths are 70 ft, could handle tankers up to 265,000 DWT, but the main channel depth is not sufficient to allow passage into Pier E. (See Figure 3 for channel reference.)

In the short term, our planned dredging of the main channel to a width of approximately 1,000 ft and a depth of 76 ft will be necessary to provide improved access by modern cargo vessels presently constrained by draft limitations. Wharfside deepening will also be necessary to fully utilize deeper channels. This dredged material will be used to create approximately 150 acres of new land for the Port (Port of Long Beach 1983).

2020 PLAN

Through extensive studies, it has been determined that San Pedro Bay, including both the Port of Long Beach and the Port of Los Angeles, will need an additional 2,600 acres of land to help handle the anticipated 223,210,000

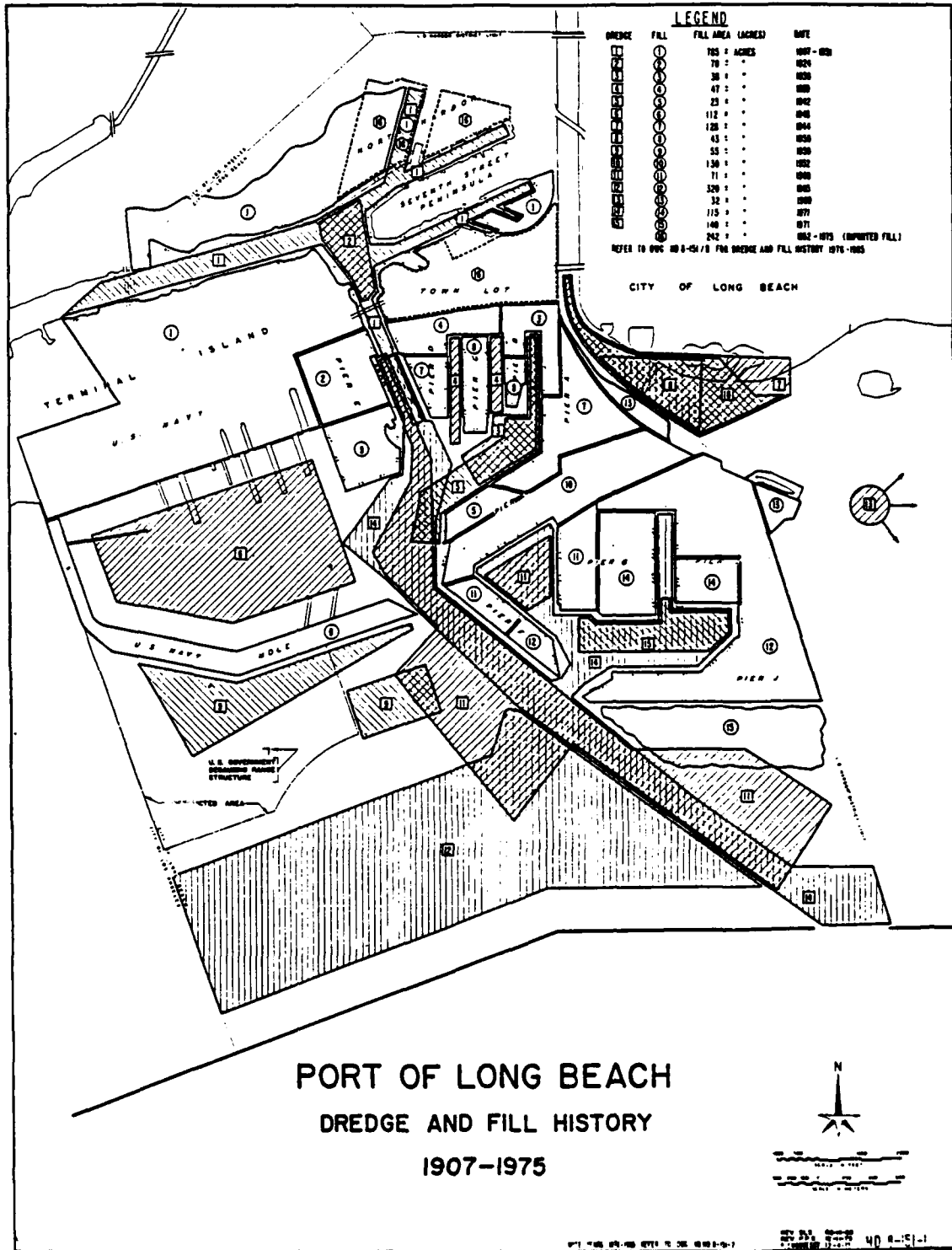


Figure 6. Dredge and fill history, 1907-1975

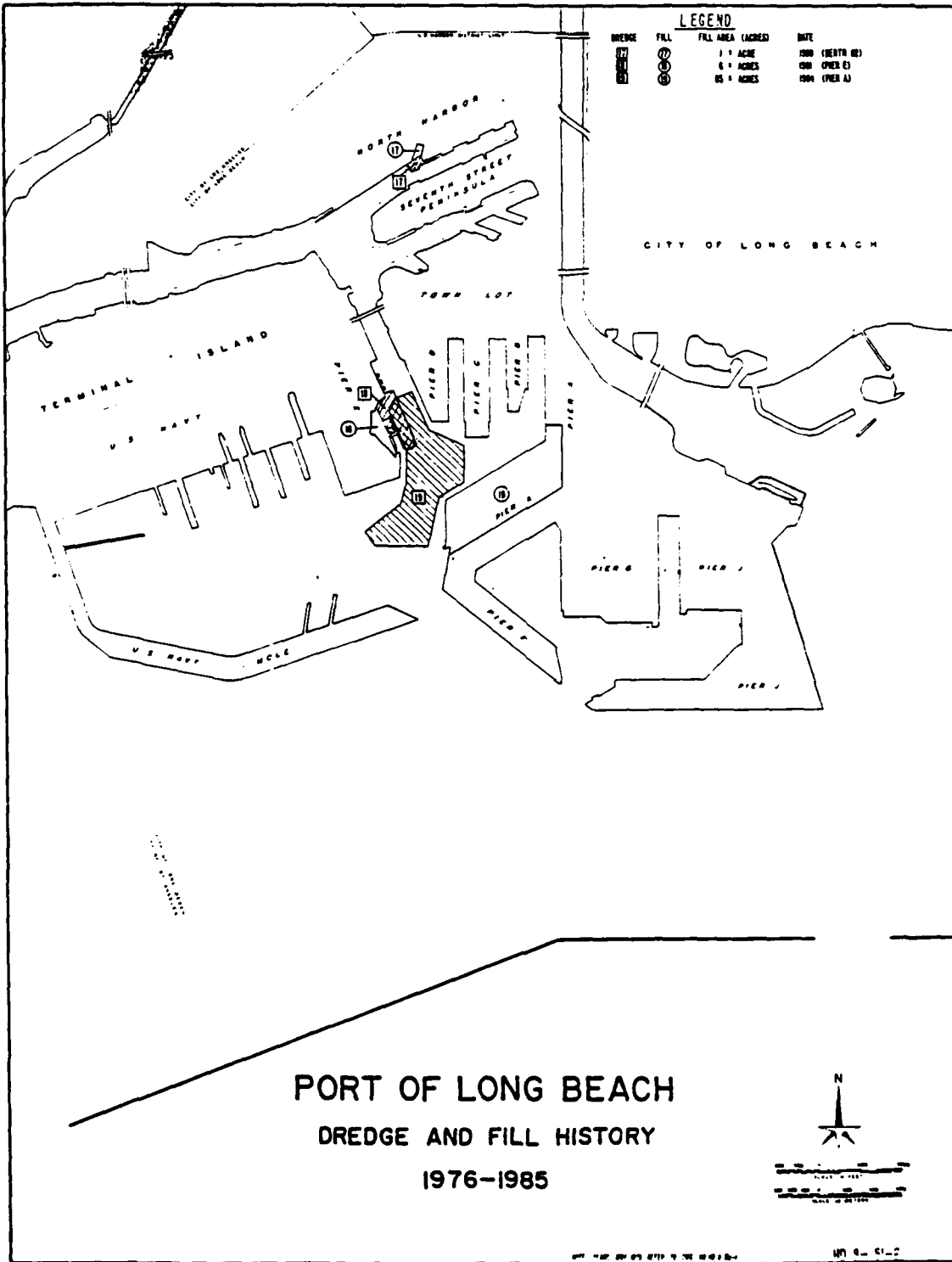


Figure 7. Dredge and fill history, 1976-1985

short tons of cargo passing through the Ports by the year 2020. Not only is the unavailability of land for expansion a problem, but the expected increase in the size of cargo vessels will also require deeper port facilities.

Modernization of the existing facilities would result in a maximum capacity of only 150,127,000 short tons. This would require the ports to turn away 73,000,000 short tons of cargo because of a shortage of land (Port of Long Beach 1984).

A master plan, called the 2020 Plan, has been developed to outline a program to meet the projected cargo handling needs of the Ports of Long Beach and Los Angeles through the year 2020. The plan is a program of dredging, landfill, and facilities construction providing 2,600 additional acres of cargo-handling facilities for both ports when completed. The plan also provides for improvements of navigation channels in and around the outer harbor to allow access by the largest vessels in the Pacific fleet. Channel dredging is proposed to a maximum depth of -82 ft, providing sufficient depths for liquid bulk carriers in excess of 300,000 DWT. The dredged material would be disposed of in various harbor landfills, again using this valuable resource to accommodate the maritime industry.

The 2020 Plan has been developed in conjunction with the US Army Corps of Engineers and with the cooperation of both state and Federal agencies and numerous experts in dozens of specialties.

The Port of Long Beach will implement the plan in three phases: Pier K, 435 acres by the year 1995; Pier L, 450 acres by the year 2010; and Pier M, 300 acres by the year 2020 (Figure 8). The Port of Los Angeles will construct its portion of the plan in similar phases.

CONSTRUCTION DESIGN

In the design of any shipside facility, a matter of major consideration is the type of fill retention and wharf to be used. Several possible wharf and fill retention schemes should be investigated to find which one best fits the situation. This must take into consideration present and future shipping conditions such as ship sizes, cargo loads, and cargo-handling methods.

Although we investigate new techniques each time a project is planned, we generally conclude that a staged rock dike and hydraulic fill, along with a concrete pile-supported wharf, is the most efficient and economic procedure. This is principally because rock is readily available and within 22 miles by marine transportation.

The general construction sequence at the Port is as follows: first, construct a dike or bulkhead to enclose a proposed pier in sheltered water; second, hydraulically fill the pier; and third, construct the wharf. Hydraulic dredges pump the fill material for the piers from the ocean bottom. Rock used in the Port is quarried from two quarries on Catalina Island. The weight of the rock varies from approximately 135 pcf to 165 pcf, depending from which pit the rock is quarried (Smongesky 1983).

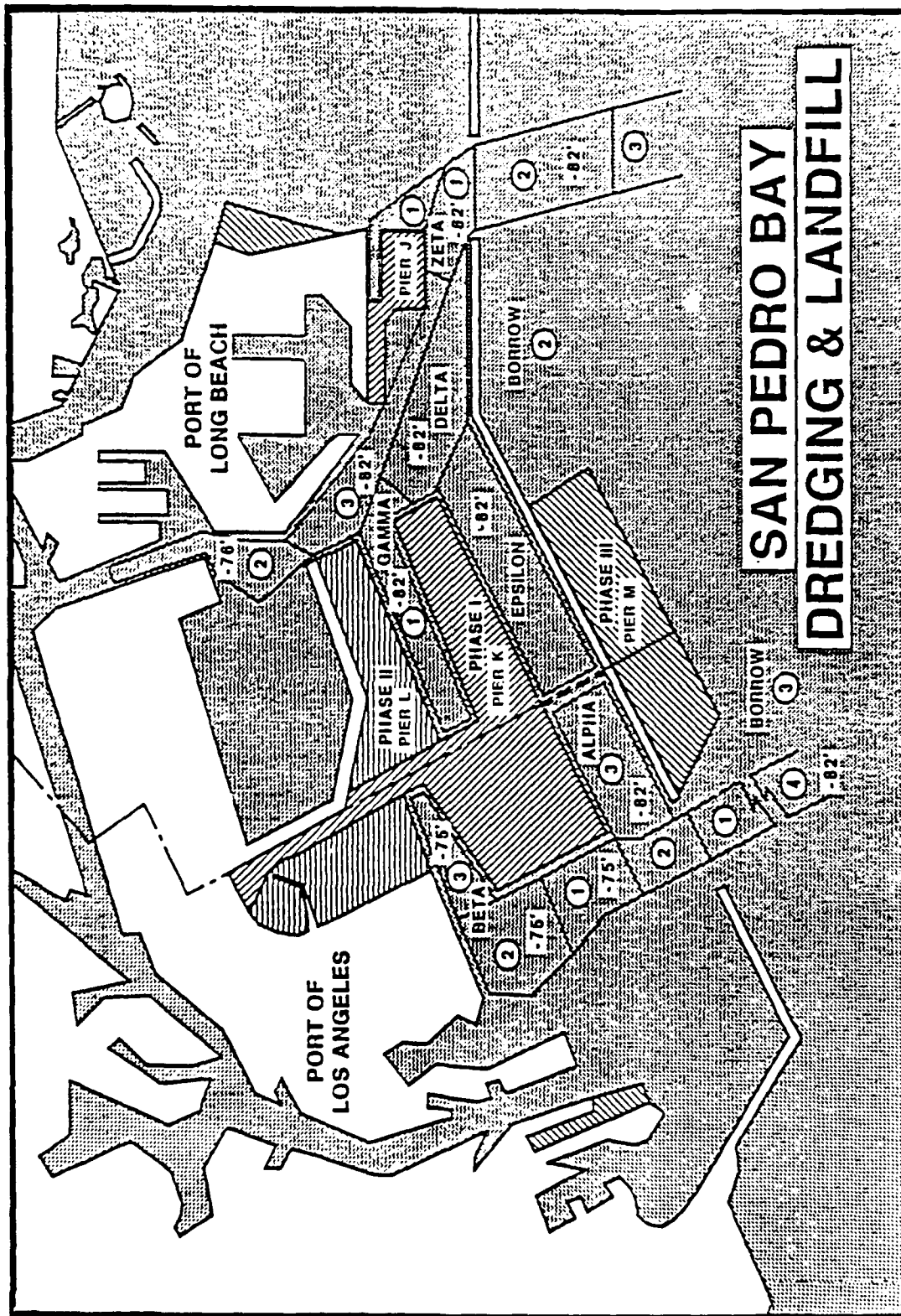


Figure 8. San Pedro Bay dredging and landfill.

Wharf on Concrete Piling Over Rock Dike

The pile wharf over a rock dike, shown in Figure 9, is the design we generally follow. The landfill that is constructed serves as a barrier to waves and creates a quiet basin for wharf construction. This type of construction is particularly economical when large quantities of rock and sand are readily available.

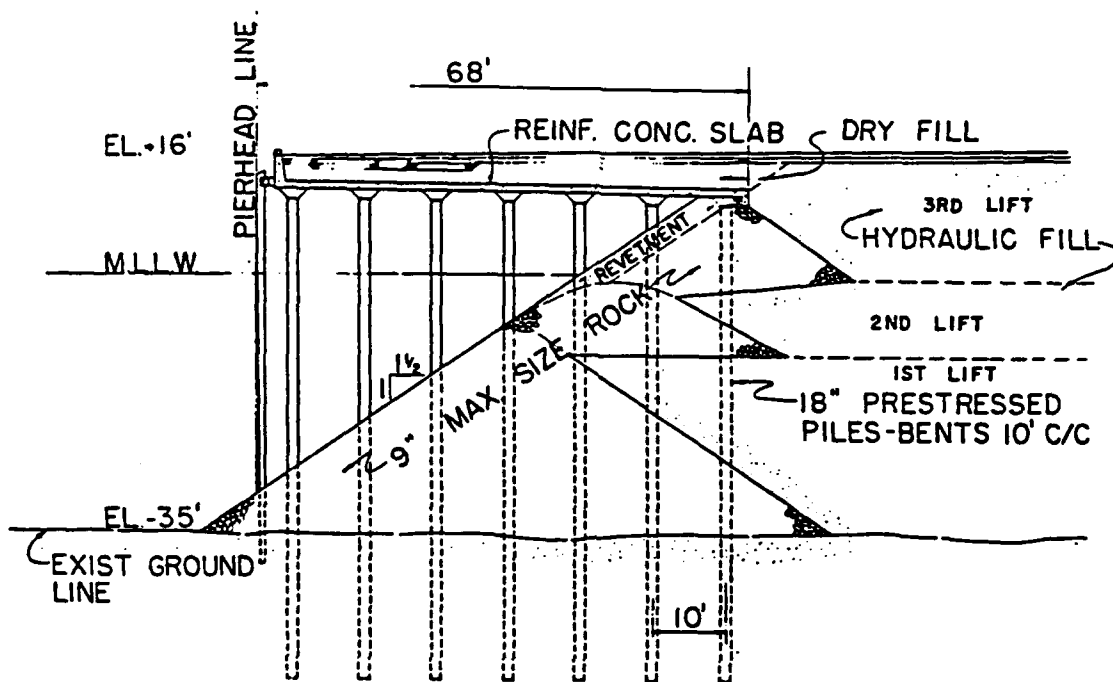


Figure 9. Pile wharf over a rock dike

Other Wharf Combinations

One of the wharf and bulkhead combinations that has been successfully used at two piers in the Port is the pile wharf in combination with the sheet pile bulkhead, as shown in Figure 10.

Steel sheet pile cellular bulkheads, another construction method, can be constructed in open water. The steel sheet pile cellular bulkheads constructed in the Port are approximately 62 ft in diameter and are connected with steel sheet pile arcs (Figure 11). The cells are constructed with 15-in.-wide sheets, 1/2 in. thick on the waterside of the cells and 3/8 in. thick in the fill. The arcs are connected to the sheets with 1/2-in.-thick tee-piles and are constructed of 1/2-in.-thick sheets.

The Port has constructed over a mile of concrete quay wall. This type of wharf and bulkhead combination is shown in Figure 12. A careful analysis of the soil should be made to determine possible slippage planes and allowable toe pressures before determining whether a concrete quay is feasible as a wharf and bulkhead structure for a particular site. Economics of marine concrete construction has prevented the Port from using this construction technique for many years.

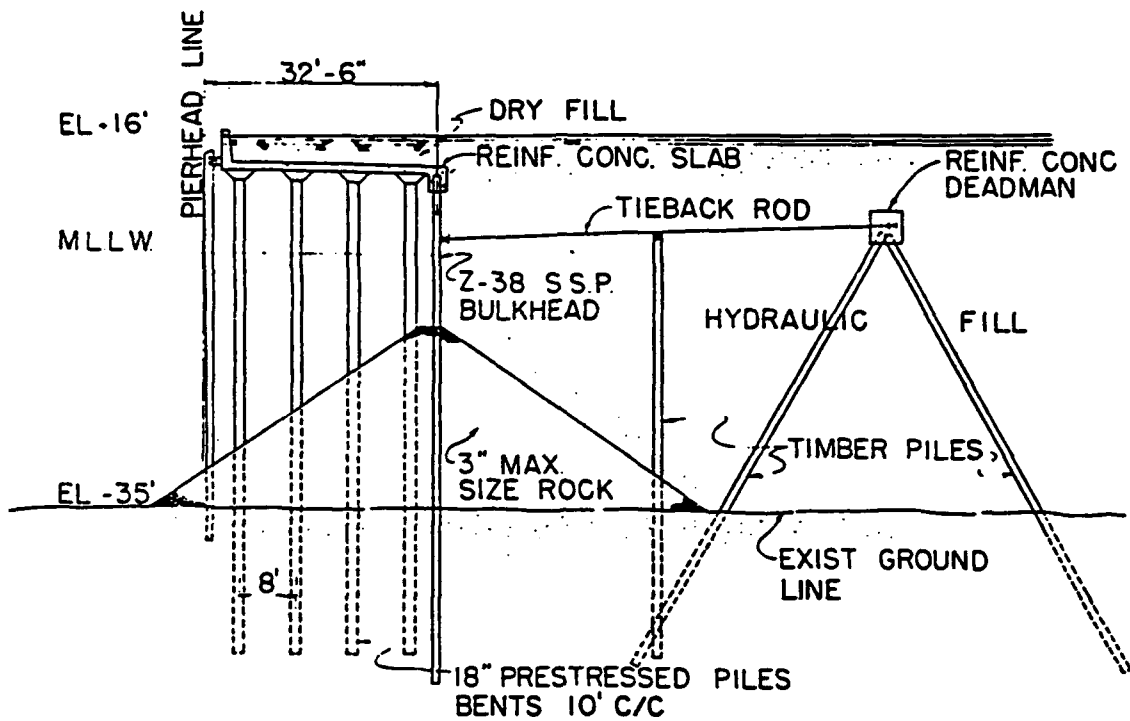


Figure 10. Pile wharf in combination with steel pile bulkhead

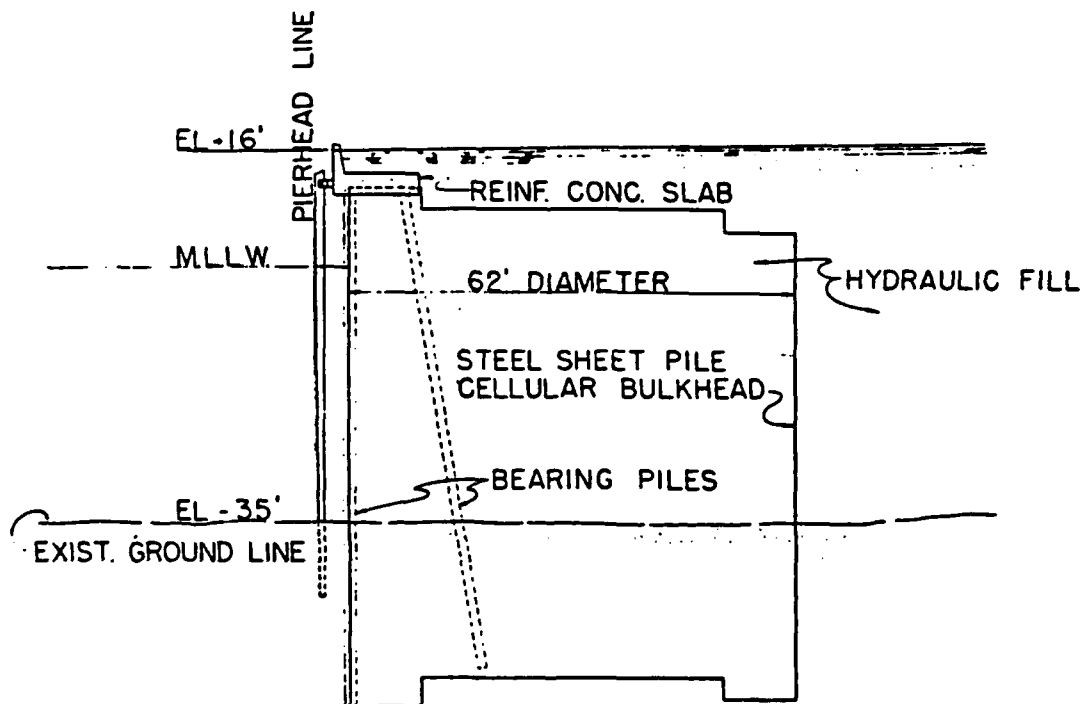


Figure 11. Steel sheet pile cellular bulkhead

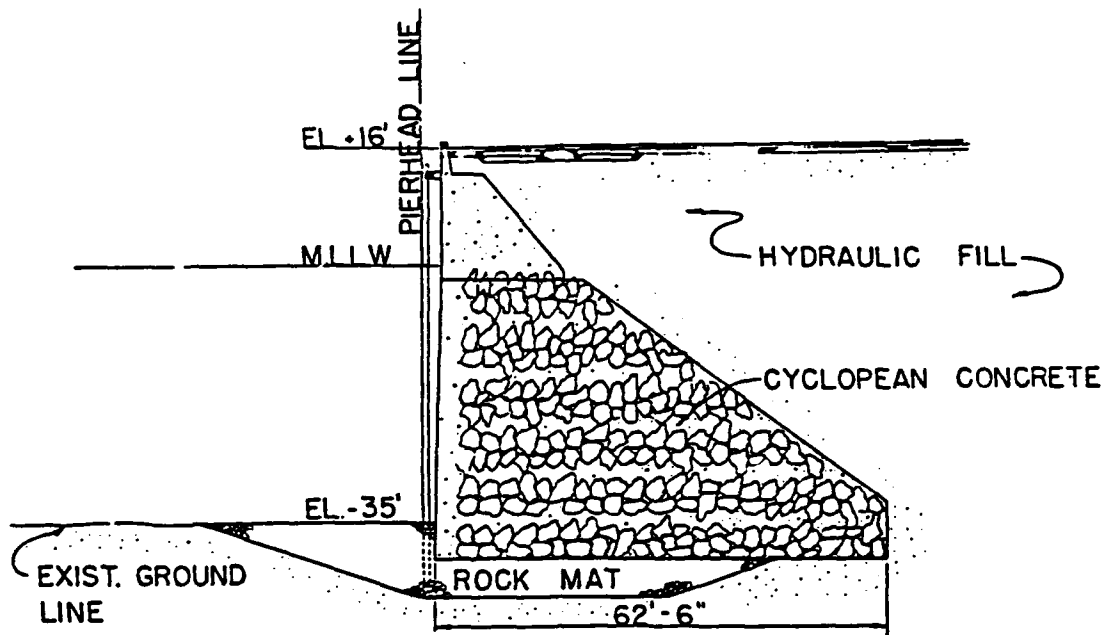


Figure 12. Concrete quay wall

PIER J LANDFILL EXPANSION

The Pier J landfill expansion is the next anticipated major landfill project in the Port (Figure 13). Work on this project is planned to begin in early 1987. This expansion will add 147 acres of new land to the seaward side of Pier J by placing 14 million cubic yards of dredged material behind multi-lift rock dikes (Figure 14). This construction will allow the Port to further increase its container handling capability by providing backland to an existing container terminal and adding four new berths and an eighth container terminal to the Port (Port of Long Beach 1985).

As a result of the landfill, it is believed that some valuable coastal marine habitat will be eliminated. The Port of Long Beach is undertaking a wetlands restoration project to mitigate for these losses to fisheries. A plan is under consideration to use the excavated material of the mitigation site in the landfill of the Pier J expansion.

PIER A EXPANSION

A recently completed landfill expansion in the Port is the Pier A Project. This \$50 million project added 24 acres of fill to the existing pier and cleared the adjacent land of transit sheds and warehouses, converting the breakbulk facility into an 88-acre container terminal. The landfill was supplied by the dredging of the Port's East Basin to a depth of 76 ft.

This expansion project involved the testing of a unique compaction technique. Following many years of laboratory experimentation and testing, a technique for compacting dredged fill materials as they are being placed has been developed. A full-scale field-testing program was conducted during the hydraulic filling performed as part of the Pier A Expansion Project.

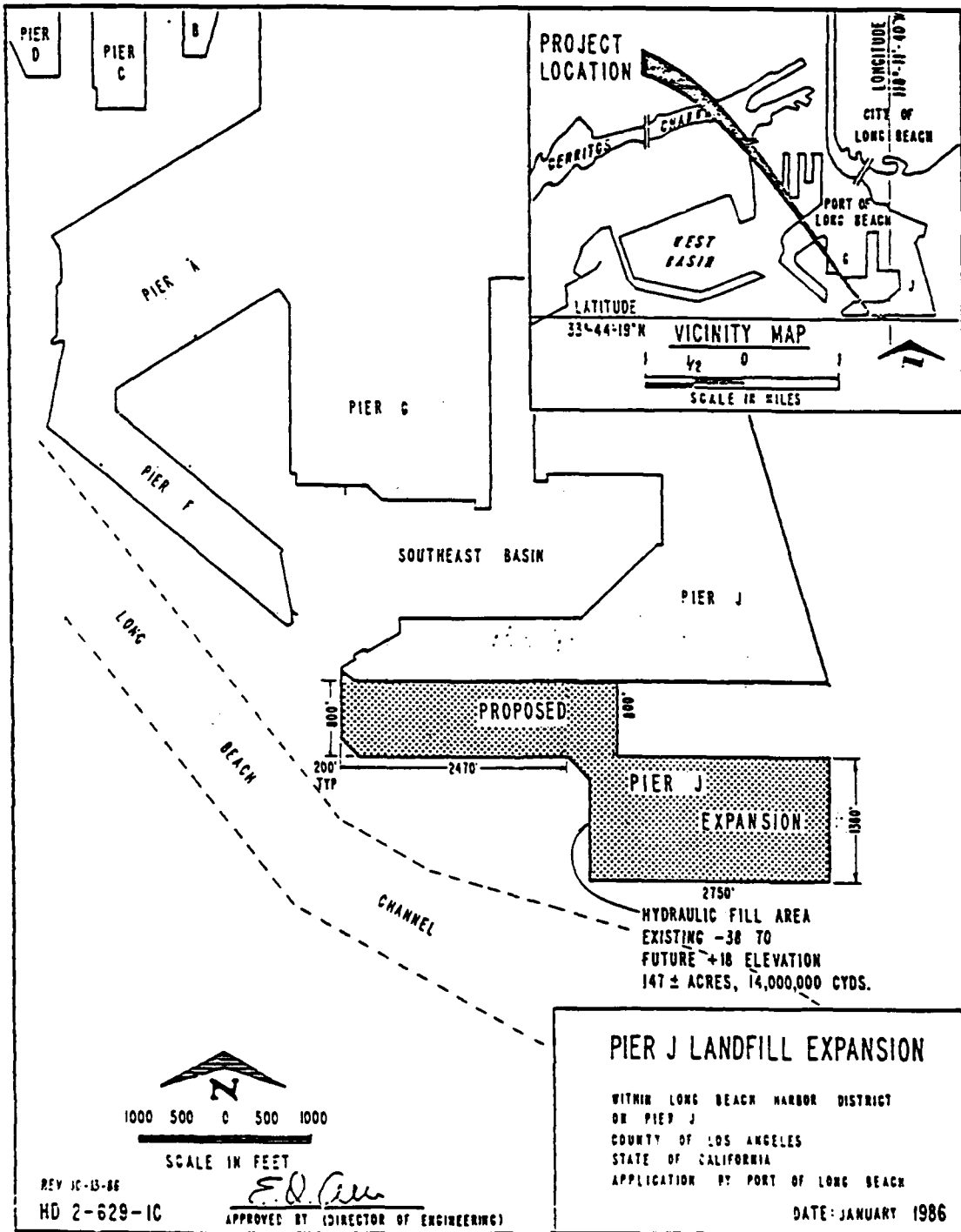
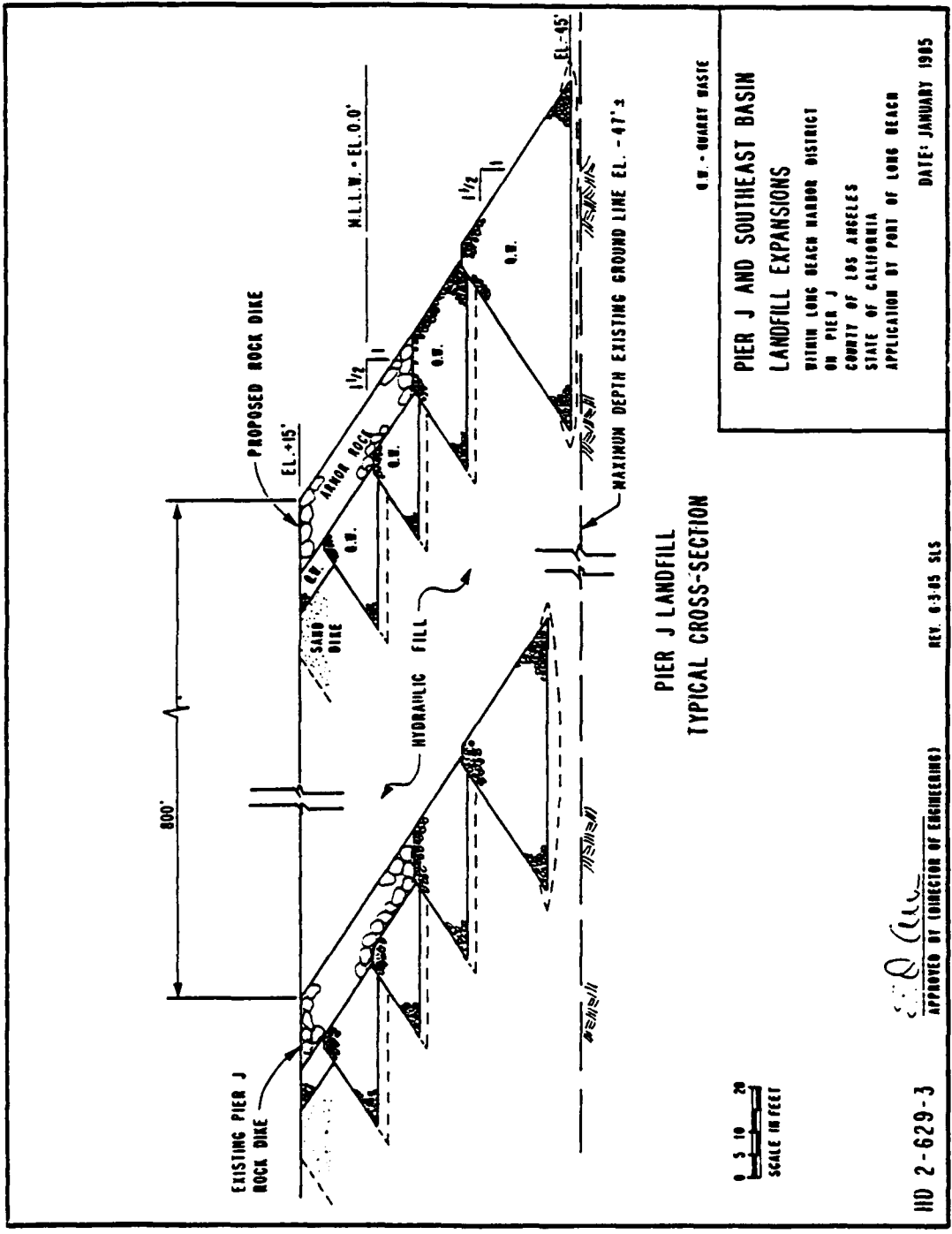


Figure 13. Pier J landfill expansion



PIER J AND SOUTHEAST BASIN
 LANDFILL EXPANSIONS
 WITHIN LONG BEACH HARBOR DISTRICT
 ON PIER J
 COUNTY OF LOS ANGELES
 STATE OF CALIFORNIA
 APPLICATION BY PORT OF LONG BEACH
 DATE: JANUARY 1995

HD 2-629-3
 APPROVED BY (DIRECTOR OF ENGINEERING)
 [Signature]
 REV. 6-3-85 SLS
 SCALE IN FEET
 0 5 10 20

Figure 14. Pier J landfill typical cross section

Cort T. Johnson was the consulting engineer for this test, and GEOFON, another engineering consulting firm, was retained by Mr. Johnson to independently assess the geotechnical effects of the compaction test program (Johnson 1985).

This new system has been developed to compact sand, either clean or containing limited amounts of silt, in lifts several feet in thickness during underwater placement. The method is similar to that of dry landfills where compaction is performed in intervals rather than at the completion of filling.

The system provides a method of compacting fill material which underlies successive lifts of dike rock in a multitiered perimeter dike (Johnson and Burke 1986). Figure 15 represents a typical perimeter fill cross section.

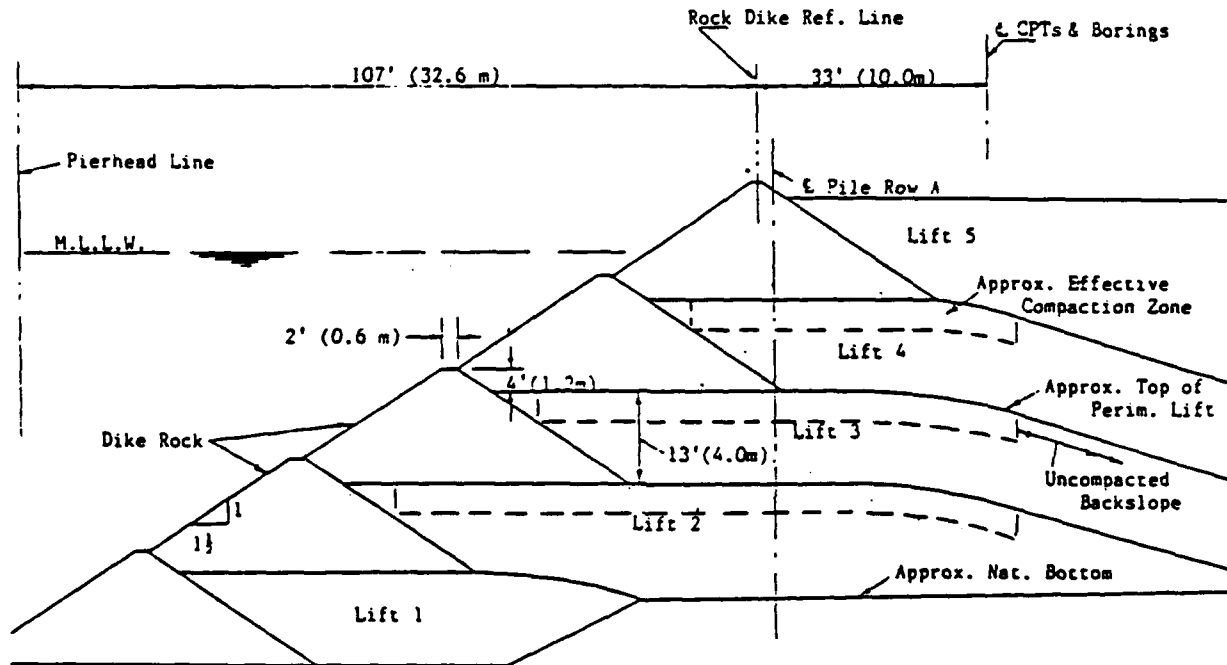
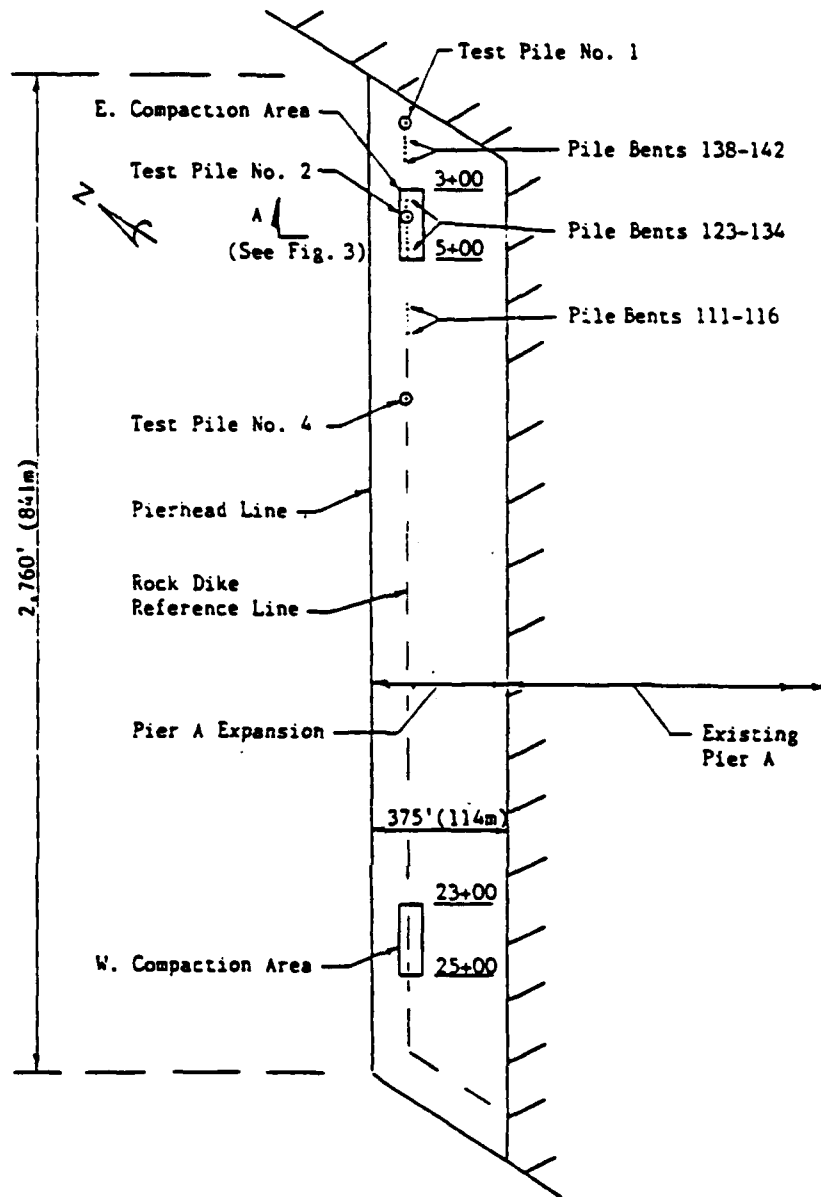


Figure 15. Dike cross section

The test was performed on the tops of three 13-ft lifts in a test area 200 ft long by about 100 ft wide, and two lifts in an area of the same size on the opposite end of the expansion site (Figure 16). The results of follow-up testing were most favorable, indicating that the relative densities of the compacted areas were improved 14 to 32 percent over the uncompacted areas (GEOFON, Inc. 1985).

It is believed by the Port of Long Beach that the effectiveness, practicality, and economy of the compaction system have been sufficient to warrant its consideration for specific future projects (Johnson and Burke 1986).

Compaction is critical to the Port of Long Beach for several reasons. The Port is situated in a seismically active area. Many experts believe that if the fill can be adequately densified, especially along the dike, then the potential for liquefaction to occur can be greatly reduced.



COMPACTION AREA LOCATIONS

Figure 16. Sand fill compaction area locations

The increase in soil strength caused by compaction is also a very important consideration. The Port plans to construct container terminals which require wharves capable of sustaining container stacking cranes and heavy truck loads. Liquid and dry bulk terminals also require high soil strength due to the heavy soil loads of the bulk warehouses and storage tanks. A cost savings also exists in wharf construction when the soil strength is greater under and immediately behind the dike. This extra strength allows for a steeper rock face and a narrower, less costly wharf (Johnson and Burke 1986).

The potential savings in dike rock by reducing its penetration into the supporting fill material have been estimated to amount to be about 7 percent.

This savings would approximately offset the entire estimated cost of compacting the perimeter fill, except for any added cost for placing the fill in thinner lifts (GEOFON, Inc. 1985). This estimated saving is based on the situation existing with the Pier A Expansion Project.

As mentioned earlier, the Port is running out of quality landfill material. By creating densified soil under perimeter dikes, the interior area of the landfill could be made up of a lesser quality landfill material; however, ideally, a sandy material would be used throughout.

The application of this underwater compaction system would be contingent upon the availability of sandy material for use in the landfill.

MITIGATION EFFORTS FOR HABITAT LOSSES

Port development, unavoidably, may permanently destroy habitat areas. The Port of Long Beach seeks to minimize these habitat losses within its boundaries whenever possible. Since there are no natural terrestrial habitats of significant value, most efforts in this area are focused on marine habitats (Port of Long Beach 1983).

The Port works with Federal and state wildlife agencies in determining appropriate mitigation projects. Trade-off ratios for various marine habitats and a formula for determining net biological impact from port developments are considered.

Mitigation for marine habitat losses does not occur within the Harbor District because this creation of a new habitat value may need to be destroyed in the future due to further port development. It has been determined that shallow, estuarine coastal embayment habitat in southern California, with its relatively high value to marine fishes and migratory birds, has been reduced in area at a greater rate than that of deepwater habitat (Board of Harbor Commissioners 1984). The Port, therefore, chooses to consider these shallow sites for mitigation purposes.

Port development and related mitigation projects consume and create "habitat value credits" which are banked and subsequently debited from and credited to, with balances being carried over to future projects. The Port entered into its first mitigation bank in March 1984 as a result of involvement with a two-phased restoration project totaling 30 acres in a location approximately 20 miles south of the Port called Newport Bay. The Newport Bay restoration site was a largely barren floodplain area above the reach of tides, providing minimal habitat value. The fishery and waterfowl habitat improvement at the site after restoration was predicted to be virtually 100 percent (Board of Harbor Commissioners 1984). The mitigation project is now completed, and preliminary indications are that the project has been successful.

The above-mentioned project was to mitigate habitat losses created by the 24-acre landfill expansion of Pier A. The expansion destroyed less marine habitat than was created by the Newport Bay restoration project; therefore, the Port now has a balance of 16 acres of habitat credits in its "bank."

In January 1986, the Port entered into its second mitigation banking agreement to provide mitigation for the 147-acre Pier J Project. This mitigation project consists of restoration of approximately 110 acres of the National Wildlife Refuge in Seal Beach Naval Weapons Station (Figure 17). This mitigation project is now in the design stage with construction costs estimated at \$6 million.

Three areas will be excavated to suitable elevations, and tidal influence will be restored to these areas. Natural colonization will then lead to the development of a wetland community in those areas. Approximately one half of the restored acreage will be permanently covered by water, providing valuable fish habitat and resting and feeding areas for many species of birds, among them the brown pelican. The remaining half of the area will consist of about 42 acres of intertidal mudflats and about 18 acres of isolated higher elevation areas covered by only the highest tides. The intertidal mudflats should develop infaunal invertebrate populations and serve as feeding areas for fish during high tides and for birds during low tides (Port of Long Beach 1985).

The task of finding new mitigation sites for future port development is a difficult one. A review of wetlands in southern California reveals that there are few places readily available for mitigation. Most of the larger areas that have been identified as being suitable for mitigation of port projects by the Federal and state wildlife agencies are either privately owned or military properties. At present, the Port is awaiting the outcome of state and local government negotiations with private property owners so that it can proceed with future port development.

The shortage of available mitigation sites is a serious problem; port expansion may come to a halt if the wildlife habitat restoration requirements are not met.

CONCLUSIONS

The Port of Long Beach has gone through many changes over the years, with many more changes planned for the future. The need for extensive future development of the Port has been investigated thoroughly and confirmed. In order for the plans to be carried out, however, the problems of a lack of suitable landfill and a scarcity of usable mitigation sites must be overcome. The Port is developing feasible solutions to these problems, and it will pursue implementation of these solutions with a high regard for design and construction, maritime service, and environmental conditions, as well as a consideration of local, national, and international interests.

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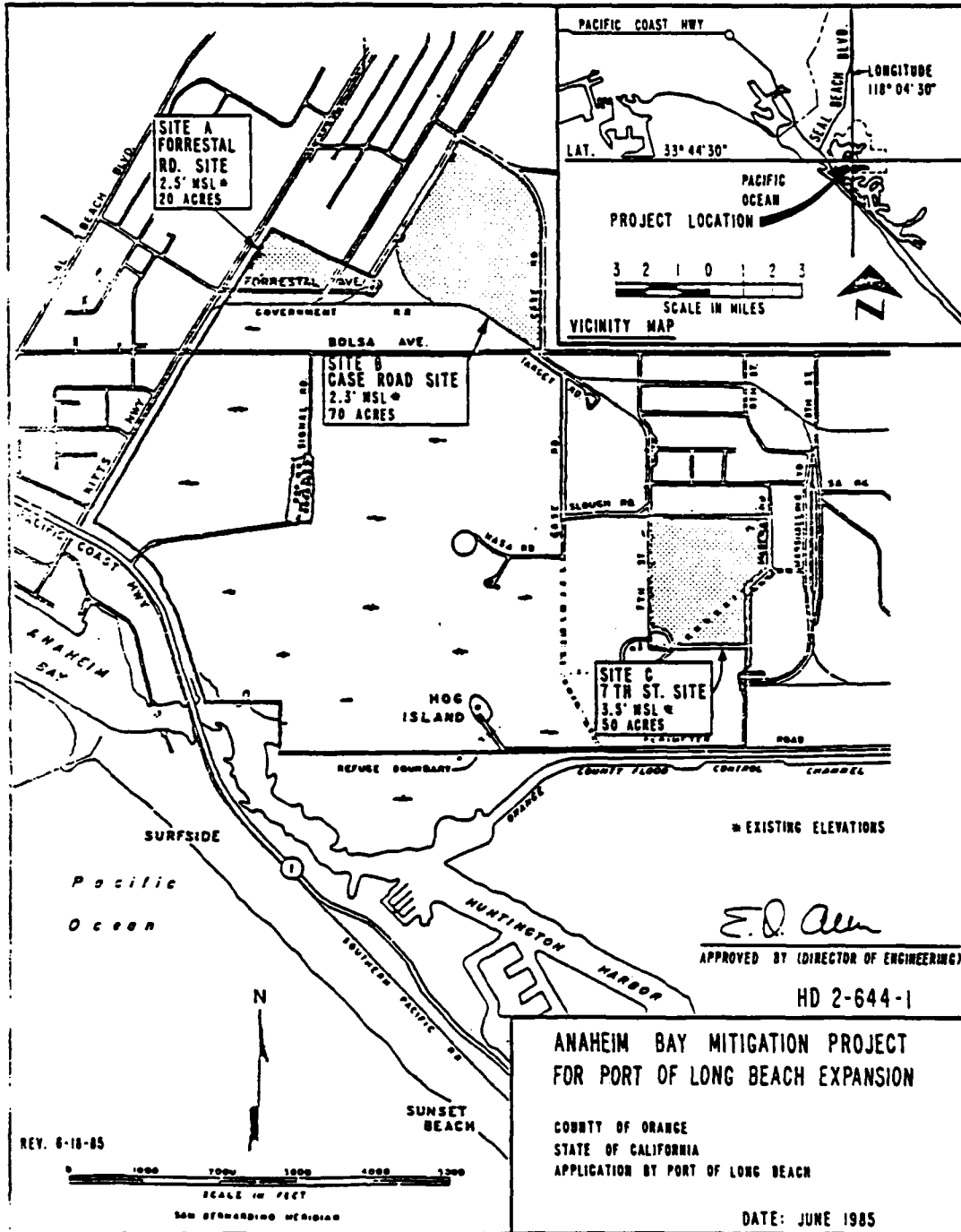


Figure 17. Anaheim Bay mitigation project

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AD-P006 912



IMPACT ASSESSMENT OF IN-PLACE CONTAMINATED SEDIMENTS
ON WATER QUALITY: A CONCEPT

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92-17645



ABSTRACT

In the past, environmental assessments of US Army Engineer Civil Works water resource projects that involve dredging of contaminated sediments have addressed only the consequences of dredged material disposal. The environmental benefits that may be incurred from the removal of in-place contaminated sediments have not been considered. The deficiency in evaluating the environmental consequence of sediment removal has been due to the lack of a logical delineated approach to quantifying the impacts of contaminated sediments on water quality as well as the reluctance of regulatory agencies to consider contaminated sediment removal as a benefit. This paper presents a conceptual approach developed for Civil Works operations and maintenance navigation projects for assessing the environmental consequences of in-place contaminated sediments. The approach and the potential benefits are delineated.

INTRODUCTION

The Corps of Engineers (CE), as one of the Nation's major construction agencies, is tasked with the responsibility for the construction and maintenance of over 25,000 miles of Federal channels, over 100 commercial harbors, and 400 small boat harbors. The CE annually removes 250 to 300 million cubic yards of dredged material. A small, although significant, portion of the sediments dredged are contaminated and require special handling and disposal. These sediments may contain a multitude of toxic materials, including polyaromatic hydrocarbons (PAH's), polychlorinated biphenyls (PCB's), heavy metals, pesticides, and by-products of industry. These contaminated sediments are normally associated with the Nation's urban industrial waterways, where Federal channels act as catchment basins for sediments.

In constructing and maintaining these channels, the CE must comply with Federal and state environmental legislation and evaluate the environmental consequences of dredging and disposal of these sediments. In evaluating the environmental impacts of dredging and disposal of contaminated sediments, the major emphasis has been on the short-term impacts on water quality from dredging and the short- and long-term impacts on the environment from disposal.

Minimum attention has been given to either the environmental effects on the water body of contaminated sediments in Federal channels or the environmental benefits that may accrue from sediment removal.

In the past, as well as today, strong public opinion is often expressed against the removal and disposal of contaminated sediments. These objections are based on the belief that the environmental consequence of sediment removal and subsequent disposal exceed the social and economic benefits gained from the project. Although polluted sediments are often located in highly industrial and urban areas, they are neither stationary nor inert. These sediments continue to exert an influence on the water quality of the waterway through oxygen demand and release of nutrients and toxic substances, as well as supporting only a limited pollutant-tolerant community of poor species diversity. Further, the resuspension of these sediments by natural or man-induced causes will not only affect the overlying water column, but may also impact water quality for miles outside the immediate area of concern.

Polluted sediments are not confined to Federal channels and major harbors in our urban and industrial areas. Spills from chemical and petroleum barges and small chemical and petroleum terminals have resulted in localized pockets of contaminated sediments in areas far removed from our major metropolitan areas. Runoff from agricultural and timber lands also introduces pesticides and nutrients to the Nation's waters, often resulting in river, reservoir, and estuary sediments becoming contaminated. Point discharges from feedlots, chemical companies, and city sewage treatment plants have also been a source of nutrients and contaminants that have affected sediments. Many of these polluted sediments are located within Federal water resource projects built and maintained by the CE. The effects of these sediments on the continued operation and maintenance of these projects, as well as proposed modifications to current use, can be significantly affected by the presence of contaminated sediments.

For a more complete assessment of cost versus benefits associated with maintaining specific water resource projects, the impacts of in-place sediments on the environment must be considered. If the CE can determine and quantify the impacts, the "true" cost of the project's social, economic, and environmental attributes can be determined.

This paper presents an approach that can be used in the planning and assessment of water resource projects to evaluate the environmental impact of contaminated sediments on water quality. This approach will be presented along with a list of potential activities and associated benefits.

BASIC CONCEPT

The basic idea is to allow the project planner the ability to identify and quantify environmental impacts from in-place sediments on the water quality of the water body, thereby allowing for an accurate evaluation of the project cost versus benefits from removing the existing sediments. Various factors will influence this assessment, including sediment characteristics, concentration and type of contaminants in the sediments, system hydrodynamics, contaminant sources, and existing and proposed uses of the water body. These and other factors will significantly affect the identification and quantification of impacts. Although many of these factors can be determined through

existing data and information, others will require data collection and analysis. Without a delineated approach that provides a logical and systematic procedure to assess in-place sediments, the results may be of limited value due to misdirected efforts or inadequate data and analysis.

ACTIVITIES

Assessment of in-place contaminated sediments has a number of applications. Listed below are potential applications where the CE has direct responsibility or is involved as a cooperating agency.

Federal operation and maintenance navigation
Permit activities
Superfund sites
Reservoirs
Toxic spills and hotspots

BENEFITS

Five basic benefits can be incurred from the assessment of in-place sediments. These benefits, although not associated with each of the above-cited activities, will provide for an improvement in the CE's ability to construct and maintain water resource projects. These basic benefits are:

- a. Quantify existing conditions, thereby providing a baseline for decisionmaking.
- b. Provide a baseline to assess the potential benefits and costs associated with each proposed alternative.
- c. Quantify project benefits incurred from removal or modification of in-place contaminated sediments.
- d. Provide for quantification of project costs and benefits gained for the proposed action.
- e. Provide a baseline for assessing not only short-term but long-term benefits and costs from various long-term management strategies.

APPROACH

The conceptual process of developing and implementing an assessment protocol for in-place sediments is presented as a three-phase approach. Each phase consists of a series of steps or essential activities that lead to the development of specific data and information necessary to determine the impact of the sediments on water quality. The intent of this process is to quantify costs and benefits associated with specific alternatives as they relate to impacts from in-place contaminated sediments.

Phase I

Phase I is intended to serve as the first level of assessment. At a minimum, this phase requires the defining of assessment objectives (e.g., cleanup and alternative evaluation), thereby determining the level of data requirement

and analysis for decisionmaking. Once the objectives for the assessment are defined, the next step is the identification and collection of existing data. There is usually a wealth of information available from various Federal and state sources. The intent under Phase I is to minimize field data collection activities; so, a decision is needed as to the sufficiency of the existing data for evaluating the impact of in-place sediments. If existing data are insufficient, data gaps are identified, validated, and screened based on factors such as potential for development and time and resources needed to fill the gaps. If the needs are valid, a data collection effort is planned. Unvalidated requirements result in either no further evaluation of the in-place sediments or a reassessment of the study objectives. Once the initial data requirements are met or, if necessary, additional data requirements identified, Phase I is completed.

Phase II

Phase II consists of collecting required additional data and conducting a data analysis of sufficient detail to evaluate the impact of in-place sediments as they relate to the objectives delineated under Phase I.

Phase III

Phase III uses the results obtained under Phases I and II to conduct the detailed evaluation as required to address the study objectives. This process will occur in conjunction with alternative evaluations that consider engineering, economic, social, and environmental cost and benefits.

Alternatives

In addition to Phases I through III, consideration must also be given to implementation of any alternative or proposed actions and monitoring of the results. Without a feedback mechanism, the validity of the analysis of both the impact of in-place sediments on water quality and the benefits from their removal or alteration cannot be fully known.

SUMMARY

The paper presents a concept that can be used by CE planners to identify and quantify environmental impacts on a water body from in-place contaminated sediments. This concept is applied through a three-phase approach that allows quantification of cost and benefits associated with in-placed sediments and proposed actions. By using a three-phase approach that incorporates the maximum use of existing data during Phase I, cost for field data collection and analysis can be kept at a minimum. Further, by using a phase approach, decision points can be incorporated during each phase for assessing proposed benefits to be gained from collection and analysis of additional data which may be disproportionate to the incurred cost.

PRELIMINARY STUDY OF THE ECOLOGICAL STRUCTURE OF
BENTHIC ORGANISMS IN TOKYO BAY

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Second District Port Construction Bureau
Ministry of Transport

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Japan Bottom Sediments Management Association

ABSTRACT

The benthic organisms and environmental conditions of Tokyo Bay were surveyed to determine the ecological structure of the bottom layer of the Bay. The results of the surveys showed that water and bottom sand qualities tended to worsen toward the inner part of the Bay, and benthic fauna became more simplified. A study on the correlation between benthic organisms and environmental factors such as chemical oxygen demand and total sulfide of the bottom deposits indicated a specific range of environmental conditions for each benthic organism. The result of the trawling survey indicated a relatively simple food chain relationship at the innermost part of the Bay: bottom detritus → polychaeta → marbled sole. The relationship tended to be more complicated toward the outer part of the Bay.

BACKGROUND

Tokyo Bay is a calm water area of 120,000 ha with an average depth of 15 m situated at the center of Honshu, the mainland of Japan. Figure 1 shows the geographical features of the Bay. The metropolitan area with its population of 28 million surrounds the Bay. The Bay serves as fishing grounds, navigation routes, and recreation.

The environmental condition of the Bay has been deteriorating with the rapid expansion of the metropolitan area, especially since 1959 (Figure 2) though many countermeasures such as extension of the sewage systems have been taken.

Chemical factors have been primarily used in the environmental assessment; few attempts have been made to assess changes in the environmental condition from the biological point of view. In the present study, a biological approach was taken to tackle the environmental problem in the Bay. Ecological aspects of benthic organisms in the Bay were studied as the first step.

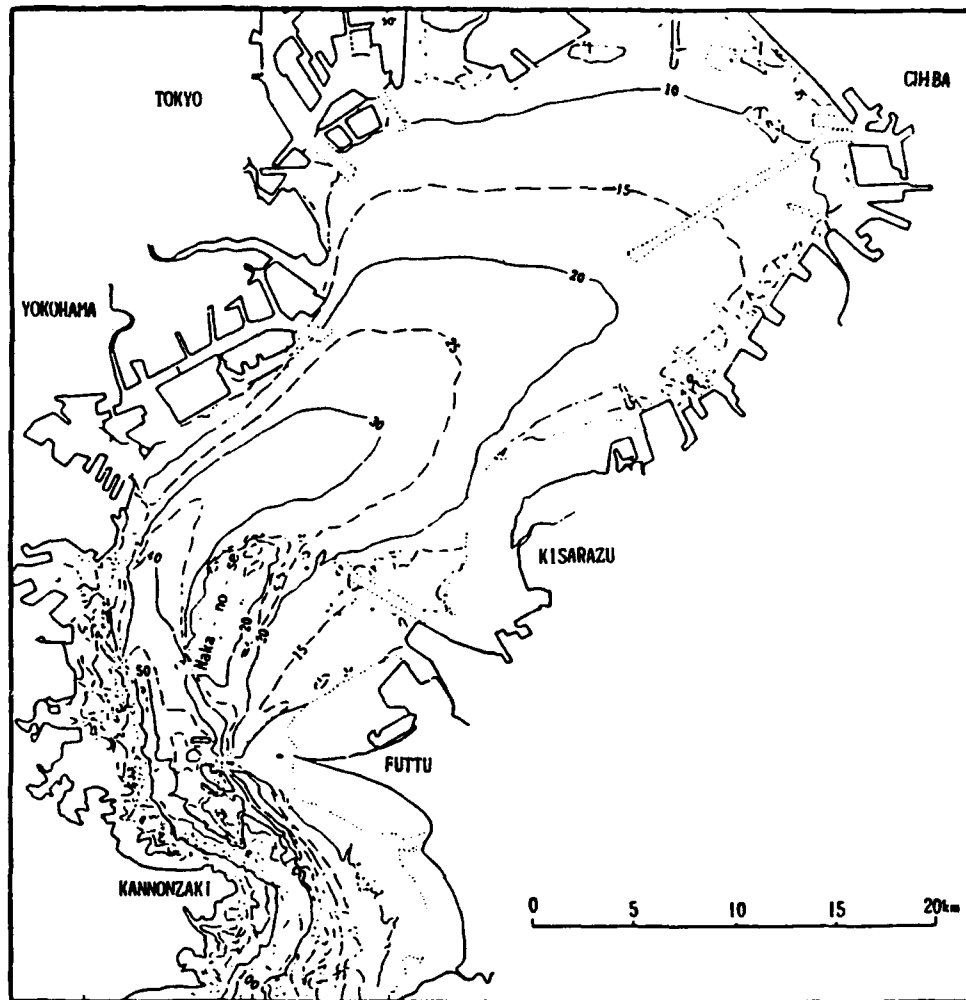


Figure 1. Geographical features of Tokyo Bay

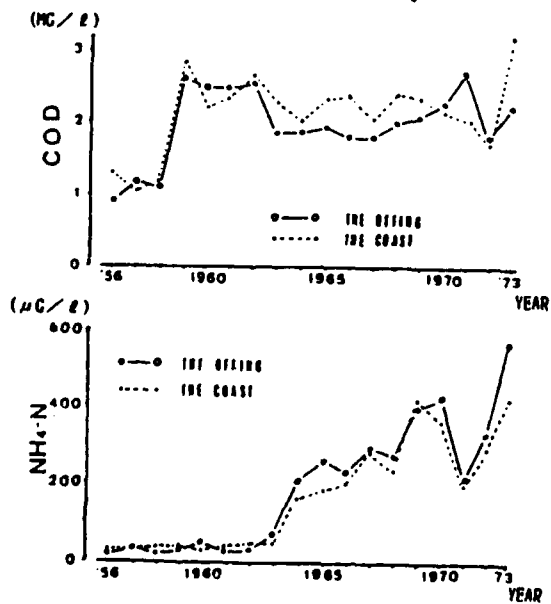


Figure 2. Changes in water quality of Tokyo Bay

METHODS AND RESULTS

Field surveys were made on water and bottom qualities and on benthic organisms at four stations: the innermost part of the Bay, the central part of the Bay, the mouth of the Bay, and the tidal flat. The depth was 10 m for the first three stations (Figure 3). Figure 4 illustrates the survey method, and Figure 5 shows the flowchart of analysis of collected data. Surveys were carried out from 21-26 January 1986.

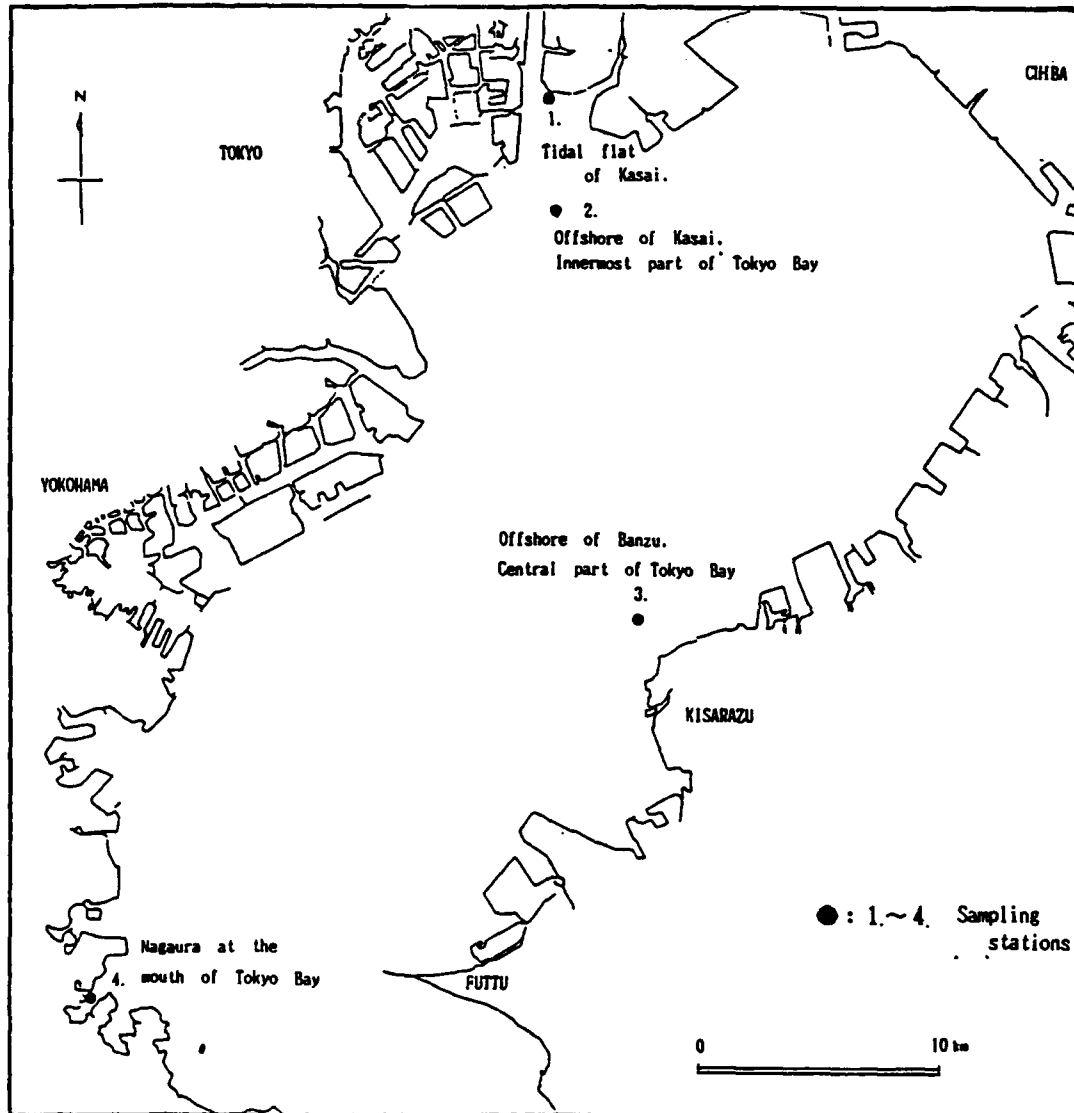


Figure 3. Survey stations

Tables 1 and 2 show the results of the surveys on water and bottom qualities.

For meiobenthos and macrobenthos collections, bottom sediments sampled by a core sampler were sifted through sieves (mesh size: 105 μ for meiobenthos and 1 mm for macrobenthos). Organisms found were identified, counted, and weighed (wet weight). Results are shown in Table 3.

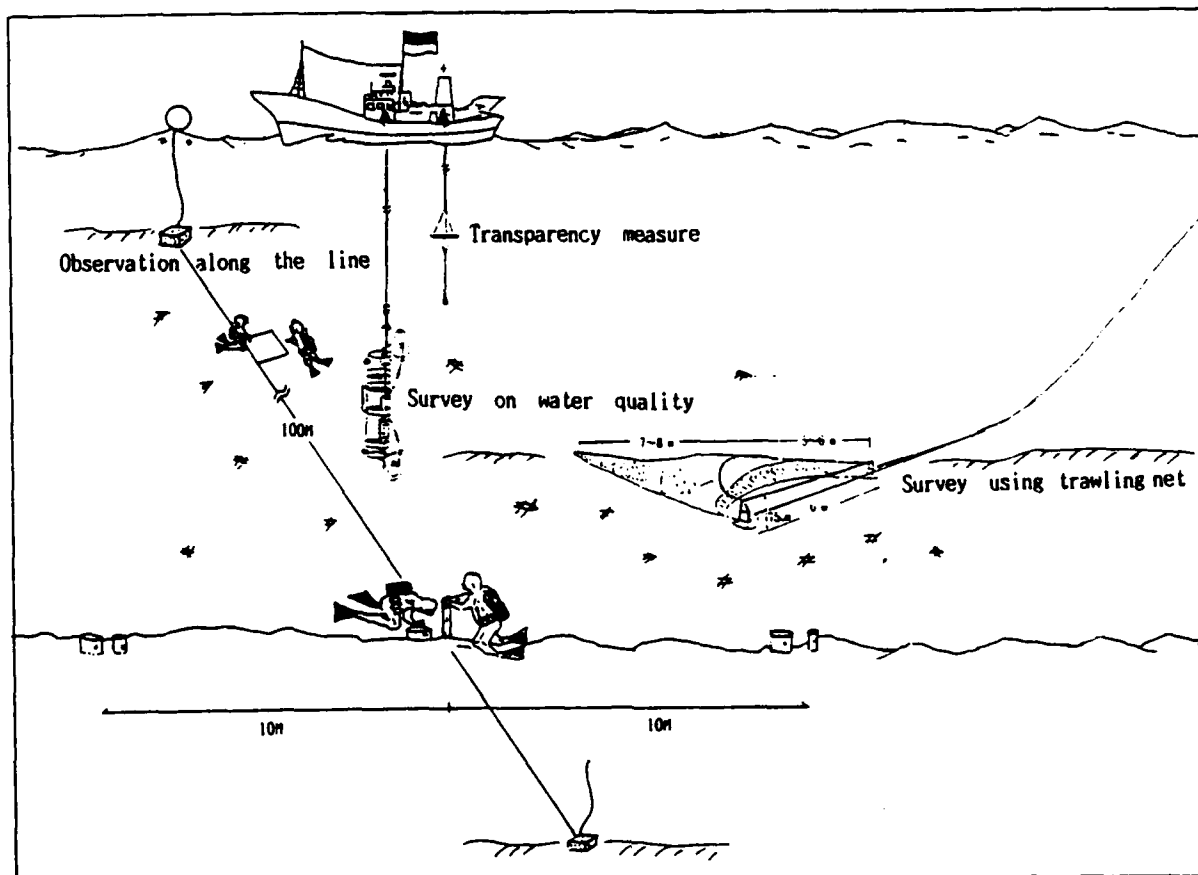


Figure 4. Schematic diagram of the field survey

Megalobenthos and benthic fish were collected with a trawling net to study species composition, standing stock of each species, and their food habits. The organisms collected were identified, counted, and weighed (wet weight). For benthic fish, stomach contents were also analyzed to study food chain relationships between other bottom organisms. Table 4 shows the dominant species of fish collected at each sampling station. Food habits of representative fish species are shown in Table 5.

ANALYSIS

Relationship Between Environmental Factors and Organisms

The correlations were studied between three properties of bottom sand: median value of sand particle diameter, COD, and T-S (Table 6). Some of these are illustrated in Figure 6. Among the dominant species of macrobenthos and meiobenthos, the species with open circles in Table 6 are dominant only in limited ranges (shown in parentheses) of bottom sand properties. Positive or negative correlations were observed between some items and the bottom sand properties and are indicated with + or - in Table 6.

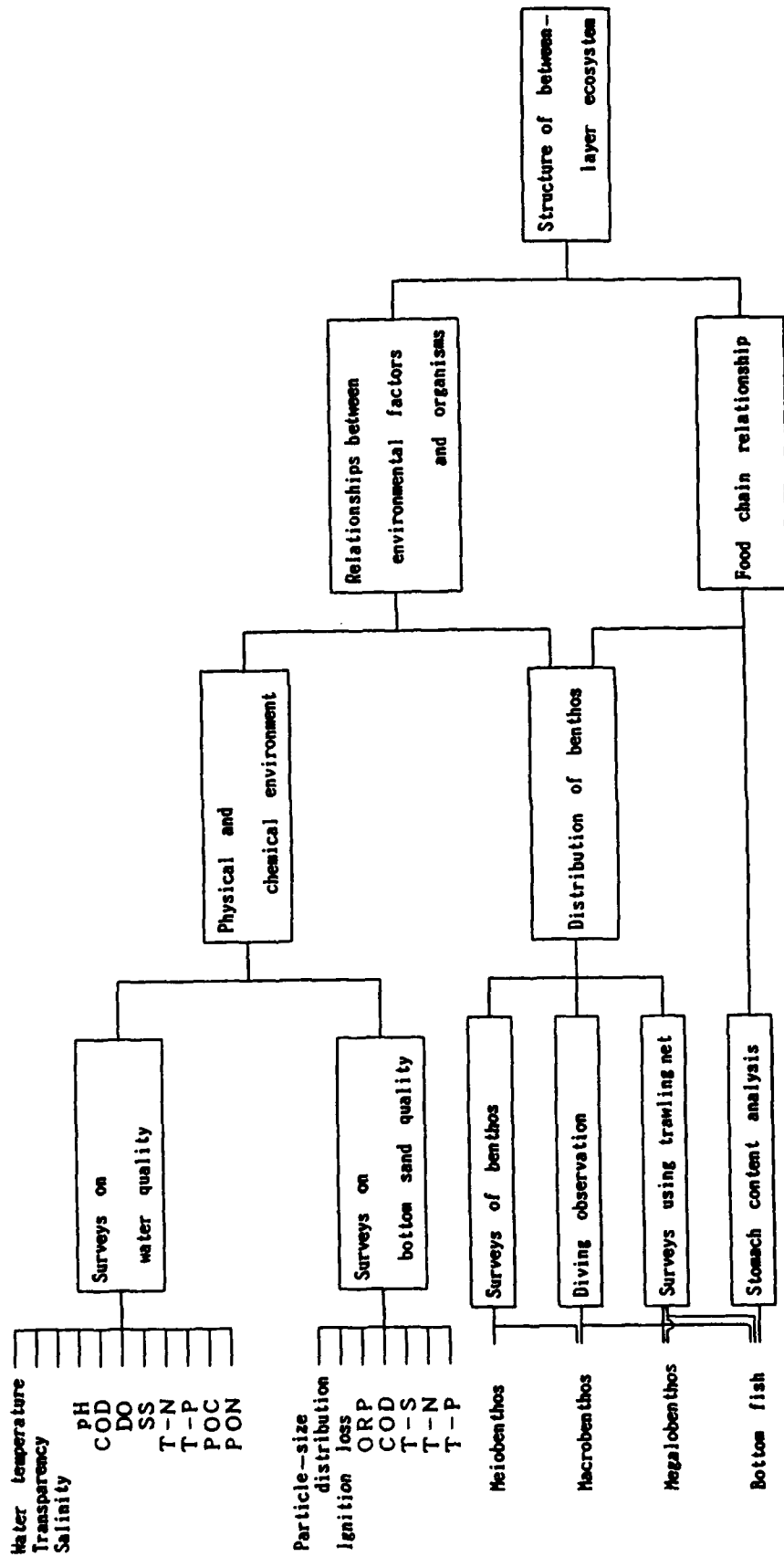


Figure 5. Flowchart of analysis of collected data

TABLE 1. WATER QUALITY (DATE OF THE SURVEY: 21-26 JAN. 1986)

Area	Depth m	Trans- parency m	Water temp. °C	Salinity	pH	COD mg/l	DO mg/l	SS mg/l	T-N mg/l	T-P mg/l	POC mg/l	PON mg/l
Tidal flat	1.0	1.8	8.4	30.38	8.3	3.4	11.3	14	1.52	0.095	1.17	0.19
	1.8		8.4	30.37	8.3	2.9	11.1	7	1.45	0.100	1.23	0.21
	2.6		9.4	30.72	8.3	2.8	10.1	7	1.46	0.090	1.05	0.15
Inner- most	1.0	1.9	8.1	30.61	8.4	3.1	11.6	7	1.55	0.090	1.07	0.18
	5.7		8.1	31.99	8.5	2.9	12.9	9	1.05	0.050	0.96	0.14
	10.3		10.7	33.25	8.1	2.2	7.0	4	0.76	0.064	0.71	0.10
Central	1.0	2.2	9.0	32.55	8.2	2.1	11.5	5	0.72	0.057	0.53	0.06
	4.2		9.0	32.57	8.3	2.2	10.5	3	0.79	0.054	0.70	0.11
	7.3		9.0	32.44	8.3	2.2	10.6	4	0.70	0.055	0.80	0.13
Mouth	1.0	3.7	8.6	32.56	8.3	1.7	10.7	4	0.78	0.062	0.59	0.09
	5.2		8.6	32.59	8.3	1.7	10.0	4	0.77	0.058	0.87	0.13
	9.3		8.6	32.58	8.3	1.7	10.7	5	0.78	0.053	0.74	0.13

TABLE 2. BOTTOM QUALITY (AVERAGE OF THREE SAMPLES)

Area	Median of particle mm	Ignition loss %	ORP, mV		COD mg/g*	T-S mg/g*	T-N mg/g*	T-P mg/g*
			0-5 cm	5-10 cm				
Tidal flat	0.21	1.7	-94	-95	2.5	0.06	0.21	0.33
Inner- most	0.0042	9.5	-104	-120	32.5	1.74	1.85	0.84
Central	0.24	2.5	-136	-150	7.0	0.09	0.73	0.18
Mouth	0.073	7.2	-103	-113	23.0	0.53	1.77	0.43

* Dry weight

The dominant species of macrobenthos and meiobenthos appeared in specific ranges of bottom sand properties. They are classified as follows according to the eutrophication scale by Yoshida (1983):

- a. Mainly in saprobic areas (COD: more than 30 mg/g, sulfide content: more than 1.0 mg/g):

Paraprionospio pinnata (Figure 6a)
Foraminiferida
Prionospio cirrifera

TABLE 3. RESULTS OF BENTHOS SURVEY*

Unit: macrobenthos; per 0.28m²
meiobenthos ; per 30cm²

Area	Number of species	Number of indiv.	Wet weight	Dominant Species (ratio)
Macrobenthos				
Tidal flat	38	1,428	51.58 g	<u>Armandia sp.</u> (25.1%) <u>Capitellidae</u> (15.2%) <u>Ancistrosyllis hanaokai</u> (10.4%)
Inner-most	15	1,391	48.58 g	<u>Paraprionospio pinnata</u> (92.4%)
Central	42	5,862	1066.72 g	<u>Musculus senhousia</u> (94.3%)
Mouth	44	642	27.34 g	<u>Lumbrineris longifolia</u> (36.4%) <u>Polydora sp.</u> (7.8%) <u>Cirratulidae</u> (6.2%)
Meiobenthos				
Tidal flat	10	5,038	33.4 mg	Nematoda (91.8%)
Inner-most	7	2,527	44.9 mg	Nematoda (80.8%) Foraminiferida (14.0%)
Central	15	2,128	26.4 mg	Nematoda (88.3%) Harpacticoida (7.1%)
Mouth	16	1,600	26.8 mg	Nematoda (79.2%) Harpacticoida (14.1%)

TABLE 4. DOMINANT SPECIES COLLECTED BY TRAWLING

Area	Fish	Other Organisms
Tidal flat	<u>Limanda yokohamae</u>	<u>Ciona robusta</u> , etc.
Innermost	<u>Limanda yokohamae</u>	<u>Styela plicata</u> , etc.
Central	<u>Limanda yokohamae</u> <u>Repomucenus valencienni</u>	<u>Mytilus edulis</u> , etc.
Mouth	<u>Leiognathus nuchalis</u> <u>Acanthogobius flavimanus</u> <u>Ditrema temmincki</u>	<u>Crangon affinis</u> , etc.

TABLE 5. STOMACH CONTENTS OF THE DOMINANT FISH SPECIES COLLECTED

Area	Fish	Stomach Contents
Tidal flat	<u>Limanda yokohamae</u>	<u>Neanthes japonica</u> Other Polychaeta
Innermost	<u>Limanda yokohamae</u>	<u>Paraprionospio pinnata</u> Other Polychaeta
Central	<u>Repomucenus valenciennesi</u> <u>Hexagrammos otakii</u> <u>Limanda yokohamae</u>	<u>Theora lata</u> Sardine and other fish Sepiolidae <u>Paraprionospio pinnata</u> <u>Lumbrineris</u> sp., etc.
Mouth	<u>Repomucenus valenciennesi</u> <u>Acanthogobius flavimanus</u> <u>Ditrema temmincki</u>	<u>Theora lata</u> , etc. <u>Theora lata</u> Polychaeta, etc. Gammaridea Caprellidea

- b. Mainly in hypertrophic areas (COD: more than 25 mg/g, sulfide content: 0.3-3.0 mg/g):

Lumbrineris longifolia (Figure 6b)

- c. Mainly in eutrophic areas (COD: 5-25 mg/g, sulfide content: 0.03-0.3 mg/g):

Musculus senhousia (Figure 6c)

- d. Mainly in oligotrophic areas (COD: less than 5 mg/g, sulfide content: less than 0.03 mg/g):

Ancistrosyllis hanaokai
Aonides sp. (Figure 6d)
Cirriiformia sp.
Capitellidae

The number of species (Figure 6e) and species diversity index (Figure 6f) tend to decrease with increments of COD and T-S. As the bottom sand becomes seriously polluted with organic matter, common bottom organisms disappeared and, instead, highly tolerant species such as *Paraprionospio pinnata* become dominant, thereby simplifying the fauna of the area.

Estimation of Food Chain Relationships

Prey-predator relationships at each sampling station are estimated from the results of the surveys (Figure 7). In the figure, the thickness of lines represents frequencies of relations.

The fundamental food chain relationship at the innermost part of the Bay was that of marbled sole (*Limanda yokohamae*) on polychaetes such as

TABLE 6. CORRELATIONS BETWEEN BENTHOS AND BOTTOM SAND PROPERTIES

Item		Median of Particle Size mm	COD mg/g	T-S mg/g
Dominant species of macro-benthos	<u>Ancistrosyllis hanaokai</u>	○ (0.20-0.21)	○ (less than 4)	○ (less than 0.1)
	<u>Nectoneanthes latipoda</u>	x	x	x
	<u>Lumbrineris longifolia</u>	○ (0.05-0.10)	○ (15-30)	○ (less than 0.7)
	<u>Aonides sp.</u>	○ (0.21)	○ (less than 5)	○ (less than 0.4)
	<u>Paraprionospio pinnata</u>	○ (less than 0.01)	○ (more than 30)	○ (more than 1.0)
	<u>Cirriformia sp.</u>	x	-	○ (less than 0.6)
	Capitellidae	○ (0.20-0.21)	○ (less than 4)	○ (less than 0.2)
	Sabellidae	x	x	x
Dominant species of meio-benthos	<u>Musculus senhousia</u>	○ (0.23-0.25)	○ (3.5-15)	○ (0.05-0.3)
	<u>Ophiura linbergi</u>	○ (0.05-0.10)	○ (15-26)	○ (0.3-0.6)
	Foraminifera	○ (less than 0.01)	○ (more than 30)	○ (more than 1.0)
Macro-benthos	Nematoda	○ (0.15-0.25)	x	x
	<u>Prionospio cirrifera</u>	-	+	○ (0.2-2.0)
	Harpacticoids	x	○ (less than 30)	○ (less than 0.7)
Meio-benthos	Number of species	+	-	-
	Number of individuals	+	x	x
	Wet weight	+	x	x
Indexes for macro-benthos	Number of species	x	x	x
	Number of individuals	x	x	x
	Wet weight	x	x	x
Indexes for macro-benthos	Biotic index	x	x	x
	Average diversity index; D_n	x	x	x
	Relative diversity index; D_m	x	x	x
	Total diversity index; I	+	-	-

NOTE: + = positive correlation; - = negative correlation; ○ = dominant species only in limited ranges (shown in parentheses); x = ?

Indexes are calculated as follows: Biotic index = N/S $D_n = \frac{1}{N} \log_e \frac{N!}{n_1! n_2! \dots n_S!}$

$$D_m = \frac{1}{N \log_e S} \log_e \frac{N!}{n_1! n_2! \dots n_S!} = \frac{D_n}{\log_e S} \quad I = \log_e \frac{N!}{n_1! n_2! \dots n_S!} = N \times D_n$$

Where N = total number of individuals, n_i = number of individuals of i-th species, S = number of species. All numbers are per 0.09 m².

Neanthes japonica and *Paraprionospio pinnata* which feed on organic deposits on the bottom. However, at the central and mouth sections of the Bay, fat greenling (*Hexagrammos otakii*) feeding on nektons and surf fish (*Ditrema temmincki*) feeding on crustaceans appeared as important predators, and the variety of preys increased accordingly. It is clear that predators and their food habits become more diversified from the innermost to the mouth of the Bay.

For megalobenthos different animals with different food habits were also observed at the innermost and outer sections of the Bay. At the innermost Bay, mussel (*Mytilus edulis*) and sea squirts (Ascidacea) which fed on suspended matters were abundant, whereas starfish, shrimps, and crabs with carnivorous or omnivorous food habits appeared at the central and mouth sections of the Bay.

The route of the organic matter and nutrient circulation system is proposed to be: organic matter deposits at the bottom → macrobenthos →

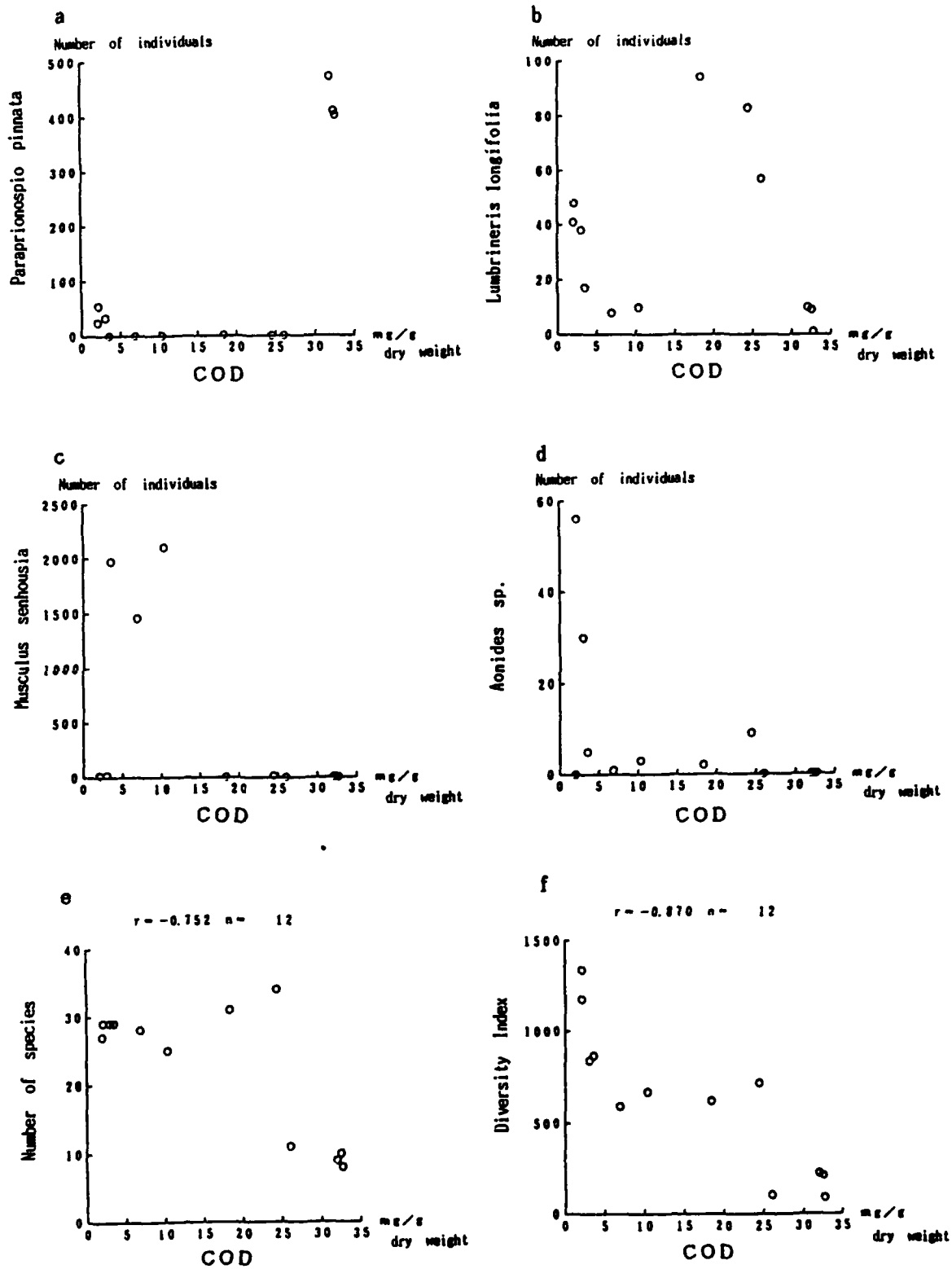


Figure 6. Correlation between benthos and bottom sand properties

benthic fishes such as marbled sole (*Limanda yokohamae*), store flounder (*Kaeius bicoloratus*), whipfin dragonet (*Repomucenus valenciennesi*) and goby (*Acanthogobius flavimanus*) in Tokyo Bay.

FUTURE PROSPECTS

Few ecosystem studies have been made on Tokyo Bay. The present study was conducted to determine the structure of the benthic ecosystem in the Bay as part of an integrated assessment of the area.

Biological surveys are influenced by more sources of errors in sampling procedures or in dealing with the results, when compared with physical or chemical surveys. Thus, the first step to be taken would be to decrease errors through repeating surveys. In practice, sampling should be done more frequently to decrease the error due to time factors. Most benthic organisms propagate in spring, and low oxygen content of the water mass occurs in the summer in Tokyo Bay. It would be necessary, therefore, to closely repeat surveys in these seasons, together with the year-round surveys.

Indicator species corresponding to a certain ecological feature may be established based on the results of the present study and other surveys.

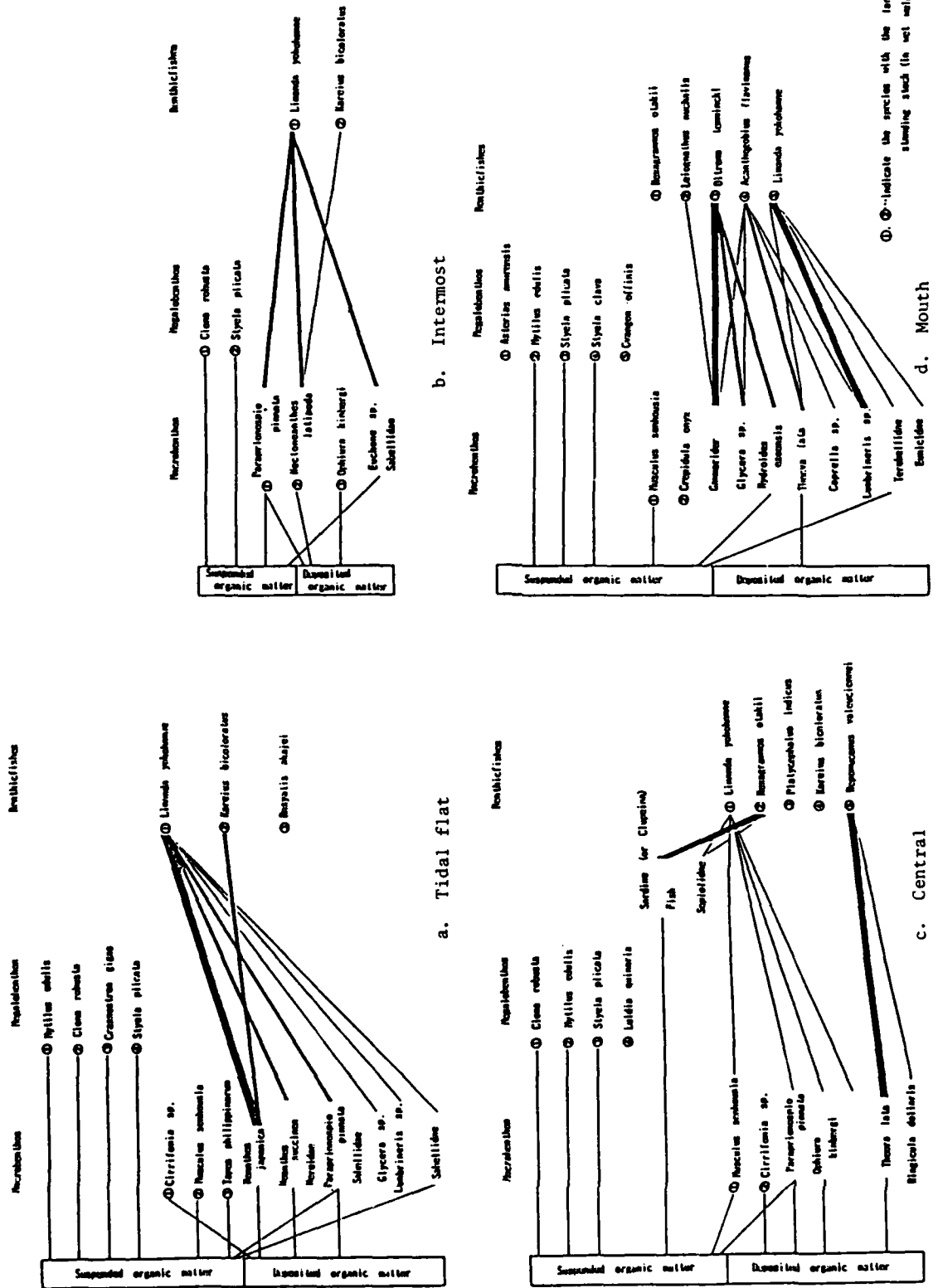


Figure 7. Estimated prey-predator relationships