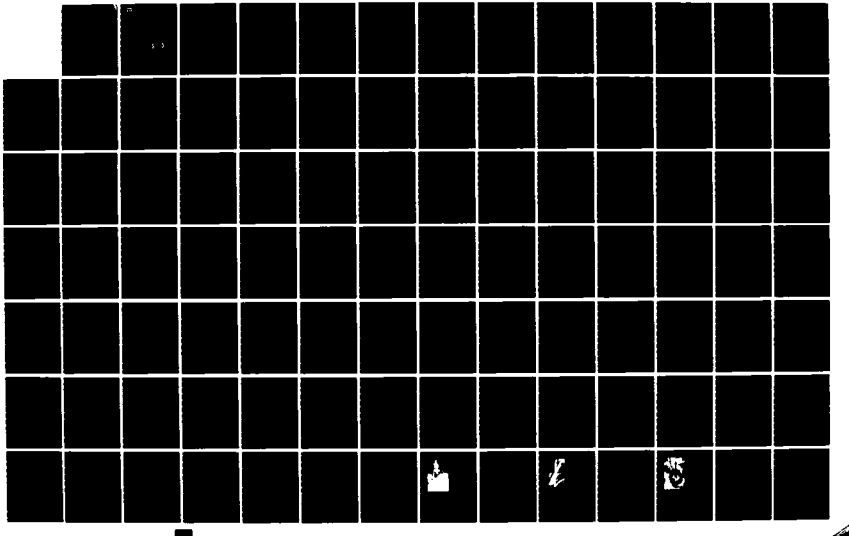
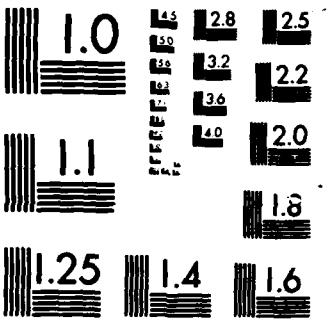
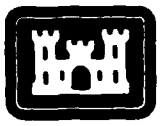
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US Army Corps
of Engineers
Water Resources
Support Center

Management of Bottom Sediments Containing Toxic Substances

Proceedings of the 10th U. S./Japan
Experts Meeting

30-31 October 1984
Kyoto, Japan

Thomas R. Patin, Editor

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The 10th U. S./Japan Meeting on Management of Bottom Sediments Containing Toxic Substances was held 30-31 October 1984 in Kyoto, 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.

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PREFACE

The 10th U. S./Japan Experts Meeting on Management of Bottom Sediments Containing Toxic Substances was held 30-31 October 1984 in Kyoto, 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. Ikuhiko Yamashita, Ministry of Transport, Tokyo, Japan, was the Japanese Chairman.

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

10th ANNUAL MEETING
U.S./JAPAN EXPERTS MEETING

U. S. Delegation

COL George R. Kleb	U. S. Co-Chairman, Corps of Engineers, Water Resources Support Center
Mr. W. Frank Bohlen	Associate Professor, Department of Marine Sciences, University of Connecticut
Mr. Jack H. Clifton	Japan District, Corps of Engineers
Mr. Charles C. Calhoun	Program Manager, Waterways Experiment Station, Corps of Engineers
Mr. Norman R. Francingues	Chief, Water Supply and Waste Treatment Group, Waterways Experiment Station, Corps of Engineers
Mr. Herbert R. Haar Jr.	Assistant Port Director, Port of New Orleans
Mr. Jerome L. Mahloch	Program Manager, Waterways Experiment Station, Corps of Engineers
Mr. James M. Mansky	Assistant Chief, New York District, Corps of Engineers
Mr. Thomas Patin	Assistant Program Manager, Waterways Experiment Station, Corps of Engineers
Mr. James R. Reese	Environmental Specialist, Portland District, Corps of Engineers
Mr. Richard C. Semonian	Chief, Dredged Material Management Section, New England Division, Corps of Engineers
Mr. Larry S. Slotta	President, Slotta Engineering Associates, Inc.
Mr. Patrick M. Tobin	Director, Criteria and Standards Division, U. S. Environmental Protection Agency
Mr. Sachio Yamamoto	Head, Environmental Science Division, Naval Ocean Systems Center

Japanese Delegation

Mr. Shunich Onodera	Director-General of the Ports and Harbours Bureau, Ministry of Transport (MOT)
Mr. Teiichi Aramaki	Vice-Governor of Kyoto Prefecture
Mr. Takao Hirota	Director General of Port and Harbour Research Institute, MOT
Mr. Terumi Iijima	Director-General of the 3rd District Port Construction Bureau, MOT
Mr. Ikuhiko Yamashita	Director, Environment Division, Ports and Harbours Bureau, MOT
Mr. Toshio Ikeda	Environment Division, Ports and Harbours Bureau, MOT
Mr. Takeshi Horie	Port and Harbour Research Institute, MOT
Mr. Ryuji Nakamura	Engineering Deputy Director General of the 3rd District Port Construction Bureau, MOT
Mr. Morito Kawaguchi	Senior Officer for Environmental Technology of the 3rd District Port Construction Bureau, MOT
Mr. Tsutomu Fukude	Director of the Environmental Protection Section of the 3rd District Port Construction Bureau, MOT
Mr. Tomio Sasaki	The 3rd District Port Construction Bureau, MOT
Mr. Hitoshi Noguchi	The 3rd District Port Construction Bureau, MOT
Mr. Toshiyasu Uno	Director of the Kobe Port Construction Office, MOT
Mr. Masashi Kamon	Associate Professor, Department of Civil Engineering, Kyoto University
Mr. Ryuichi Sudo	Laboratory of Freshwater Environment, National Institute for Environmental Studies
Mr. Masaaki Hosomi	Laboratory of Freshwater Environment, National Institute for Environmental Studies
Mr. Mitumasa Okada	Laboratory of Freshwater Environment, National Institute for Environmental Studies
Mr. Hiroshi Nakanishi	Professor, Department of Civil Engineering, Yamaguchi University

Mr. Motoshi Fujiki	Professor, Institute of Community Medicine, Tsukuba University
Mr. Atsushi Yamada	Professor, Department of Science and Engineering, Ritsumeikan University
Mr. Akira Kawai	Professor, Department of Fisheries, Kinki University
Mr. Susumu Kawashima	Professor, Osaka Institute of Technology
Mr. Genta Uno	Professor, Osaka Institute of Technology
Mr. Iwao Takada	Osaka Institute of Technology
Mr. Zenya Yoshino	Professor, Science University of Tokyo
Mr. Tadashi Tanimoto	Director, Seacoast Division, River Bureau, Ministry of Construction
Mr. Masanori Kobayashi	Kanto Regional Construction Bureau, Ministry of Construction
Mr. Fumio Kodama	Kanto Regional Construction Bureau, Ministry of Construction
Mr. Masaharu Taguchi	Kanto Regional Construction Bureau, Ministry of Construction
Mr. Koji Tsuchinaga	Osaka City Institute of Public Health and Environmental Science
Mr. Hiromi Koba	Japan Dredging and Reclamation Engineering Association
Mr. Michio Miyake	Japan Dredging and Reclamation Engineering Association
Mr. Kozo Furuya	Japan Dredging and Reclamation Engineering Association
Mr. Hiromichi Yamashita	Japan Dredging and Reclamation Engineering Association
Mr. Makoto Natori	Japan Sediments Management Association
Mr. Tatsuo Yoshida	Japan Sediments Management Association
Mr. Kiyoshi Nikaido	Japan Sediments Management Association
Mr. Takahiro Inoue	Japan Sediments Management Association
Mr. Ikuo Sakamoto	Japan Sediments Management Association

Mr. Toshihide Mishima	Japan Sediments Management Association
Mr. Yoji Kurimoto	Japan Sediments Management Association
Mr. Masahiro Tagawa	Coastal Development Institute of Technology
Mr. Shingo Kato	Vice-Councilor, Civil Engineering Department, Mie Prefecture
Mr. Shoichi Hamaguchi	Director of Governor of Minamiise District, Mie Prefecture
Mr. Minoru Tomita	Director of Civil Engineering and Construction Department, Kyoto Prefecture
Mr. Tamaki Matsuoka	Director, Port and Harbour Division, Kyoto Prefecture
Mr. Masato Yasui	Chief Editor, Port and Harbour Division, Kyoto Prefecture

AGENDA

10th U. S./JAPAN EXPERTS MEETING ON MANAGEMENT OF BOTTOM SEDIMENTS CONTAINING TOXIC SUBSTANCES

Kyoto, Japan

30-31 October 1984

Tuesday, October 30, 1984

8:30-9:00	Welcoming Remarks
8:30-8:40	Director-General of the Ports and Harbours Bureau, Ministry of Transport
8:40-8:45	Vice-Governor of Kyoto Prefecture
8:54-8:50	Director-General of the 3rd District Port Construction Bureau, MOT Chairman Kleb
8:50-9:00	Chairman Yamashita
9:00-9:30	"Legal System and Works for Preservation of the Port, Harbour, and Marine Environment in Japan" by Toshio Ikeda, Ports and Harbours Bureau, MOT
9:30-10:00	"United States Environmental Protection Agency's Program for Managing Contaminated Sediments in the Environment" by Patrick M. Tobin, Criteria and Standards Division, U. S. Environmental Protection Agency
10:00-10:30	"A Model of Nitrogen Release from Bottom Sediments" by Masaaki Hosomi, Laboratory of Freshwater Environment, National Institute for Environmental Studies
10:30-11:00	"Study on Phosphorus Behavior Between Sediment and Upper Water" by Hiroshi Nakanishi, Dept. of Civil Engineering, Yamaguchi University, Japan
11:00-11:30	"Evaluation of the Factors Governing the Mobility of Dredged Material Placed at Open Water Disposal Sites" by W. Frank Bohlen, Department of Marine Sciences, University of Connecticut

11:30-12:00	"Environmental and Water Quality Operations Studies (EWQOS) Application to Lake Restoration" by Jerome L. Mahloch, Waterways Experiment Station, Corps of Engineers
12:00-13:30	Luncheon (Sponsored by Vice-Governor of Kyoto Prefecture)
13:30-14:00	Movie
14:00-14:30	"Numerical Prediction on the Effect of Phosphorus Release Control by Sediment Dredging" by Takeshi Horie, Port and Harbour Research Institute, Ministry of Transport
14:30-15:00	"Ecological Evaluation of Organotin Contaminated Sediments" by Sachio Yamamoto, Environmental Science Division, Naval Ocean Systems Center
15:00-15:30	Coffee Break
15:30-16:00	"Characterization of Sediment Contamination Utilizing Bioassay/Bioaccumulation Testing" by James M. Mansky, New York District, Corps of Engineers
16:00-16:30	"Identification of Promising Concepts for Treatment of Contaminated Sediments" by Norman R. Francingues, Waterways Experiment Station, Corps of Engineers
16:30-17:00	"The Influence of Accumulating Organic Sediment on the Seawater Environment in Osaka Bay" by Hitoshi Noguchi, The 3rd District Port Construction Bureau, Ministry of Transport
17:00-17:30	"Results of Sedimentation and Elutriate Tests from Osaka Bay Sediment" by Kiyoshi Nikaido, Japan Sediments Management Association, Joint Research with the 3rd District Port Construction Bureau, Ministry of Transport
18:30-20:30	RECEPTION

Wednesday, October 31, 1984

8:30-9:00	"Partial Review of Dredged Mud Disposal Techniques" by Hiromi Koba, Japan Dredging and Reclamation Engineering Association
9:00-9:30	"Lime-Cement Hardening of Very Soft Freshwater Clay" by Masashi Kamon, Department of Civil Engineering, Kyoto University
9:30-10:00	"Dewatering of Bottom Sediments" by Zenya Yoshino, Science University of Tokyo
10:00-10:30	"Dredge Cutterhead Flow Processes" by Larry S. Slotta, Slotta Engineering Associates, Inc.
10:30-11:00	"Selection and Management of a New Ocean Disposal Site at Coos Bay, Oregon, to Receive Contaminated Dredged Materials" by James R. Reese, Portland District, Corps of Engineers

11:00-11:30	"Wastewater Treatment by Soil" by Tadashi Tanimoto, River Bureau, Ministry of Construction
11:30-12:00	Movie
12:00-13:30	Luncheon
13:30-14:00	"In Situ Stream Water Clarifying Tests" by Tatsuo Yoshida, Japan Sediments Management Association (Joint Research with Civil Engineering Research Institute, Ministry of Construction)
14:00-14:30	"Purification of the Nho River by the Gravel Contact Method" by Masanori Kobayashi, Kanto Regional Construction Bureau, Ministry of Construction
14:30-15:00	"The Mussel Watch Program/Monitoring the Long-Term Effects of Dredged Material Disposal in New England Waters" by Richard C. Semonian, New England Division, Corps of Engineers
15:00-15:30	Coffee Break
15:30-16:00	"Principles and Practices for the Ocean Disposal of Dredged Material-The London Dumping Convention Re-examined" by Herbert R. Haar, Jr., Port of New Orleans, New Orleans, Louisiana
16:00-16:30	"Progressive Trenching Using a Scraper Digger" by Takahiro Inoue, Japan Sediments Management Association
16:30-17:00	"Environmental Conservation in the Port of Maizuru" by Tamaki Matsuoka, Construction Department, Kyoto Prefecture
17:00-18:00	DISCUSSION AND CLOSING SESSION (Joint Communique)
18:30	RECEPTION

JOINT COMMUNIQUE

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

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

This tenth meeting was highly successful and fruitful. The information exchange was very effective. The conference attendees generally agreed that the dredging technology and sediment management programs of both countries will significantly benefit. The Co-Chairman agreed that the next meeting will be in the United States in 1985 and that the date will be decided jointly.

Director, Environment Division
Ports and Harbours Bureau
Ministry of Transport

Ikuhiko Yamashita
October 31, 1984

Commander and Director, Water Resources
Support Center, U.S. Army Corps of
Engineers

Col. George R. Kleb
October 31, 1984

LEGAL SYSTEM AND WORKS FOR PRESERVATION OF THE PORT, HARBOUR, AND MARINE ENVIRONMENT IN JAPAN

T. Ikeda

Environmental Division, Ports and Harbours Bureau
Ministry of Transport

JAPANESE ENVIRONMENTAL CONDITIONS

Environmental pollution in Japan progressed rapidly during the period of high economic growth beginning in 1960. In response to strong social demand for the prevention of pollution, pollution control measures progressed rapidly in the 1970s and the worst cases of environmental pollution were eased. The conservation of energy and resources also progressed while the economy shifted from high growth to stable growth. Today, environmental pollution is gradually lessening.

This paper will outline the current situation of water pollution in Japan. The situation of water pollution can be expressed by the rate of achievement of environmental water quality standards (health items and living environment items).

As for health items, the results of measurement of water quality in public marine areas in FY 1982 show that the proportion of samples not meeting the environmental quality standards (percentage of nonconformity) was 1.4% in FY 1970 but decreased to 0.3% in FY 1982, thereby showing a considerable improvement (Figure 1).

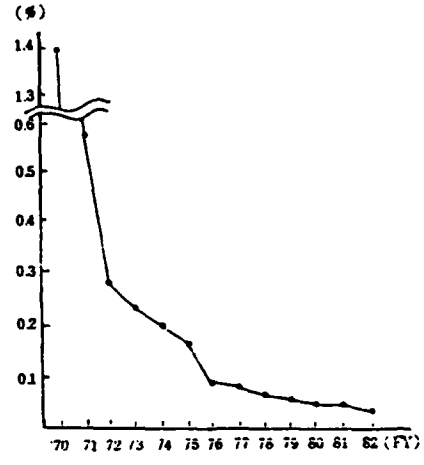


Figure 1. Percentage of nonconformity of the environmental water quality standards on health items

As for the living environment, 67.5% of the marine areas achieved the environmental quality standard for the typical water quality index of biochemical oxygen demand (BOD) or chemical oxygen demand (COD). In detail, 81.3% of the sea areas, 65.3% of the rivers, and 41.7% of the lakes achieved the quality standard. Even though the sea areas achieved a better quality

standard than the other marine areas and the water quality of sea areas tends to improve (Figure 2), the water quality is still at a low level in Tokyo Bay, Ise Bay, and other areas where the seas are mostly closed and large pollution sources exist in the hinterland (Figure 3).

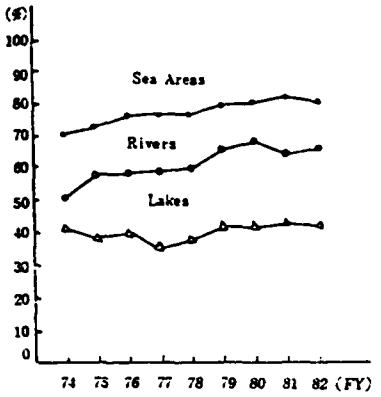


Figure 2. Rate of achievement of the environmental water quality standards on the living environmental items

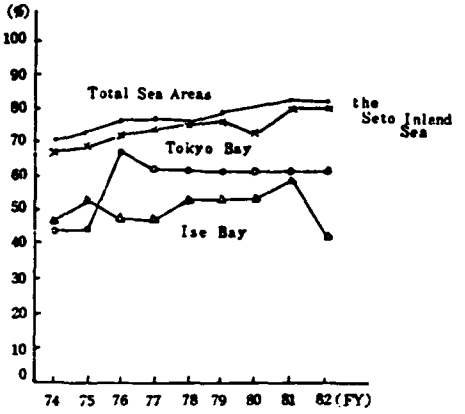


Figure 3. Rate of achievement of the environmental water quality standards (COD) in large closed sea areas

LEGAL SYSTEM FOR THE PRESERVATION OF THE PORT, HARBOUR, AND MARINE ENVIRONMENT

Twenty anti-pollution laws were established during the 6-month session of the Diet beginning in late 1970 which was called the "Anti-pollution Diet." Thereafter, various additional laws were formulated and many legal restrictions enacted. The legal system covering the environmental conservation of ports, harbours, and marine environment (Figure 4) will be discussed below.

Basic Law for Environmental Pollution Control (Established 1967)

Contents of the Law

This is the basic law for various laws and regulations related to pollution in Japan. Its purpose is to comprehensively promote anti-pollution measures, protect the public health, and conserve the living environment. To achieve this purpose, the following items are prescribed: (1) items for clarifying the responsibility of enterprises, the Government, and local public bodies for the prevention of pollution; (2) basic items related to measures for the prevention of pollution; (3) items related to the bearing of costs and financial measures; and (4) items related to the Conference on Environmental Pollution Control and the Council for Control of Environmental Pollution. Substantial restrictions and measures are executed based upon the individual laws established for the fulfillment of the basic law, except the establishment of environmental quality standards for the unification of policies (Article 9) and the preparation of an environmental pollution control programme (Article 19), which are covered below.

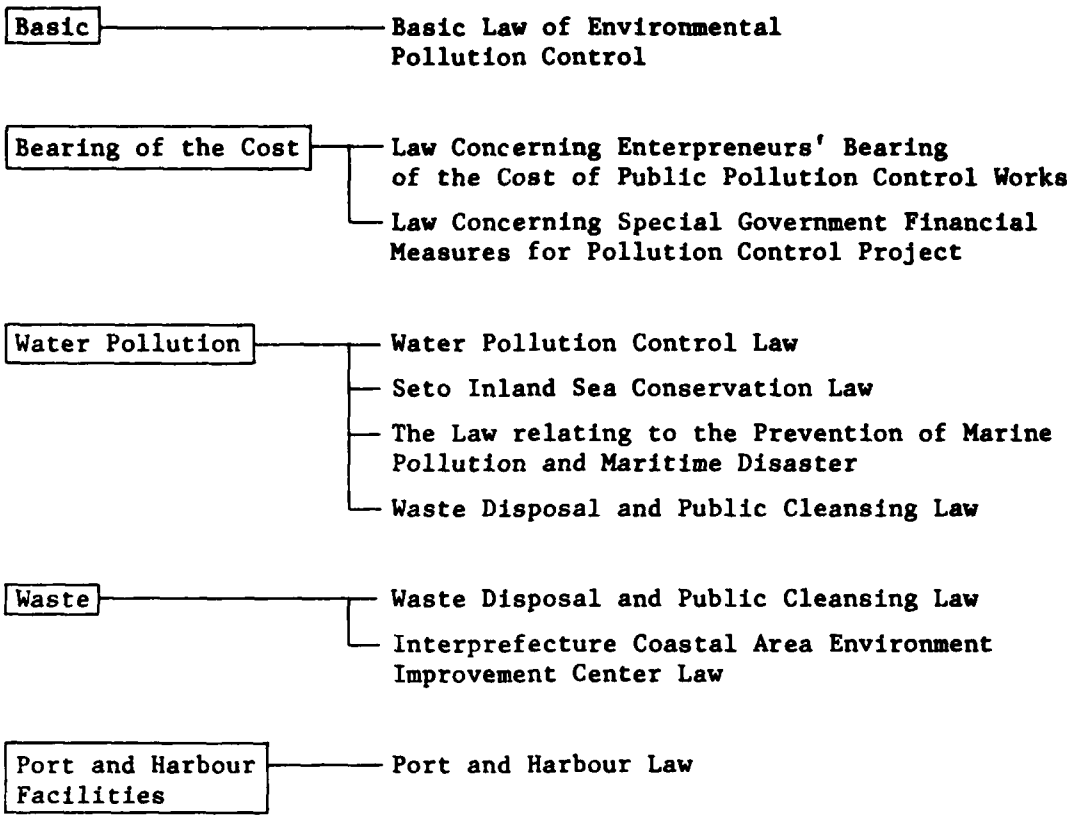


Figure 4. Legal system for the preservation of the ports, harbours, and marine environment in Japan

Establishment of Environmental Water Quality Standards

These standards are the targets for the water quality preservation administration and were established as desirable standards to be maintained for the water quality of public marine areas such as sea areas, rivers, and lakes. They comprise the standards for protecting the people health (health items) and the standards for preserving the living environment (living environment items). For the health items, nine substances harmful to public health were uniformly established for all public water areas. However, for the living environment items, the types of water areas were established by water utilization for rivers, lakes, and sea areas, and standard values for items such as BOD, COD, and dissolved oxygen (DO) were established for each type of water body. Thus, the environmental quality standard for the relevant public water body can be firmly established by designating the type of water body.

Preparation of Environmental Pollution Control Programme

This programme is formulated in accordance with Article 19 of the Basic Law for Environmental Pollution Control for areas where considerable pollution already exists or areas where the pollution is likely to continue to worsen due to the high concentration of people or industry and the pollution cannot

be controlled unless comprehensive measures are taken. The first of these areas was recognized in FY 1970, with several additional areas recognized sequentially thereafter; the seventh such area was recognized in FY 1976. Since then programmes have been reexamined, integrated, and abolished. Today there are programmes for 46 areas in Japan, and environmental pollution control programmes have been prepared for all major metropolitan areas and industrial cities across the country (Figure 5).

Law Concerning Enterpreneurs' Bearing of the Cost of Public Pollution Control Works (Established 1970)

This law was established to enact Article 22 of the Basic Law for Environmental Pollution Control which obligates corporations to share the burden of all or part of the cost of works conducted by the Government or local public bodies for preventing pollution expected from the operations of the corporations. This law sets forth the details concerning the burden of costs of the pollution prevention works to be shared by the corporations. The law states that a corporation is obligated to share the burden of all or part of the cost of pollution prevention works, the total cost depending upon the degree of pollution considered to be created by the corporation's operating activities. The costs cover dredging for pollution prevention, earth borrowing for polluted agricultural land, and forming buffer green tracts. From the enactment of this law to the end of 1982, this law was applied to 62 cases of pollution prevention works. The total cost of the pollution prevention works was about ¥136,300,000,000, and the total amount borne by corporations was about ¥68,100,000,000; the average percentage of costs borne by corporations was 50.0%.

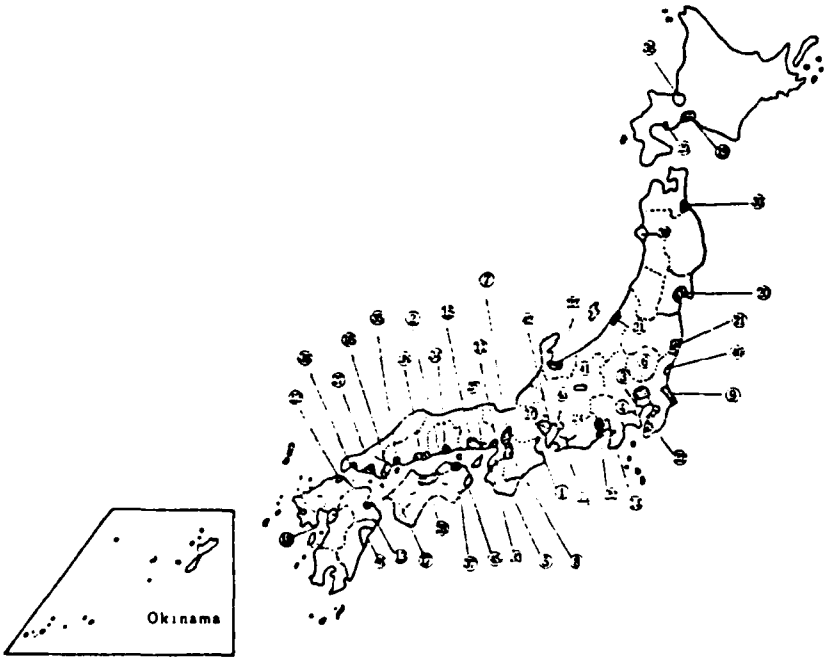


Figure 5. Formulated areas of Environmental Pollution Control Programmes (Continued)

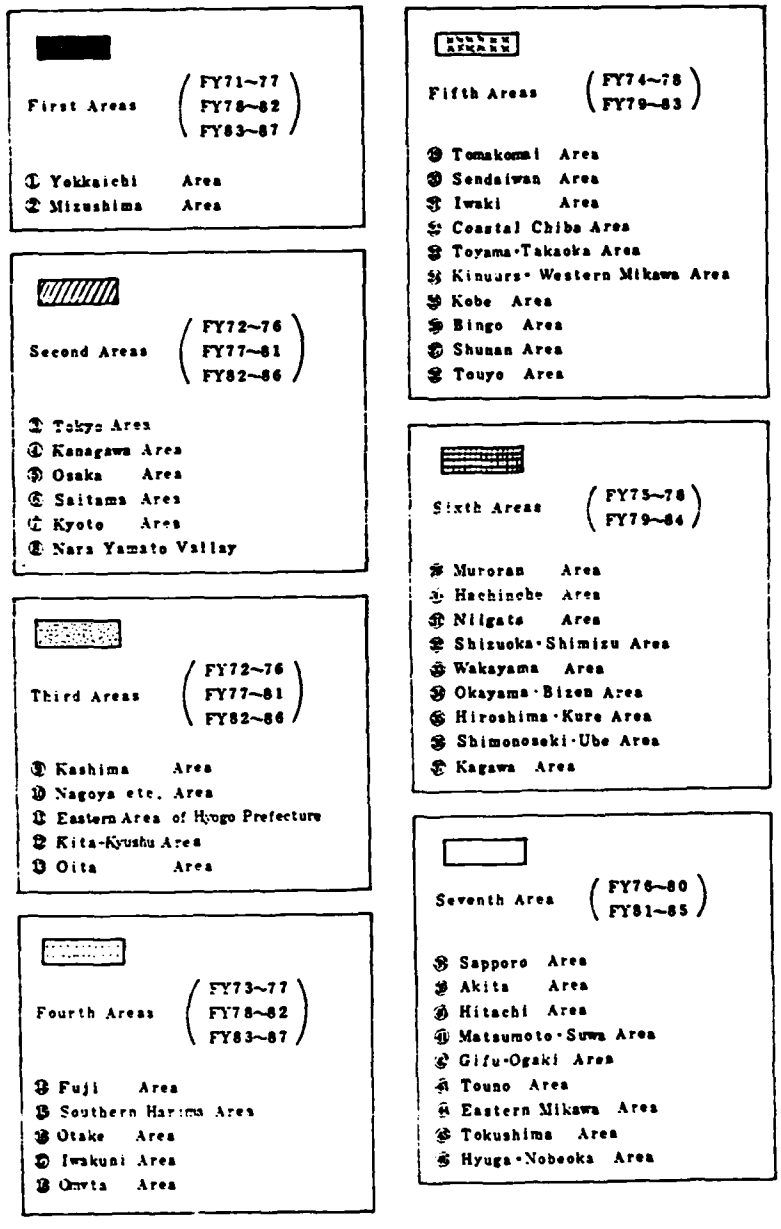


Figure 5 (Concluded)

Law Concerning Special Government
Financial Measures for Pollution
Control Projects (Established 1971)

Past financial measures by the Government for pollution prevention were not necessarily sufficient, and because of this, the above law was established to clarify the financial responsibility of the Government and to further promote anti-pollution measures by local public bodies.

This law applies mainly to areas covered by the environmental pollution control programmes set forth in Article 19 of the Basic Law for Environmental Pollution Control. This law also details the special measures related to the shares of government subsidies, local government securities, and subsidy taxes. The shares of government subsidies give 50% aid (burden) instead of the ordinary ratio of aid (ratio of burden) for all dredging, water conveyance, and works of installation of waste disposal facilities related to the ports and harbours. Special measures for local government securities are intended to expand these works to give preferential treatment for work to which special ratios of burden and aid of the law have been applied.

Water Pollution
Control Law (Established 1970)

This law was established to protect the public health and conserve the living environment by properly maintaining the water quality in public marine areas by restricting the discharge of water from factories, etc.

Effluents to be Restricted

Effluent from factories and places of work ("specified places of work") where "specified facilities" are installed in the public marine areas such as rivers, lakes, ports and harbours, and coastal sea areas is restricted by this law.

The "specified facilities" are the facilities so designated by Cabinet Orders which discharge wastewater or liquids containing harmful substances or substances that are likely to cause damage to the living environment. The facilities of 91 types of industry have been already designated as specified facilities.

Effluent Standards

Effluent standards are the allowable limits established for each pollution index with respect to the existing pollution of the effluent in order to restrict the effluents from the specified places of work. There are three types of standards as shown below.

Uniform effluent standards. Uniform effluent standards have been established by the Government in the form of maximum values as a rule for harmful substances such as cadmium and cyanogen and for items such as pH, COD, and BOD, and are uniformly applied to all the specified places of work.

Strict standards. For marine areas for which the environmental quality standards cannot be achieved or maintained by the uniform effluent standards

alone due to natural and social conditions, the prefectural governments can establish more strict standards by means of prefectural ordinances. All the prefectural governments established these strict standards after FY 1975.

Standards for regulation of total emission. In public marine areas with a large volume of effluent resulting from the high concentration of people and industries, it is difficult to achieve the environmental quality standards by uniform effluent standards or strict standards alone. In these areas the standards for the regulation of total emission are established. Standards for the regulation of total emission have been established for Tokyo Bay and Ise Bay in accordance with the total emission reduction plans formulated by the relevant prefectural governments in response to the basic policies for reducing total emission concerning COD established by the prime minister in June 1979. Measures for reducing the COD emission are now being taken.

Seto Inland Sea
Conservation Law (Established 1978)

Though the Seto Inland Sea has favorable natural conditions, water pollution developed quickly since it is a closed water area with a high concentration of people and industry in the nearby areas.

Therefore, the "Extraordinary Seto Inland Sea Conservation Law" was established in 1973. However, in 1978 new measures were added to this law and it was revised to "Seto Inland Sea Conservation Law." This is a special law of the Water Pollution Control Law that promotes the following environmental conservation measures:

(1) Relevant prefectural governments are required to make environmental conservation plans in accordance with the Basic Plan for the Environmental Conservation of Seto Inland Sea (established by a Cabinet conference in April 1978). The relevant prefectural governments formulated the conservation plans in July 1981.

(2) When installing the specified facilities set forth in the Water Pollution Control Law, an environmental influence assessment is required, and the installation must be approved by the appropriate governors.

(3) A system regulating the total emission of COD not only for industrial effluent, but also for household effluent is prescribed (the mechanism of the system is prescribed in the Water Pollution Control Law). The relevant prefectural governments formulated the total emission reduction plans in March 1980 (target year: FY 1984).

(4) The Director of the Environment Agency ordered the formulation of measures against eutrophication in July 1979. The relevant prefectural governments accordingly formulated the reduction guidance policies for phosphorus and its compounds in April and May 1980 (target year: FY 1984).

(5) Natural beach areas that can be used for recreation such as sea bathing now and in the future were designated as natural preservation areas. Various activities are permitted there with proper prior notification.

(6) The perculiarities of the Seto Inland Sea must be considered when approving reclamation work (basic policies to be considered by the relevant prefectures were established in June 1974).

Law Relating to the Prevention of Marine Pollution and Maritime Disaster (Established 1970)

This law ensures the proper disposal of waste oil by restricting the discharge of oil and waste from ships. This restriction prevents marine pollution and maritime disasters by taking proper measures against discharged oil and wastes, and thereby conserves the marine environment and protects people's lives, health, and other assets. This law has been revised four times to respond to the international trends in the prevention of maritime pollution; its restrictions were strengthened and new restrictions were added.

Emission of Oil from Ships

As a rule, ships are not permitted to discharge oil in sea areas. Oil may be discharged in an emergency and if the effluent standards established by tonnage and type of ship are strictly observed.

Discharge of Waste from Ships

The discharge of waste from ships in sea areas is also prohibited as a rule. Discharge is permitted in an emergency and in the cases listed below if certain requirements are met:

a. Discharge and sewage resulting from the daily life of seamen in ships.

b. Discharge to reclaimed land and so forth.

c. Discharge of wastes (part of sludge, sewage, cinders, etc., for which the disposal by ocean dumping is unavoidable and thus permitted by "Enforcement Ordinance of the Waste Disposal and Public Cleansing Law").

d. Discharge of wastes for which the disposal by ocean dumping is unavoidable and thus permitted.

Waste Oil Disposal Works

To ensure the proper disposal operations at waste oil disposal facilities, an approval system has been formed for these and other facilities (excluding port management bodies and fishing port management bodies). A notification system is in effect for port management and fishing port management facilities and also for privately owned facilities.

Waste Disposal and Public
Cleansing Law (Established 1970)

This law conserves the living environment and improves the public health by properly disposing of wastes and maintaining a clean living environment.

Definition of Wastes

Wastes are defined in this law as discarded items such as rubbish, bulky refuse, and cinders and are classified into ordinary waste and industrial waste. Dredged sediment, surplus soils produced by inland construction and radioactive wastes, etc., are not included in the wastes set forth in this law (Figure 6).

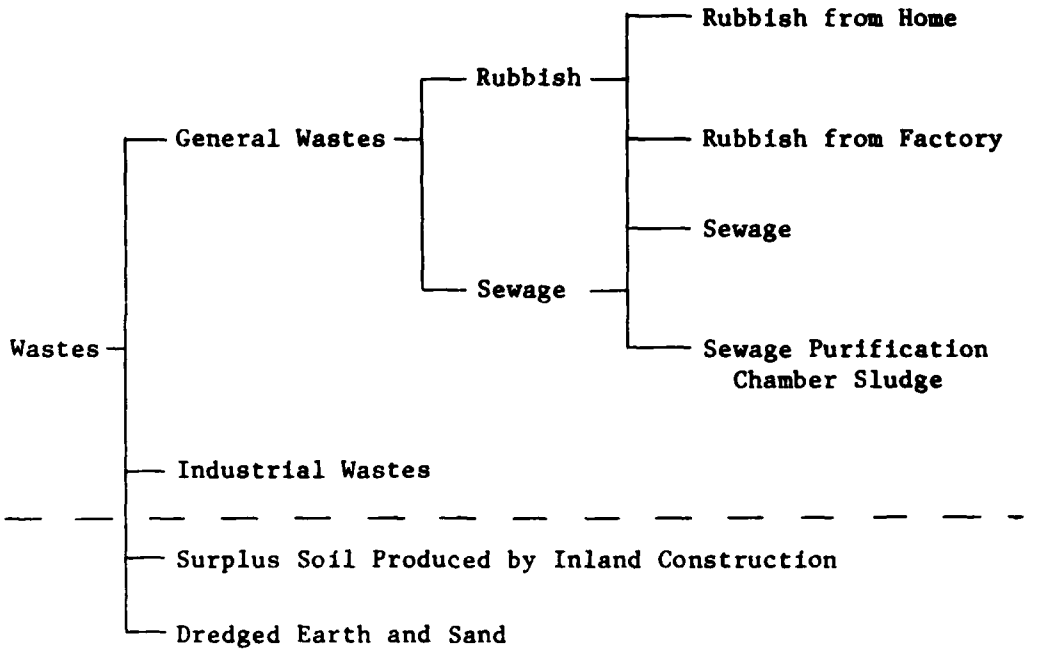


Figure 6. Classification of wastes

Method of Disposing of Wastes

Wastes may be disposed of by incineration or by disposal on a reclamation site, but must be disposed of without polluting the living environment in accordance with the disposal standards prescribed by Cabinet Order.

Interprefecture Coastal Area
Environment Improvement Center Law

An extremely strong demand exists today for securing coastal waste disposal sites (interprefecture coastal disposal sites shown in Figure 7) in metropolitan areas that can be used both by local public bodies and by developing ports and harbours to support the sound development and activities of cities. Therefore, this law was established to prescribe the foundation and scope of work of the organization called the "Interprefecture Coastal Area Environment Improvement Center" (hereinafter called "the Center"). Duties of the Center

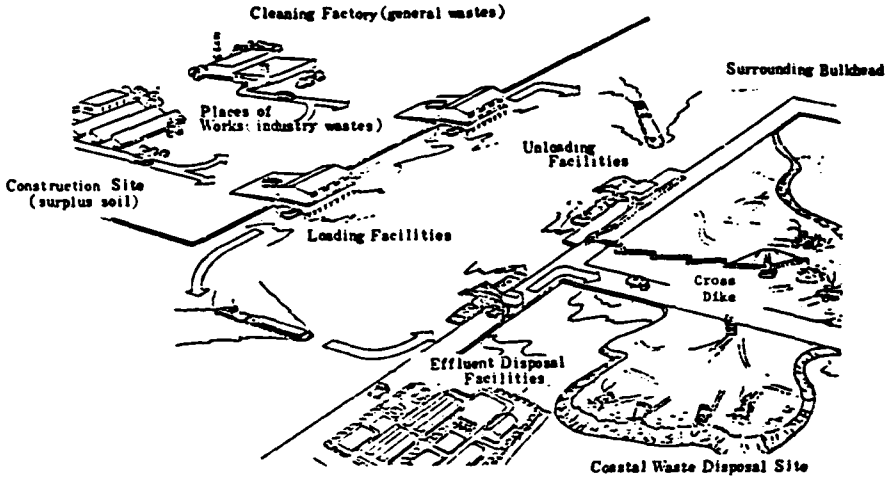


Figure 7. Outline of interprefecture coastal waste disposal site

are (1) to develop bulkheads for coastal waste disposal sites commissioned by port management bodies and to develop coastal reclaimed land on the sites by using waste, (2) to develop final disposal facilities commissioned by local public bodies for ordinary waste and industrial waste prescribed by Cabinet Orders and to develop coastal reclaimed land by using these wastes, and (3) to develop final disposal sites for industrial waste as the Center's own project and to develop coastal reclaimed land by using these wastes. The Center is required to formulate a master plan and an execution plan based on the master plan for the duties stated above. Because of the nature of the work assigned to the Center, the relevant port management bodies and local public bodies are deeply involved in the operation of the Center, and the Government is involved in the establishment of the Center, the preparation of the master plan, etc. (Figure 8).

For the Osaka Bay zone, the Osaka Bay Interprefecture Coastal Area Environment Improvement Center was established in March 1982 in accordance with this law, and this Center has been making a survey required for developing interprefecture coastal disposal sites.

Port and Harbour
Law (Established 1950)

This law controls the orderly development and proper operation of ports and harbours. Improvement of the port and harbour environments was amended in 1973. Because of this, port pollution prevention facilities (water conveyance facilities, buffering zones for pollution prevention, etc.), waste disposal facilities (bulkheads for coastal waste disposal sites, waste incineration facilities, etc.), and port environment improvement facilities (beaches, green tracts, etc.) are designated as port and harbour facilities, and legal and financial measures necessary for the development of these facilities are being taken.

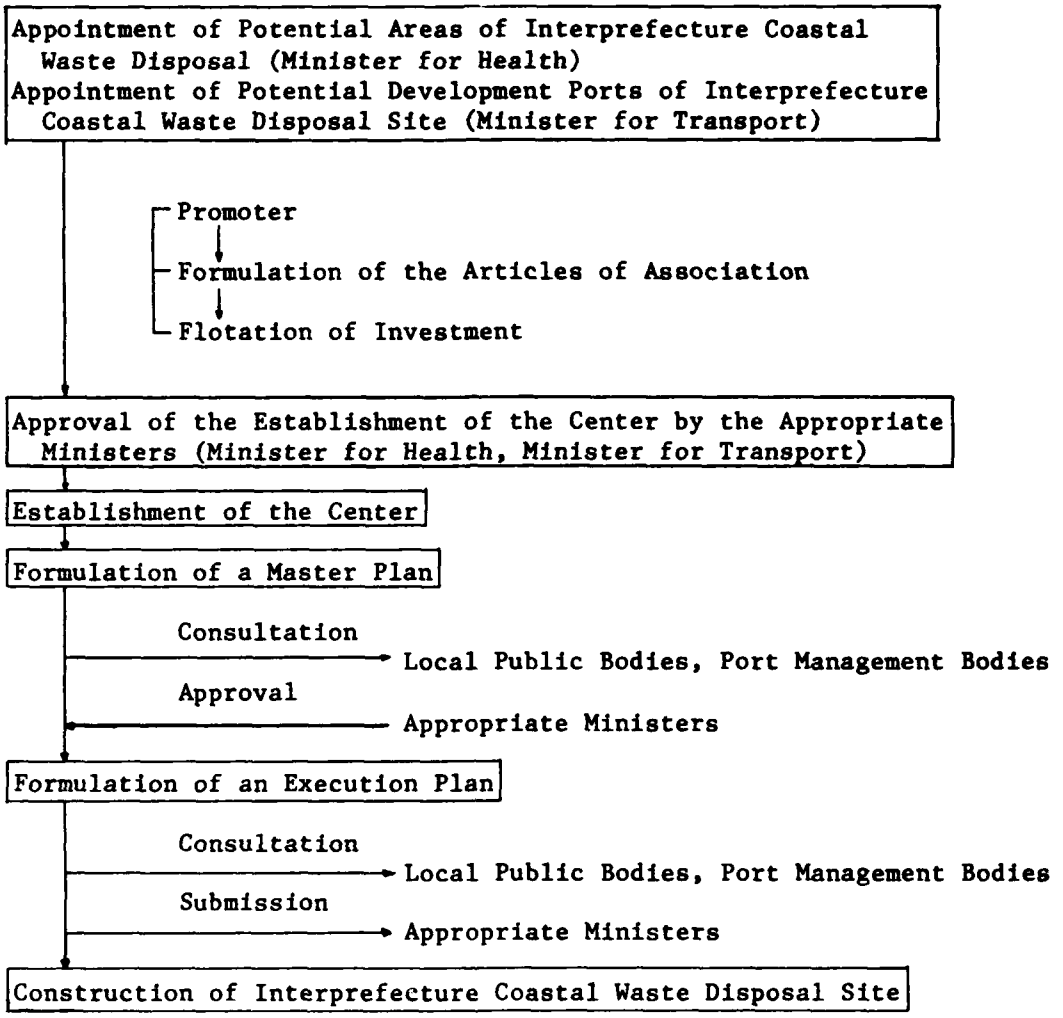


Figure 8. Flowchart of interprefecture coastal waste disposal works by Interprefecture Coastal Area Environment Improvement Center Law

PRESERVATION OF THE PORT, HARBOUR, AND MARINE ENVIRONMENT

As part of the port and harbour improvement works, various tasks are being carried out to preserve the port, harbour, and marine environment and to prevent pollution. These tasks are being particularly aggressively carried out since the partial amendment of the Port and Harbour Law in 1973. As shown in Figures 9 and 10, the total project cost was ¥4,000,000,000 in FY 1972 and increased to ¥17,000,000,000 in FY 1973, with a peak in FY 1979 of ¥58,000,000,000. Today the cost tends to remain around ¥50,000,000,000 since the development of bulkheads for coastal waste disposal sites in metropolitan areas is almost completed and the financial situation is very severe. The various kinds of work to preserve the port, harbour, and marine environment are outlined in Table 1 and discussed in detail below.

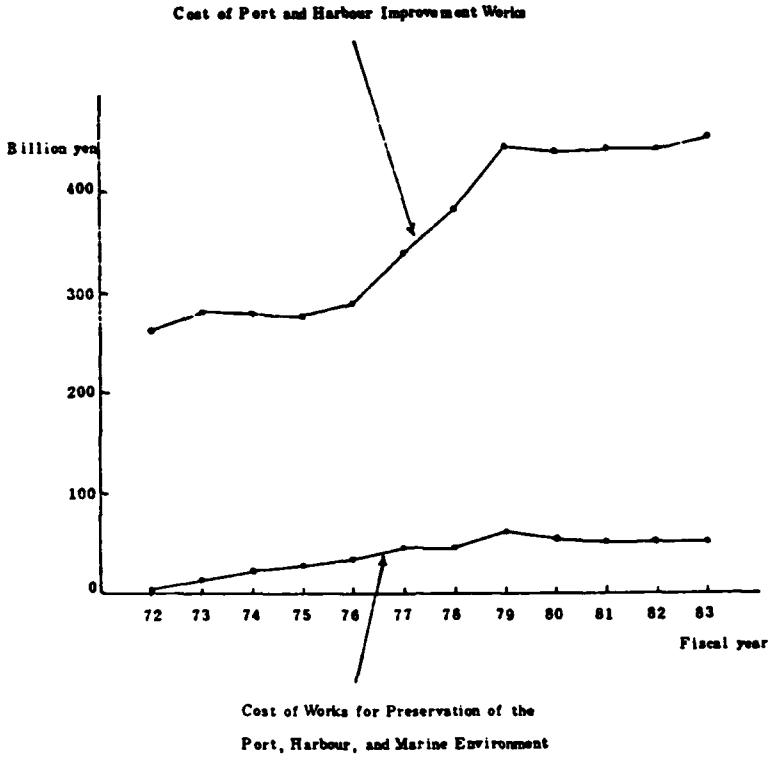


Figure 9. Project costs, FY 1972 to FY 1983

Works to Provide Facilities for
Preventing Oil Pollution of Seawater

The disposal of effluent oil generated by ships to the ocean is restricted by the law relating to the Prevention of Marine Pollution and Maritime Disasters. Waste oil reception facilities for receiving and properly disposing of this waste oil on land are being developed by petroleum refiners, waste oil disposal experts, port management bodies, and fishing port management bodies. As of August 1984, Japan had 140 facilities in 85 ports (Figure 11). Government subsidies were offered so that the above facilities could be constructed and improved by the port management bodies beginning in FY 1967, and subsidized projects were executed in 54 ports through FY 1983.

Port Pollution Prevention Works

These works are performed in ports and harbours where contaminated sediments exist which may cause pollution, or where water quality would be degraded by removing the sediments. Port pollution prevention can be divided into two sections: (a) dredging and earth covering performed in accordance with the Law Concerning Special Government Financial Measures for Pollution Control Projects, (b) constructing or improving water conveyance facilities, dust control fences, etc., performed in accordance with the Port and Harbour Law. Government subsidies have been offered since FY 1972 to the above works to be performed by the port management bodies, and subsidized works were carried out in 30 ports by FY 1983 (Figure 12).

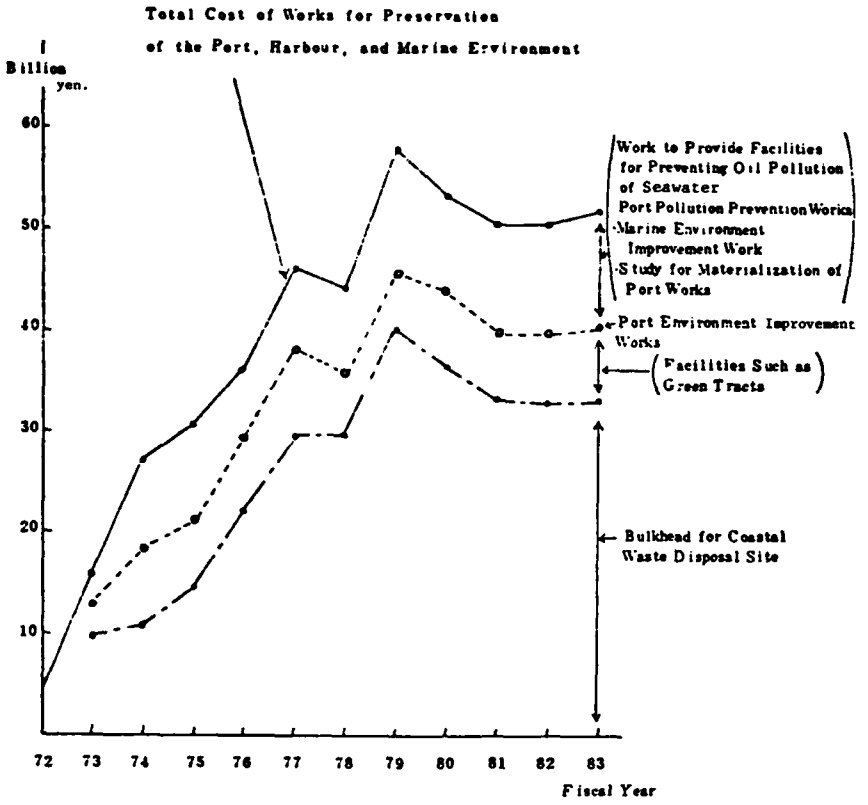


Figure 10. Costs of works for preservation of the port, harbour, and marine environment

Port Environment Improvement Works

Bulkheads for Coastal Waste Disposal Sites

Though measures for reducing the weight of wastes such as incineration and recycling are being taken, there is still a large volume of waste that requires a final disposal such as disposal on reclamation sites. Also, it is becoming more difficult to secure final inland waste disposal sites due to urbanization. Because of this, a strong demand exists for securing coastal waste disposal sites. In addition, more sophisticated port functions and redeveloped urban seashore areas are needed to support the sound development and activities of the cities behind the ports. Therefore, a strong demand for securing necessary land has arisen.

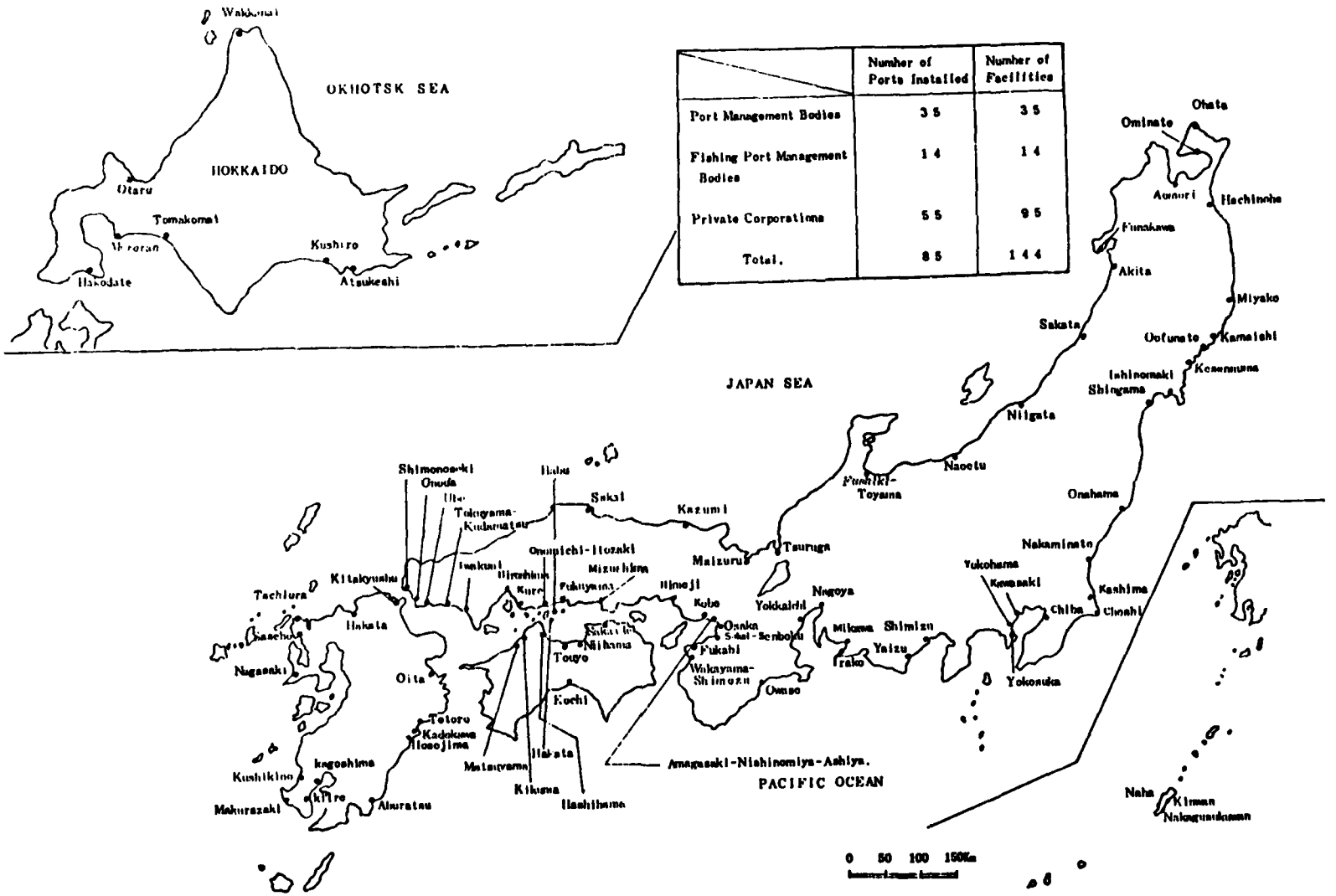
In order to respond to the social needs stated above, government subsidies have been offered since FY 1973 for the development of bulkheads for coastal waste disposal sites by port management bodies in Tokyo Bay and Osaka Bay. Work has begun to systematically secure the required space within ports and to properly dispose of the wastes. By FY 1983 works of this kind were carried out in 24 areas in 19 ports.

TABLE 1. COSTS FOR PRESERVATION OF THE PORT, HARBOUR, AND MARINE ENVIRONMENT

(Unit:Million yen)

	FY 72	FY 73	FY 74	FY 75	FY 76	FY 77	FY 78	FY 79	FY 80	FY 81	FY 82	FY 83	FY 84
1. Works to Provide Facilities for Preventing Oil Pollution of Seawater	907	167	393	585	114	396	61	20	80	80	78	138	110
2. Port Pollution Prevention Works	3,288	3,656	6,948	8,188	5,678	6,880	6,742	7,792	6,192	7,860	7,746	7,986	12,679
3. Port Environment Improvement Works	0	12,944	17,559	20,936	29,812	38,411	36,682	48,123	44,385	39,835	39,855	40,918	35,978
(1) Bulkheads for Coastal Waste Disposal Sites	0	9,894	10,989	14,536	22,865	29,763	28,550	40,470	36,647	32,865	32,782	33,544	28,595
(2) Facilities such as Green Tracts	0	2,452	5,802	5,106	5,608	7,110	7,545	7,383	7,272	6,728	6,995	7,274	7,313
(3) Building Cleaning Ships	0	0	189	80	200	55	94	100	192	113	78	0	70
(4) Others	0	598	579	1,214	1,139	1,483	493	170	274	129	0	100	0
4. Marine Environment Improvement Works	0	0	410	562	847	802	1,034	1,423	1,503	1,710	1,845	1,906	1,947
5. Study for the Materialization of Port Works	0	0	0	0	0	0	0	500	650	786	774	780	790
(1) Bottom Sediment Purification	0	0	0	0	0	0	0	300	500	675	699	705	715
(2) Interprefecture Coastal Waste Disposal Sites	0	0	0	0	0	0	0	200	150	111	75	75	75
Grand Total	4,195	16,767	25,310	30,271	36,451	46,489	44,519	57,858	52,810	50,307	50,298	51,728	51,504

Note: Each cost is a total of each work. Government subsidy is included.



	Number of Ports Installed	Number of Facilities
Port Management Bodies	35	35
Fishing Port Management Bodies	14	14
Private Corporations	55	95
Total.	85	144

Figure 11. Ports with waste oil disposal facilities (as of Aug. 1, 1984)

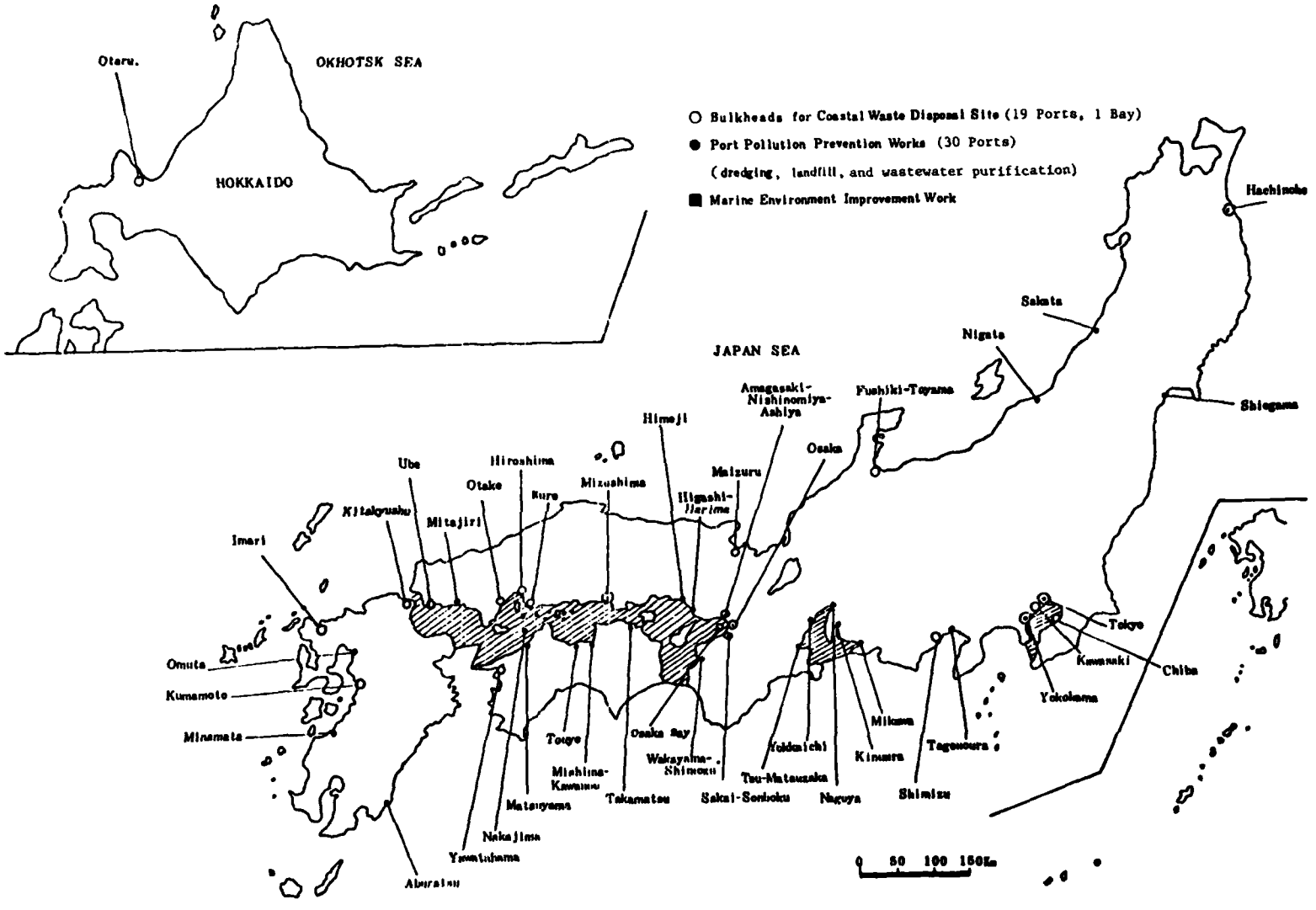


Figure 12. Ports, harbours, and sea areas where works for preservation of the port, harbour, and marine environment were conducted (through FY 1983)

Facilities Such as Green Tracts

In order to secure a pleasant environment and provide recreational areas within ports, government subsidies have been offered since FY 1973 to develop port environment improvement facilities such as green tracts, open spaces, and vegetation. These facilities have been developed by the port management bodies. A gross project cost of about ¥69,000,000,000 has been invested through FY 1983 and green tracts of about 520 ha have been developed.

Building Cleaning Ships

Rubbish such as refuse from rivers or articles dropped during cargo handling in ports may float on the water in port areas and spoil the beauty of the port or obstruct the navigation of ships. Port management bodies recover such rubbish and some ports have cleaning ships for the efficient and quick removal of rubbish. Government subsidies have been offered for building these cleaning ships for the ports since FY 1974. Seventeen ships were built through FY 1983.

Marine Environment Improvement Works

The ocean is polluted by rubbish, oil, and other refuse flowing from rivers. Especially severe pollution occurs in inland areas and bays that greatly affects marine activities, living environment, marine resources, etc.

Because of this, the removal of rubbish and oil by ships in three sea areas, Tokyo Bay, Ise Bay, and the Seto Inland Sea, for which the preservation of marine environment is greatly needed, has been being carried out as a government project since FY 1974.

Study for the Materialization of Port Works

Bottom Sediment Purification

In order to drastically improve the marine environment in Tokyo Bay, Ise Bay, and Seto Inland Bay where eutrophication has greatly progressed, it is necessary to reduce the inflow load and decrease the elution of organic matters and nutrient salts from sludge deposited on sea bottom, thereby eliminating the vicious cycle of seawater pollution. For this purpose, the bottom sediment must be purified by removing the sludge. In FY 1979 a government project for examining technical problems to be solved in executing the sediment purification of Tokyo Bay, Ise Bay, and the Seto Inland Sea was begun.

Interprefecture Coastal Waste Disposal Sites

In order to respond to the demand for the development of "interprefecture coastal disposal sites" explained earlier, a survey required to establish the Interprefecture Coastal Area Environment Improvement Center has been conducted since FY 1979 as a study for the materialization of port works as a government project.

In response, the Osaka Bay Interprefecture Coastal Area Environment Improvement Center was established in the Osaka Bay area in March 1982. In addition, a continuous study for the materialization of port works is currently being performed in the Tokyo Bay area.

A MODEL OF NITROGEN RELEASE FROM BOTTOM SEDIMENTS

M. Hosomi, M. Okada, and R. Sudo

Laboratory of Freshwater Environment
National Institute for Environmental Studies
Yatabe-machi, Tsukuba, Ibaraki 305, Japan

ABSTRACT

A multi-component dynamic model for nitrogen in the bottom sediment-water system of Lake Kasumigaura, a highly eutrophic and shallow lake, was developed to obtain quantitative information on the sediment-water interactions. The model was calibrated and the sensitivity analysis of the model parameters conducted to identify the deficiency of our knowledge at present and to determine future research needs. Good agreement between the model calculations and observed data of nitrogen release flux and nitrogen concentrations in the sediment was noted. The nitrogen dynamics of the upper layers (0-6 cm) of the sediment seem to regulate the nitrogen transportation between sediment and water. Further studies on the vertical distribution of ammonification rate were found to be necessary from the sensitivity analysis. The effectiveness of eutrophication control programs was evaluated using the calibrated model. The reduction of nitrogen deposition rate to the sediment and the bottom sealing were more effective than the sediment dredging for the control of NH_4-N release from bottom sediment.

INTRODUCTION

Bottom sediments are known to play an important role in the lake eutrophication process (3, 7, 11, 14). Particularly in lake restoration programs, we cannot ignore the effects of the sediments on nitrogen and phosphorus budgets. In restoration programs, such as wastewater diversion and advanced wastewater treatments, that have been completed, there are examples where water quality did not improve to the desired levels because of nutrient release from bottom sediments (21, 37). In implementing lake restoration programs effectively, it is necessary to quantify the role of bottom sediments.

In this paper, a multi-component dynamic model for nitrogen in the sediment-water system of Lake Kasumigaura was developed with two purposes. The first purpose is to obtain quantitative information on the sediment-water interactions and to identify the deficiency of our knowledge at present and to

determine future research needs. The second purpose is to estimate the effectiveness of eutrophication control programs such as sediment dredging and lake bottom sealing using the calibrated model.

Nitrogen is closely related to lake eutrophication. Particularly, nitrogen is considered to be the limiting nutrient for algal growth during the summer in Lake Kasumigaura (33). Table 1 shows previous models on nitrogen release from bottom sediments. One box model (5, 16, 33) cannot give us quantitative information on nitrogen dynamics in bottom sediments. Additionally, one box model uses an assumed sediment thickness or volume which may be unrealistic. Previous models did not consider denitrification, whereas denitrification plays the most important role in nitrogen budget in shallow lakes (3). Although some studies presupposed steady-state for NH_4-N and NO_3-N profiles in the sediments, it is well known that nitrogen concentration in the sediments shows seasonal changes, especially in shallow lakes. Previous models as shown in Table 1 were not used for the objective of estimating the effectiveness of eutrophication control programs.

The proposed model for nitrogen dynamics in the sediment-water system includes not only nitrogen transformation processes such as ammonification, nitrification, and denitrification, but also nitrogen transportation processes such as diffusion, sediment accumulation, and compaction, thereby enabling us to predict the seasonal changes of the release flux of NH_4-N and $NO_2 + NO_3$ - N (NO_X-N) at the sediment-water interface and the vertical distributions of NH_4-N and NO_X-N in the pore water, and particulate organic nitrogen (PON)* in the sediments.

MATERIALS AND METHODS

The model was developed for Lake Kasumigaura, the second largest lake in Japan. The lake is highly eutrophic. Annual mean concentrations of total nitrogen and total phosphorus are 1.5 mg l^{-1} and 0.09 mg l^{-1}, respectively. In summer, chlorophyll-a concentrations often exceed 200 mg m^{-3}. The lake consists of four basins with a total surface area of 171 km^2 with a mean and maximum depth of 3.9 and 7 m, respectively. The study station shown in Figure 1 is 6 m deep. In this study, NH_4-N and NO_x-N release fluxes were estimated at this station during 1981-1984 by the laboratory sediment core method and mathematical model method used by Hosomi et al. (15). At the same time (1980-1982), NH_4-N and NO_x-N concentrations in interstitial water of the sediments (12), PON in the sediments (unpublished data), and limnological data in lake water (2) were measured.

A numerical procedure by the Runge-Kutta-Gill method was employed to solve the differential equations to define the model (HITAC M-150, Computer Center, National Institute for Environmental Studies) using t = 0.02 days.

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* For convenience, symbols and abbreviations are listed in the Notation.

TABLE 1. PREVIOUS MODELS ON NITROGEN RELEASE FROM BOTTOM SEDIMENTS

Researchers	State Variable	Structure	Process
Chen & Orlob (5) Park et al. (33)	Organic sediment	Box model (1 layer)	Deposition of detritus and phytoplankton, release by decay (1st order) of organic sediment
Nyholm (27)			Release rate = a × (deposition rate) a; constant; that is, release rate is proportional to deposition rate
Jorgensen (16)	Nitrogen in sediment (PON)	Box model (1 layer)	Deposition of detritus and phytoplankton, release rate = b × (nitrogen concentration in sediment), empirical model
Ukita (35)	BDN, NH_4-N	Box model (20 layer)	Deposition of detritus and phytoplankton, mineralization of BDN, diffusion of NH_4-N
Kamiyama (18)	PON, NH_4-N	Analytical model (steady-state)	Sedimentation process, generation rate of NH_4-N, adsorption, diffusion of NH_4-N
Vanderborght, Wollast, and Billen (36)	NH_4-N, NO_3-N, O_2, SO_4	Analytical model (steady-state), two layer mass transfer model	Ammonification, nitrification, denitrification, diffusion of NH_4-N, NO_3-N, and O_2

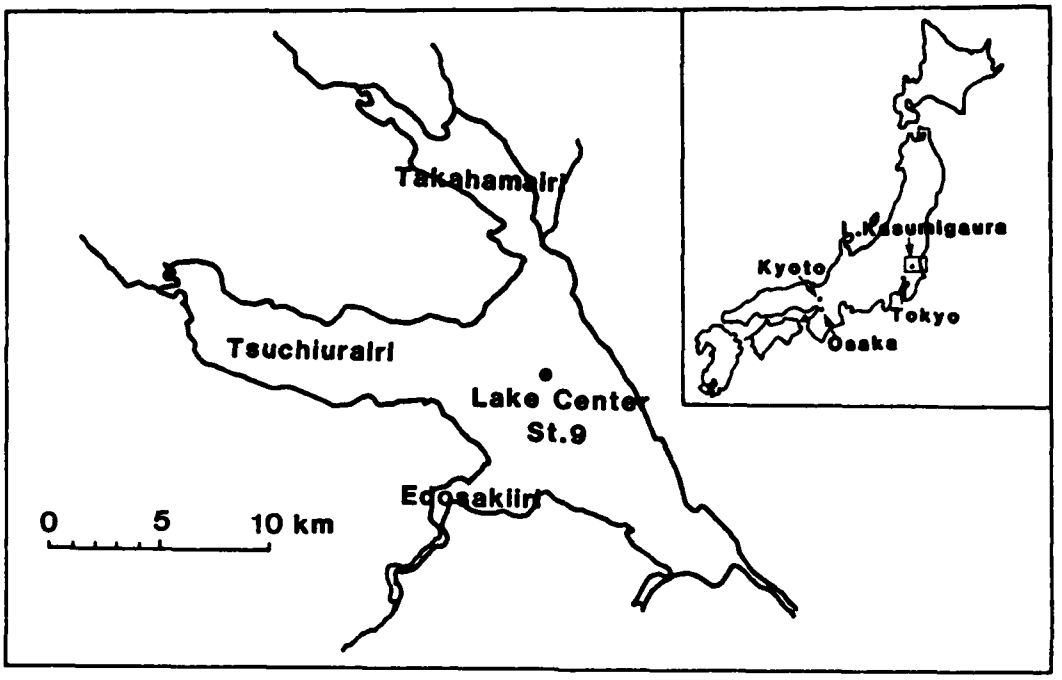


Figure 1. Study station in Lake Kasumigaura

MODEL STRUCTURE

The proposed model in this study consists of homogeneous overlying water and the sediment divided into 20 layers along the depth with the thickness of 1 cm. The model includes transformation processes of nitrogen compounds and sedimentation processes.

Sedimentation Process

Assumptions to formulate sedimentation processes are as follows:

a. The sediment consists of pore water and solid sediment.

b. Deposited materials from the overlying water form the solid sediment.

c. The volume and density of solid sediment are conserved in sedimentation processes.

d. The porosity at a definite depth does not change with time.

e. Porosity decreases with depth by sediment compaction and levels off at the specific depth.

Sediment accumulation by continuous deposition of particulate matter at the top layer of the sediment causes the sediment-water interface to move upward in reference to a fixed datum. In addition, sediment compaction due to sedimentation processes induces the advection of pore water. In order to adjust this moving interface, the space coordinate system in the model was

transformed to move with the interface so that the sediment characteristics are observed relative to the interface (z = 0). As a result, a velocity term is introduced in the sediment equations which represents the relative motion of the sediment with respect to the interface. We call this velocity term sedimentation rate. Naturally advection induced by compaction is also affected by the sedimentation rate.

Volumetric balance of solid sediment and pore water of the i^{th} layer in the sediment was established based on the assumptions a, b, and c. From assumption d, sedimentation rate of the i^{th} layer, S_i, is:

$$S_i = f_{ss}/(1 - \phi_i) \tag{1}$$

In the depth where porosity approaches its ultimate value, advection velocity of pore water by compaction approaches zero (assumption e). Then, the following equation is obtained:

$$AV_i = [1/(1 - \phi_i) - 1/(1 - \phi_\infty)] \cdot f_{ss} \tag{2}$$

From Equations 1 and 2, the net velocity of pore water is:

$$SW = \phi_\infty \cdot f_{ss}/(1 - \phi_\infty) \tag{3}$$

Nitrogen Kinetics

Major transformation processes of nitrogen compounds in the sediment-water system are as follows:

a. Deposition of PON in the overlying water to the top layer of the bottom sediment.

b. Ammonification based on the decay of PON in the sediment.

c. Adsorption equilibrium between dissolved NH_4-N in the pore water and adsorbed NH_4-N onto the solid sediment phase.

d. Nitrification of dissolved NH_4-N in the pore water.

e. Denitrification of dissolved NO_x-N in the pore water.

f. Diffusion of dissolved NH_4-N and NO_x-N in the pore water along concentration gradients.

Both nitrogen transformation processes mentioned above and sedimentation processes are incorporated into the model. Figure 2 is a schematic diagram of the structure of the model.

Sedimentation flux of PON in the overlying water to the top layer of the sediment is obtained by multiplying PON concentration in the overlying water with deposition rate V_N of PON in the overlying water. PON in the overlying water is expressed as forcing functions. PON cannot be decomposed completely (31). A fraction, 1-E, of PON is not attacked by bacteria.

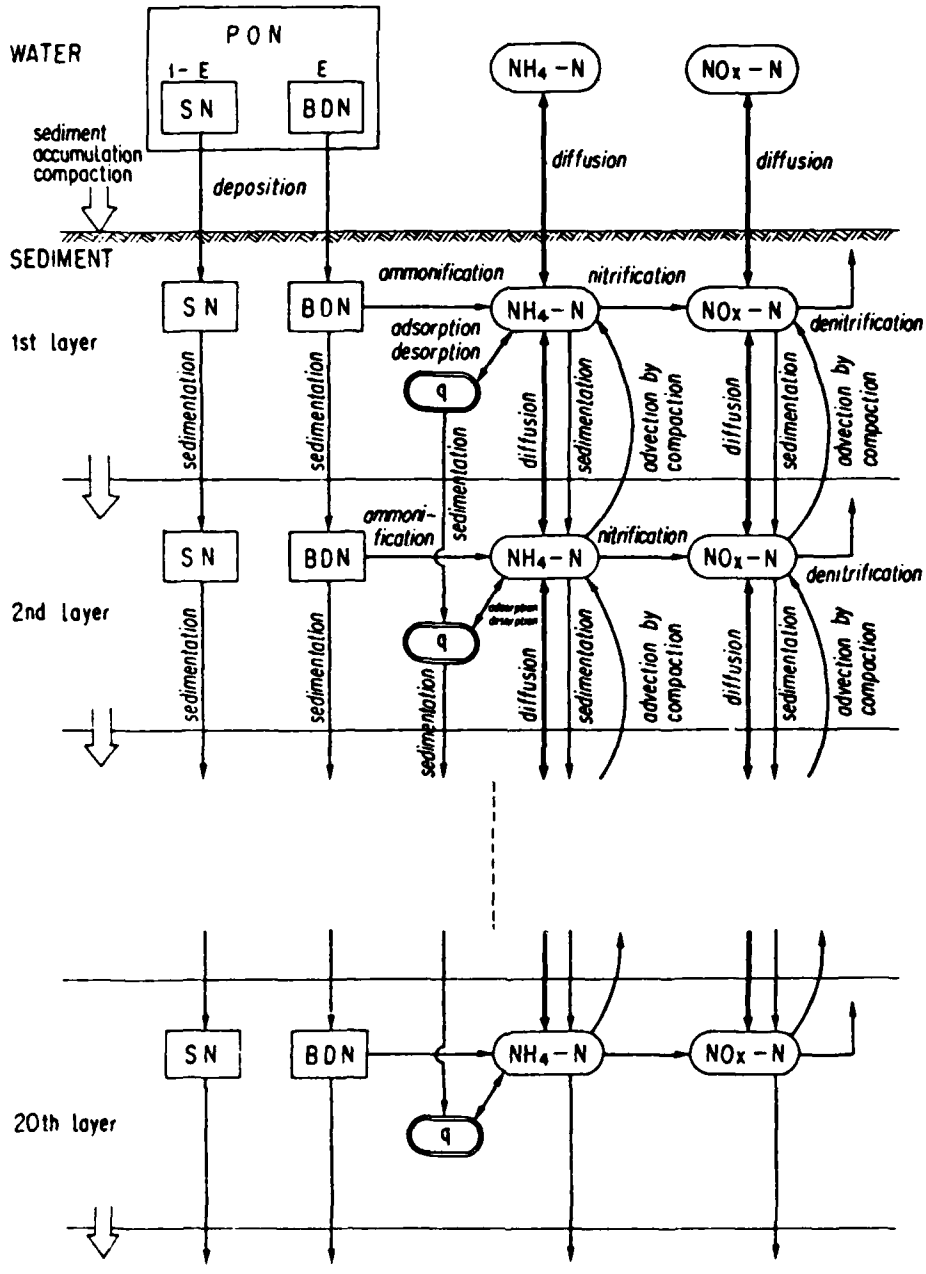


Figure 2. A schematic diagram of the structure of the model proposed

PON in the sediment can be divided into biodegradable nitrogen (BDN) and stable nitrogen (SN) (12, 35). The former is mineralized, while the latter is not. Ammonification in the sediment is caused by mineralization of BDN. This process was simulated as the first-order reaction. BDN is defined from the vertical distribution of PON in the sediment as shown by the hatched area in

Figure 3. Ammonifying bacteria decrease exponentially with the depth of sediments (13, 20). Ammonification rate is also strongly dependent upon temperature. Ammonification rate in the i^{th} layer g_i is:

$$g_i = g_{30} \cdot BDN_i \cdot \theta g^{T-30} \cdot \exp(-\alpha \cdot i) \quad (4)$$

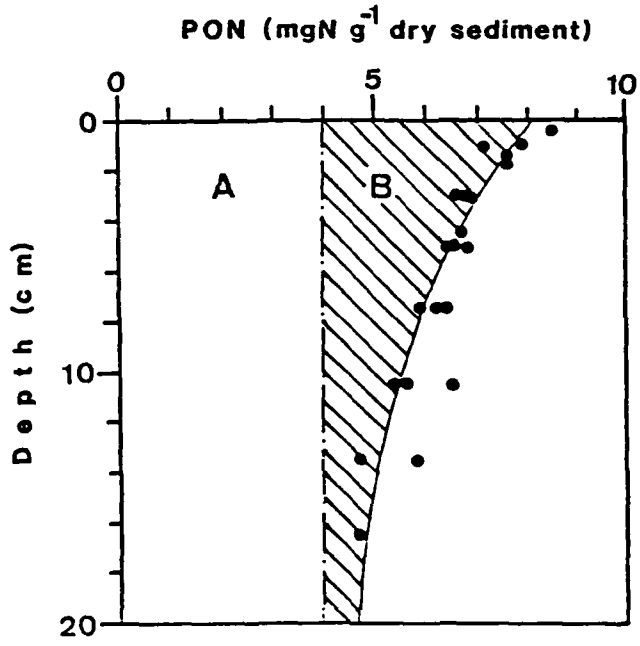


Figure 3. Vertical distribution of PON (SN(A) + BDN (B)) in the sediment in Lake Kasumigaura

Figure 4 shows a relationship between NH_4-N concentration in dissolved form and in adsorbed form in the sediments in Lake Kasumigaura. A similar simple relationship was confirmed in Lake Biwa (19) and Lake Suwa (25). Adsorbed NH_4-N onto the solid sediment phase of the i^{th} layer in the sediment q_i can be expressed by NH_4-N concentration in the pore water:

$$q_i = K \cdot NH_i \quad (5)$$

Ammonium nitrogen is transformed to nitrate in a series of bacteria-mediated oxidation reactions within the sediments superimposed by oxygenated water. Nitrifying bacteria (*Nitrosomonas* and *Nitrobacter*) decrease exponentially with the depth of sediments (1). Nitrification reactions need oxygen, but the processes can go on at dissolved oxygen (DO) concentrations as low as 0.3 mg l^{-1} (10). Nitrification rate is described by the Michaelis-Menten

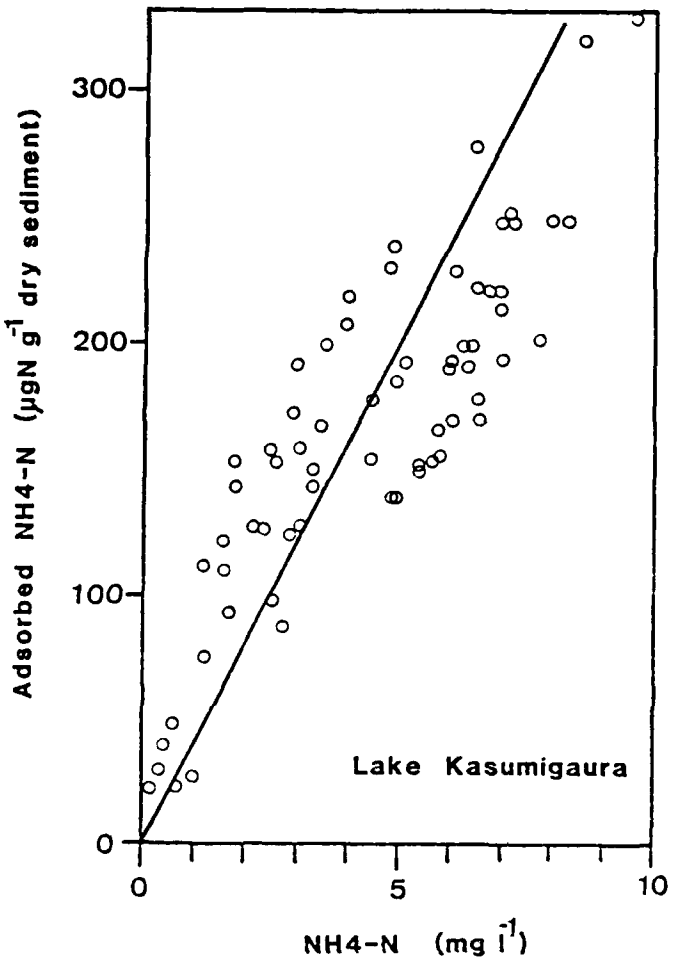


Figure 4. Relationship between NH_4-N concentration in dissolved form and in adsorbed form in the sediments in Lake Kasumigaura

equation as a function of NH_4-N and DO concentrations in the pore water. Nitrification also depends upon temperature. Nitrification rate in the i^{th} layer k_{Ni} is:

$$k_{Ni} = k_{N30} \cdot \theta_N^{T-30} \cdot \frac{NH_i}{K_{NH4} + NH_i} \cdot \frac{DO_i}{K_{DO} + DO_i} \cdot \exp(-\beta \cdot i) \qquad (6)$$

DO concentration in the sediment is:

$$\frac{\partial DO(z)}{\partial t} = D_{DO} \cdot \frac{\partial^2 DO(z)}{\partial z^2} - k_{DO} \cdot DO(z) \qquad (7)$$

In steady-state condition,

$$DO = DOW \cdot \exp\left(-\sqrt{k_{DO}/D_{DO}} \cdot z\right) \tag{8}$$

DOW is DO concentration in the sediment-water interface and assumed to be nearly equal to DO concentration in the overlying water.

Rate of DO consumption in the sediment is:

$$k_{DO} = k_{30} \cdot \theta_{DO}^{T-30} \tag{9}$$

Therefore, from Equation 8 and Equation 9 DO concentration in the i^{th} layer DO_i is:

$$DO_i = DOW \cdot \exp\left(-\sqrt{k_{30} \cdot \theta_{DO}^{T-30} / D_{DO}} \cdot i\right) \tag{10}$$

Denitrification in the sediment in Lake Kasumigaura can be described by the Michaelis-Menten equation kinetics as a function of NO_x-N concentration in the pore water (26). Nakajima (25) defined the function to express the inhibition of denitrification by DO concentration in the sediment:

$$\begin{aligned} f(DO_i) &= 1 && \text{for } DO_i < 0.48 \\ &= 1/[1 + 32(DO_i - 0.48)] && \text{for } DO_i \geq 0.48 \end{aligned} \tag{11}$$

Denitrifiers are known to decrease exponentially with the depth of the sediment in Lake Kasumigaura (1). Denitrification rate in the i^{th} layer k_{Di} is:

$$k_{Di} = k_{D30} \cdot \theta_D^{T-30} \cdot \frac{NO_i}{K_{NOx} + NO_i} \cdot f(DO_i) \cdot \exp(-\gamma i) \tag{12}$$

Dissolved nitrogen compounds in the pore water are transported by diffusion. Also, the mechanism responsible for release of NH_4-N and NO_x-N in the pore water from the sediment into the overlying water is diffusion across the sediment-water interface. Many researchers have attempted to evaluate the diffusion coefficient in sediment using various methods, but the estimated values range widely from 2×10^{-4} to 10^{-7} cm^2 s^{-1} (7, 22, 23). The nutrient exchange across the sediment-water interface is a combination of physical, chemical, and biological processes such as molecular diffusion, bioturbation, physical mixing, and adsorption-desorption. These processes are complex and difficult to measure separately. Accordingly, at present, it is difficult to evaluate effective diffusion or transport coefficients incorporating the above-mentioned processes. Freedman and Canale (7) applied the correlation of diffusion coefficient with porosity shown by Manheim (23) to the sediment of White Lake and evaluated flux from the sediment. Lerman (22) presented a

relationship between diffusion coefficients and porosity using data from Manheim (23). The value of the diffusion coefficient D_i is in proportion to ϕ_i^2.

$$D_i = D_0 \cdot \phi_i^2 \tag{13}$$

where D_0 is the value of the diffusion coefficient in the bulk (that is = 1). Diffusion coefficients increase proportionally with temperature:

$$DT_i = D_i \cdot (1 + a \cdot T) \tag{14}$$

MODEL FORMULATION AND PARAMETER EVALUATION

Table 2 shows state variables used in the model of nitrogen release from bottom sediments. A list of forcing functions is shown in Table 3.

The differential equations and adjunct equations to define this model containing sedimentation processes and nitrogen transformation processes are shown in Table 4. The equations with associated descriptors are essentially self-explanatory.

Field data from Lake Kasumigaura and laboratory data using sediments of Lake Kasumigaura were used extensively to evaluate the parameters defined in the model formulation. If sufficient information was not available, literatures were also consulted to provide a range of values for the parameter. Table 5 shows the values of parameters used in the model.

Ammonification rate g_i is determined by decomposition experiments using deposited materials and sediments taken from various depths of the sediment (Hosomi, unpublished data). It is also estimated from the vertical distribution of BDN and the sedimentation rate.

Ukita (35) used 0.05 d^{-1} as the first-order kinetic coefficient in the sediment on decomposition of BDN. Otsuki and Hanya (31) obtained 0.02 d^{-1} as the decomposition rate of dead algae at 20°C.

The estimate values ranged widely from 10^{-4} to 10^{-2} d^{-1}. Accordingly, ammonification rates were determined by the model calibration in this study in the range of estimated values.

MODEL CALIBRATION

Concentrations of nitrogen compounds in the overlying water and the sediment measured in January were averaged for 4 years (1981-1984) and used as initial values of state variables in the model.

The results of model calibration using the 1980-1982 data as forcing function are shown in Figure 5. Measured data of NH_4-N release flux during 1981-1984 were averaged each month and shown by mean values and standard deviations. Maximum NH_4-N release flux (about 70 mgN m^{-2} d^{-1}) was observed

TABLE 2. STATE VARIABLES USED IN THE MODEL OF NITROGEN RELEASE FROM LAKE SEDIMENTS

Symbol	Definition	Unit
NH_i	NH_4-N concentration in the pore water of the i^{th} layer in the sediment	gN m^{-3}
q_i	Adsorbed NH_4-N onto the solid sediment phase of the i^{th} layer in the sediment	gN g^{-1} dry sediment
NO_i	NO_x-N concentration in the pore water of the i^{th} layer in the sediment	gN m^{-3}
BDN_i	Biodegradable nitrogen (BDN) in the solid sediment of the i^{th} layer in the sediment	gN g^{-1} dry sediment
SN_i	Stable nitrogen (SN) in the solid sediment of the i^{th} layer in the sediment	gN g^{-1} dry sediment

TABLE 3. FORCING FUNCTIONS USED IN THE MODEL OF NITROGEN RELEASE FROM LAKE SEDIMENTS

Symbol	Definition	Unit
NHW	NH_4-N concentration in the overlying water	gN m^{-3}
NOW	NO_x-N concentration in the overlying water	gN m^{-3}
DOW	Dissolved oxygen concentration in the overlying water	g m^{-3}
PON	Particulate organic nitrogen concentration in the overlying water	gN m^{-3}
T	Water temperature	°C

TABLE 4. THE DIFFERENTIAL EQUATIONS AND ADJUNCT EQUATIONS TO DEFINE THE MODEL OF NITROGEN RELEASE FROM LAKE SEDIMENTS

1) *NH_4-N concentration in the pore water of the i-th layer in the sediment, NH_i* (gN m^{-3})

for i=1

$$\phi_1 \cdot \frac{dNH_1}{dt} = \underbrace{\frac{fss \cdot \phi_1 \cdot NHW}{\Delta Z \cdot (1-\phi_1)}}_{\text{gain due to moving boundary}} - \underbrace{\frac{NH_1 \cdot \phi_1 \cdot S_1}{\Delta Z}}_{\text{sedimentation}} + \underbrace{\frac{NH_1 \cdot AV_1}{\Delta Z}}_{\text{advection by compaction}} - \underbrace{\frac{DT_1 \cdot (1+\phi_1) \cdot (NH_1 - NHW)}{(\Delta Z)^2}}_{\text{diffusion across the sediment-water interface}}$$

$$+ \underbrace{\frac{DT_2 \cdot (\phi_1+\phi_2) \cdot (NH_2 - NH_1)}{2 \cdot (\Delta Z)^2}}_{\text{diffusion across the 1st layer-2nd layer interface}} + \underbrace{\rho \cdot (1-\phi_1) \cdot g_1}_{\text{ammonification}} - \underbrace{\phi_1 \cdot k_{N1}}_{\text{nitrification}}$$

for $i \geq 2$

$$\phi_i \cdot \frac{dNH_i}{dt} = \underbrace{\frac{NH_{i-1} \cdot \phi_{i-1} \cdot S_{i-1}}{\Delta Z}}_{\text{sedimentation}} - \underbrace{\frac{NH_{i-1} \cdot AV_{i-1}}{\Delta Z}}_{\text{advection by compaction}} - \underbrace{\frac{NH_i \cdot \phi_i \cdot S_i}{\Delta Z}}_{\text{sedimentation}} + \underbrace{\frac{NH_i \cdot AV_i}{\Delta Z}}_{\text{advection by compaction}}$$

$$- \underbrace{\frac{DT_i \cdot (\phi_{i-1}+\phi_i) \cdot (NH_i - NH_{i-1})}{2 \cdot (\Delta Z)}}_{\text{diffusion across the upper interface}} + \underbrace{\frac{DT_{i+1} \cdot (\phi_i+\phi_{i+1}) \cdot (NH_{i+1} - NH_i)}{2 \cdot (\Delta Z)^2}}_{\text{diffusion across the lower interface}}$$

$$+ \underbrace{\rho \cdot (1-\phi_i) \cdot g_i}_{\text{ammonification}} - \underbrace{\phi_i \cdot k_{Ni}}_{\text{nitrification}}$$

2) *Adsorbed NH_4-N onto the solid sediment phase of the i-th layer in the sediment, q_i.* (gN g^{-1} dry sediment)

$q_i = K \cdot NH_i$

for $i=1$

$$\rho \cdot (1-\phi_1) \cdot \frac{dq_1}{dt} = \underbrace{\frac{-\rho \cdot (1-\phi_1) \cdot q_1 \cdot S_1}{\Delta Z}}_{\text{sedimentation to the 2nd layer}}$$

for $i \geq 2$

$$\rho \cdot (1-\phi_i) \cdot \frac{dq_i}{dt} = \underbrace{\frac{\rho \cdot (1-\phi_{i-1}) \cdot q_{i-1} \cdot S_{i-1}}{\Delta Z}}_{\text{sedimentation from the upper layer}} - \underbrace{\frac{\rho \cdot (1-\phi_i) \cdot q_i \cdot S_i}{\Delta Z}}_{\text{sedimentation to the lower layer}}$$

(Continued)

TABLE 4 (CONTINUED)

3) *NO_x−N concentration in the pore water of the i-th layer in the sediment, NO_i* (gN m^{-3})

for $i=1$

$$\phi_1 \cdot \frac{dNO_1}{dt} = \underbrace{\frac{fss \cdot \phi_1 \cdot NOW}{\Delta Z \cdot (1-\phi_1)}}_{\text{gain due to moving boundary}} - \underbrace{\frac{NO_1 \cdot \phi_1 \cdot S_1}{\Delta Z}}_{\text{sedimentation}} + \underbrace{\frac{NO_1 \cdot AV_1}{\Delta Z}}_{\text{advection by compaction}} - \underbrace{\frac{DT_1 \cdot (1+\phi_1) \cdot (NO_1 - NOW)}{(\Delta Z)^2}}_{\text{diffusion across the sediment-water interface}}$$

$$+ \underbrace{\frac{DT_2 \cdot (\phi_1+\phi_2) \cdot (NO_2 - NO_1)}{2 \cdot (\Delta Z)^2}}_{\text{diffusion across the 1st layer-2nd layer interface}} + \underbrace{\phi_1 \cdot k_{N1}}_{\text{nitrification}} - \underbrace{\phi_1 \cdot k_{D1}}_{\text{denitrification}}$$

for $i \geq 2$

$$\phi_i \cdot \frac{dNO_i}{dt} = \underbrace{\frac{NO_{i-1} \cdot \phi_{i-1} \cdot S_{i-1}}{\Delta Z}}_{\text{sedimentation}} - \underbrace{\frac{NO_{i-1} \cdot AV_{i-1}}{\Delta Z}}_{\text{advection by compaction}} - \underbrace{\frac{NO_i \cdot \phi_i \cdot S_i}{\Delta Z}}_{\text{sedimentation}} + \underbrace{\frac{NO_i \cdot AV_i}{\Delta Z}}_{\text{advection by compaction}}$$

$$- \underbrace{\frac{DT_i \cdot (\phi_{i-1}+\phi_i) \cdot (NO_i - NO_{i-1})}{2 \cdot (\Delta Z)^2}}_{\text{diffusion across the upper interface}} + \underbrace{\frac{DT_{i+1} \cdot (\phi_{i+1}+\phi_i) \cdot (NO_{i+1} - NO_i)}{2 \cdot (\Delta Z)^2}}_{\text{diffusion across the lower interface}}$$

$$+ \underbrace{\phi_i \cdot k_{Ni}}_{\text{nitrification}} - \underbrace{\phi_i \cdot k_{Di}}_{\text{denitrification}}$$

4) *BDN (Biodegradable Nitrogen) in the solid sediment phase of the i-th layer in the sediment, BDN_i* (gN g^{-1} dry sediment)

for $i=1$

$$\frac{dBDN_1}{dt} = \underbrace{\frac{fss \cdot PON \cdot \phi_1 \cdot E}{\Delta Z \cdot (1-\phi_1)^2 \cdot \rho}}_{\text{gain due to moving boundary}} - \underbrace{\frac{BDN_1 \cdot S_1}{\Delta Z}}_{\text{sedimentation to the 2nd layer}} - \underbrace{g_1}_{\text{ammonification}} + \underbrace{\frac{V_s \cdot PON \cdot E}{\Delta Z \cdot (1-\phi_1) \cdot \rho}}_{\text{PON deposition from the overlying water}}$$

for $i \geq 2$

$$\frac{dBDN_i}{dt} = \underbrace{\frac{BDN_{i-1} \cdot S_{i-1} \cdot (1-\phi_{i-1})}{\Delta Z \cdot (1-\phi_i)}}_{\text{sedimentation from the upper layer}} - \underbrace{\frac{BDN_i \cdot S_i}{\Delta Z}}_{\text{sedimentation to the lower layer}} - \underbrace{g_1}_{\text{ammonification}}$$

(Continued)

TABLE 4 (CONCLUDED)

5) *SN (Stable Nitrogen) in the solid sediment phase of the i-th sediment layer, SN_i* (gN g^{-1} dry sediment)

for $i = 1$

$$\frac{dSN_1}{dt} = \underbrace{\frac{fss \cdot PON \cdot \phi_1 \cdot (1-E)}{\Delta Z \cdot (1-\phi_1)^2 \cdot \rho}}_{\text{gain due to moving boundary}} - \underbrace{\frac{SN_1 \cdot S_1}{\Delta Z}}_{\text{sedimentation to the 2nd layer}} + \underbrace{\frac{V_N \cdot PON \cdot (1-E)}{\Delta Z \cdot (1-\phi_1) \cdot \rho}}_{\text{PON deposition from the lake water}}$$

for $i \geq 2$

$$\frac{dSN_i}{dt} = \underbrace{\frac{SN_{i-1} \cdot S_{i-1} \cdot (1-\phi_{i-1})}{\Delta Z \cdot (1-\phi_{i-1})}}_{\text{sedimentation from the upper layer}} - \underbrace{\frac{SN_i \cdot S_i \cdot (1-\phi_i)}{\Delta Z}}_{\text{sedimentation to the lower layer}}$$

6) *Adjunct equations*

Sedimentation rate of the i-th sediment layer, S_i (m d^{-1})

$$S_i = \frac{fss}{1-\phi_i}$$

Advection induced by compaction in the i-th layer, AV_i (m d^{-1})

$$AV_i = \left(\frac{1}{1-\phi_i} - \frac{1}{1-\phi_\infty}\right) \cdot fss$$

Ammonification in the i-th layer, g_i (gN g^{-1} dry sediment d^{-1})

$$g_i = g_{30} \cdot BDN_i \cdot \theta_G^{T-30} \cdot \exp(-a \cdot i)$$

Nitrification in the i-th layer, k_{Ni} (gN m^{-3} d^{-1})

$$k_{Ni} = k_{N30} \cdot \theta_N^{T-30} \cdot \frac{NH_i}{K_{NH4} + NH_i} \cdot \frac{DO_i}{K_{DO} + DO_i} \cdot \exp(-\beta \cdot i)$$

DO concentration in the i-th layer, DO_i (g m^{-3})

$$DO_i = DOW \cdot \exp\left(-\frac{\sqrt{k_{30} \cdot \theta_{DO}^{T-30}}}{D_{DO}} \cdot i\right)$$

Denitrification in the i-th layer, k_{Di} (gN m^{-3} d^{-1})

$$k_{Di} = k_{D30} \cdot \theta_D^{T-30} \cdot \frac{NO_i}{K_{NOX} + NO_i} \cdot f(DO_i) \cdot \exp(-\gamma \cdot i)$$

$$\text{where } f(DO_i) = 1 \qquad \text{for } DO_i < 0.48$$

$$= \frac{1}{1+32 \cdot (DO_i - 0.48)} \qquad \text{for } DO_i \geq 0.48$$

TABLE 5. VALUES OF PARAMETERS USED IN THE MODEL OF NITROGEN RELEASE FROM BOTTOM SEDIMENTS

	Parameter	Unit	Values	Found by
ϕ_i	Porosity	$m^3\ m^{-3}$	0.90+(0.96-0.90) exp (-0.117·i)	Measurements
f_{ss}	Volumetric flux of sedimentation from the water to sediment surface	$m^3\ m^{-2}\ d^{-1}$	0.0000010	Estimated value on basis of Otsubo et al. (30)
D_O	Molecular diffusion coefficient at 0°C	$m^2\ d^{-1}$	0.0000864	Lerman (22)
D_{DO}	Diffusion coefficient of dissolved oxygen	$m^2\ d^{-1}$	0.0000475	Revsbech et al. (34)
a	Empirical constant		0.04	Lerman (22)
ρ	Density of solid	$g\ m^{-3}$	2400000	Measurements
g_{30}	Ammonification rate (BDN decomposition rate) in the top layer of the sediment at 30°C	d^{-1}	0.02	Calibration in the range of estimated values on basis of Hosomi and Sudo (12), Otsuki and Hanya (31), and Ukita (35)
K_{N30}	Nitrification rate in top layer of the sediment at 30°C	$gN\ m^{-3}\ d^{-1}$	5.0	Measurements
K_{D30}	Denitrification rate in the top layer of the sediment at 30°C	$gN\ m^{-3}\ d^{-1}$	100	Estimated value on basis of Nakajima and Aizaki (26)
K_{30}	DO consumption rate in the top layer of the sediment at 30°C	d^{-1}	30	Estimated value on basis of Okada and Sudo (29)
Θ_g	Temperature coefficient		1.15	Measurements
Θ_N	Temperature coefficient		1.07	Measurements
Θ_D	Temperature coefficient		1.08	Measurements

(Continued)

TABLE 5. (CONCLUDED)

	Parameter	Unit	Values	Found by
θ_{DO}	Temperature coefficient		1.08	Edberg and Hofsten (6)
α	Empirical constant to express the vertical distribution of ammonifiers		0.8	Calibration
β	Empirical constant to express the vertical distribution of nitrifying bacteria		0.4	Estimated value on basis of Aizaki and Nakajima (1)
γ	Empirical constant to express the vertical distribution of denitrifiers		0.4	Estimated value on basis of Aizaki and Nakajima (1)
K_{NH4}	Michaelis constant for NH_4-N concentration on nitrification	gN m^{-3}	0.1	Barnes and Bliss (4)
K_{NOx}	Michaelis constant for NO_x-N concentration on denitrification	gN m^{-3}	0.5	Nakajima and Aizaki (26)
K_{DO}	Michaelis constant for DO concentration on nitrification	g m^{-3}	0.3	Barnes and Bliss (4)
K	Adsorption coefficient	m^3 g^{-1} dry sediment	0.00004	Measurements
V_N	Settling velocity of PON in the overlying water	m d^{-1}	0.08	Estimated value on basis of Fukushima, Aizaki, and Muraoka (8) and Goda (9)
E	BDN fraction of PON in the overlying water		0.8	Estimated value on basis of Otsuki and Hanya (31)

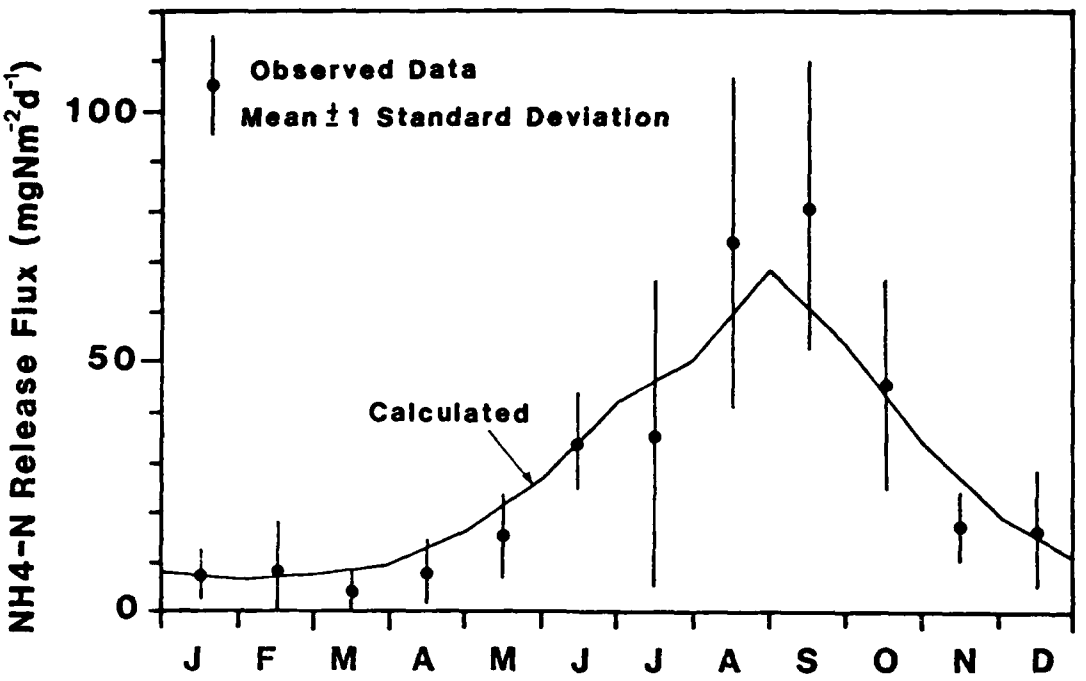


Figure 5. NH_4-N release flux from bottom sediments in Lake Kasumigaura; comparison of model calculations and observed data

during the summer and early autumn. The measured flux decreased with the decrease in water temperature, and, in most cases, was no more than 10 mgN m^{-2} d^{-1} during winter and early spring. Good agreement between the model calculations and measured release flux of NH_4-N was noted. The calculated seasonal change in NH_4-N concentrations in the pore water of the upper layers (0-6 cm) agreed fairly well with the observed profiles of NH_4-N concentrations (Figure 6), whereas the calculated NH_4-N concentrations in lower layers (6-20 cm) were lower than those of the observed data during the summer and early autumn. It must be noted that the increase in values of NH_4-N concentrations in lower layers (6-20 cm) during summer and early autumn had little effect on NH_4-N release flux. This result suggests that the nitrogen dynamics of the upper layers of the sediment regulates the nitrogen transportation between the sediment and the water.

Figure 7 shows comparisons of model calculations (PON = BDN + SN) and observed PON in the surface sediment. The calculated PON in the 1st layer increased since autumn, while the calculated PON in the 2nd layer and the 3rd layer decreased. Observed PON in the surface sediments was approximately constant within the range of sampling and analytical errors. These results seemed to be caused by the fact that this model did not consider mixing of sediment particles due to biological and physical action, e.g. organism burrowing and waves, because sufficient quantitative data were not available to evaluate the mixing rates of sediment particles in Lake Kasumigaura.

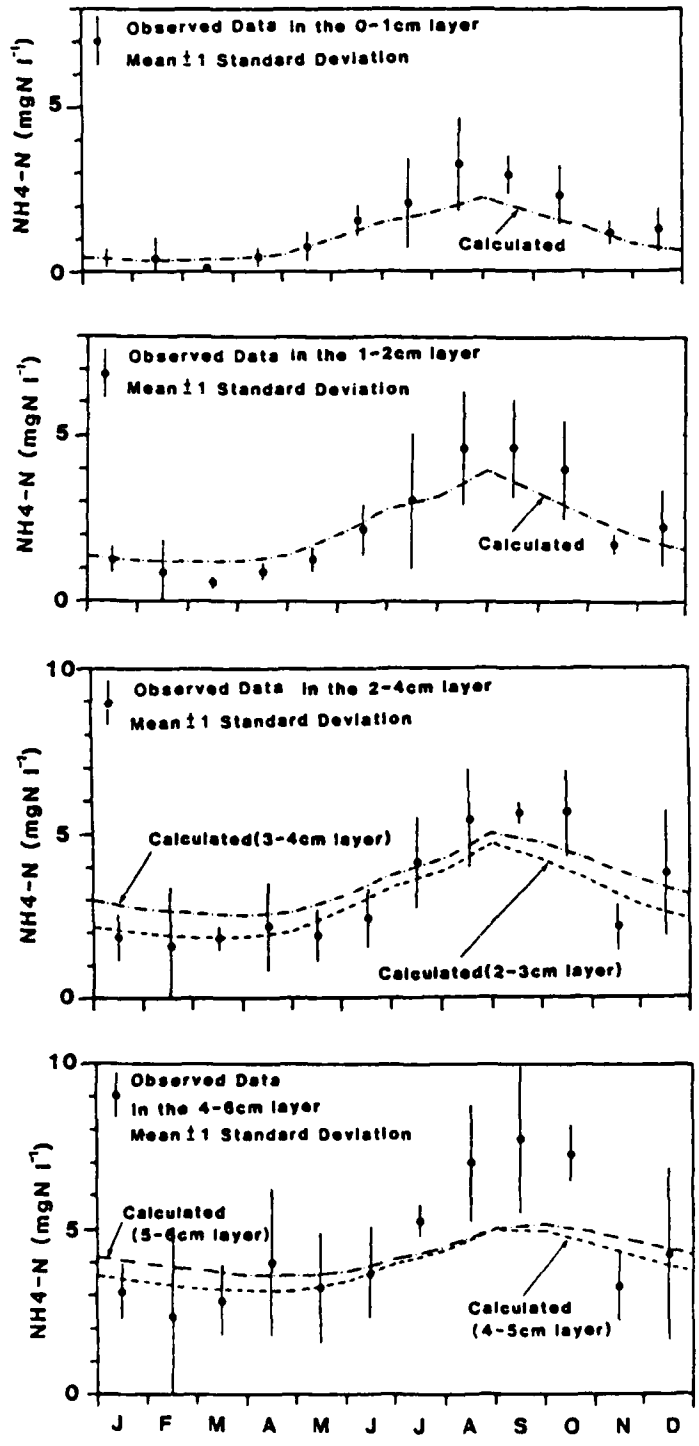


Figure 6. NH_4-N concentration in the pore water in the surface sediment of Lake Kasumigaura; comparison of model calculations and observed data

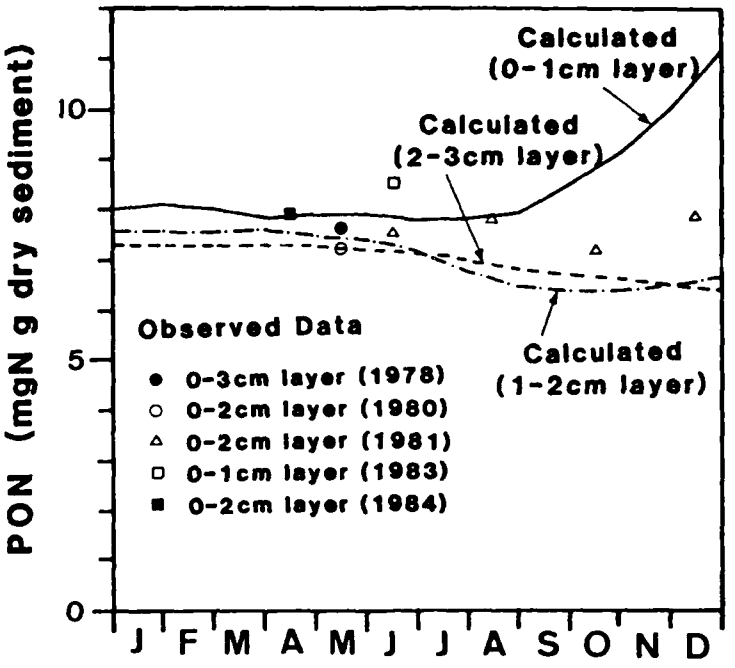


Figure 7. PON in solid sediment in Lake Kasumigaura; comparison of model calculations and observed data

Therefore, further studies on mixing effects of sediment particles is necessary. However, because observed data of PON are mean values in the 0-3 cm or 0-2 cm layers of the sediment, the average values of calculations for 0-1 cm, 1-2 cm, and 2-3 cm layer were in fair agreement with the observed values.

Figure 8 shows comparisons of calculated and observed values of NO_x-N release flux and NO_x-N concentration in the interstitial water of the 1st layer. Contrary to NH_4-N flux, NO_x-N flux was observed from overlying water to the sediment almost all the year. The agreement between the model calculations and the observed data was good, but some data deviated from the model calculations.

In order to gain additional insight into the nitrogen dynamics in the sediment-water system, characteristics of the model were examined. Long-term (10 years) calculation was carried out using the same forcing functions to affirm the stability of the model. As shown in Figure 9, seasonal variation of NH_4-N release flux was the same since the 2nd year. This result originates from the fact that BDN in the upper layer regulates the nitrogen transportation between the sediment and water and most BDN is originated from PON in the overlying water after 2 years.

Annual average flows of nitrogen in the upper layers of the sediment after the 10th year (probably steady-state) are shown in Figure 10. It should be noted that NH_4-N release flux is equivalent to 87 percent of PON

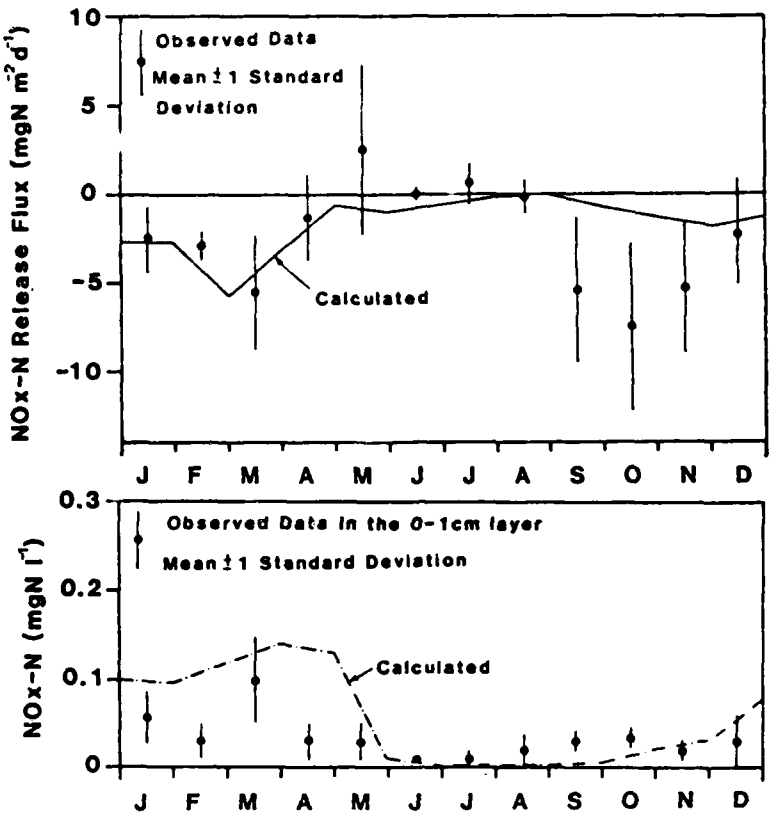


Figure 8. NO_x-N release flux from bottom sediments (a) and NO_x-N concentration in the pore water in the top layer of the sediment (b) in Lake Kasumigaura; comparison of model calculations and observed data

sedimentation flux from the overlying water to the sediment. The rate of ammonification, nitrification, and denitrification in the 1st layer of the sediment played an important role on the nitrogen transformation in the total sediment system. NH_4-N release flux depended on ammonification of BDN. The flux from the 2nd layer to the 1st layer was not negligible, but this flux also depended on the ammonification of BDN in the 2nd layer.

SENSITIVITY ANALYSIS

A sensitivity analysis of the model parameters for annual mean NH_4-N release flux was carried out to provide further insight into the mechanism of nitrogen release from the sediment and to identify further research needs.

By doubling or halving the parameter values of the control simulation obtained by the calibration mentioned above, the ratios of the annual mean NH_4-N release fluxes estimated both from perturbed and control simulations were determined (Table 6). Short-term and long-term effects of parameter sensitivity on annual mean NH_4-N release flux were estimated by the simulation of the 1st year and the 10th year, respectively.

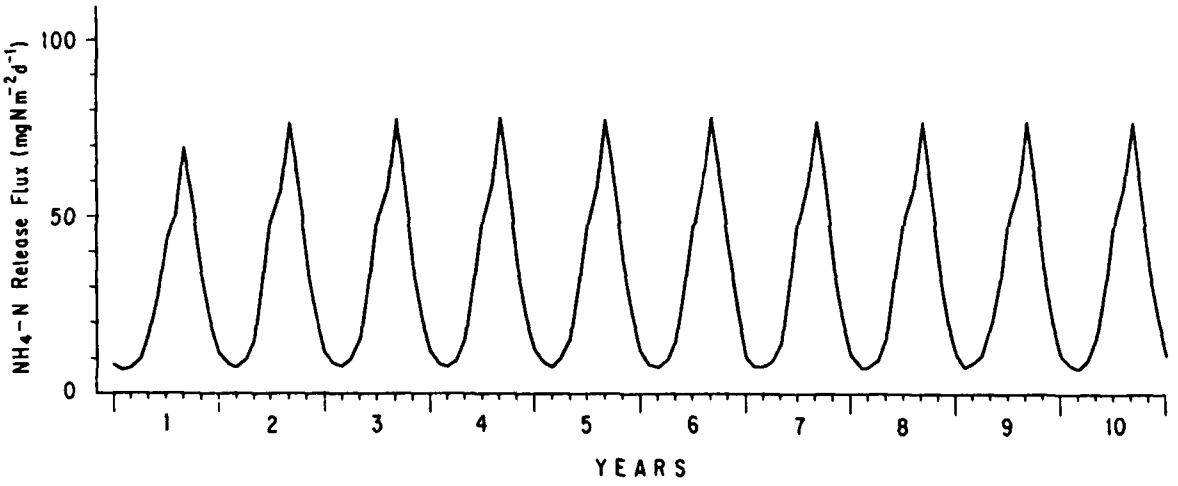


Figure 9. Simulation of NH_4-N release flux from bottom sediments in Lake Kasumigaura

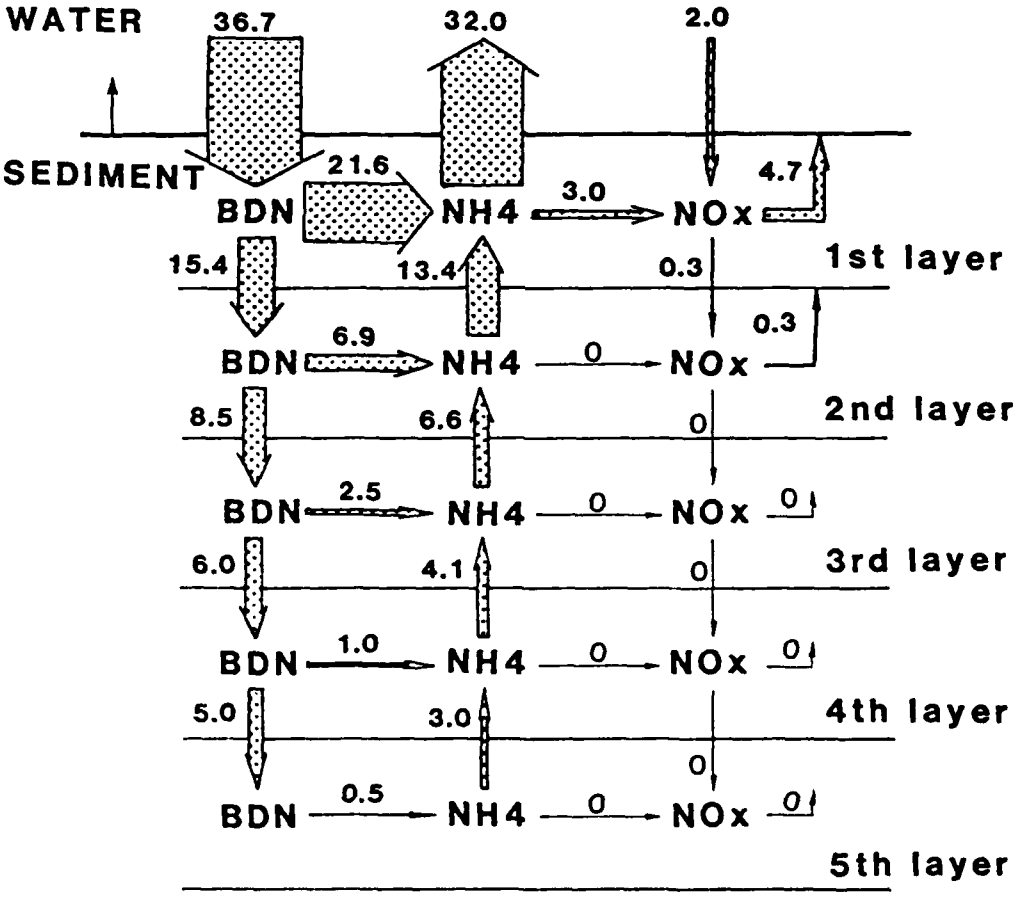


Figure 10. Annual average flows of nitrogen in the upper layers of the sediment in the 10th year

TABLE 6. SENSITIVITY OF ANNUAL MEAN NH_4-N RELEASE FLUX FROM BOTTOM SEDIMENT RESPONSE TO PARAMETER VALUES (THE RATIOS OF ANNUAL NH_4-N RELEASE FLUXES ESTIMATED BOTH FROM PERTURBED AND CONTROL SIMULATIONS.)

	Ratio of perturbed/control simulation			
	Short term	(1st year)	Long term	(10th year)
Parameter	× 1/2*	× 2**	× 1/2*	× 2**
V_N	0.79	1.42	0.52	1.96
g_{30}	0.66	1.42	0.80	1.10
α	1.27	0.82	1.12	0.84
k_{N30}	1.05	0.91	1.05	0.92
k_{D30}	1.00	1.00	1.00	1.00
K	1.01	0.98	1.01	0.99
D_O	0.84	1.23	0.93	1.13
f_{ss}	1.08	0.87	1.13	0.77

* Halving parameter values of the control simulation.
** Doubling parameter values of the control simulation.

Ammonification rate, g_{30}, was the most sensitive parameter for NH_4-N release flux in the short term. But in the long term, settling velocity of PON in the overlying water, V_N, was more sensitive than g_{30}. The model was also sensitive to the parameter α to define the vertical distribution of ammonification rate. Therefore, further studies on the vertical distribution of ammonification rate in the sediment and PON sedimentation flux from the overlying water to the sediments would be necessary.

The results also show that increasing or decreasing the diffusion coefficient D_O does not change the annual mean NH_4-N release flux more significantly than V_N and g_{30}.

EUTROPHICATION CONTROL PROGRAMS

The effectiveness of eutrophication control programs on the nitrogen release flux rate from sediment was evaluated using the calibrated model. Sediment dredging, bottom sealing, artificial aeration, and reduction of external loading as eutrophication control programs were studied.

The same rates of ammonification, nitrification, and denitrification in the top layer in the sediment were used in the eutrophication control simulations and control simulation based on the following.

a. The number of bacteria attached on the deposited materials is several times as large as the bacteria in the surface sediment (0-2 cm layer) (1).

b. BDN in the top layer is formed by BDN originating from the overlying water except for the first year.

Accordingly, even if the sediment-water interface would be changed significantly as a result of sediment dredging or bottom sealing, bacterial activity and BDN in the top layer in the sediment are not different from those of the control simulation. However, ammonification, nitrification, and denitrification rates in the layers lower than the 2nd layer were assumed to be zero in sediment dredging and bottom sealing simulations.

Initial conditions in sediment dredging and bottom sealing simulations are shown in Figure 11. In the bottom sealing simulation, initial values in the layer (0-10 cm) covered with nitrogen free fly ash and sand were all zero.

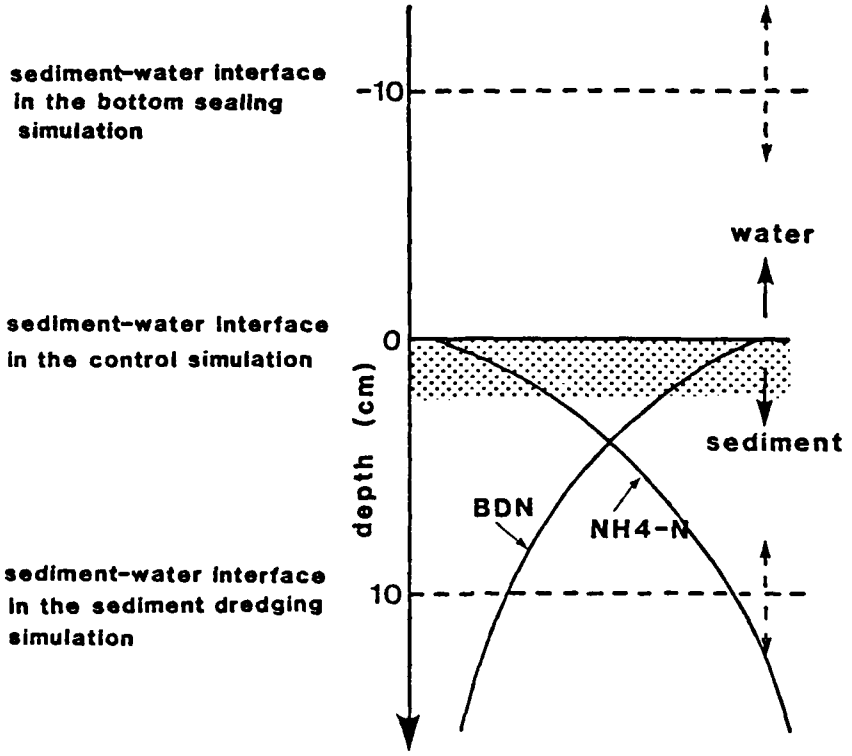


Figure 11. Schematic diagram on the initial conditions of the control, sediment dredging, and bottom sealing simulations

Artificial aeration not only increases DO concentration in the overlying water, but also stimulates nitrification activity in the sediment-water interface. In this simulation, the nitrification rate in the top layer at 30°C, k_{N30}, was multiplied four times.

Reduction of external nitrogen loading would reduce the rate of deposition from the overlying water to the sediment. In this simulation, deposition rate V_N was halved irrespectively of the amount of external loading reduction. Figures 12-15 show the comparison of control simulation and the results of eutrophication control programs.

In the case of sediment dredging, a large flux of NH_4-N from sediment was observed immediately after the sediment dredging. This flux was caused by the transfer of the highly accumulated NH_4-N in the pore water and adsorbed onto the sediment particle in the lower layer to the surface sediment as a result of dredging. Sediment dredging contributed only slightly to the reduction of NH_4-N release flux with the exception of the summer season.

Sediment sealing seems to be a more effective countermeasure for NH_4-N release flux than dredging. This result showed that highly accumulated dissolved and adsorbed NH_4-N in the surface layer of the sediment affected the NH_4-N release flux.

Figure 14 shows that artificial aeration is an effective rehabilitation technique for preventing NH_4-N release from the bottom sediment. Experiments have confirmed that nitrogen did not release from the sediment to the overlying water under aerobic conditions except for a few days after initiation of the experiments. NO_x-N may be the limiting factor in the denitrification in the sediment-water system because a large amount of organic matter was contained in the sediment. Aeration accelerated nitrification in the sediment-water system and, as a result, denitrification occurred in the surface sediment.

The 50-percent reduction of nitrogen deposition rate to the sediments was most effective for the control of NH_4-N release from the sediment among four cases studied.

CONCLUSIONS

A multi-component dynamic model for nitrogen in the bottom sediment-water system of Lake Kasumigaura, a highly eutrophic and shallow lake, was developed. Nitrogen transformation and transportation processes were incorporated into the model. In order to obtain quantitative information on the sediment-water interactions and to identify the deficiency of our knowledge at present and to determine future research needs, the model was calibrated and the sensitivity analysis of the model parameters was conducted.

Good agreement between the model calculations and observed data of NH_4-N release flux from sediment, and NH_4-N and NO_x-N concentrations in the pore water of the upper layer was noted.

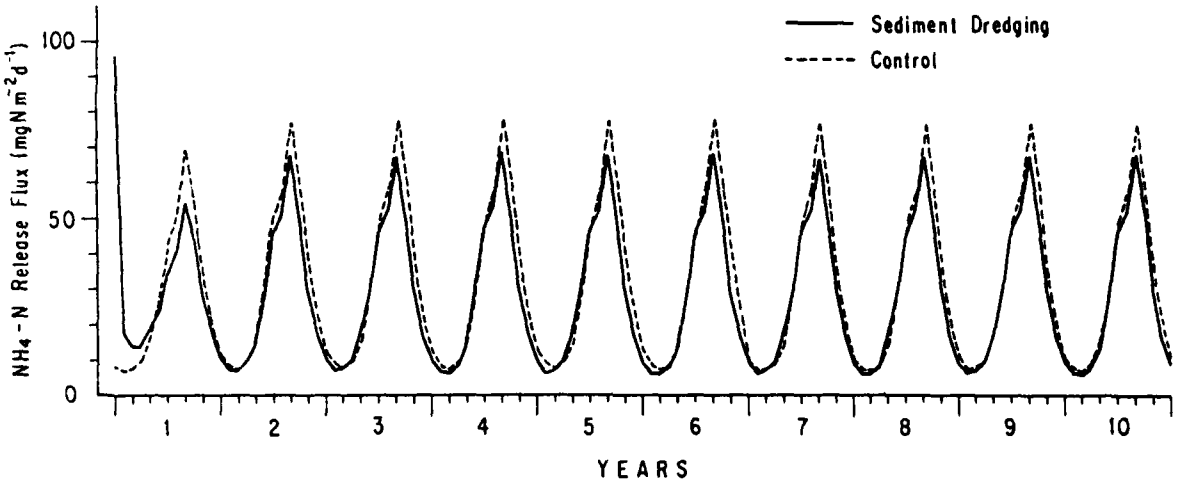


Figure 12. Effects of sediment dredging on NH_4-N release flux; control vs. sediment dredging

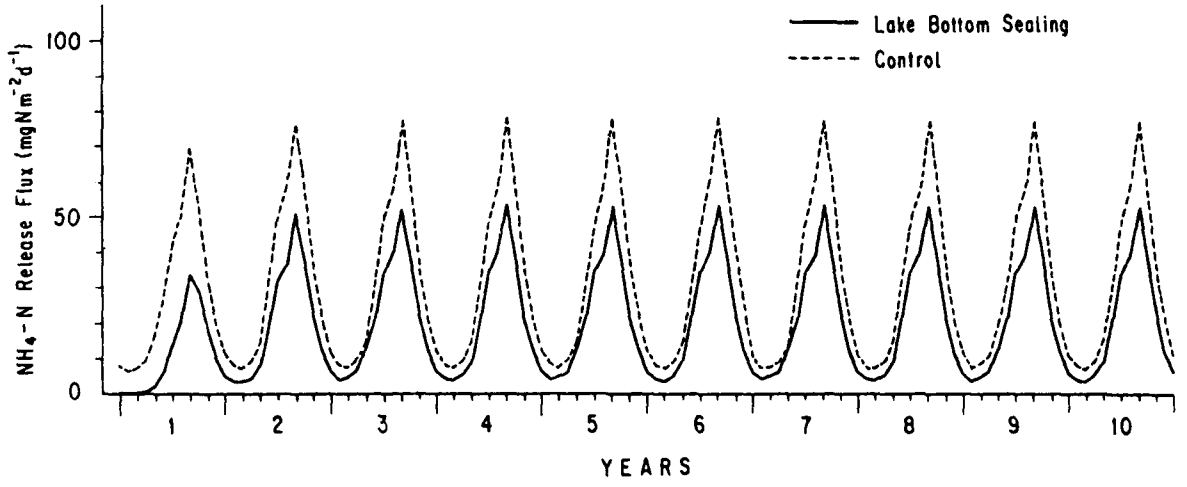


Figure 13. Effects of bottom sealing on NH_4-N release flux; control vs. bottom sealing

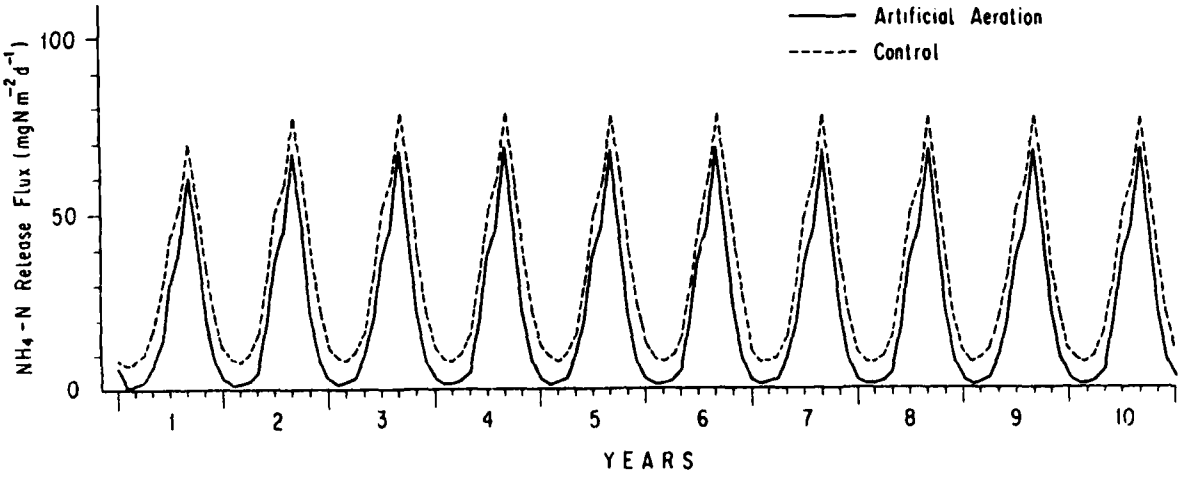


Figure 14. Effects of artificial aeration on NH_4-N release flux; control vs. artificial aeration

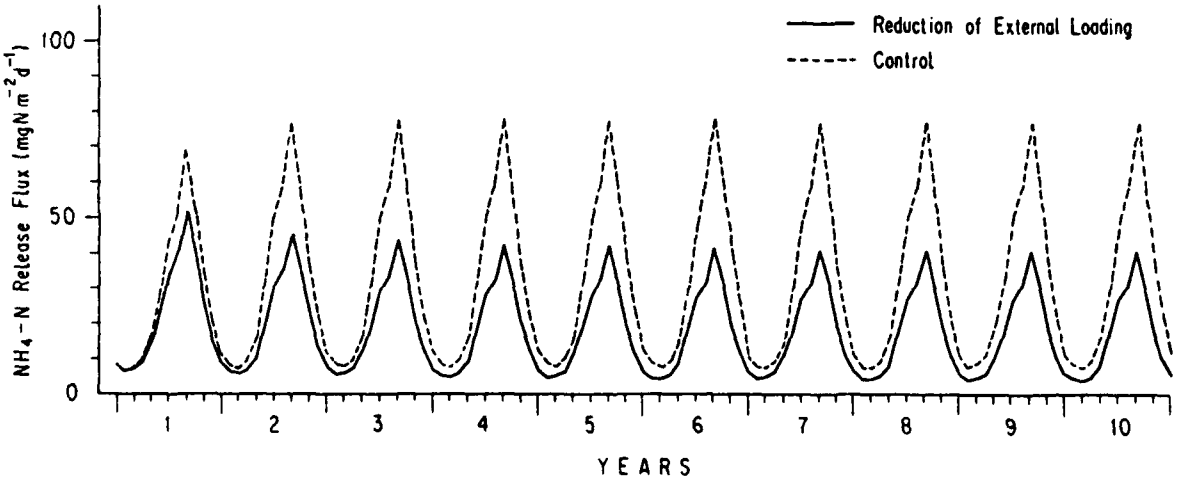


Figure 15. Effects of 50-percent reduction of external nitrogen loading on NH_4-N release flux; control vs. 50-percent reduction of external nitrogen loading

Annual average flows of nitrogen in the upper layers of the sediment showed that the rates of ammonification, nitrification, and denitrification in the 1st layer of the sediment played an important role on the nitrogen dynamics in the total sediment system. Further studies on the vertical distribution of ammonification rate were found to be necessary from the sensitivity analysis.

The effectiveness of eutrophication control programs on the nitrogen release flux from sediment was evaluated using the calibrated model. The reduction of nitrogen deposition rate to the sediment and the bottom sealing were more effective than the sediment dredging for the control of NH_4-N release from the sediment.

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NOTATION

a	Empirical constant
AV_i	Advection velocity induced by compaction in the i^{th} layer, m d^{-1}
BDN_i	Biodegradable nitrogen in the solid sediment phase of the i^{th} layer, gN g^{-1} dry sediment
D_0	Molecular diffusion coefficient of NH_4-N and NO_x-N at 0°C and in the bulk, m^2 d^{-1}
D_{DO}	Molecular diffusion coefficient of dissolved oxygen at 0°C and in the bulk, m^2 d^{-1}
D_i	Diffusion coefficient of NH_4-N and NO_x-N in the i^{th} layer in the porosity ϕ_i, m^2 d^{-1}
DO_i	Dissolved oxygen concentration in the i^{th} layer, g m^{-3}
DOW	Dissolved oxygen concentration in the overlying water, g m^{-3}
DT_i	Diffusion coefficient of NH_4-N and NO_x-N in the i^{th} layer at T°C, m^2 d^{-1}
E	BDN fraction of PON in the overlying water
$f(DO_i)$	Function to express inhibition of denitrification by DO concentration in the i^{th} layer
f_{ss}	Volumetric flux of sedimentation from the overlying water onto the sediment surface, m^3 m^{-2} d^{-1}
g_{30}	Rate of BDN decomposition at 30°C in the surface sediment, d^{-1}
g_i	Rate of ammonification in the i^{th} layer, gN g^{-1} dry sediment d^{-1}
K	Adsorption coefficient, m^3 g^{-1} dry sediment
k_{30}	Rate of DO consumption at 30°C, d^{-1}
k_{D30}	Rate of denitrification at 30°C in the surface sediment, gN m^{-3} d^{-1}
k_{Di}	Rate of denitrification in the i^{th} layer, gN m^{-3} d^{-1}
k_{DO}	Michaelis constant for DO concentration on nitrification, g m^{-3}
k_{N30}	Rate of nitrification at 30°C in the surface sediment, gN m^{-3} d^{-1}
K_{NH4}	Michaelis constant for NH_4-N concentration on nitrification, gN m^{-3}

k_{Ni} Rate of nitrification in the i^{th} layer, gN m^{-3} d^{-1}

K_{NOx} Michaelis constant for NO_x concentration of denitrification, gN m^{-3}

NH_i NH_4-N concentration in the pore water of the i^{th} layer, gN m^{-3}

NHW NH_4-N concentration in the overlying water, gN m^{-3}

NO_i NO_x (=NO_2+NO_3-N) in the pore water of the i^{th} layer, gN m^{-3}

NOW NO_x-N concentration in the overlying water, gN m^{-3}

PON Particulate organic nitrogen concentration in the overlying water, gN m^{-3}

q_i Adsorbed NH_4-N onto the solid sediment phase in the i^{th} layer, gN g^{-1} dry sediment

S_i Sedimentation rate of the i^{th} layer in the sediment. This velocity term is caused by the combined effects of sediment accumulation and compaction, m d^{-1}

SN_i Stable nitrogen in the solid sediment phase of the i^{th} layer, gN g^{-1} dry sediment

SW Net velocity of pore water, m d^{-1}

T Water temperature, °C

V_N Rate of PON deposition from the overlying water, m d^{-1}

Δz Thickness of the sediment layer, m

α Empirical constant to express the vertical distribution of ammonifiers

β Empirical constant to express the vertical distribution of nitrifying bacteria

γ Empirical constant to express the vertical distribution of denitrifiers

θ_g Temperature coefficient

θ_N Temperature coefficient

θ_D Temperature coefficient

θ_{DO} Temperature coefficient

ϕ_i Porosity of sediment in the i^{th} layer, i.e. volume fraction of pore water per unit volume of bulk sediment, m^3 m^{-3}

ϕ_∞ Ultimate porosity, i.e. porosity decreases with the depth by the sediment compaction and levels off at the specific depth, $m^3\ m^{-3}$

ρ Density of solid sediment phase, $g\ m^{-3}$

STUDY ON CHEMICAL BEHAVIOR BETWEEN SEDIMENT AND UPPER WATER

H. Nakanishi, Professor
M. Ukita, Assistant Professor
Y. Kawai

Division of Sanitary Engineering
Department of Civil Engineering, Yamaguchi University
Tokiwadai 2557, Ube-city, Yamaguchi Prefecture, Japan 755

ABSTRACT

This study investigated modeling for the assessment of eutrophic water and sediment quality, with a focus on the exchange of nutrients between water and sediments and on the primary production of phytoplankton.

This model was applied to a simplified core system consisting of two layers of water and two layers of sediment. The fundamental elements were phosphorus, nitrogen, oxygen, and iron. Sulfide ions were regarded as debts of dissolved oxygen (DO), and ferric oxide as a source of DO, respectively. The oxidized zone (ferric oxide), which has a certain capacity of phosphate adsorption, is formed by the reaction of active iron in the upper layer of sediment with DO in the lower layer of water. The oxidized zone is reduced by the sulfide ions in the upper layer of sediments and, thus, an equal part of the phosphate is released.

The supply of phosphorus and nitrogen to the sediment depends upon the settling of phytoplankton from water. The phytoplankton decompose and are partially adsorbed to the clay surface or to the oxidized zone.

The released nutrients diffuse upward toward the water or downward toward the lower layer of sediments. The nutrients which diffuse upward are assimilated by phytoplankton which settle onto sediments and recycle in the same manner.

The main parameters were changed seasonally as a function of temperature.

Using this model, the characteristic patterns of seasonal variation of phosphate discharge from sediments, the DO concentration in water, and the rise and fall of the oxidized zone were satisfactorily simulated.

The method proposed here would be useful as a significant submodel for prevailing water quality assessments.

INTRODUCTION

In the recent past various efforts have been made in eutrophic water areas to restore water quality: for example, by reducing the pollutant load or by dredging polluted bottom sediment. With the rapid progress of computer technology, many eutrophication models have been developed (2,6,7,10,12).

In order to evaluate the effect of these efforts on water quality, the prediction of sediment quality is essentially needed. However, in most cases of water quality assessment, the influence of sediment against overlying water is generally expressed by constant terms. The complicated mechanism of phosphate release from sediment especially has not been satisfactorily simulated (1,3,8,9,11,15).

This study, based on experiments and field surveys in eutrophic coastal sea areas, investigated the numerical model of the behavior of nutrients between water and bottom sediment.

WATER AND SEDIMENT QUALITY SURVEY FOR THE MODELING

Methods

Release Rate of Nitrogen (N) and Phosphorus (P) from Sediments

By using the specially made core sampler, undisturbed sediment samples 10-20 cm deep were collected in acryl pipes 10 cm in diameter (Figure 1).

The experimental procedures of N and P release rate are indicated in Figure 2. These experiments were made at a constant temperature and DO concentration similar to field condition. The release rate of N and P was calculated by measuring the increase of ammonium-N and orthophosphate in overlying water for 3-5 days.

Determination of Soluble Nitrogen (SN), Soluble Phosphate (SP), Biodegradable Nitrogen (BDN), and Estimation of Decomposition Rate Coefficient of Organic Matter in Sediment (KD3)

SN and SP were determined as follows: ca. 5 g of wet sediment samples were weighed into wide-mouth incubation bottles, filled with deoxygenated seawater and tightly closed, and then mixed vigorously. After 30 min, the supernatant was filtered with GFC filter paper and analyzed for ammonium-N and ortho-P.

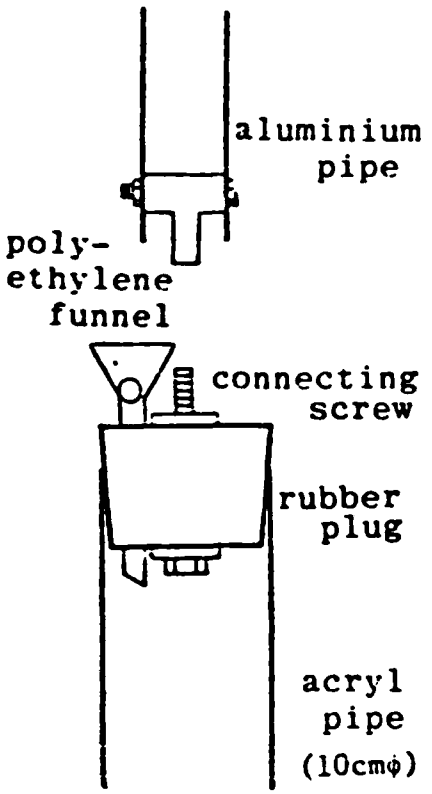


Figure 1. Core sampler

BDN was determined by measuring SN after long-term incubation in filled and tightly closed bottles. The decomposition rate coefficient K of organic matter in sediment was estimated by the following first-order reaction equation:

$$K = (1/t)\ln[(Sd - So)/(Sd - St)] \quad (1)$$

where

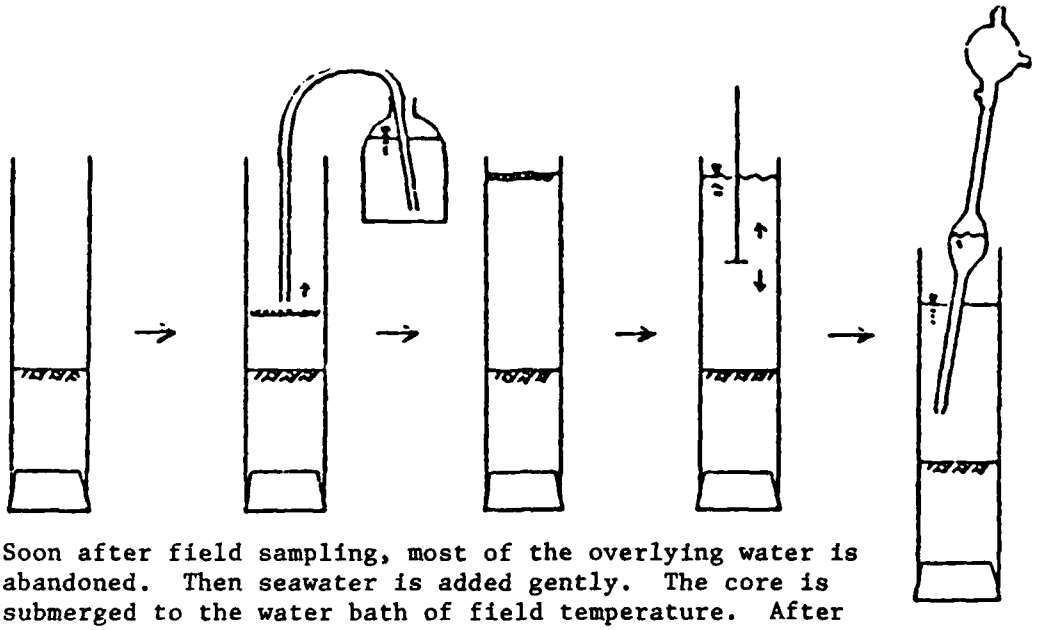
K = decomposition rate coefficient, day^{-1}

t = incubation period, days

Sd = ultimate BDN at 35° C for 30 days, μg/g wet sediment

So = initial Sn, μg/g wet sediment

St = SN after t days with the same temperature as field condition, μg/g wet sediment



Soon after field sampling, most of the overlying water is abandoned. Then seawater is added gently. The core is submerged to the water bath of field temperature. After 2 or 3 hr, a fixed volume of sample is taken for measurement of ammonium-N and ortho-P along a specified time series.

Figure 2. Experiment procedures for N and P release rate from bottom sediments

Maximum Specific Production Rate Coefficient in the Upper Layer of Water (KP1)

In order to obtain the value of KP1, production and respiration rate and the ratio of chlorophyll-a/Pss*, etc., were measured in the western part of Seto Inland Sea in 1979 by using the pseudo-in-situ DO method. The incubation period was 24 hr for both production and respiration. The former was expressed as gross production rate.

Cultivation of Phytoplankton by Microcosm Using Core Sample of Polluted Bottom Sediment

Using the apparatus indicated in Figure 3, various samples of phytoplankton were obtained. These were filtered with the GFC filter paper and analyzed for CODcr (Standard Method), CODmn (JIS K0102), TKN, and TP.

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* Pss = particulate phosphorus. This rate is used when we convert the respiration or production rate per unit amount of chlorophyll-a into per unit amount of particulate phosphorous, i.e. phytoplankton. By converting the value of O_2 respirated or produced into phosphorous, then we can calculate the values of KP1 or KD1.

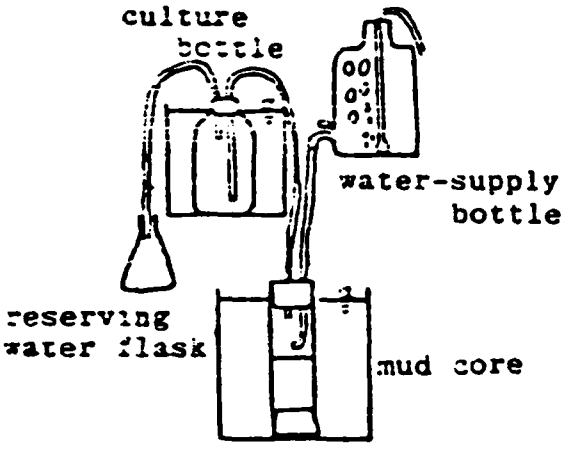


Figure 3. Apparatus of microcosm for cultivation of phytoplankton

Results

Seasonal Variation of N and P Release Rate (RN, RP)*

The characteristic patterns of seasonal variation of RN and RP are presented in Figure 4. Both RN and RP have notable variations, but RP changes more markedly. The peak position of RN is shifting toward late summer. Table 1 indicates the values of RN, RP, and RO measured in a similar manner.

Season Changes of the Vertical Distribution of SN and SP in Sediment

Figure 5 indicates the summer and winter pattern of vertical distribution of SN and SP in the sediment of some fishery harbors. It shows that SN and SP concentrations in surface layers rise in summer due to active degradation of organic matter; thereafter they diffuse upward or downward according to the concentration gradient.

Season Variation of Degradable Organic Nitrogen (FN), Decomposition Rate Coefficient (KD3), and DO Concentration of Water

Figure 6 shows the seasonal variation of FN and KD3 in the upper and lower layers of Maruo Harbor sediment. It is noted that (a) KD3 has a value ranging from 0.005 to 0.112 (day^{-1}) and higher in summer than in winter, (b) FN has a value ranging from 37 to 126 ($\mu g/g$ wet sediment) and higher in winter than in summer, and (c) both KD3 and FN have larger values and a more remarkable variation in the upper layer than in the lower layer.

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* RN, RP = released nitrogen and phosphorus from sediment to the overlying water, respectively.

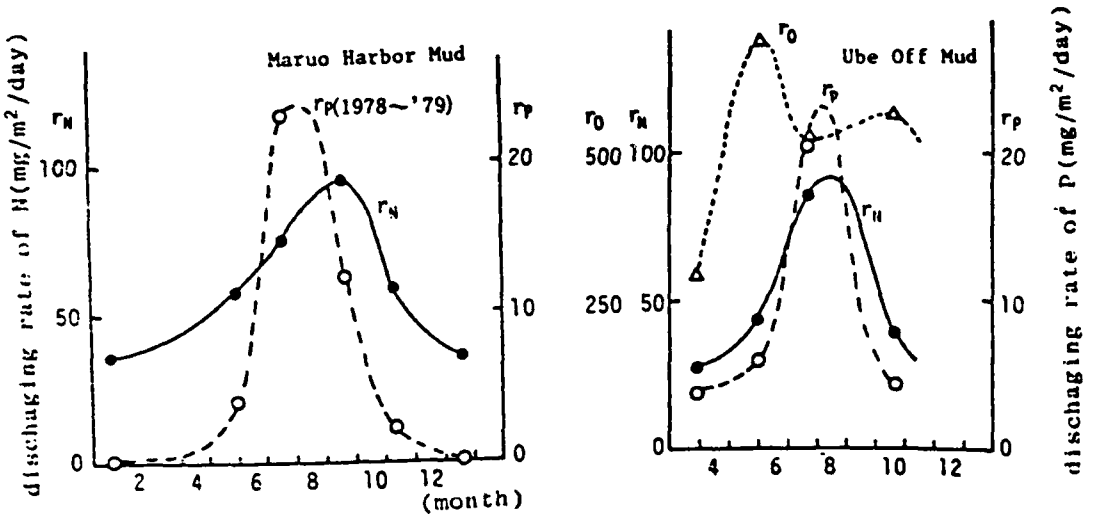


Figure 4. Seasonal variation of N and P release rate from sediments

TABLE 1. VALUES OF N AND P RELEASE RATE AND DO CONSUMING RATE (RN, RP, RO) ($mg/m^2/day$)

Sea Area	Sampling Time	n	RN	RP	RO
Ube Off	4 times '77-'78	7	85	6.9	510
Tokuyama Bay	Aug. Nov. '77	3	38	4.0	470
Fishery Ports near Ube	Summer '75	7	194	36	
	Autumn-	7	86	10	
	Winter '77	7	75	7.4	
West Suo Nada	Aug. '79	40	23	3.3	310
	Mar. '80	34	8	tr	170
Yamaguchi Bay	Aug. '81	6	34	3.8	640
	Mar. '81	3	4	tr	220
Tokyo Bay	Sep. '76*	11	150	24	1800
Osaka Bay	Aug. '76*	5	38	23	800
Hakata Bay	Aug. '81**	27	76	5.0	
	Mar. '81**	27	11	0.8	

* Data of the Ministry of Transport.
** Data of Kyushu Envir. Eval. Association.

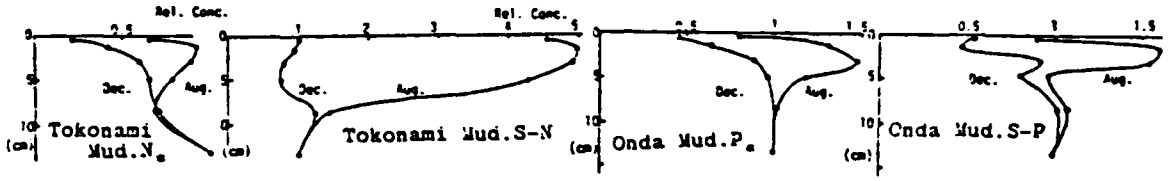


Figure 5. Vertical distribution of soluble ammonium-N and ortho-P

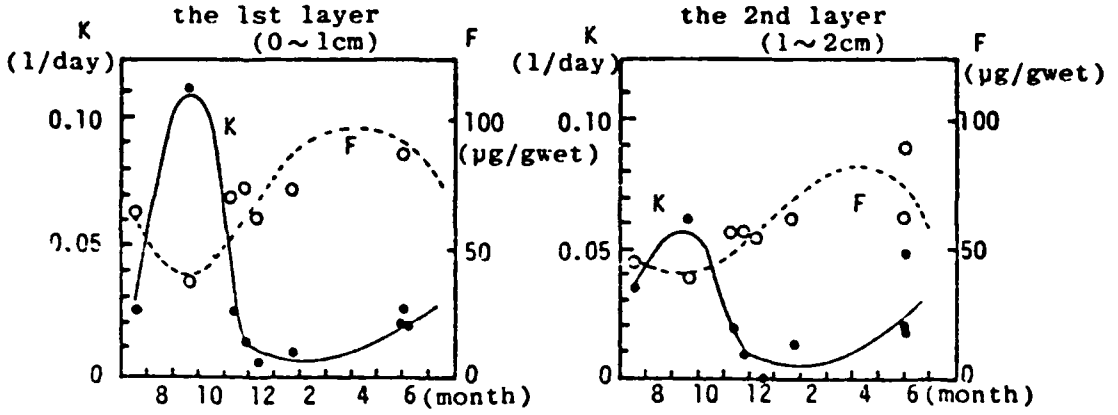


Figure 6. Seasonal variation of degradable organics and decomposition rate constant in sediment

Figure 7 shows the seasonal variation of DO in the upper and lower water at the anchor site in Tokuyama Bay.

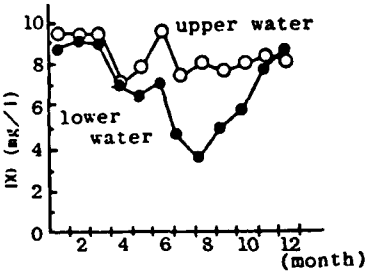


Figure 7. Seasonal variation of DO in Tokuyama Bay

Maximum Specific Production Rate Coefficient (KP1) a.
Respiration or Decomposition Rate Coefficient in Water (KD1,2)

The following values were obtained concerning production and respiration of phytoplankton. The maximum specific production rate is 0.1, 0.5, and 1.0 mg O_2/μg chl-a/day in winter, spring, and summer, respectively. In contrast, the respiration rate coefficient does not have as notable a seasonal variation. Though the data have a slightly large fluctuation, the constant value of 0.1 mg O_2/μg chl-a/day was tentatively adopted. Assuming that

chl-a/Pss = 0.8, and 1 mg of P is equivalent to 143 mg of O_2, KP1 and KD1,2 are tentatively expressed as follows: KP1 = 0.56 (winter), 2.8 (spring), 5.6 (summer) (day^{-1}), and KD1,2 = 0.56 (day^{-1}).

Composition and Conversion Factor of Phytoplankton in Eutrophic Coastal Sea Area

The values of CODcr/N and CODcr/P of phytoplankton obtained are 17.5 (n = 15, r = 0.96) and 131 (n = 25, r = 0.84), respectively. Assuming that the percentage of oxidizable fraction of phytoplankton for CODcr would be approximately 90 percent, the conversion factors TOD/N = 19.7* and TOD/P = 143, derived from the generally prevailing formula $C_{106}H_{263}O_{110}N_{16}P$, would also be available for phytoplankton in eutrophic coastal sea areas. The TOD/COD is considered to be ca. 3 based on CODcr/COD = 2.63 (n = 9, r = 0.99).

THE MODEL FOR PREDICTION OF P, DO, N, AND COD CONCENTRATION IN A SIMPLIFIED CORE SYSTEM

Outline of the Model

Figure 8 shows the conceptual outline of the core model consisting of two layers of water and two layers of sediment. The fundamental equations and the explanation of symbols are given in Tables 2 and 3, respectively.

The mechanism of the nutrient regeneration or recycling system is composed of several steps, as follows:

1) Assimilation of nutrient by phytoplankton.

2) Settling and sedimentation of phytoplankton.

3) Decomposition of phytoplankton in the upper layer of sediment and regeneration of nutrient.

4) Adsorption and desorption or precipitation and dissolving of nutrient between interstitial water and sediment.

5) Diffusion of nutrient toward the overlying water.

The flux of phosphate through the interface between sediment and water is influenced markedly by the ORP condition of the sediment surface, namely with the rise and fall of the oxidized zone. Therefore, the DO prediction cannot be eliminated.

The DO in the surface water is changed with the decomposition or production of phytoplankton. It is also consumed with the decomposition of organic matter in sediment. If sufficient DO is supplied to sediment, the oxidized zone, i.e. ferric oxide, is formed by the reaction of DO in the lower layer of

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* TOD = theoretical oxygen demand.

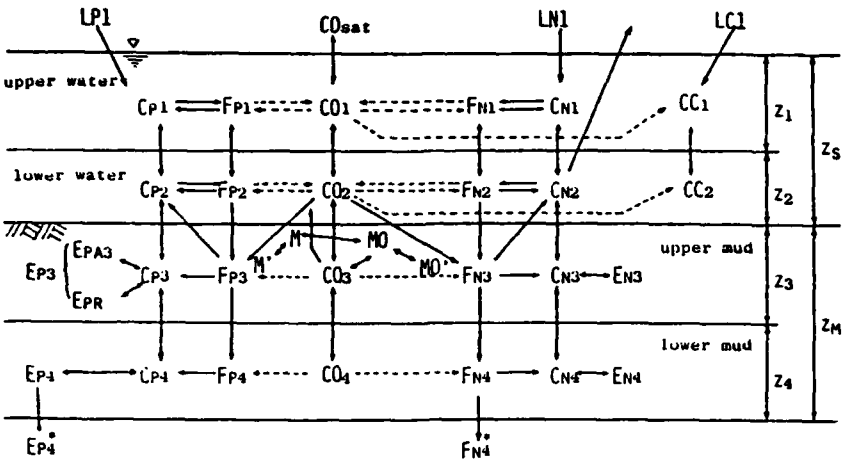


Figure 8. Outline of the P, N, and COD model

water and ferrous iron in the upper layer of sediment. It is assumed that the oxidized zone has a specific capacity to precipitate phosphate.

In contrast, if DO is deficient, sulfide, i.e. debt of DO, is formed and reduction of ferric oxide occurs in the upper layer of sediment.

Setting of Various
Factors and Coefficients

The values of various factors and coefficients are listed in Table 3. In adopting these values, both the experimental results described above and informative literatures were referenced. The sensitivity studies by the numerical model were achieved to evaluate those factors which are important but insufficiently investigated.

Coefficients Having Seasonal Variation

If possible, the coefficients were expressed as a function of temperature.

Temperature. Temperature data of the upper and lower water measured at the anchor site in Tokuyama Bay from 1977 to 1981 were used and fitted with sine and cosine curves (Figure 9). Temperature of sediment was assumed to be delayed 20 days from that of the lower water.

TABLE 2. FUNDAMENTAL EQUATIONS

	Equation	Equation No.
(P)	dCP1/dt=Dz(CP2-CP1)/(0.5Zs*Z1) -PRO1*FP1 +KD1*FP1 +LP1/Z1	(1)
	dCP2/dt=Dz(CP1-CP2)/(0.5Zs*Z2) +r*D(CP3-CP2)/(0.5Z3*Z2) +r*KDB(Z3/Z2)FP3 +KD2*FP2 -PRO2*FP2	(2)
	dCP3/dt=D(CP4-CP3)/(0.5Zm*Z3) -D(CP3-CP2)/(0.5Z3*Z3) +KD3*FP3 +(KE3/w)*(EPA3-(CP3/a)) +(KR3/w)*(EPR3-1.3*b*w*MO)	(3)
	dCP4/dt=D(CP3-CP4)/(0.5Zm*Z4) +KD4*FP4 +(KE3/w)*(EP4-(CP4/a)) +s*KH*FP4	(4)
	dFP1/dt=Rdz*Dz*(FP2-FP1)/(0.5Zs*Z1) +PRO1*FP1-KD1*FP1-Ws*FP1/Z1	(5)
	dFP2/dt=Rdz*Dz*(FP1-FP2)/(0.5Zs*Z2) +PRO2*FP2 -KD2*FP2 +Ws*FP1/Z2 - Ws*FP2/Z2	(6)
	dFP3/dt=Ws*FP2/(r*Z3) -Wm*FP3/Z3 -KD3*FP3 -KDB*FP3	(7)
	dFP4/dt=Wm*FP3/Z4 -KD4*FP4 -KH*FP4	(8)
	dFP4*/dt=(1-s)*KH*FP4	(9)
	dEPA3/dt= -KE3(EPA3-(CP3/a))	(10)
	dEPR3/dt= -KR3(EPR3-1.3*b*w*MO)	(11)
	dEP4/dt= -KE3(EP4-(CP4/a)) -KF*EP4	(12)
	dEP4*/dt=KF*EP4	(13)
	*1) PRO1=KP1(CP1+n*FP1)/(CPo+CP1+n*FP1) n=n'*CP1/(CP1+CPo')	(a)
	PRO1=KP1(CN1+n*FN1)/(CNo+CN1+n*FN1) n=n'*CN1/(CN1+CNo')	(b)
	The smaller value of the result from (a) and (b) is adopted	
	*2) CO2<0; KDB=0	(c)
	CO2>0; KDB=KDB'*CO2/(CO2+COo)	(d)
(DO)	dCO1/dt=Dz(CO2-CO1)/(0.5Zs*Z1) +Rop*PRO1*FP1 -Rop*KD1*FP1 -Roc*KCD1(CC1-CCB) +KA(COsat-CO1)/Z1	(14)
	dCO2/dt=Dz(CO1-CO2)/(0.5Zs*Z2) +Rop*PRO2*FP2 - Rop*KD2*FP2 -Roc*KDC2(CC2-CCB) + *Rdm*D(CO3-CO2)/(0.5Z3*Z2) -e*KMP*M*CO2 -r*Rop*KDB*Z3/Z2	(15)
	dCO3/dt=Rdm*D(CO4-CO3)/(0.5Zm*Z3) -r*Rdm*D(CO3-CO2)/(0.5Z3*Z3) -Rop*KD3*FP3 -KMD*MO*CO3	(16)
	dCO4/dt= -Rdm*D(CO4-CO3)/(0.5Zm*Z4) -Rop*KD4*FP4	(17)
	dMO/dt= e*KMP*M*CO2 +KMD*MO*CO3 +KOM(MO'-(MO/po))	(18)
	dM/dt= -e*KMP*M*CO2 -KMD*MO*CO3 -KAM(M'-(M/pa))	(19)
	dMO'/dt= -KOM(MO' -(MO/po))	(20)
	dM'dt= -KAM(M'-(M/pa))	(21)
	*3) CO2<0; KMP=0	(e)
	CO3>0; KMD=0	(f)
(N)	dCN1/dt=Dz(CN2-CN1)/(0.5Zs*Z1) -PRO1*FN1 +KD1*FN1 +LN1/Z1	(22)
	dCN2/dt=Dz(CN1-CN2)/(0.5Zs*Z2) +r*D(CN3-CN2)/(0.5Z3*Z2) +r*KDB*FN3(Z3/Z2) +KD2*FN2 -PRO2*FN2 -KN2*CN2	(23)
	dCN3/dt=D(CN4-CN3)/(0.5Zm*Z3) -D(CN3-CN2)/(0.5Z3*Z3) +KD3*FN3 +(KE3/w)(EN3-(CN3/a))	(24)
	dCN4/dt=D(CN3-CN4)/(0.5Zm*Z4) +KD4*FN4 +(KE3/w)(EN4-(CN4/a))	(25)
	dEN3/dt= -KE3(EN3-(CN3/a))	(26)
	dEN4/dt= -KE3(EN4-(CN4/a))	(27)
	FNi=Rnp*FPi (i=1,2,3,4)	(28)
	dFN4*/dt=KH*FN4	(29)
(COD)	dCC1/dt= -KCD1(CC1-CCB) +Dz(CC2-CC1)/(0.5Zs*Z1) +LC1/Z1	(30)
	dCC2/dt= -KCD2(CC2-CCB) -Dz(CC2-CC1)/(0.5Zs*Z2)	(31)
	TCOD1=CC1 +FP1(Rop/Roc)	(32)
	TCOD2=CC2 +FP2(Rop/Roc)	(33)

TABLE 3. COEFFICIENT'S AND VALUES OF THE P, N, COD MODEL FUNDAMENTAL EQUATIONS

Coefficient	Explanation	Value	Units
CP1	Inorg-P conc. in the upper water layer	0.01	µg/ml
CP2	Inorg-P conc. in the lower water layer	0.03	µg/ml
CP3	Inorg-P conc. in the upper sediment layer	0.2	µg/ml
CP4	Inorg-P conc. in the lower sediment layer	2.0	µg/ml
FP1	Org-P conc. in the upper water layer	0.03	µg/ml
FP2	Org-P conc. in the lower water layer	0.02	µg/ml
FP3	Org-P conc. in the upper sediment layer	10.0	µg/ml
FP4	Org-P conc. in the lower sediment layer	2.0	µg/ml
FP4*	Humic-P conc. in the lower sediment layer	10.0	µg/ml
EPA3	Adsorbed-P conc. in the upper sediment layer	0.8	µg/g dry
EPR3	Stored-P conc. in the upper sediment layer	9.2	µg/g dry
EPA4	Adsorbed-P conc. in the lower sediment layer	10.0	µg/g dry
EP4*	Residual-P conc. in the lower sediment layer	10.0	µg/g dry
CN1	Inorg-N conc. in the upper water layer	0.2	µg/ml
CN2	Inorg-N conc. in the lower water layer	0.5	µg/ml
CN3	Inorg-N conc. in the upper sediment layer	2.0	µg/ml
CN4	Inorg-N conc. in the lower sediment layer	5.0	µg/ml
FN1 4	Org-N conc. in each layer	Rnp*FP1 4	µg/ml
FN4*	Humic-N conc. in the lower sediment layer	50.0	µg/ml
FP4*	Humic-P conc. in the lower sediment layer	0.0	µg/ml
EN3	Adsorbed-N conc. in the upper sediment layer	8.0	µg/g dry
EN4	Adsorbed-N conc. in the lower sediment layer	20.0	µg/g dry
CC1	Alochtonus COD conc. in the upper	1.0	µg/ml
CC2	or the lower water layer	1.0	µg/ml
CCB	COD conc. of background	0.5	µg/ml
MO	Ferric oxide conc. in the upper sediment layer	75	µg/ml
MO'	(inactive)	25	µg/ml

(Continued)

(Sheet 1 of 3)

TABLE 3 (CONTINUED)

Coefficient	Explanation	Value	Units
M	Ferrous ion conc. in the upper sediment layer	75	µg/ml
M'	(inactive)	25	µg/ml
CO1	Dissolved oxygen conc. in the upper	9.0	µg/ml
CO2	or the lower water layer	8.0	µg/ml
CO3	Dissolved oxygen conc. in the upper	0.0	µg/ml
CO4	or the lower sediment layer	-10.0	µg/ml
KP1	Production rate coefficient of FP1	Fig. 11	day^{-1}
KP2	Production rate coefficient of FP2	0.3KP1	day^{-1}
KD1	Decomposition rate coefficient of FP1	Fig. 10	day^{-1}
KD2	Decomposition rate coefficient of FP2	Fig. 10	day^{-1}
KD3	Decomposition rate coefficient of FP3	Fig. 10	day^{-1}
KD4	Decomposition rate coefficient of FP4	0.1KD3	day^{-1}
KDB'	Decomposition rate coefficient by benthos	0.04	day^{-1}
KN	Denitrification rate coefficient	0.005	day^{-1}
KCD1	Decreasing rate coefficient of CC1	0.02	day^{-1}
KCD2	Decreasing rate coefficient of CC2	0.02	day^{-1}
KE3	Adsorption or desorption rate coeff.	$1*10^{-4}$	sec^{-1}
KR3	Precipitation or dissolving rate coeff.	$1*10^{-5}$	sec^{-1}
KMP	Production rate coefficient of MO	$3*10^{-7}$ml*	μg^{-1}/sec
KMD	Diminishing rate coefficient of MO	$3*10^{-8}$ml*	μg^{-1}/sec
KOM	Inactivation (activation) coeff. of MO	$2*10^{-3}$	day^{-1}
KAM	Inactivation (activation) coeff. of M	$2*10^{-3}$	day^{-1}
KH	Production rate coefficient of FN4*	$2*10^{-7}$	sec^{-1}
KF	Production rate coefficient of EP4*	$1.5*10^{-7}$	sec^{-1}
KA	Reaeration rate coefficient	Fig. 12	cm/sec
Dz	Vertical mixing coefficient	Fig. 12	cm^2/sec
D	Diffusion constant in the sediment layer	$0.5*10^{-5}$	cm^2/sec
Ws	Settling velocity of FP1 (FN1), FP2 (FN2)	15	cm/sec
Wm	Sedimentation rate of sediment layer	0.05	cm/day
COsat	Saturated DO concentration	Fig. 12	µg/ml
CNo	Michaelis constant of inorg-N for KP1,2	Rnp*CPo	µg/ml

(Continued)

TABLE 3 (CONCLUDED)

Coefficient	Explanation	Value	Units
CNo'	Michaelis constant of inorg-N for n	Rnp*CPo'	µg/ml
CPo	Michaelis constant of inorg-P for KP1,2	0.02	µg/ml
CPo'	Michaelis constant of inorg-P for n	$5*10^{-4}$	µg/ml
COo	Michaelis constant of oxygen for KDB	0.1	µg/ml
a	Adsorption equivalent constant	0.25	g/ml
w	Volume/weight ratio of interstitial water and dry sediment	1.6	ml/g
b	Activity of MO	0.3	
r	Porosity of sediment layer	0.78	
n'	Maximum availability of org-N,P	0.04	
s	Decomposing fraction of FP4*	0.8	
po	Inactivation equibrium constant of MO	4.5	
pa	Inactivation equibrium constant of M	4.5	
Rdm	Conversion factor DM for oxygen/inorg-P	2	
Rdz	Conversion factor Dz for org-P/inorg-P	0.5	
Rop	Conversion factor O/P	143	
Ron	Conversion factor O/N	19.7	
Rnp	Conversion factor N/P	Ron/Rop	
Roc	Conversion factor O/COD	3	
LP1	Inflowing load of inorg-P	0.03	µg*cm/day
LN1	Inflowing load of inorg-N	0.3	µg*cm/day
LC1	Inflowing load of COD	5.0	µg*cm/day
Z1	Width of the upper water layer	400	cm
Z2	Width of the lower water layer	100	cm
Z3	Width of the upper sediment layer	1.0	cm
Z4	Width of the lower sediment layer	4.0	cm

Decomposition rate coefficients. Coefficients KD1, KD2, and KD3 were determined by the following equations (Figure 10):

$$KD1,2 = 0.47 \text{ Exp } [0.04(T1,2 - 18)] \quad (2)$$

$$KD3 = 0.025 \text{ Exp } [0.052(T3 - 15)] \quad (3)$$

where

KD1,2 = decomposition rate coefficient in the upper or lower water, day^{-1}

KD3 = decomposition rate coefficient in the upper sediment layer, day^{-1}

T1,2,T3 = temperature of the upper water, the lower water, and the upper sediment, respectively, °C

Production rate coefficient KP1 (day^{-1}). Coefficient KP1 (Figure 11) was determined by the following equation:

$$KP1 = 3.6 \, [S/(S + 120)] \text{ Exp } [0.073(T1 - 18)] \quad (4)$$

where S = relative light intensity considering Oswald's solar energy and weather condition (Figure 9).

Vertical mixing coefficient (Dz) and diffusion constant in sediment (D). With reference to several reports, Dz and D were expressed by the following equations (Figure 12):

$$Dz = 2.0 - 3.3 \left(|T1 - T2|/6 \right) \quad (5)$$

$$D = 5 * 10 \, (1 + 0.04 \, T3) \quad (6)$$

where

Dz = cm^2/sec

T1,T2 = °C

D = cm^2/sec

T3 = °C

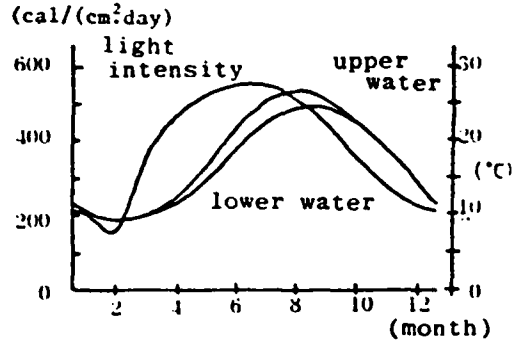


Figure 9. Seasonal variation of water temperature at the anchor site in Tokuyama Bay

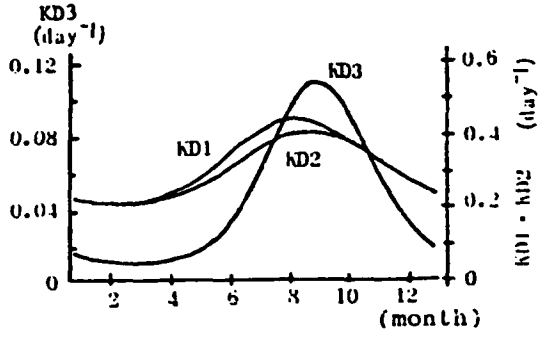


Figure 10. Seasonal variation of KD1, KD2, and KD3 used in the calculation

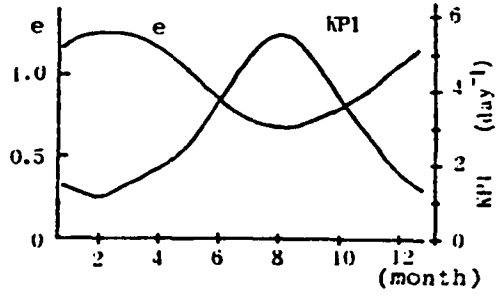


Figure 11. Seasonal variation of KP1 and complementary factor for KMP•e

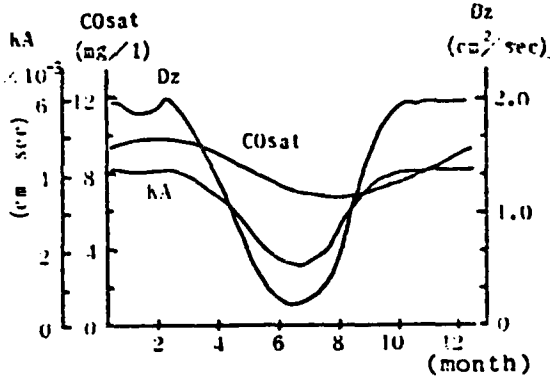


Figure 12. Seasonal variation of KA, COsat, and Dz used in the calculation

Method of Calculation

The fundamental formulas were reformed to differential equations and solved by the explicit and rear method with dt = 30 min. The calculation was evaluated up to 5 years. The stable results were then printed out concerning the seasonal variation of DO, RN, RP, and other parameters.

Results and Discussion

Basic Results of Calculation

Figure 13 shows representative results obtained. The simulated seasonal variation of RN, RP, and DO concentration compares well with the data of RN and RP shown in Figure 3 and DO in Tokuyama Bay shown in Figure 8. The difference between the variation pattern of RN and RP is especially well described. The rise and fall of the oxidized zone are also of reasonable variation, in view of our experience, although the data have not been proposed.

Sensitivity Analysis on Several Important Parameters

Effect of Rdm. Rdm is a parameter introduced to express the probable effect of benthos, stimulating the flux of DO more than that of nutrients. For the larger value of Rdm, the decrease of DO in late summer to autumn becomes more severe. The rise and fall of the oxidized zone becomes more striking as well. Rdm exerts little effect on the flux of phosphate (RP) or RN (Figure 14).

Effect of KMD and KMP. The diminishing rate coefficient of the oxidized zone (KMD) indicates an intense effect on RP, RN, DO, and on the variation of the oxidized zone. For a small value of KMD, RP remains at a relatively low level in summer due to the insufficient decrease of the oxidized zone. A large value of KMD also lowers RP because of a delayed formation of sulfide in early summer (Figure 15). Though the KMP value does not have as intense an effect as KMD, for the smaller value the peak of RP delays toward autumn.

Effect of β. Activity of the oxidized zone (i.e., precipitating capacity β) intensely affects RP, RN, DO, etc. Especially for the larger value of β, RP is seriously suppressed and accordingly delayed (Figure 16).

Effect of KA and Dz. Although both values of KA (reaeration rate coeff.) and Dz (vertical mixing rate coeff.) were varied to half or 1.5 times the base value, notable effects on RN and RP were not induced. For the smaller values, restoration of the oxidized zone causes a slight delay and the peak of RP shifts toward autumn (Figure 17).

It should be noted that in all cases RN changes in almost the same manner as RP, in spite of the fact that release of nitrogen from the sediment has no relation to the oxidized zone.

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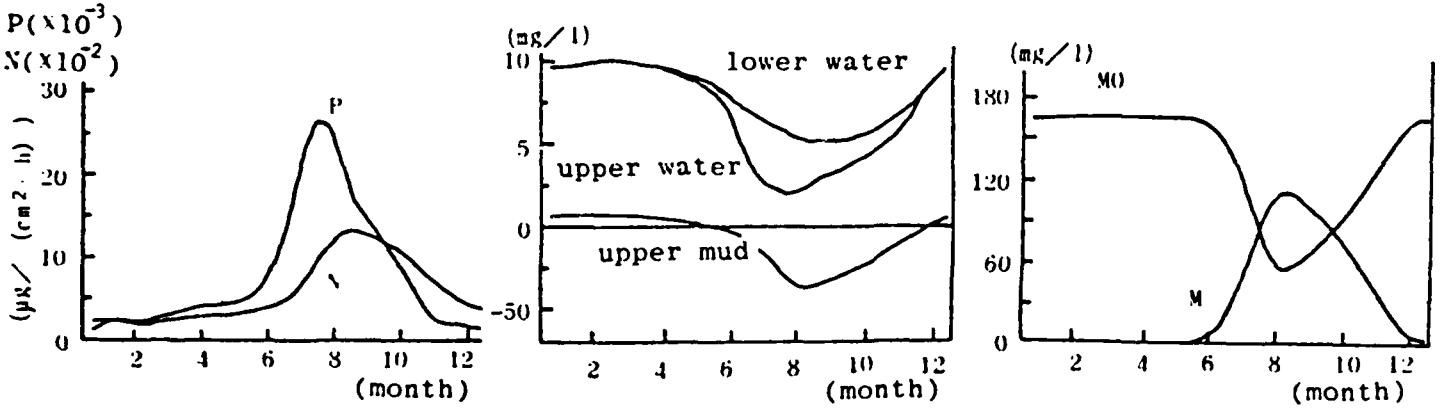
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It should be noted that in all cases RN changes in almost the same manner as RP, in spite of the fact that release of nitrogen from the sediment has no relation to the oxidized zone.



(a) Seasonal variation of N, P discharging rate (calculated)

(b) Seasonal variation of DO concentration (calculated)

(c) Seasonal variation of ferric oxide and ferrous iron (calculated)

Figure 13. Representative results of calculation by P, N, COD model

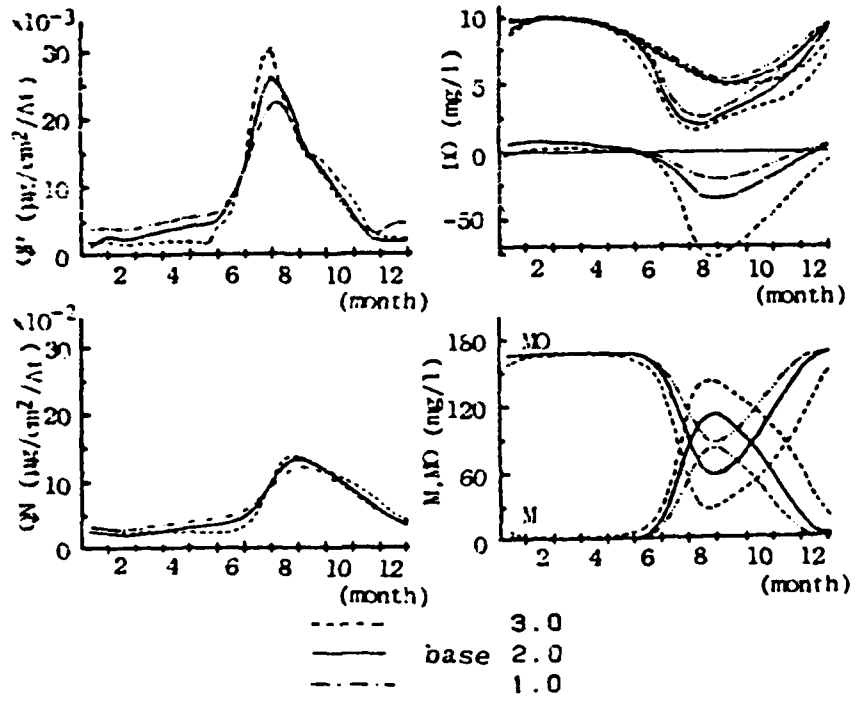


Figure 14. Effect of Rdm on the results of calculation

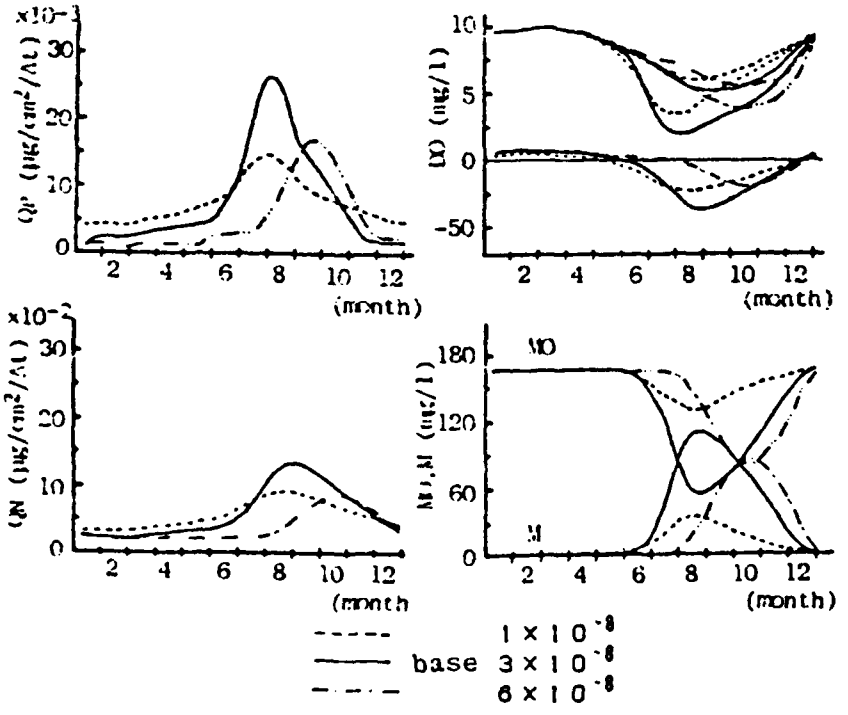


Figure 15. Effect of KMD

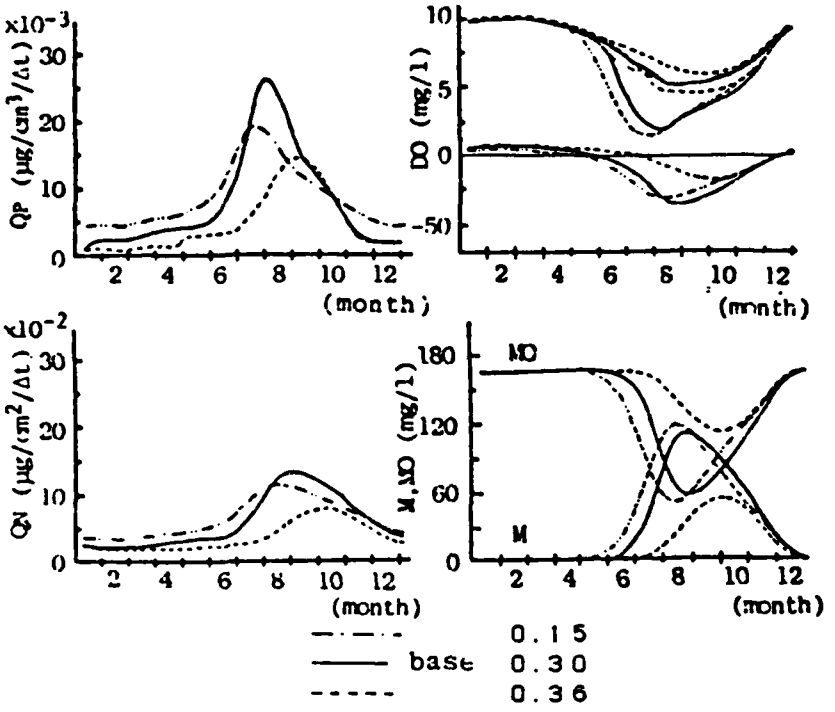


Figure 16. Effect of β

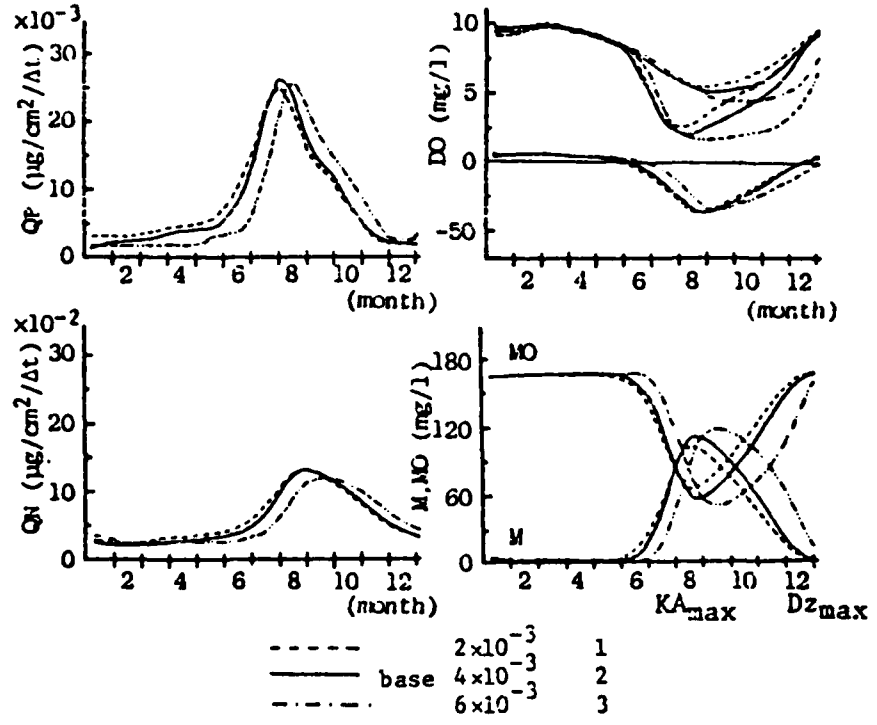


Figure 17. Effect of KA and Dz

Discussion

Until recently, the characteristic relationship between the interstitial phosphate and the oxidized zone in surface sediment has not been well simulated. It appears that the main cause of the instability in calculation is that the oxidized zone has been assumed to be formed by consuming DO from the upper layer of sediment. Therefore, it is an important feature of this model that the oxidized zone is assumed to be formed by consuming DO in the lower layer of water. This assumption might not be unreasonable considering that formation of the oxidized zone is highly stimulated by curling up to the overlying water.

It should be noted that composition of the organic matter is assumed to remain constant in all stages of decomposition and that the decomposition rate of organic matter is assumed to remain at the same level either in the aerobic or anaerobic condition.

Many parameters exist in this model, including uncertain ones. Therefore, more extensive studies of experimental or numerical analysis have to be conducted. For example, the upper limit of the oxidized zone, which is 200 mg O_2/ℓ, seems to be too small compared with the concentration of ferric oxide that actually exists. The parameters KMD, b, or even relatively popular KA and Dz and others admit more discussion.

Furthermore, the model might be expected to be simpler when applied to practical assessments of water and sediment quality.

CONCLUSIONS

The fundamental components of this model are P, N, DO, and COD. Sulfide is regarded as a debt of DO for simplification, and oxidized zone (ferric oxide) as a source of DO. The oxidixed zone, which has a specific capacity of precipitating interstitial phosphate, is assumed to be formed by the reaction of DO in the lower layer of water and iron in the upper layer of sediment.

Using this model, the characteristic seasonal variation of phosphate release from sediment to water, and the seasonal changes of DO, COD, etc., were well simulated.

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Takashi Hirajou, Mitsunori Amatani, Kikuo Miyake, Kazunori Matsumura, and other members of our laboratory.

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EVALUATION OF THE FACTORS GOVERNING THE MOBILITY OF DREDGED MATERIAL PLACED AT OPEN WATER DISPOSAL SITES

W. F. Bohlen
University of Connecticut
Marine Sciences Institute
Avery Point
Groton, Connecticut 06340

ABSTRACT

Long-term management of open water disposal areas receiving dredged material requires an understanding of the physical and biological factors affecting local sediment transport. Within most continental shelf, nearshore, and estuarine areas these factors display a high degree of temporal variability with the majority of significant sediment displacements occurring during aperiodic storm events. The short-lived nature of these events establishes particular sampling constraints and favors the use of in-situ systems to monitor transport-related processes. To satisfy these constraints a bottom-mounted instrumentation array has been developed and deployed. This array contains optical transmissometers to monitor suspended material concentrations, temperature-conductivity probes, an electromagnetic current meter, a wave gage, a lapsed-time camera and strobe system, and a sequenced mechanical water sampler. The entire system is controlled and the resultant data recorded by a digital data logger. The unit is capable of unattended operation for periods in excess of 30 days at sampling rates of four scans of the entire array each hour. Reviews of selected data obtained at several locations within Long Island Sound reveal a range of hign and low frequency variations in sediment movements. Low frequency variations appear to be primarily the result of biologically mediated factors while the higher frequency components occur in response to persistent tidal forcing and aperiodic storm events. The latter displacements appear to be nonlinear in character and not simply related to storm intensity. In addition to these temporal characteristics, array observations indicate evident spatial variability associated primarily with sediment characteristics and the presence or absence of near-bottom fluff layers. These layers of organic-mineral aggregates appear to modify the structure of the flow field affecting the sediment-water interface and may serve to limit erosion and transport. Determination of the dynamics associated with these layers required consideration of the role

of the overlying water column as a source of materials to the near-bottom areas. A system sufficient to provide these observations is outlined.

INTRODUCTION

The effective, long-term management of open water disposal areas intended to receive dredged material requires an understanding of the variety of factors affecting local sediment transport. Such information is of particular concern when the dredged sediments are associated with moderate to high concentrations of toxic contaminants. Present practice within the United States requires effective containment of such materials by either placement within areas characterized by low natural dispersion or through the use of special handling procedures such as "capping" wherein the contaminated materials are isolated from the adjacent water column by a layer of clean sediments (3). Preproject specification of disposal site suitability and postproject monitoring of natural containment and/or the effectiveness of capping procedures are complicated by the generally high degree of temporal variability characterizing the sediment transport system active within most continental shelf, nearshore, and estuarine areas. Within this coastal zone studies have shown that the majority of significant sediment displacements occur during aperiodic, high energy storm events (e.g., Bohlen (2)). Evaluation of the effects of such events on sediment stability within a given disposal area is difficult to realize using standard shipboard sampling techniques due to the limited accuracy of efforts to predict the times of storm occurrence and the inherent difficulty of ship operation during storms. These factors, in combination with the generally short-lived nature of storm-induced sediment resuspension, establish some particular sampling constraints and clearly favor the use of in-situ systems sufficient to provide reliable, quantitative measurements of sediment transport and associated governing factors over reasonably long deployment periods.

In early 1980 a bottom-mounted instrumentation array designed specifically to detail the effects of aperiodic storm events on the sediments placed within active dredged material disposal areas was constructed and deployed within Long Island Sound, a major commercial and recreational waterway along the northeastern coast of the United States (Figure 1). Over the past 4 years this array has been deployed at several locations within the Sound as part of a Corps of Engineers sponsored monitoring program (DAMOS - Disposal Area Monitoring System). The data obtained during these deployments serve to document the response to the near-bottom suspended material field to aperiodic storm events as well as the lower frequency response to the more persistent tidal forcing and the variety of seasonal factors including streamflows and biological activity. In addition, the observations provide initial indication of the probable range of spatial variability characterizing the local sediment transport system and complement efforts to design optimum sampling schemes sufficient to permit calibration and verification of predictive analytical models. The combination finds application within several areas of the overall management of the regional dredged material disposal sites.

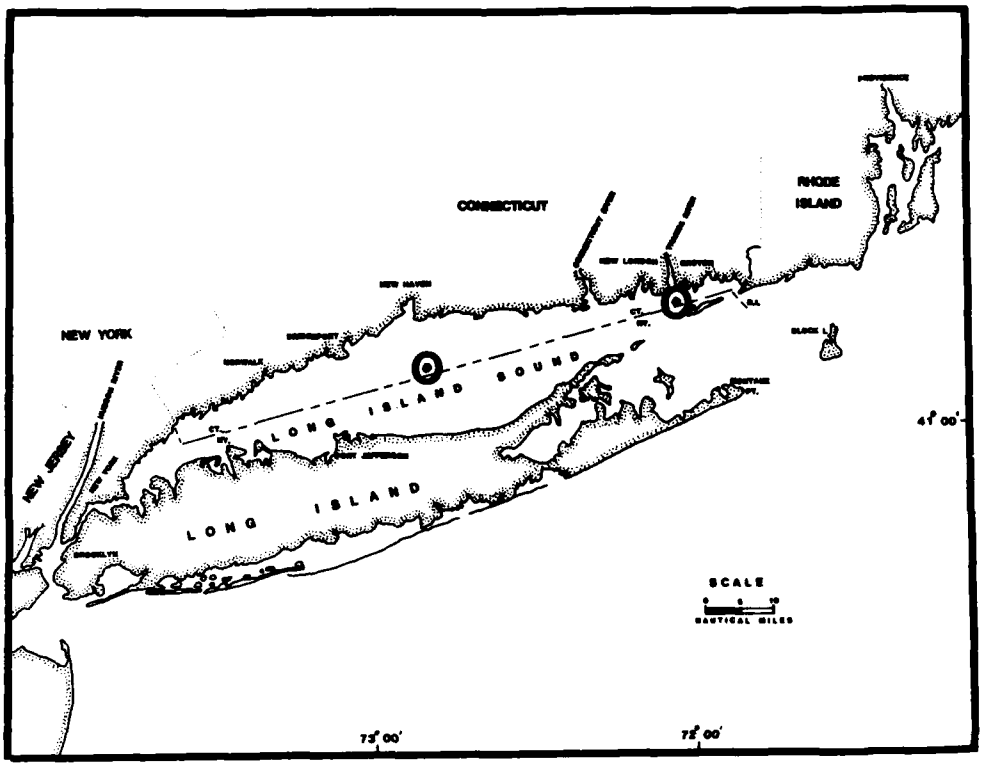


Figure 1. Long Island Sound. Circled areas designate study sites

INSTRUMENTATION ARRAY

Review of the variety of monthly data obtained via shipboard sampling over a period of more than 4 years in combination with experience associated with the employment and subsequent servicing of a moored instrumentation array centered around a large surface buoy (1) indicates that detailing of the high frequency characteristics of the coastal suspended material field is best realized using a bottom-mounted instrumentation array. This array would be self-contained with integral power and data recording sufficient to permit reasonably long-term in-situ observation of near-bottom suspended material concentrations and associated hydrographic factors including water temperature, salinity, and current speed and direction. Physically the array was to be free-standing and configured so as to minimize flow disturbance. To avoid vandalism and potential upset of the array, surface buoys were not to be included and recovery routines were developed around an acoustic command and location system. The array constructed to satisfy these criteria is shown in Figure 2.

As configured, the array consists of a digital control network and four basic subsystems in combination with supplementary temperature and conductivity sensors (Figure 3). On occasion these components are supplemented by a self-recording package intended to monitor variations in the near-bottom pressure field induced by tidal variations and the surface wave field. All instruments are mounted rigidly to the aluminum frame. As specified, the frame outline is designed to minimize flow disturbance while providing reasonable



Figure 2. Bottom-mounted instrumentation array

stability and ease of handling from a 20-m research vessel. The circular base is approximately 2.0 m in diameter and is semicircular in cross section. The base is half-filled with lead. The remaining area is filled with concrete to prevent lead contamination of the local waters or nearby sediments and to provide a rugged footing and some additional ballast for the array. This combination has proved to be extremely rugged and shows no signs of deterioration despite 4 years of extensive use.

In the vertical the array frame is approximately 4 m in height. Three aluminum pipes (~6 cm in diam.) forming the outer legs are bolted to the periphery of the base and extended upwards to be joined by a circular horizontal plate. This plate provides a central strength member and serves to support a section of heavy walled aluminum pipe (~10 cm in diam.) extending downward approximately 0.5 m and a support tie-rod configuration extending upwards 1.0 m. At their upper limit the tierods are joined by a second horizontal aluminum plate. A stainless bail is bolted to the upper side of this plate to provide a secure lifting point for the array. When completely equipped, housing all primary systems, the array weighs approximately 230 kg.

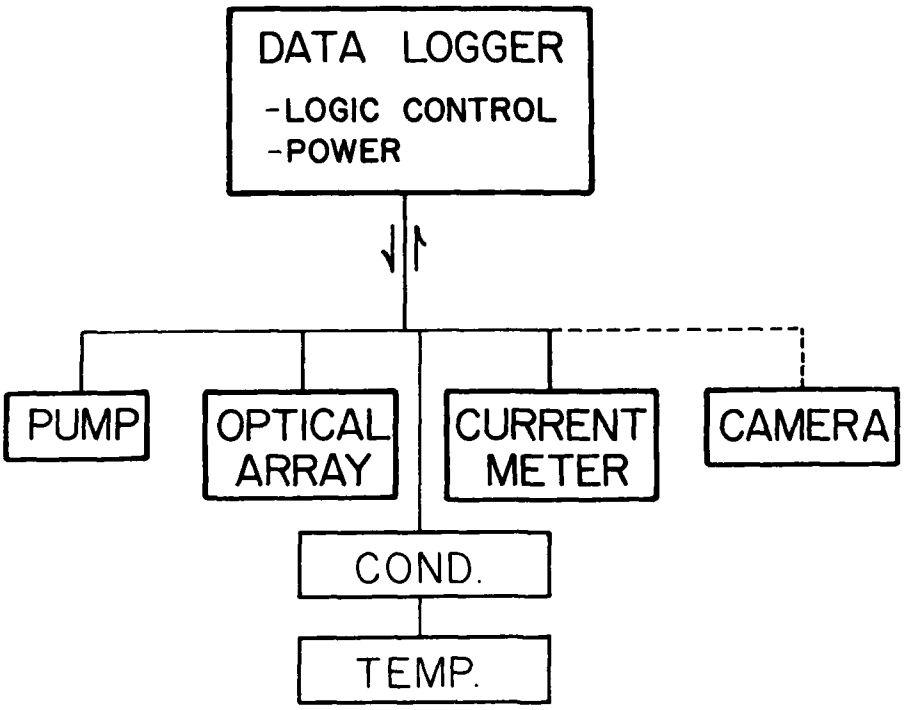


Figure 3. Primary and secondary subsystems; bottom-mounted instrumentation array

The primary control of the array is provided by a digital data logger (Sea Data Model 651-8). This unit, located near the top of the array (Figure 2), also supplies power to select instruments and serves to record the output signals from the current meter, optical array, and the temperature and salinity sensors. These data are stored on magnetic tape cassettes with capacity sufficient to permit sampling of all instruments four times each hour for 36 days. The format of the stored data is system specific and requires conversion prior to analysis. Conversion is accomplished using a Sea Data Model 12A reader in combination with a Digi-Data magnetic tape drive. The resultant nine-track tape contains data in an IBM compatible format and is simply read using standard programming procedures.

The optical array is designed to provide direct samples of suspended material concentrations at at fixed point above the sediment-water interface. The array consists of two red-light transmissometers each mounted rigidly to adjoining outer legs of the array (Figure 4). Designed to be relatively insensitive to dissolved materials (particularly color-rich organics) and water matrix variations, each of these sensors provides an analog output proportional to the concentration of sediments suspended over a fixed 10-cm path. Path length was specified following an evaluation of instrument sensitivity and examination of ambient suspended material concentrations and the characteristic variations expected at several locations within Long Island Sound. Output characteristics were defined following individual laboratory calibrations of each instrument over a range of suspended material concentrations prepared using natural sediments found in the vicinity of each deployment



Figure 4. Optical sensor-nephelometer; bottom-mounted instrumentation array

site. This calibration procedure is carried out prior to each instrument deployment and repeated immediately following recovery of the instrument. The results of this procedure, intended to permit correction of the initial calibration for instrument drift and variations induced by biofouling of the optical windows during the deployment, yield an algorithm detailing the relationship between output voltage and the equivalent suspended material concentrations in milligrams per liter. For the majority of the instruments used on the array this relationship is only slightly nonlinear and can be adequately described by a fourth order polynominal.

The current meter system consists of a two-axis electromagnetic current meter (Marsh-McBirney Model 585 OEM) aligned so that the sensors are confined to the horizontal plane. This unit, mounted at the lower end of the section of perforated heavy-walled piping (Figure 2), is positioned to sense velocities at a point approximately 1.0 m above the sediment-water interface. Flow direction is calculated vectorially using the output from each axis of the EM sensor as referenced to the orientation of the array. The latter orientation is referenced to magnetic north and is provided by diver observations obtained during the immediate postdeployment check-out and reconfirmed just prior to array recovery. For both speed and direction calculations output voltage is converted to equivalent flow using a manufacturer-supplied algorithm.

The camera system mounted near the top of the legs, in the vicinity of the central strength member (Figure 2), consists of a modified super-8mm movie camera and a strobed lighting unit (Vivatar). Both components are mounted

separately in watertight polyvinyl chloride (PVC) housings. The front windows for each unit are clear Plexiglas with no optical correction. Shutter speed and lens aperture are adjusted to provide a sufficient depth of field to permit simultaneous viewing of selected instruments on the array and the sediment-water interface. This combination is of particular value within efforts to resolve the cause or causes of anomalous peaks within the output of the optical sensors and in addition provides direct visual evidence of an event's ability to erode or perturb the local bottom. This latter observation assists in the resolution of the extent to which the suspended materials passing the optical sensor represent locally suspended materials or the product of far-field erosion being advected past the array. These questions are often of primary concern within efforts to manage a given dredged material disposal area.

The final subsystem within the primary array consists of a drawn water sampler intended to provide direct samples of the suspended material field to supplement the optical data and to permit instrument calibration and possible determination of the composition of the suspended load. The system consists of an array of 12 syringe-like ports mounted concentrically around the central support column just above the EM current meter (Figures 2 and 5). Each port is spring-loaded and armed by means of a burn-wire strap. Sample sequencing is controlled by a counting network sensing pulses from the central data logger. After receiving a predetermined number of pulses the circuit causes a current to pass through the burn-wire rupturing the wire and activating the selected sample port. The resultant sample can be either filtered or unfiltered depending on the port entry configuration.

Supplementing the primary array systems are two subsystems designed to provide concurrent measurements of water temperature and salinity. Temperature is measured at two points on the vertical using protected thermistors housed in watertight PVC pressure vessels. One unit is mounted near the upper end of the stainless steel tierods restraining the data logger while the other is attached to one of the vertical legs just above the annular base. Each unit was individually calibrated against laboratory standards prior to deployment.

Salinity is monitored using a single conductivity sensor mounted within the perforated segment of the vertical heavy-walled aluminum piping just above the attachment point for the mechanical water sampler. The sensor consists of a three electrode flow-through cell and associated electronics (Sea-Bird Model 4-01). This combination provides a frequency output proportional to conductivity. Conversion is accomplished using a counting network located within the data logger. These counts are subsequently converted to equivalent conductivity and then salinity using manufacturer-supplied algorithms.

On several occasions special purpose deployments required measurement of surface wave characteristics coincident with the primary array observations. To provide these measurements a self-contained digital wave and tide recorder (Sea Data Model 635-11) was added to the array. This unit uses a quartz sensor to monitor the near-bottom pressure field providing an inherent resolution of approximately 0.05 cm in water depths of 20 m. The system is entirely self-contained housing all necessary power and data storage and was rigidly mounted in a vertical position adjacent to the array data logger. For all measurements the unit functioned in a stand-alone mode. No attempt was made

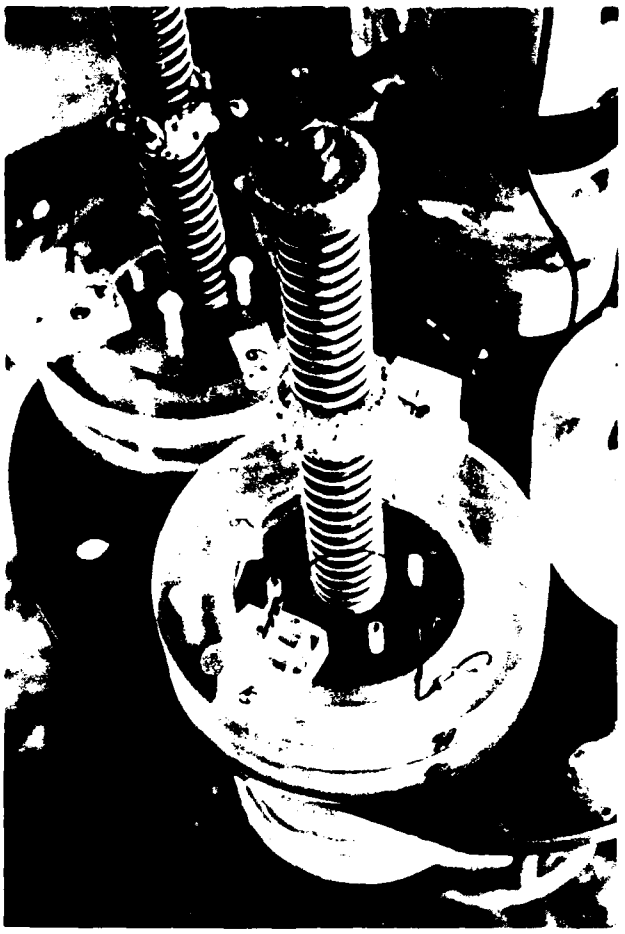


Figure 5. Mechanical water sampler

to synchronize the internal clocks contained in the array data logger and the wave sensor although both units were initialized at approximately the same time. This timing arrangement appears satisfactory for the majority of the observations required of the array.

SELECTED FIELD OBSERVATIONS

In January 1980 the instrumentation array was deployed at a location along the western limit of the dredged material disposal area sited just to the south of New London, Connecticut (Figure 1). The array was retained at this location for a period of approximately 24 months. Following refurbishment, after the recovery, the array was redeployed in April 1982 at a site along the northeastern limit of the Central Long Island Sound disposal area located approximately 10 km to the south of the harbor of New Haven, Connecticut (Figure 1). The initial observations at the central Sound site were intended to complement efforts to define a study area to be used within the Field Verification Program (FVP), an interdisciplinary research program sponsored jointly by the Corps of Engineers (CE) and the Environmental Protection Agency (EPA). At the completion of this site selection phase a long-term monitoring program was initiated in August 1982. This program, designed primarily to define the characteristics of the near-bottom suspended material

field so as to permit evaluations, both field and laboratory, of the range of potential effects on the local benthic biological community, is still in progress. A brief review of some selected segments from these data sets provides clear indication of the degree of temporal variability characterizing the suspended material field in the central and eastern Sound as well as information concerning the response of the field to aperiodic storm events and some indication of site-specific differences in the factors dominating material concentrations at each location.

Eastern Long Island Sound

For the initial deployment series the instrumentation array was placed along the western limit of the New London dredged material disposal area in water depths of approximately 20 m. Circulation in this area is dominated by the M2 tidal component with near-surface currents displaying a peak value of approximately 75 cm/sec. Freshwater discharge from local streams is small and generally exerts a minimal influence on the regional water density structure and associated circulation. Surface waves can affect local circulation and mixing with influence varying primarily in response to wind speed and direction. The wind field characteristic of the eastern Sound varies seasonally with winter months dominated by northwesterly winds and summer months by southwesterlies. Occasional high energy storm events can occur throughout the year although maximum frequency of occurrence is usually confined to the late fall-winter-early spring period.

Since approximately 1975 the New London disposal area has received more than $5 \times 10^6 m^3$ of dredged material. The majority of these materials were removed from the Thames River and consist of fine-grained cohesive sediments having mean grain sizes of approximately 62μ. Materials are generally removed using a clam-shell bucket dredge and placed via hopper barge.

Data developed during the initial deployment period (January 1980) show average suspended material concentrations at the study site ranging from 1 to 3 mg/ℓ. This progressive increase in average concentration over the 10-day period appears to be primarily the result of material accumulations on the windows of the optical sensors rather than any persistent increase in actual ambient levels. Concentrations vary about this mean by approximately ±1 mg/ℓ at rates essentially coincident with the twice daily tidal period (Figure 6). These variations display only minor sensitivity to the monthly spring-neap cycle.

In addition to the low frequency variability associated with local tidal cycling, the suspended material record displays a number of high frequency variations during which indicated concentrations significantly exceeded the average background levels. Such perturbations occurred on Day 2 (~26 hr), Day 3 (~76 hr), Day 7 (~198 hr), Day 10 (~240 hr), Day 11 (~260 hr), and Day 12-13 (~300 hr). With the exception of the event which began on Day 7, these perturbations were short lived and persisted for less than 5 hr. Review of the primary factors governing suspended material concentrations in coastal waters including streamflows, winds, and tidal currents indicates that, for the period of observation, only wind-induced factors and tidal currents displayed variability sufficient to affect the observed perturbations. Streamflows remained essentially constant throughout the period and were

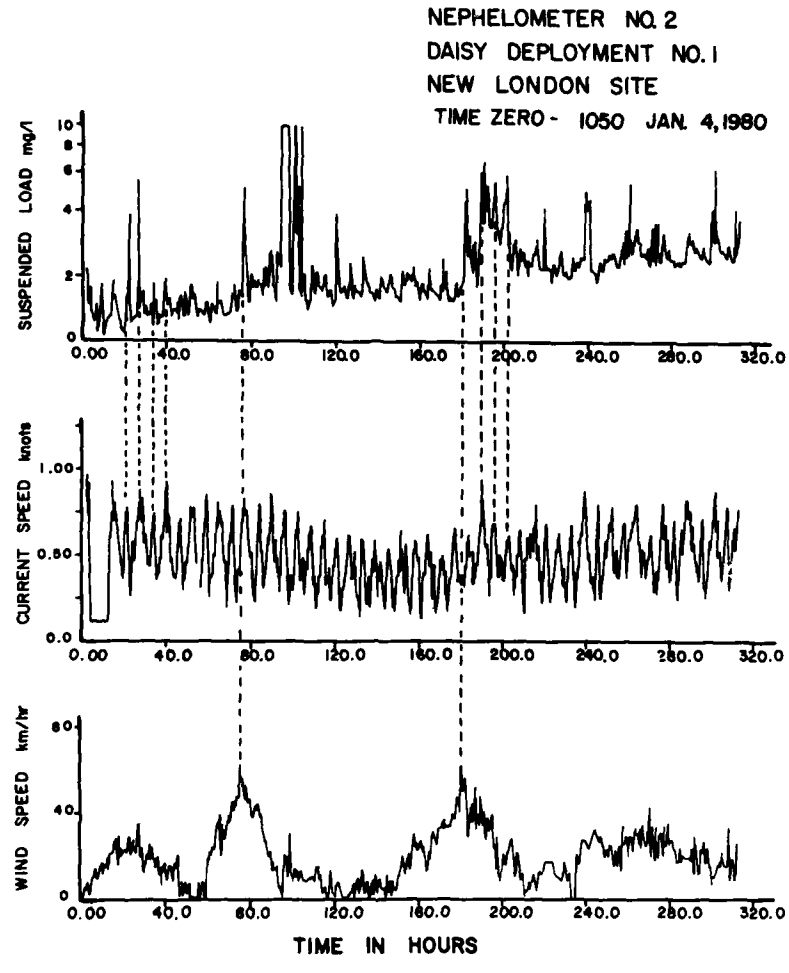


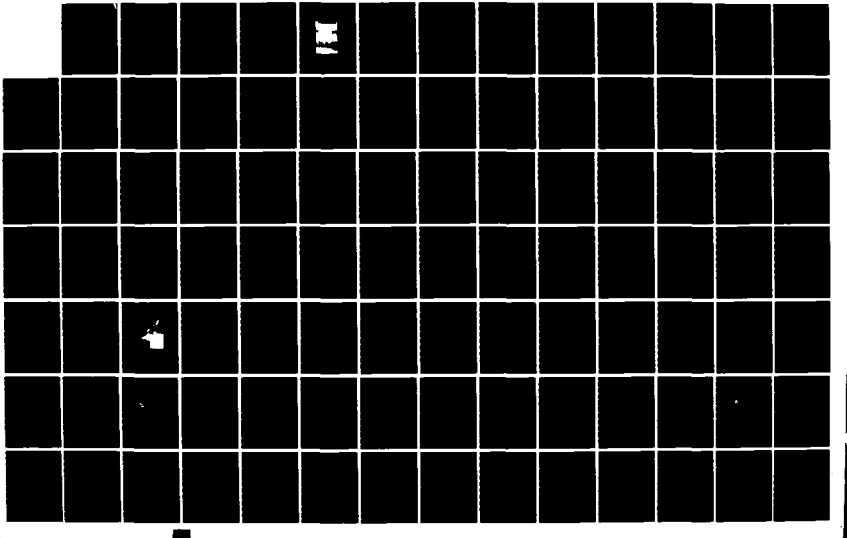
Figure 6. Output data from bottom-mounted instrumentation array and associated wind speeds: Eastern Long Island Sound deployment, January 1980

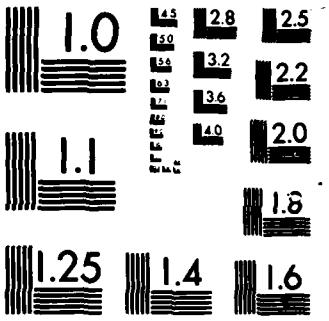
characterized by below average discharge. The wind and tidal current speed records for the deployment period (Figure 6) contain several prominent aperiodic variations, several of which are coincident with the observed perturbations in suspended material concentrations. The majority of these variations are contained within the wind record. Only during the Day 7 (~198 hr) event was there a significant variation in the tidal current system. Wind speed peaks are clearly associated with the Day 3 and Day 7 events with secondary smaller peaks evident during the Day 2 and Day 11 events. The remaining perturbations within the suspended material record show little relation to tidal current or wind speed and appear to be primarily the result of biogenic detritus, including seaweed and swimming and planktonic organisms passing through the sampling volume of the optical sensors.

Each resuspension event displays some significant similarities and differences. Events induced by relatively low wind speeds (<30 km/hr) tend

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to appear as a well-defined peak essentially coincident in time with the onset of maximum tidal speeds. Increasing wind speed above 45 km/hr induces only minor increases in the magnitude of resuspension but favors a significantly longer duration (e.g. compare Day 2 to Day 7). Even during the longer duration events however the pattern remains "spikey" with each peak essentially coincident with the tidal speed maxima. This continuing coincidence suggests that the observed resuspension is not simply the result of wind-induced factors but is rather due to the combined effects of winds and tidal currents. The observations suggest that the sediment-water interface in the study area is in essential equilibrium with the prevailing tidal stream and experiences significant disturbance only during times in which available transport energy is significantly increased by the added influence of wind-stress-induced factors.

In addition to the onset of significant perturbations, the suspended material observations also show significant differences in the response of the sediment transport system to given wind stress events. This difference is particularly evident in comparisons of the Day 3 with the Day 7 events and appears to be primarily the result of differences in the duration of the associated wind events. The remaining characteristics are essentially similar with both events displaying peak speeds of approximately 60 km/hr and a predominantly south to southwest directionality. The duration for the above average wind speeds for the Day 3 event was approximately 35 hr. For the Day 7 event high velocity winds persisted for approximately 55 hr. In addition, winds during this latter period displayed more intense high frequency variability sufficient to produce a generally more energetic system.

Differences in wind duration and energy levels will affect primarily the stress-induced wave field with some secondary influence within the associated mass transport system. At equivalent peak speeds, longer duration favors generation of larger amplitude, longer period surface waves. The observed response suggests that the transport field in the eastern Sound is particularly sensitive to this generalized behavior. This sensitivity is most probably the result of the relatively sheltered nature of the Sound resulting from its long, narrow configuration. Such a form serves to significantly reduce the effective distance or "fetch" over which wave generation occurs making the surface wave field particularly sensitive to wind direction and duration. As a result, slight variations in dominant wind direction and event duration can produce significant variations in surface wave characteristics and associated material resuspension. This variety of observations complements determination of the cause of the apparent nonlinear response of the suspended material field in the eastern Sound to aperiodic forcing noted in previous investigations (2) and assists in the selection of the proper modeling form to be used in the analytical prediction of system response to the passage of high energy storm events.

Central Long Island Sound

Within the central Sound the instrumentation array has been deployed near the northeastern corner of the New Haven dredged material disposal area (Figure 1) since August 1982. This location represents the primary study site within the open water aquatic phase of the FVP. Bottom contours in the vicinity of the deployment site are essentially planar and display average water depths of approximately 20 m. Circulation in the region is again dominated by

the M2 tidal component with local streamflows representing a generally minor influence. The resultant current field is less energetic than that found within the eastern Sound and displays maximum speeds that are typically less than 50 cm/sec. Since the initiation of FVP this site has received approximately 70,000 m^3 of fine-grained sediments dredged from within Black Rock Harbor, a small embayment adjacent to Bridgeport, Connecticut (Figure 1). These materials, characterized by a generally moderate to high water content and a low density of infaunal organisms, were removed using clam-shell bucket techniques and placed at the disposal site by a hopper barge. Precision placement techniques resulted in a coherent conical mound centered on the disposal buoy at the completion of the dredging phase.

Observations provided by the instrumentation array indicate that the suspended material concentrations in the areas adjacent to the FVP site are typically higher than those observed in the eastern Sound with values averaging approximately 6-8 mg/ℓ. In addition to higher average values, the optical data also indicate a higher degree of temporal variability in the central Sound suspended material field. As shown in Figure 7, a representative set of data obtained just prior to the initiation of dredged material disposal, this variability is reasonably persistent with peak concentration values exceeding the average by nearly a factor of two. This pattern persisted after the completion of the disposal operation with only minor perturbations observed during the immediate postdisposal period.

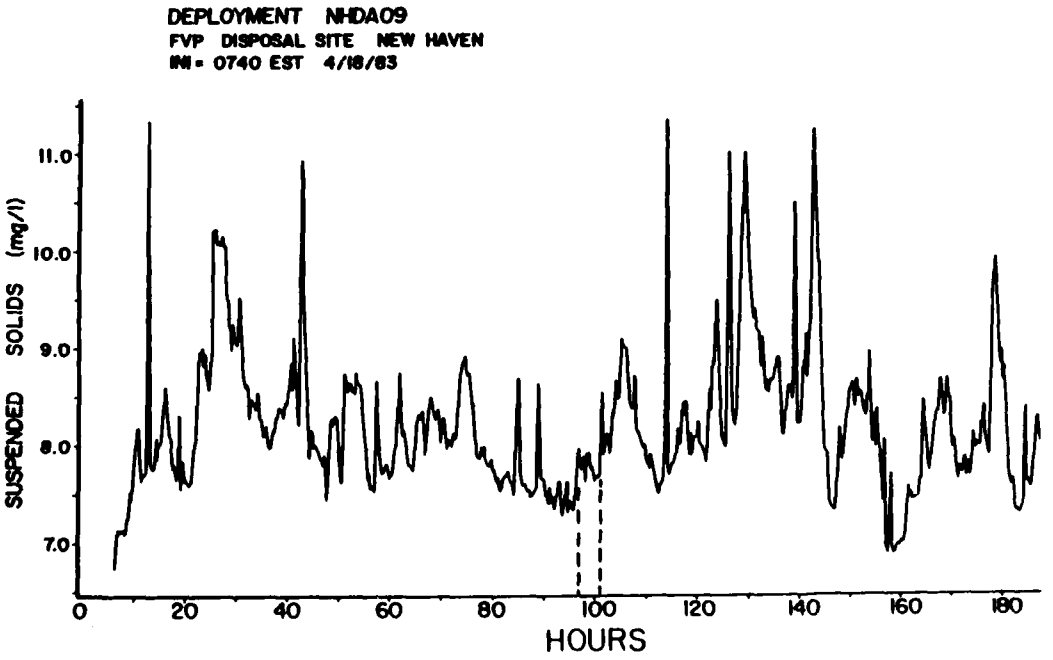


Figure 7. Suspended material concentrations, central Long Island Sound, 1983

The observed variations in suspended material concentrations display a clear correlation with the local tidal phase. In contrast to the eastern Sound distributions, peak concentrations are typically found during periods of

low velocity with concentrations characteristically decreasing as speed increases. This response suggests that the near-bottom suspended material field in the central Sound is dominated by an assemblage of fine-grained materials with settling characteristics that favor the formation of a "fluff layer" that alternately disperses and consolidates over the course of the tidal cycle. These observations are consistent with those discussed in previous studies that have noted the presence of high concentrations of organic-mineral aggregates within the near-bottom waters of the central Sound (4).

The source of the materials forming the fluff layer is difficult to accurately specify. Observations during the spring of 1983 intended to determine the importance of the water column relative to the sediment-water interface indicate a progressive variation in light attenuation over the vertical with the water column divided into two layers by a pycnocline located approximately 10 m below the water surface (Figure 8). Within the lower layer light

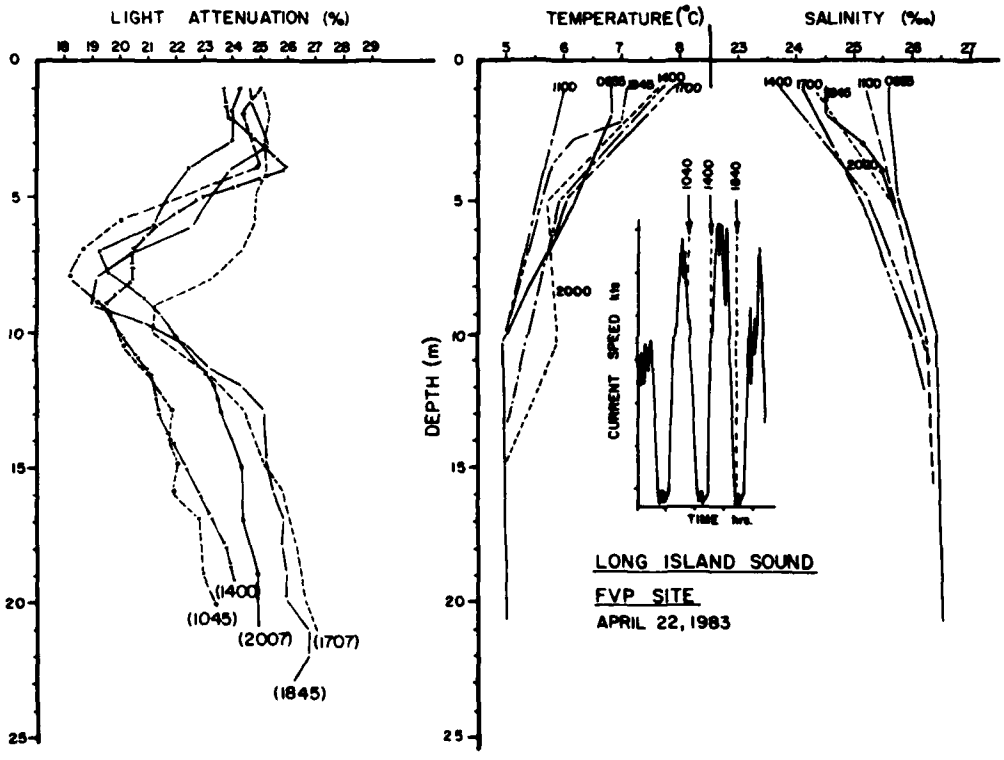


Figure 8. Selected hydrographic data, central Long Island Sound, 1983 (all times indicated are EST)

attenuation progressively increased as flow velocity decreased, a trend generally consistent with the observed increase in suspended material concentrations at low velocities. Although these light attenuation patterns fail to display a well-defined cross-over point indicative of a simple mass balance over the vertical, they appear to be reasonably representative of a system in which continuing erosion from the sediment-water interface is limited by a variety of factors and may be of secondary importance as a source of materials to the fluff layer relative to the overlying water column. For the study site the factors limiting erosion of the sediment-water interface include the

primary sediment characteristics, biological stabilization, and possible dissipation of tidal energy, sufficient to effectively reduce the boundary shear stress, resulting from the present of the fluff layer. Surveys of interface characteristics using a specially designed camera system (4) in and adjacent to the disposal site indicate reasonably high compaction rates in the placed materials and characteristically high densities of burrowing organisms immediately before and very shortly after completion of the disposal operation (Figure 9). These preliminary observations suggest that the presence of the fluff layer may represent the primary factor controlling the short-term erodibility of the sediment-water interface. Studies intended to resolve this question are presently in progress.



Figure 9. Interface camera profile, central Long Island Sound FVP site, 1983

Beyond the tidal frequencies the central Sound observations provide only limited indication of the character of system response. The 1982-1983 deployment period was characterized by an absence of significant storm events. Maximum wind velocities were associated with the prevailing westerlies and none were sufficient to measurably perturb the near-bottom suspended material field. The present extension of monitoring through 1984 will hopefully provide a more typical range of conditions and permit initial examination of the response of the transport system to storm passage under varying seasonal

conditions. Such observations would nicely complement the above studies of the effects of the fluff layer on the erodibility of the sediment-water interface.

DISCUSSION

The observations provided by the instrumentation array show the near-bottom suspended material field in eastern and central Long Island Sound to be characterized by a moderate to high degree of temporal variability. These variations are typically short lived and essentially confined to the period of the dominant forcing system. In the case of the tidal cycle this period is essentially coincident with the M2 component and the semimonthly neap-spring cycle. For significant storm events the perturbation period increases typically approaching the duration of the storm. Tidal effects continue to appear within these later events, however, with the superposition evident within the observed suspended material distributions providing evidence of the significance of wave-current interactions within these storm-induced perturbations.

The temporal characteristics displayed by the near-bottom suspended material field suggest that the majority of the sediments in suspension have high settling velocities despite their fine-grained nature. This evident enhancement in settling velocities, most probably resulting from a combination of biologically mediated aggregation and concentration/neighbor effects, complicates sampling and establishes criteria that are often difficult to satisfy using standard shipboard sampling procedures. Observations at both study sites indicate the utility of relatively high-frequency, in-situ sampling, particularly within efforts to develop analytical modeling schemes. Both the variations in system response to wind stress events differing only in duration, observed at the eastern Sound site, and the effects associated with the presence of the fluff layer, found in the central Sound, would be extremely difficult to assess using a few discrete samples.

In addition to the temporal characteristics the observations provided by the instrumentation array assist in the resolution of various spatial characteristics. The available data suggest that these differences are primarily the result of the presence of a fluff layer overlying the sediment-water interface within the central Sound. As noted, this layer may serve to modify near-bottom shear stress levels and thereby effectively armor the sediment-water interface or shield it from progressive erosion. Test of this hypothesis requires determination of the relationships between the fluff layer and the overlying water column. The addition of this vertical coordinate adds a sampling criterion that cannot be simply satisfied using the present instrumentation array. For the moment standard shipboard procedures are being used to obtain selected profiles over the vertical. If these data cannot disprove the importance of the water column as a source of materials to the fluff layer, future observations will most probably make use of a sensor array contained within a buoyancy capsule tethered to the bottom-mounted instrumentation array. The buoyancy system would, on command, cause the sensor package to migrate over the vertical, obtaining nearly continuous samples during transit. The resultant data could be stored internally, passed to the master data logger on the instrumentation array, or telemetered to a shore station during the short interval during which the capsule penetrated the air-sea surface. The measurements provided by such a system would provide a valuable extension to our present two-dimensional measurements of sediment

transport and in addition may serve to complement a variety of other studies including both physical and chemical characteristics. The technology required for this system is presently available. Similar vertical cycling systems have been used in open ocean studies although none have been applied in studies specifically related to sediment transport. The observations obtained within Long Island Sound suggest that within the more variable near-shore areas such systems may prove of particular value and significantly extend the utility and capabilities of the existing bottom-mounted instrumentation arrays. Such an extension can only improve our abilities to monitor and model the sediment transport processes affecting materials placed within coastal open water disposal areas.

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NUMERICAL PREDICTION ON THE EFFECT OF PHOSPHORUS RELEASE CONTROL BY SEDIMENT DREDGING

T. Horie

Chief of Purification Hydraulics Laboratory
Hydrodynamics Division
Port and Harbour Research Institute
Ministry of Transport
1-1, 3-chome, Nagase
Yokosuka, Japan, 239

ABSTRACT

This paper discusses the development of a numerical model on phosphorus behavior in sediment, behavior that is closely related with internal production of phytoplankton. Several assumptions are applied to the model formulation such that PO_4-P release is caused by the dispersion in interstitial water, and that PO_4-P in interstitial water is determined by the decomposition and dissolution of organic substance in sediment, adsorption or desorption to sediment particles, and dispersion. Each process included in the model was confirmed by field experiments or laboratory tests. Sediment thickness effective to release was decided as 60 cm, considering settling velocity, vertical distribution of substance in sediment, etc. Seven layers were totally prepared for the analysis. The model verification was performed to the annual verification of PO_4-P release rate, SS accumulation rate, I-P, O-P in sediment, and PO_4-P in interstitial water, by applying the model to the sediment sampled at off-Nishinomia in Osaka Bay. Furthermore, this model was applied to the prediction for the cut-off effect of PO_4-P release rate and its duration time. According to this analysis, in the case of 50-cm dredging thickness, the cut-off effect of PO_4-P release rate was estimated to be 70% just after the dredging, 50% after 20 years, and 20% after 50 years.

INTRODUCTION

Organic pollution in seawater is mainly due to the effluent, the internal production, and the release from sediment. These processes greatly influence

the seawater quality and complicatedly interact. Release flux from sediment is affected by sediment property, temperature, dissolved oxygen, deposited substances supplied from seawater, etc. According to the in-situ or laboratory experiments, the influence of settling substances on release flux is not negligible, and occasionally plays an important role. For the complete understanding of water quality distribution in seas, the mechanism of release from sediment should be analyzed with its connection to the influence of settling substances.

The release from sediment is also influenced by seawater quality as well as various processes in sediment. Therefore, the release mechanism should be simultaneously analyzed with seawater quality. However, under the present situation where the release mechanism is not well understood, it is necessary to first formulate the release process, then confirm the model reproducibility, and finally combine the release model and the seawater quality model.

Because of the above, a mathematical model on phosphorus behavior is formulated, based on water quality, sediment, interstitial water, and release rate data obtained at off-Nishinomia.

The release control effect due to dredging was numerically investigated by using this model.

MATHEMATICAL MODEL

All the essential materials composing organics, as well as temperature and illumination, can influence marine eutrophication. By limiting the group to chemical aspects, nitrogen, phosphorus, carbon, iron, manganese, vitamins, etc., are more influential. Among these elements, iron and manganese are sufficiently supplied from sediment. Therefore, the content of nitrogen and phosphorus is more sensitive to the eutrophication.

With the consideration of water quality distribution in Osaka Bay, the following assumptions and approximations are adopted:

a. PO_4-P release (R) is caused by dispersion in interstitial water.

b. PO_4-P concentration in interstitial water (C*) varies with bacterial decomposition and dissolution of sediment organic substances, adsorption to sediment particle, and release into seawater.

c. PO_4-P is allocated with a certain ratio (α) into interstitial water and sediment particles.

d. PO_4-P production in sediment is caused by the decomposition of newly accumulated substances and existing sediment.

e. The O-P supply from seawater to sediment is carried by settling substances, which is regarded as conservative.

f. Phosphorus behaves in the sediment within a certain depth.

g. PO_4-P in overlying seawater mixes with lower seawater in a certain exchange ratio.

The mathematical model formulated with the above assumptions is illustrated in Figure 1. (Each process shown in Fig. 1 is expressed by Equations 1 through 5).

O-P in n-th layer:

$$OP_n^{t+1} - OP_n^t = \left[\underbrace{-\tau_n^t}_{\text{decomposition}} + \underbrace{DS \times \left(OP_{n-1}^t - OP_n^t\right)}_{\text{accumulation}} + \underbrace{DSL \times \left(OP_{n-1}^t - 2 \times OP_n^t + OP_{n+1}^t\right)}_{\text{agitation}}\right] \times \Delta t \qquad (1)$$

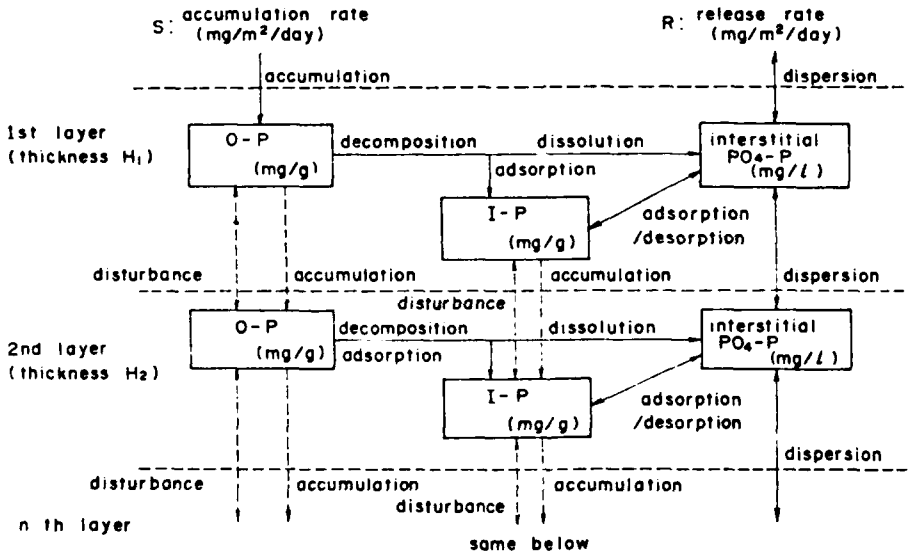


Figure 1. Illustration of the model on phosphorus behavior in sediment

I-P in n-th layer:

$$IP_n^{t+1} - IP_n^t = \left[\underbrace{\frac{\frac{1}{\phi_n \alpha_n}}{1 + \frac{1}{\phi_n \alpha_n}} \times \tau_n^t + q_n^t}_{\text{ad-, de-sorption}} + \underbrace{DS \times \left(IP_{n-1}^t - IP_n^t\right)}_{\text{accumulation}} + \underbrace{DSL \times \left(IP_{n-1}^t - 2 \times IP_n^t + IP_{n+1}^t\right)}_{\text{agitation}}\right] \times \Delta t \qquad (2)$$

allocation

C* in 1-st layer:

$$C_{*1}^{t+1} - C_{*1}^{t} = \left\{ \underbrace{\frac{1}{1 + \frac{1}{\phi_1' \sigma_1}} \times \frac{1}{\phi_1'} \times \tau_1^t}_{\text{allocation}} - \underbrace{\frac{1}{\phi_1'} \times q_1^t}_{\text{ad-, de-sorption}} + \underbrace{\left[\phi_1' \times D \times \frac{C_w^t - C_{*1}^t}{H_1/2} + \phi_1' \times D \times \frac{C_{*2}^t - C_{*1}^t}{(H_1 + H_2)/2} \right] \times \frac{1}{H_1}}_{\text{dispersion}} \right\} \times \Delta t \qquad (3)$$

C* in n-th layer:

$$C_{*n}^{t+1} - C_{*n}^{t} = \left\{ \frac{1}{1 + \frac{1}{\phi_n' \sigma_n}} \times \frac{1}{\phi_n'} \times \tau_n^t - \frac{1}{\phi_n'} \times q_n^t + \left[\phi_n' \times D \times \frac{C_{*n-1}^t - C_{*n}^t}{(H_n + H_{n-1})/2} + \phi_{u+1}' \times D \times \frac{C_{*n+1}^t - C_{*n}^t}{(H_{n+1} + H_{n-1})/2} \right] \times \frac{1}{H_n} \right\} \times \Delta t \qquad (4)$$

C_w in overlying water:

$$C_w^{t+1} = \left(C_w^t + \frac{1}{1000} \times R^t \times \Delta t \right) \times DL \qquad (5)$$

where

DS = accumulation rate

DSL = agitation factor

D = diffusion coefficient

ϕ = interstitial water ratio in dry sediment

ϕ' = void ratio

H = layer thickness

α = adsorption-desorption ratio of produced PO_4-P

q = adsorption or desorption rate

t = time

n = layer

The details of these equations and parameters are explained in the foregoing sections.

FORMULATION OF EACH PROCESS

Effective Thickness of Sediment and Layer Partition

The analysis of each parameter included in Equations 1 through 5 was performed on sediment sampled at a certain point in Osaka Bay. Figure 2 shows the experimental release rate results of sediment taken by a core sampler 200 mm in diameter. These experiments were conducted at 25°C under anaerobic conditions. According to these results, release rate is higher in layers less than 50 cm deep, and quite low in deeper layers. Matsumoto and Yokota (5) estimate an accumulation rate of 0.18-0.16 cm/year in the inner bay, and an annual mean of 0.41 cm by using ^{210}Pb vertical distribution in sediment (Figure 3). For this reason, the effective sediment thickness to release is taken to be 60 cm, which corresponds to an accumulation time of 150 years.

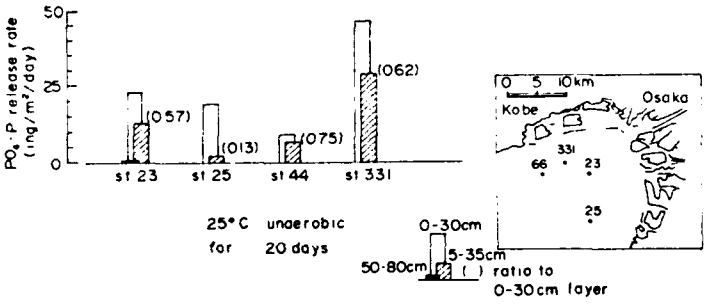


Figure 2. PO_4-P release rate in Osaka Bay

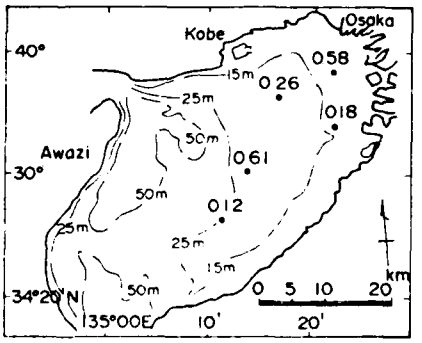


Figure 3. Distribution of accumulation rate (cm/year)

The vertical structure of sediment is schematically shown in Figure 4. The very thin surface layer consists of fresh deposits with high water content and easily mixes or moves along the surface. This layer has high T-P and PO_4-P production rate is great. This layer plays a major role in nutrient

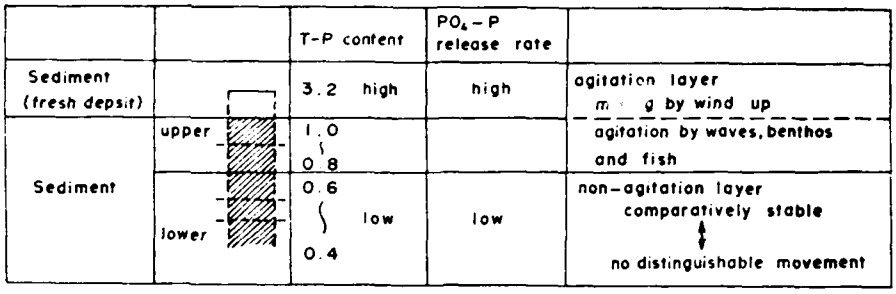


Figure 4. Vertical structure of sediment

release. In the lower layer, however, the water content decreases asymptotically under long-term consolidation, and less vertical agitation occurs. The T-P content and PO_4-P production rate in the lower layer are lower than in the upper layer. In the middle layer, water content is nearly constant, and the sediment in the upper layer is agitated by waves, fishery, and benthos activity. Agitation was presumed to occur within 8-30 cm in depth, according to the vertical distribution of excess ^{210}Pb by Matsumoto and Yokota (5). Kikuchi (4) concludes that sediment disturbance by benthos is limited within 20 cm in depth. Considering the results, the upper 15 cm in depth was regarded as the agitation layer in this model and assumed to be completely mixed in 4 years.

Thus, the effective thickness of sediment, 60 cm, was divided into an agitation layer of 15 cm and a lower nonagitation layer of 45 cm. Furthermore, the upper layer was divided into four layers and the lower layer into three layers.

The first layer was defined as the surface layer of 0 to 0.41 cm in depth, which corresponds to a 1-year deposit by Matsumoto and Yokota (5). The second layer was defined as 0.41 to 5 cm in depth where sufficient data were obtained by observation. The agitation layer lower than 5 cm was divided into the third and fourth layers. The nonagitation layer of 15 to 60 cm was divided into the fifth, sixth, and seventh layers. These layer partitions are illustrated in Figure 5.

The PO_4-P release processes were analyzed by Equations 1-5 for the seven layers.

Settling and Accumulation

Accumulation rate is related to settling rate as follows:

$$T = S \times \frac{1}{\text{sediment ratio}} \times \frac{1}{\text{unit weight}} \times 10^{-4} \times 365 \qquad (6)$$

where

T = accumulation rate (cm/year)

S = settling rate (SS g/m^2/day)

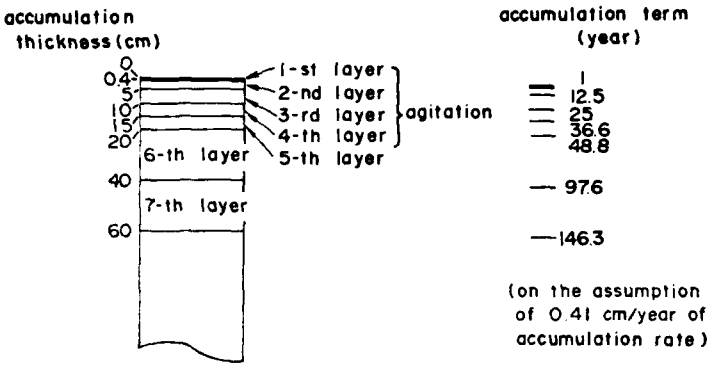


Figure 5. Layer partition of sediment

By substituting sediment ratio = 0.29 , unit weight = 1.12 g/cm^3 , and T = 0.41 cm into Equation 6, accumulation S is calculated by 3.9 SS g/m^2/ day. Assuming settling SS for t days as ΔS_t , stock in the first layer as $S1_t$, and allowable capacity of SS in the first layer as TS1, then the stock in the first layer is counted in the following manner (see Figure 6):

$$\left.\begin{aligned} S1_t &= S1_{t-1} + \Delta S_t \quad (\text{for } S1_t < TS1) \\ S1_t &= TS1 \\ S2_t &= \Delta S'_t \\ \Delta S'_t &= S1_t + \Delta S - TS1 \end{aligned}\right\} \left(\text{for } S1_{t-1} + \Delta S_t \geq TS1\right) \qquad (7)$$

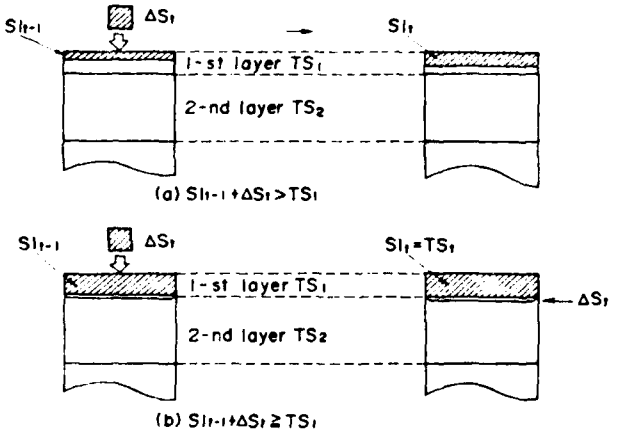


Figure 6. Estimation of SS stock

The SS stock in the other layers was counted in a similar manner. In the analysis, only SS was related with accumulation rate, and O-P and I-P were transported by SS. O-P and I-P contents in SS were considered to vary with dissolution and adsorption/desorption.

PO_4-P Production

PO_4-P production in sediment is mainly due to the bacterial decomposition of organics. Considering the property of sediment shown in Figure 4, PO_4-P production was estimated to both the fresh deposit and the old sediment for the first layer, and to only the old sediment for the second through fourth layers. The relation among SS, O-P, and PO_4-P production in sediment is illustrated in Figure 7.

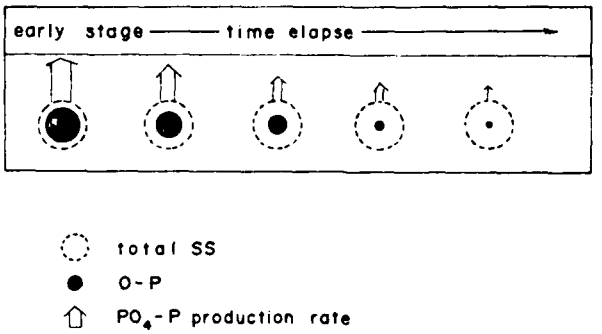


Figure 7. O-P decomposition and PO_4-P production

The SS shown in Figure 7 is time-dependent, and the O-P contained in SS decreases with time due to PO_4-P production by O-P decomposition. PO_4-P production rate decreases with time. PO_4-P production in the fresh deposit was obtained by the decomposition rate in an incubation test of sediment and O-P/SS ratio, with the assumption that O-P in test water decreases due to O-P in SS. Denoting dissolution rate constant as μ, the soluble PO_4-P for 1 day A is calculated as:

$$A = P_o - P_1 = P_o \times (1 - e^{-p \times 1}) \qquad (8)$$

where

P_o = initial O-P concentration, mg/ℓ

P_1 = O-P concentration after 1 day, mg/ℓ

Dividing A by SS, the PO_4-P production rate per 1 mg of SS is obtained. PO_4-P production rates for 17 samples in seawater and 7 rivers obtained by this method are plotted to O-P/SS ratio in Figure 8.

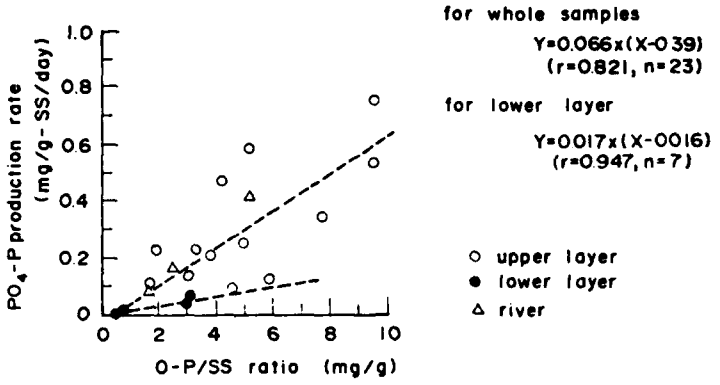


Figure 8. PO_4-P production rate from organic substances in seawater

The linear regression for the lower layer in seawater becomes

$$Y = 0.017 \times (X - 0.016) \tag{9}$$

Temperature dependence of PO_4-P production rate is expressed in Figure 9. By taking the temperature compensation factor as 1.05, the PO_4-P production rate for fresh deposit τ_F is expressed as follows:

$$\tau_F = 0.017 \times (OP/SS - 0.016) \times 1.05^{T-25} \tag{10}$$

where

OP/SS = O-P in 1 g of deposit, mg P/g SS

T = temperature, °C

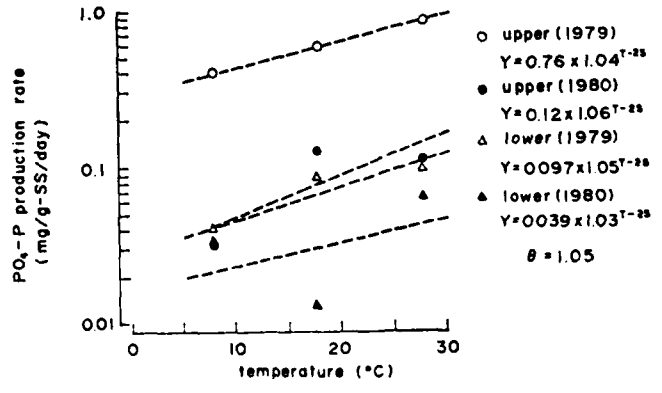


Figure 9. Temperature dependence on PO_4-P production rate

PO_4-P production rate for old sediment τ_S was obtained by a 20-day incubation test of sediment as follows:

$$\tau_S = \frac{\Delta C}{\Delta t} \times \frac{w}{V_d} \times \frac{1}{1 - w} \times \frac{1}{1000} \qquad (11)$$

where

$\Delta C/\Delta t$ = interstitial PO_4-P increase, mg/ℓ/day

τ_S = PO_4-P production rate per 1 g of dry sediment, mg/g dry sediment/day

w = water content

V_d = unit weight g/cm^3

The values of τ_S thus obtained by the incubation test at 25°C are plotted with O-P in sediment in Figure 10.

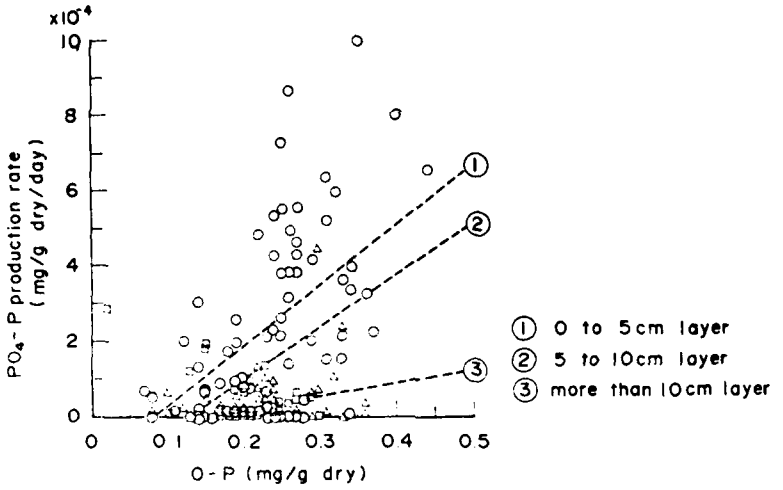


Figure 10. PO_4-P production rate and O-P in sediment

The average temperature compensation factor was estimated as 1.06 by using Figure 11.

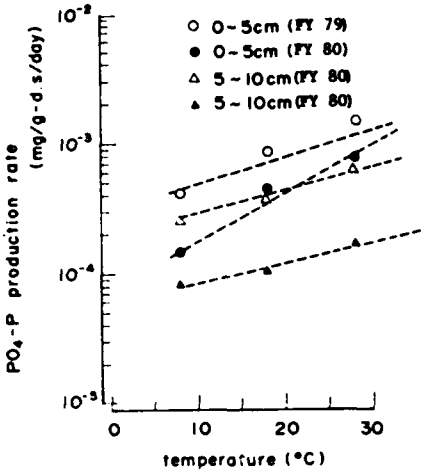


Figure 11. Temperature dependence on PO_4-P production rate

Accordingly, PO_4-P production rates for old sediment were expressed for each layer as:

$$\tau_s = 0.0016 \times (OP/SS - 0.08) \times 1.06^{T-25}$$

(for 0 to 5 cm depth)

$$\tau_s = 0.0014 \times (OP/SS - 0.13) \times 1.06^{T-25} \quad (12)$$

(for 5 to 10 cm depth)

$$\tau_s = 0.00035 \times (OP/SS - 0.14) \times 1.06^{T-25}$$

(for more than 10 cm depth)

where

OP/SS = O-P in 1 g of sediment, mg/g

T = temperature, °C

Adsorption and Dissolution of Produced PO_4-P

A certain amount of produced PO_4-P due to the dissolution of organic matters in sediment was assumed to dissolve into interstitial water and the rest to adsorp to sediment particles. The increase of interstitial PO_4-P,

ΔC^*, and the increase of I-P in sediment due to the adsorption ΔIP were related with the following equation for the time lapse Δt and the PO_4-P production rate τ :

$$\tau \times \Delta t = \Delta IP + \phi \cdot \Delta C^* \tag{13}$$

where

C = interstitial PO_4-P, mg/ℓ

IP = I-P in sediment, mg/ℓ

ϕ = interstitial water ratio in sediment, l/g

t = time, days

By defining allocation factor α between the above-mentioned interstitial PO_4-P, C^* , and adsorped I-P

$$\alpha = \frac{C^*}{IP} \tag{14}$$

and substituting Equation 14 into 13,

$$\Delta IP = \frac{\frac{1}{\phi\alpha}}{1 + \frac{1}{\phi\alpha}} \times \tau \times \Delta t \tag{15}$$

or

$$\Delta C^* = \frac{1}{1 + \frac{1}{\phi\alpha}} \times \frac{1}{\phi} \times \tau \times \Delta t \tag{16}$$

is reduced. Vertical distribution of the allocation factor α was obtained in Figure 12.

The variation of α with depth is comparatively large between 0 and 10 cm in depth, and nearly constant deeper than 10 cm. This factor α is dependent on temperature and dissolved oxygen. The temperature compensation factor was selected from the vertical distribution of α at the depth of 80 cm in Figure 12 as follows:

$$\alpha = \alpha_{20} \times \theta^{\tau-20} \tag{17}$$

where

α, α_{20} = allocation factor at the temperature of T°C and 20°C, respectively.

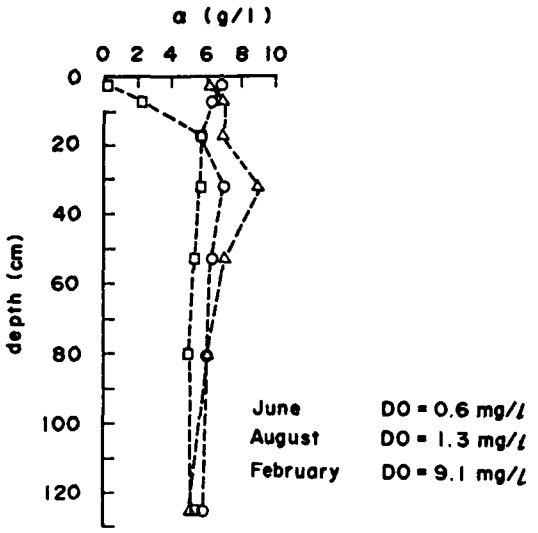


Figure 12. Vertical distribution of allocation factor

By substituting $\alpha_{20} = \alpha_{21.1}$ (g/ℓ) and $\alpha_{11.2} = 4.92$ (g/ℓ), $\theta = 1.02$ is obtained. On the other hand, at 20°C, the allocation factor α of the sediment in Osaka Bay at the depth of 0 to 5 cm is a function of the DO (0 to 2 m above sea bottom), as shown in Figure 13.

Therefore, α for 0 to 5 cm is expressed with the aid of Equation 17 as

$$\alpha = 11.5 \times 0.748^{DO} \times 1.02^{T-20} \tag{18}$$

The α for more than 5 cm in depth is a constant value of 6.1 g/ℓ based on the results of Figure 12. The α in surface layer thus varies with temperature and DO, and the following relation must be held in order to satisfy the equilibrium due to the increase of adsorption Q :

$$\frac{C^* - \frac{Q}{\phi}}{IP + Q} = \alpha \tag{19}$$

or

$$Q = \frac{\frac{C^*}{\alpha} - IP}{1 + \frac{1}{\phi\alpha}} \tag{20}$$

where

Q = adsorption of I-P, mg/ℓ.

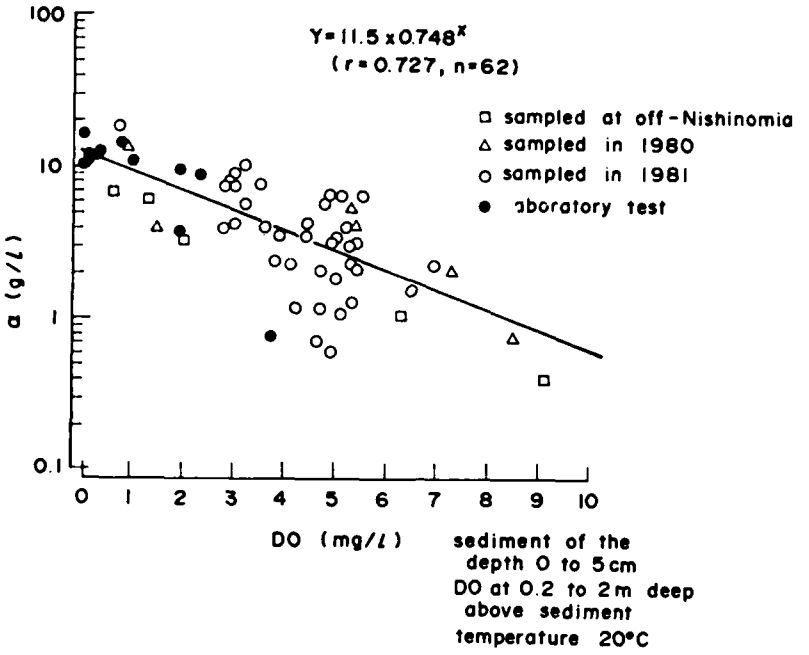


Figure 13. Allocation factor and DO

The time required to attain equilibrium d for Equation 20 is approximately 20 hours (6). Thus, d was taken as 1 day and adsorption rate q was expressed as:

$$q = \frac{1}{d} \times Q = \frac{\frac{C^*}{\alpha} - IP}{1 + \frac{1}{\phi\alpha}} \qquad (21)$$

where

q = adsorption rate, mg/g/day

d = time required for equilibrium adsorption rate, days

Movement of Substance in Sediment

O-P movement

O-P in accumulated SS gradually decreases with time within the sediment. For the disturbed lower layer, deposit ΔS_n, O-P in sediment, P_n, SS S_n, and PO_4-P production rate τ_n after the n-th day were estimated as illustrated in Figure 14 and Equation 22.

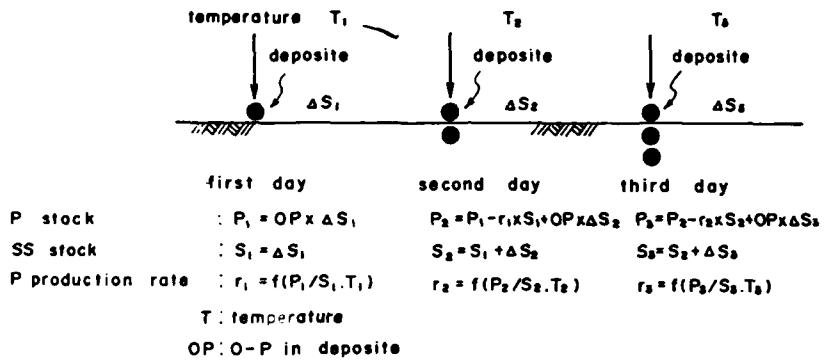


Figure 14. Estimation of SS, P, and produced PO_4-P

$$P_n = P_{n-1} - \tau_{n-1} \times S_{n-1} + OP \times \Delta S_n$$

$$S_n = S_{n-1} + \Delta S_n \qquad (22)$$

$$\tau_n = f(P_n/S_n, T_n)$$

where

OP = O-P in deposits

T = temperature

Dispersion and release

Dispersion of interstitial PO_4-P between layers was expressed as a general form of the function of vertical concentration gradient as shown in Equations 3 and 4. PO_4-P release rate R was expressed as the dispersion from sediment surface to seawater as follows:

$$R = \phi' \times D \times \frac{C^*_1 - C_w}{H_1/2} \times 10^3 \qquad (23)$$

where

R = PO_4-P release rate, $mg/m^2/day$

ϕ' = void ratio

D = dispersion coefficient, m^2/day

C^* = interstitial PO_4-P in the first layer, mg/ℓ

C_w = PO_4-P just over the sediment surface, mg/ℓ

H_1 = thickness of the first layer, m

To the samples of the 0- to 30-cm layer, the 50- to 80-cm layer, and the 100- to 130-cm layer, laboratory test results were obtained as shown in Figure 15.

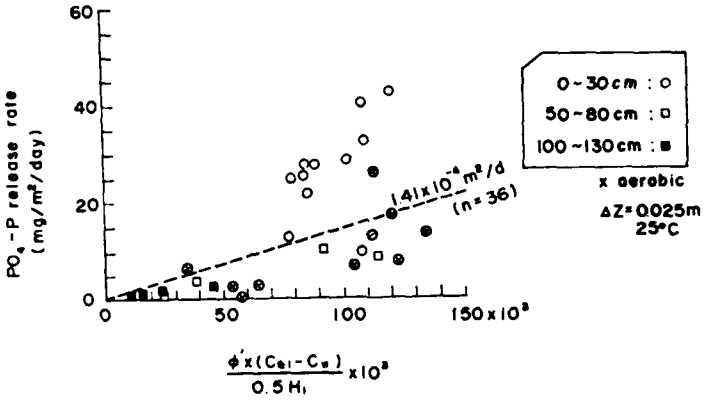


Figure 15. Dispersion coefficient

Assuming that the temperature compensation factor of the dispersion coefficient is equivalent to that of molecular dispersion coefficient of cation or anion, 1.03^{T-25}, the dispersion coefficient at arbitrary temperature D_T is expressed by

$$D_T = 1.41 \times 10^{-4} \times 1.03^{T-25} \quad (24)$$

Agitation

As stated before, the upper four sediment layers were treated as an agitation layer. The agitation coefficient DSL was expressed as

$$T = DSL \times C_{SS} \times \frac{1}{L} \quad (25)$$

where

T = SS flux, g SS/m^2/day

DSL = agitation coefficient, g SS/m^2

L = displacement of sediment, m

By regarding the physical property of sediment less than 15 cm in depth as homogeneous and assuming SS in one half of the agitation layer (i.e., 7.5 cm moves a distance of 15 cm for 4 years), then sediment in the layer is completely mixed. Thus, DSL was estimated by Equation 25 as

$$DSL = 3.4 \times 10^{-6} \ (m^2/day) \tag{26}$$

Water Ratio and Unit Weight of Sediment

The vertical distribution of water ratio w and the unit weight V_d of sediment at off-Nishinomia was obtained as shown in Figures 16 and 17.

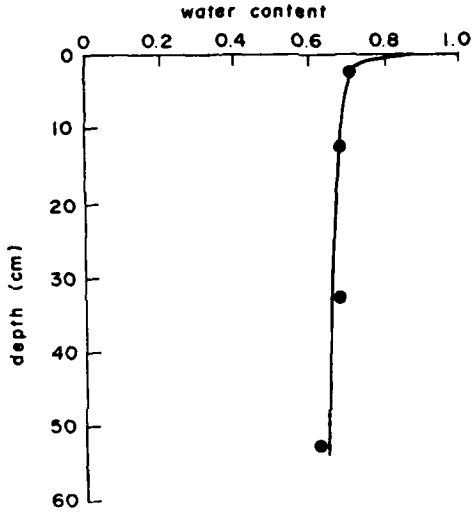


Figure 16. Vertical distribution of water content

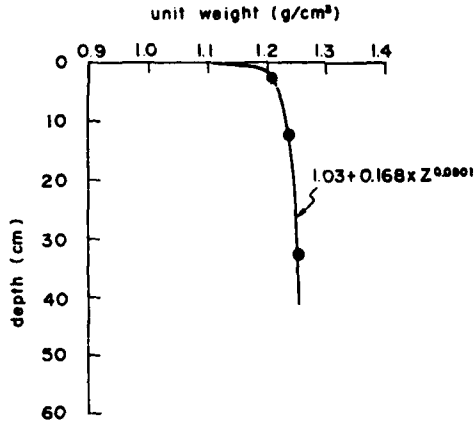


Figure 17. Vertical distribution of unit weight

Assuming that $w = 1.0$ and $V_d = 1.03\ g/cm^3$ at the sediment surface ($z = 0$), and that w and V_d vary exponentially with depth z, the following relations are obtained:

$$w = 1 - 0.268 \times z^{0.0689}$$
$$V_d = 1.03 + 0.168 \times z^{0.0801} \quad (27)$$

VERIFICATION OF MODEL

Computational Conditions

For the verification of the model, the O-P settling rate and DO in bottom water were presumed to be the term of the past 100 years. The settling rate of phosphorus was assumed to be proportional to the phosphorus in seawater. Long-term variation of phosphorus was decided on the basis of data observed by Kobe Marine Observatory in 1934-35, 1952-53, and 1974-79, and by the data obtained by prefectural governments after 1976. This result is shown in Figure 18.

Furthermore, the annual variation of O-P in the lower layer was assumed by the simulation results from the two-layer nutrient-cycle model (1). DO was also calculated at almost the same period as O-P. DO in lower layer was, therefore, given as

$$DO = DO1 - (DO1 - DO2) \times \beta \quad (28)$$

where

DO1 = DO at salinity 31.6 ppt and given temperature

DO2 = observed DO (see Figure 19)

β = unsaturation index to a standard year

β was determined from the records shown in Figure 20. Temperature variation was periodically given as shown by the line in Figure 21.

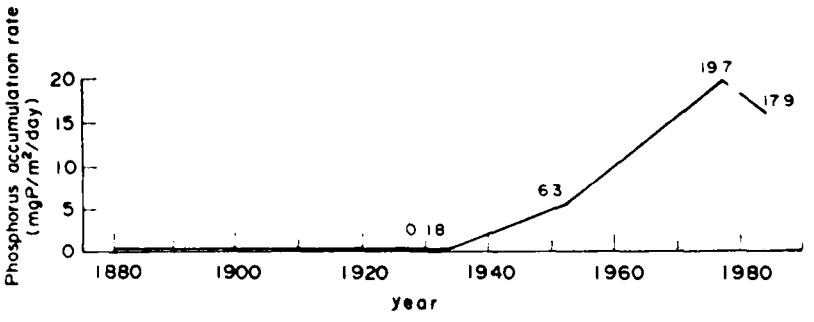


Figure 18. Long-term variation of phosphorus accumulation

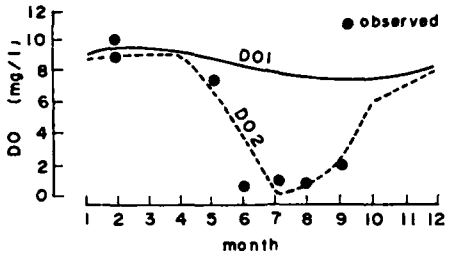


Figure 19. Observed DO and saturated DO

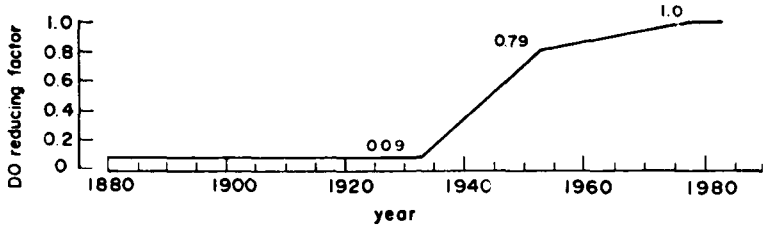


Figure 20. Long-term variation of DO reducing factor

Calculated Release Rate and Vertical Distribution of Phosphorus

A numerical simulation of Equations 1-5 was conducted for the 100-year period from 1880 to 1983 by applying the above conditions and a time increment Δt = 0.1 day. The calculated annual variation of release rate is shown in Figure 22 for the most recent year of 1983. The calculated release rate is slightly greater than the observed value, but annual variation generally coincides with the observed one. The release rate integrated with the time range of each year is shown in Figure 23. PO_4-P release has a close relation with settling load of O-P as shown in Figure 18, and rapidly increases after 1950. The observed annual release from 1979 to 1981 is in the range of 3 to 4 g P/m^2/year, and calculated value also lies on this range.

The vertical distribution of calculated T-P, O-P, I-P in sediment, and interstitial PO_4-P is shown with observed values in Figure 24. According to these results, vertical distribution of calculated phosphorus in the lower layers (from first to fourth) coincides closely with the observed value, with the exception of interstitial PO_4-P. Calculated phosphorus lower than the fifth layer is about half the observed value, and seems to be very sensitive to initial values, iteration time, and other parameters.

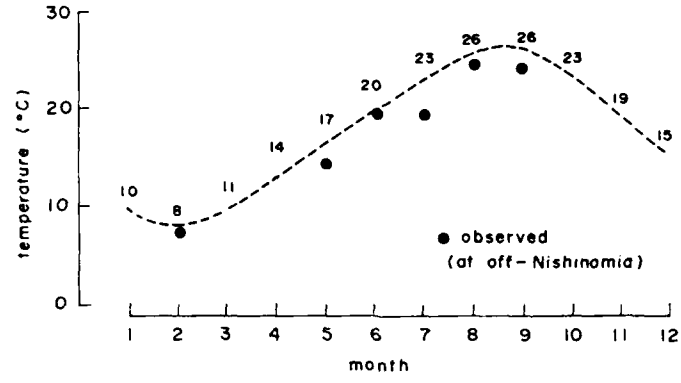


Figure 21. Seasonal variation of temperature

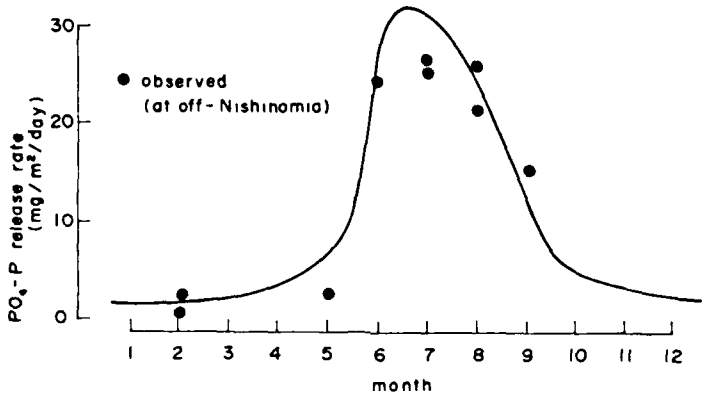


Figure 22. Seasonal variation of release rate

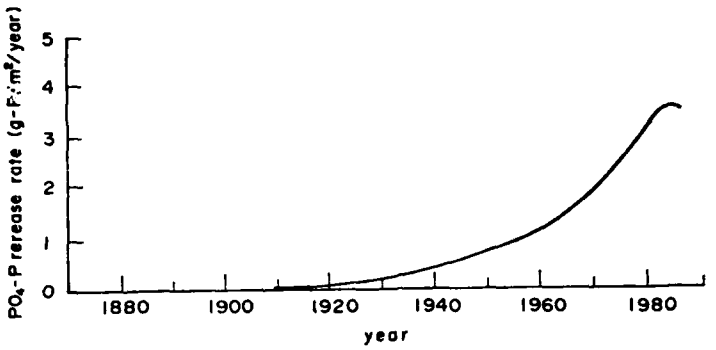


Figure 23. Long-term variation of release rate

NUMERICAL PREDICTION ON THE EFFECT OF RELEASE CONTROL DUE TO DREDGING

Dredging and Layer Partition

Assuming a surface dredging thickness of 50 cm, layer partitioning before and after dredging was executed as shown in Figure 25. By dredging, the layers from the first to fifth predredging were removed, and the sixth and seventh predredging layers corresponded to the first to fifth layers after dredging.

Treatment of O-P Load and DO After Dredging

The O-P load to sediment decreases after dredging due to improved seawater. The O-P just over the bottom is decided by both inflowing load and internal production, and these relations are rather complicated. Owing to this fact, O-P and DO were decided on the basis of simulation result of Osaka Bay, i.e., natural condition, release cutoff condition, and release and inflowing load cutoff condition. Calculated O-P values at off-Nishinomia are listed in Table 1.

TABLE 1. O-P IN LOWER LAYER CALCULATED BY THE TWO-LAYER MODEL

Condition	O-P, mg/ℓ
Natural	0.027
Release cutoff	0.016
Both release and effluent cutoff	0.009

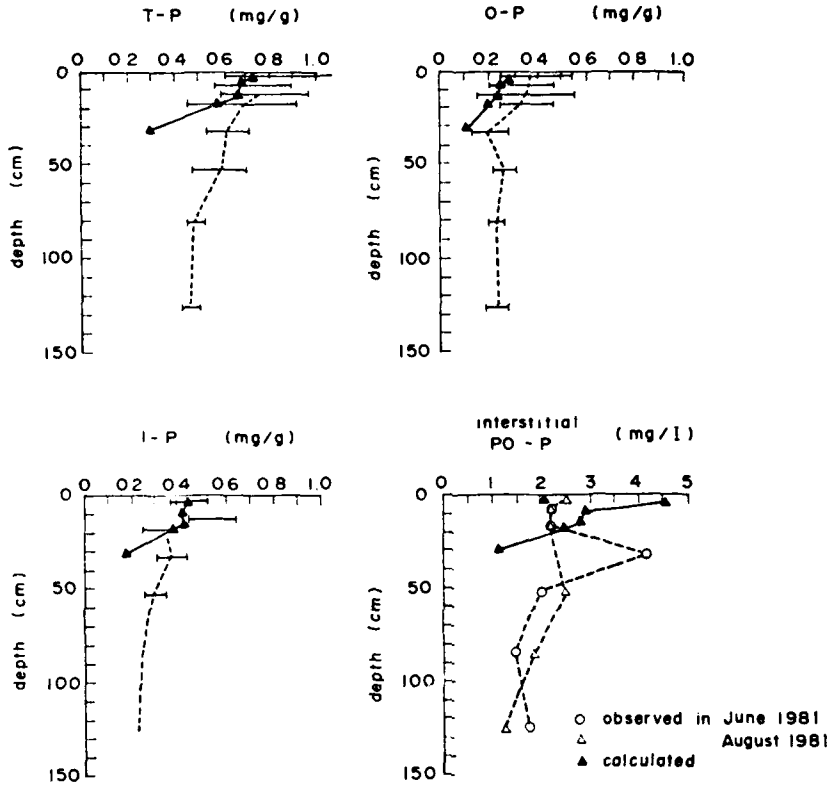


Figure 24. T-P, O-P, and I-P in sediment and interstitial PO_4-P

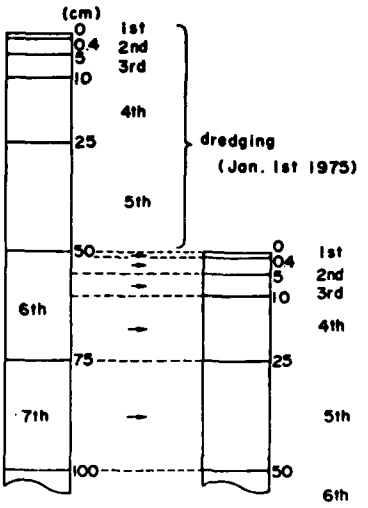


Figure 25. Layer correspondence before and after dredging

The O-P from the natural condition to the release cutoff condition was estimated as the basis of release and effluent cutoff condition as

$$(0.016 - 0.009)/(0.027 - 0.009) = 0.388$$

Therefore, supposing $O\text{-}P = 1$ and $R = 1$ in the natural condition, then the relationship between O-P and R is expressed by linear interpolation as Figure 26.

Settling $R = 1$ for before dredging and $R = R$ at a specific time after dredging, then the reducing factor of O-P load to natural condition β is expressed as

$$\beta = 0.612\ R + 0.388 \qquad (29)$$

In the same manner, DO after dredging was estimated by using above β as:

$$\text{DO after dredging} = \text{natural DO} + \text{saturated DO} \times (1 - \beta) \qquad (30)$$

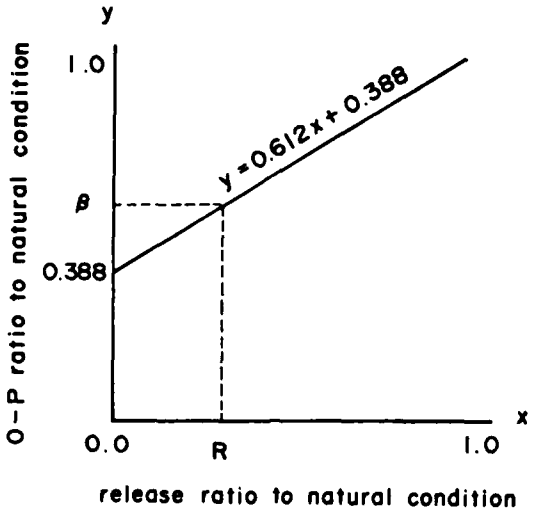


Figure 26. Relation between O-P and release rate

Release Cutoff Effect and its Duration Time

After the computation for 50 years starting with natural condition of O-P, I-P in sediment, and interstitial PO_4-P, another 50 years of computation after dredging was continued. The variation of PO_4-P release is shown in Figure 27.

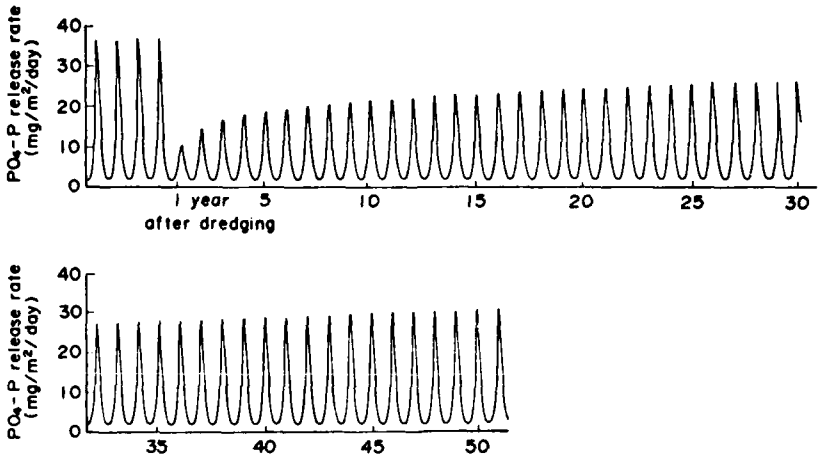


Figure 27. Release cutoff effect and its duration time

Comparing each peak value during each summer found a release cutoff ratio of 70% just after dredging, 50% after 20 years, and 20% after 50 years. Such a gradual reduction of the effect seems to depend on newly accumulated substances. The above results are, however, deduced on the basis of many uncertain parameters and assumptions. Further investigation is necessary until a satisfactory solution is obtained.

CONCLUSIONS

A numerical model was developed to predict the behavior of phosphorus in sediment. For verification of the model, O-P, I-P in sediment, interstitial PO_4-P, and PO_4-P release rates were simulated from the data of the past 100 years. Furthermore, the release cutoff effect and its durability due to dredging were investigated by this model.

These results are summarized as follows:

a. The model's variation of PO_4-P release rate and integrated release rate over the year are closely correlated with the observed values.

b. The vertical distribution of T-P, O-P, and I-P in sediment closely matched the upper layer, but the lower layer was underestimated because of the effect of initial condition, iteration time, etc. Interstitial PO_4-P was overestimated to the upper layer.

c. Release cutoff was very effective just after dredging, but reduced with time. Further investigation is necessary to obtain a satisfactory solution.

ACKNOWLEDGEMENTS

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I must express my thanks to Prof. H. Nakanishi of Yamaguchi University, Prof. T. Horiguchi of Tokyo Metropolitan University, Assistant Prof. T. Morioka of Osaka University, and Assistant Prof. M. Ukita who gave a lot of instructions on the fulfillment of this investigation.

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ECOLOGICAL EVALUATION OF ORGANOTIN-CONTAMINATED SEDIMENT

M. H. Salazar and S. M. Salazar

Environmental Sciences Division
Naval Ocean Systems Center
San Diego, California 92152

ABSTRACT

A standard dredged material bioassay was conducted on sediment with high levels of organotins to (1) assess the toxicity and bioavailability of organotins associated with sediment, and (2) determine if this sediment would qualify for ocean disposal. Since the Navy is considering fleetwide implementation of organotin coatings and some commercial fleets are currently using them, it is essential to evaluate the effects of organotins on the dredging permit process.

Particulate-phase tests were conducted with *Acanthomysis sculpta* (mysid), *Citharichthys stigmaeus* (flatfish), and *Acartia tonsa* (copepod). Solid-phase tests were conducted with *A. sculpta*, *Macoma nasuta* (clam), and *Neanthes arenaceodentata* (polychaete worm). The bioassay also included an estimate of the potential for bioaccumulation of cadmium, chromium, copper, mercury, silver, pesticides, PCBs, petroleum hydrocarbons, and organotins.

The concentration of tributyltin-oxide in sediment collected from Commercial Basin was measured to be 780 ppb. Initial concentrations of TBTO in particulate phase test water was 0.49 ppb and in solid phase test water 0.20 ppb. Butyltins measured in test water and test sediments demonstrate that monobutyl-, dibutyl-, and tributyltin were all leached off Commercial Basin sediment. Treatment clams accumulated organotins to a concentration an order of magnitude above control clams (2.82 ppm TBTO vs. 0.26 ppm TBTO) and a factor of four above treatment sediment.

Survival was high in all particulate phase and solid phase tests. There were no statistically significant differences in survival when controls were compared to treatments. As demonstrated for other contaminants in other studies, high levels of organotins in sediments do not a priori indicate a significant adverse impact on the marine environment after ocean

disposal. The sediment tested would qualify for ocean disposal under the present guidelines administered by EPA and the Army Corps of Engineers.

INTRODUCTION

Commercial Basin, San Diego Bay, California, is an area that is heavily used by both commercial and private vessels. These vessels are coated with antifouling paints that leach copper or organotins into the water column. A significant portion of these contaminants are adsorbed by the sediment. Previous measurements in the area have shown elevated copper levels in the water column, presumably from the antifouling coatings (11, 33). More recent analyses performed by this laboratory indicate the Commercial Basin water column to have the highest levels of tributyltins as tributylin-oxide (TBTO) found in San Diego Bay (30). TBTO concentrations for water collected near the bottom ranged from 0.11 to 0.55 ppb. These water column values are close to those causing effects on sensitive marine organisms like mysids and copepods in laboratory tests (20, 27, 29). Corresponding sediment samples from the same area ranged from 32 to 560 ppb TBTO. This represents a concentration factor of approximately three orders of magnitude between the water column and the sediment.

It is estimated that more than 20 percent of the vessels in Commercial Basin currently use organotin-based antifoulant coatings. The environmental impact of organotins from these coatings is of great concern to the Navy due to projected Fleet implementation of organotin-based antifouling coatings. As part of a research program on the fate and effects of organotins in the marine environment and a program to expedite Navy dredging, we performed a bioassay on sediments collected from Commercial Basin. This site was selected, even though it is not a Naval facility, because of the high concentration of TBTO found there. The purpose of this work was to (1) assess the toxicity and bioavailability of organotins associated with sediments, and (2) determine if sediment with these high organotin loads would qualify for ocean disposal under the present guidelines.

According to Section 103 of the Marine Protection, Research and Sanctuaries Act of 1972, sediments to be dumped into ocean waters must be evaluated to determine the potential environmental impact. Evaluations must be in accordance with criteria in the Federal Register, Vol 42, No. 7, Tuesday 11 January 1977. Previously, evaluations were based only on chemical analyses of dredged material. Now, sediment chemical analyses are not required and bioassays are the primary basis for determining the potential for environmental impact. The intent of the law is to prevent an adverse environmental impact from ocean disposal of dredged material.

The standard bioassay test procedures have been published by the Environmental Protection Agency (EPA) and the Corps of Engineers (COE) in an Implementation Manual (6). We conducted an ecological evaluation of Commercial Basin sediment according to the bioassay procedures outlined in this manual

even though no dredging or ocean disposal is planned for this sediment. The following are the results of the evaluation performed on sediment from Commercial Basin, San Diego.

METHODS

The procedures given in the manual for the "Ecological Evaluation of Proposed Discharge of Dredged Material into Ocean Waters" (6) were followed wherever possible and practicable for the majority of the bioassay. It should be remembered that the Implementation Manual was intended to be only a guide and need not be followed exactly. In 1983 the COE, Los Angeles District, also outlined requirements and guidance in "District Bioassay Testing Procedures." These regionally specific procedures were also followed wherever feasible. The most significant procedural modification used in these bioassays was a conceptual one. Instead of collecting sediment from three sites within each dredging area and conducting three bioassays per site as the manual suggests, we collected sediment from three different sites in the dredging area and pooled them for one bioassay. Sediment samples were pooled by mixing during collection, preparation of particulate phase slurries, and pre-sieving for the solid phase.

Although the manual suggests using 10, 50, and 100 percent test solutions of liquid and particulate phases, for this bioassay we used only the 100 percent particulate phase solutions for two reasons. First, the suspended particulate phase incorporates both a liquid and a particulate phase. Second, in all our previous bioassays we have not had a significant effect with the 100 percent particulate phase solutions. Therefore, the lesser dilutions (10 and 50 percent) are superfluous. For these reasons, only 100 percent suspended particulate phase solutions were prepared and tested. To increase statistical reliability we used five replicates instead of the recommended three. These changes were also made with agreement of EPA and the COE.

Sediment

Collection Procedures

Three samples were collected along a pier on the northwest shore in Commercial Basin (Figure 1). Test sediment was collected from the upper 20-30 cm of bottom material by SCUBA divers using plastic buckets. On the surface the sediment was dumped into 60-liter ice chests which were lined with large plastic bags to reduce potential contamination. Ice chests were used for ease in handling sediment and to maintain the temperature of sediment between collection and storage. The sediment was stored at 4°C and used 48 hours after collection. Approximately 200 liters of reference/control and 100 liters of treatment sediment were collected.

Sediment Characteristics

Sediment that would serve as both a reference and a control was collected in an uncontaminated area off North Island near the mouth of the bay (Figure 1). Sediment from this area has been used as a combined reference/control in our 19 previous dredged material bioassays on San Diego Bay sediment. It can be used as a reference because mean grain size and composition are similar to disposal site sediment. Available information suggests that

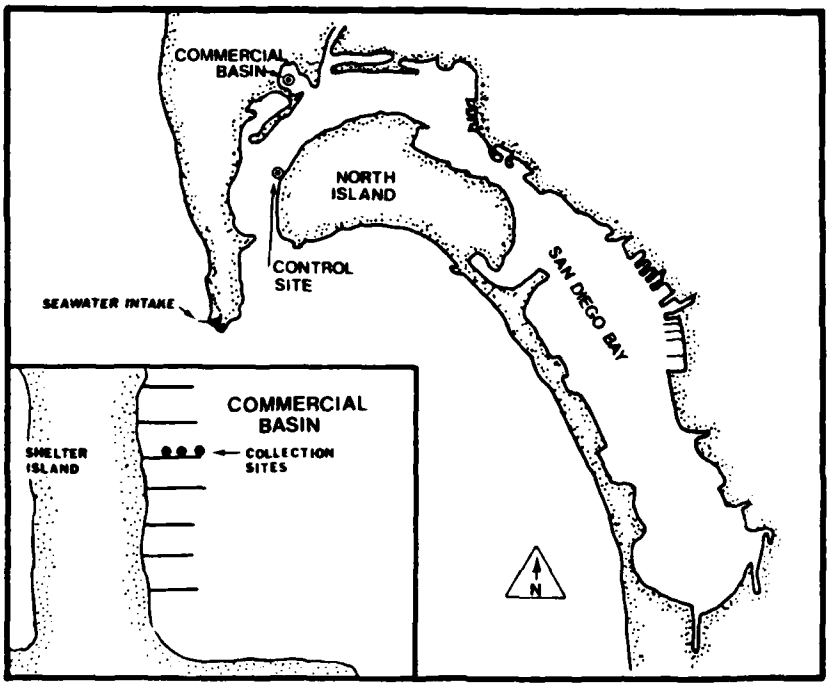


Figure 1. Location of Commercial Basin and control sediment collection sites in San Diego Bay, and the seawater intake.

the average grain size at the nearest interim EPA disposal site (LA-5) is between 0.030 mm (4) and 0.079 mm (12). The mean grain size of the control sediment used in these bioassays was 0.0981 mm. Control sediment was composed of approximately 81 percent sand and 19 percent silt and clay, and is classified as fine sand (31). This is similar to disposal site sediment composition estimated at 67 percent sand and 33 percent silt and clay (12). Grain sizes for control and treatment sediment were measured by the tube drop method.

This sediment can also serve as a control because of its relatively low contamination. As part of our previous bioassays we measured five heavy metals (Cd, Cr, Cu, Hg, Ag), petroleum hydrocarbons, PCBs, and pesticides in test sediment. Control sediment has almost always been significantly lower in these contaminants than treatment sediment (21-26). In many cases contamination was lower by approximately an order of magnitude. Relatively low contamination combined with sediment composition and grain size, which are similar to disposal site sediment, justify using North Island sediment as a control. For expediency this reference/control sediment will be referred to as "control sediment."

Seawater

The manual suggests that disposal site water be used if possible. This was impractical because of the large volume required and the distance to the site. Nearshore seawater available at the Naval Ocean Systems Center (NOSC) Marine Life Sciences laboratory was used for this bioassay. The seawater

intake for this system is approximately 250 meters from shore (Figure 1). This natural seawater was passed through large sand filters prior to delivery in the laboratory.

Periodic water analyses have shown that the five heavy metals (Cd, Cr, Cu, Hg, Ag) we routinely measure to assess bioaccumulation are present in only trace amounts. These concentrations are similar to those found in open-ocean water (2). Analyses of tissues from test animals in previous bioassays have shown no significant accumulation of heavy metals, hydrocarbons, PCBs, or pesticides (21-26).

Preparation of Test Tanks

Suspended Particulate Phase

Test slurries were prepared in a 4:1 ratio as the manual recommends by adding 120 liters of control water to 30 liters of treatment sediment. Clean, aged 50-gallon plastic trash cans were used for mixing. The slurries were mixed by vigorous bubbling with compressed air at a rate of 50 liters per minute. After 30 minutes of bubbling, the air stones were removed and the slurries were allowed to settle for one hour. After settling, the supernatant was pumped directly into test tanks for the 100 percent suspended particulate phase test.

Solid Phase

Treatment and control sediments were sieved through 1.0-mm stainless steel screen to remove endemic organisms. Solid phase test tanks were prepared with a 15-mm layer of treatment sediment on top of a 30-mm layer of control sediment. Control tanks for the solid phase tests were filled with control sediment to a depth of 45 mm. After distribution of the sediments, the tanks were slowly filled with control water. Animals were added after preparation of the solid phase tanks.

Test Animals

Suspended Particulate Phase

For the particulate phase the manual suggests using phytoplankton or zooplankton, a crustacean or mollusk, and a fish. A copepod (planktonic crustacean), a mysid (hypoplanktonic crustacean), and a fish were used in this phase of the sediment bioassays. The manual further suggests that organisms from the disposal site be used where possible. It was not practicable to use species collected from the disposal site because they are neither well known nor easily obtained.

Acartia tonsa, a copepod, was selected to satisfy the zooplankton requirement in the particulate phase. *Acanthomysis sculpta* was used as the representative crustacean species. The speckled sanddab, *Citharichthys stigmaeus*, was the third species used for the particulate phase tests. All *C. stigmaeus* surviving at the end of the 96-hour test were weighed and measured. These animals had a mean length of 62.8 mm and a mean wet weight of 2.75 g.

Solid Phase

The manual suggests using three species for solid phase tests, one filter-feeder, one deposit-feeder, and one burrowing species. It is further suggested that the species be selected to include a crustacean, an infaunal bivalve, and an infaunal polychaete. As with the particulate phase, it was not practicable to use species from the disposal site. Solid phase tests for the Commercial Basin bioassay were conducted with *A. sculpta* (mysids), *Neanthes arenaceodentata* (polychaete worms), and *Macoma nasuta* (clams).

Estimates of lengths and weights for clams were made by measuring and weighing 50 animals per treatment at the end of the bioassay. Test animals had a mean length of 40.6 mm and a mean wet weight of 8.24 g.

Bioaccumulation

The manual suggests that all biological evaluations of the dredged material include an assessment of the potential for contaminants to be bioaccumulated in the tissues of marine organisms. We used *M. nasuta* as one bioaccumulator because of the relatively large amount of tissue available for chemical analysis. We used *N. arenaceodentata* as the second required bioaccumulator.

Animal Collection

We collected copepods and mysids just outside the mouth of San Diego Bay on the first day of the test. Copepods were collected by slowly towing a net (175-μ mesh) in open water just beneath the surface. Mysids were collected in the surface canopy of the kelp with buckets. Temperature upon arrival was 18°C for both species.

Three of the five species of marine organisms used in these bioassays were purchased from two commercial suppliers. Fish and clams were collected by Brezina and Associates, Dillon Beach, California. The fish were collected near the mouth of Tomales Bay, California, in 10-15 feet of water. The clams were collected in an intertidal mudflat, also in Tomales Bay.

N. arenaceodentata is kept in continuous culture by Dr. Donald Reish, California State University, Long Beach. They were obtained one day prior to the start of the test.

Test Conditions

Polycarbonate tanks, which exhibit low adsorption of trace metals and organotins, were used as test containers in the particulate phase and solid phase tests for all species except copepods. We used Pyrex beakers for copepods because Pyrex glass is also nonadsorptive and the size is convenient.

Temperature at the 600-foot-deep disposal site probably ranges between 8 and 10°C. We used water temperatures between 13.5 and 16.0°C for this bioassay because they are closer to the optimum for maintaining these particular test animals and because of the limits of ocean temperature and the flow-through seawater system. Physical-chemical parameters were measured daily. Cool white fluorescent bulbs were used to approximate the spectral output of the sun, but irradiance in the test containers was significantly less than the

1200 microwatts/cm^2 recommended by the manual. Irradiance was measured with a photometer. For each test species the values obtained (microwatts/cm^2) were as follows: *C. stigmaeus* (100-500); *N. arenaceodentata* (1-250); *A. sculpta* (1-250); *M. nasuta* (100-250); and *A. tonsa* (5-50). The light regime consisted of a 14L:10D cycle.

Suspended Particulate Phase

The copepods were held in 400-ml Pyrex beakers, mysids in 4-liter polycarbonate tanks, and fish in 16-liter polycarbonate tanks. All particulate phase tests were conducted under static conditions for 96 hours. Five replicates were used for each treatment and control condition with 10 organisms in each replicate. The fish were vigorously aerated at a rate between 500 and 1300 ml/minute. Mysids were moderately aerated at a rate of 3-7 ml/minute. Copepods were not aerated.

Copepods were fed maintenance levels of the, phytoplankton *Isochrysis galbana*. Feeding occurred on days 1, 2, and 3 of the bioassay (1,000-10,000 cells/copepod/day). Mysids were fed maintenance levels of brine shrimp nauplii (25-30 nauplii/mysid/day). The fish were not fed during the bioassay.

Live animals were counted at the end of each test. It was impractical to count live copepods and mysids every day as suggested in the manual since the only reliable method involves removing the animals. This procedure would severely stress the delicate test organisms and could adversely affect the results. Fish were counted daily and dead animals removed. During the particulate phase tests seawater temperature was maintained at 13.5-15.3°C, salinity between 31.0-33.0 ppt, dissolved oxygen between 6.0 and 8.0 ppm, and pH between 7.4 and 7.8 for all species tested.

Solid Phase

The solid phase tests were run for 10 and 20 days. The 10-day tests utilized *A. sculpta* only. The mysids were maintained in 4-liter polycarbonate tanks under static conditions. Five replicates were prepared for each treatment and control condition with 20 organisms in each replicate. As in the particulate phase tests, mysids were gently aerated (3-7 ml/minute) and fed brine shrimp nauplii daily (25-30 nauplii/mysid/day).

Clams (*M. nasuta*) and worms (*N. arenaceodentata*) were held in test sediment for 20 days using flow-through seawater. These organisms fulfilled requirements for both the 20-day survival portion and the bioaccumulation portion of the bioassay. Clams and worms were maintained in separate 16-liter polycarbonate tanks. Five replicates were prepared for each treatment and control condition with 20 clams per replicate and 25 worms per replicate. Clams were not fed. Worms were fed TetraMin SM80 fish food on days 4, 7, 11, 14, and 18. Fish food flakes were ground with a mortar and pestle and combined with seawater to create a slurry that was injected into the water with a large syringe. Each tank received 125 mg of food (5.0 mg/worm) on feeding days. The number of live animals was determined at the end of the test instead of during the test as the manual recommends, because counting during the test disturbs the sediment regime and places additional stress on test animals.

Seawater was delivered to these tanks using a manifold system developed as part of the Navy program to evaluate the toxicity of organotin antifouling compounds (13). Flow rates were approximately 170 ml/minute. All tanks were aerated at a rate between 500 and 1000 ml/minute. For these animals seawater temperature was maintained between 14.0-16.3°C, salinity between 31.0-34.0 ppt, dissolved oxygen between 6.2-8.3 ppm, and pH between 7.6-8.0.

Bioaccumulation

At the end of the bioassay, test sediment was removed by wet-sieving. Clams and worms were counted and immediately returned to their original flow-through test containers. The clams and worms were then fed rations of TetraMin; worms were fed 5.0 mg each and clams 14.0 mg each. These animals were held for a 24-hour depuration period. The animals were fed to encourage elimination of ingested sediment. After the 24-hour depuration period, the animals were removed and tissues frozen for subsequent bioaccumulation estimates.

Tissue and sediment samples were analyzed for cadmium, chromium, copper, mercury, silver, pesticides, PCBs, petroleum hydrocarbons, and organotins. Cadmium, chromium, copper, and silver were analyzed by graphite furnace atomic absorption spectroscopy; mercury by cold vapor atomic absorption spectroscopy. Pesticides, PCBs, and petroleum hydrocarbons were measured by gas chromatography. The manual does not require chemical analysis of sediment. It was analyzed here to document the amount of contaminants available for bioaccumulation in control and treatment sediments and to provide confirmation of control sediment selection. Analytical measurements were conducted by Anatech Laboratories, Santa Rosa, California.

All organotin measurements were made by NOSC personnel. To determine the amount of organotin released from sediment upon mixing, seawater samples were taken from the suspended particulate phase preparation container just before adding the test slurry to treatment tanks. Solid phase water samples were taken from mysid tanks on the first and last days of the bioassay. Butyltin species were measured in these and control water samples using a hydride reduction method followed by hydrogen flame atomic absorption spectroscopy (30).

The total tin concentration in test sediments was measured at the beginning and end of the bioassay. Bulk samples from the collection chests (Control and Commercial Basin) and test samples from solid-phase mysid and clam tanks were taken on the first day of the bioassay. Samples were also taken from solid-phase control, mysid, worm and clam tanks on the last day of the test. The concentration of total solvent-extractable tin was measured by graphite furnace atomic absorption spectroscopy (27).

Additional clams from the 20-day bioaccumulation study were used to determine organotin uptake. Accumulated organotins were extracted from ground clam tissues with methylene chloride. Samples were placed on a rotary shaker for 24 hours to ensure complete extraction. The concentration of total solvent-extractable organotin was determined by graphite furnace atomic absorption spectroscopy. The tin values expressed as µg Sn/g dry tissue were

multiplied by 2.5 to estimate TBTO concentration and then multiplied by 0.20 to estimate a wet-weight concentration comparable to our other bioaccumulation data.

Data Analysis Procedures

Statistical methods outlined in the manual were followed wherever possible. Homogeneity of variances in bioaccumulation data was assessed with the Max-F test. Significant differences were assessed by either the Student's t-test if the variances were homogeneous, or the Mann-Whitney U-test if they were not.

Survival data for control and treatment organisms in the particulate and solid phase tests are discrete data. Therefore, instead of the t-test as recommended by the manual, we used its nonparametric equivalent, the Mann-Whitney U-test (3, 32) to compare treatment and control survival. Significant results for all statistical tests were determined by the critical values ($a = 0.05$) of the appropriate distributions.

RESULTS

The Commercial Basin bioassay was conducted in March 1984. Percent survival for the particulate phase and solid phase tests is presented in Tables 1 and 2. Results of the statistical analyses are presented in Table 3. No statistically significant differences in survival were found between controls and treatments for any of the species tested. The concentrations of contaminants measured in test animal tissues and in test sediments are presented in Table 4. The mean concentration and the results of statistical analyses on these data are included in Table 5. No statistically significant differences in bioaccumulation were found between control and treatment groups except for copper, PCBs, and organotins in *M. nasuta* and copper and silver in *N. arenaceodentata*. The concentrations of organotins in test sediments, test water, and clam tissues are given in Table 6.

Suspended Particulate Phase

In particulate phase tests (Table 1) with fish, control survival was 98 percent and treatment survival was 96 percent. Control and treatment survival for mysids was 100 percent. For copepods, control survival was 88 percent and treatment survival was 82 percent. There were no statistically significant differences in survival between control groups and the Commercial Basin treatment groups for any of the animals tested.

Solid Phase - 10-Day Tests

In 10-day solid phase tests with mysids, control survival was 95 percent and treatment survival was 86 percent. There was no statistically significant difference in survival between controls and treatments.

Solid Phase - 20-Day Tests

In 20-day solid phase tests (Table 2) with clams, both control and treatment survival was 99 percent. In 20-day solid phase tests with worms, control

TABLE 1. THE NUMBER OF FISH (*CITHARICHTHYS STIGMAEUS*), MYSIDS (*ACANTHOMYSIS SCULPTA*), AND COPEPODS (*ACARTIA TONSA*) SURVIVING AFTER 96 HOURS OF EXPOSURE TO 100 PERCENT PARTICULATE PHASE MATERIAL

Species	Replicate Number	Control	Commercial Basin
Citharichthys stigmaeus	1	9	10
	2	10	10
	3	10	10
	4	10	8
	5	10	10
		98%	96%
Acanthomysis sculpta	1	10	10
	2	10	10
	3	10	10
	4	10	10
	5	10	10
		100%	100%
Acartia tonsa	1	6	8
	2	8	8
	3	10	8
	4	10	8
	5	10	9
		88%	82%

TABLE 2. THE NUMBER OF MYSIDS (*ACANTHOMYSIS SCULPTA*), WORMS (*NEANTHES ARENACEODENTATA*), AND CLAMS (*MACOMA NASUTA*) SURVIVING THE SOLID PHASE TESTS. MYSIDS WERE EXPOSED FOR 10 DAYS; CLAMS AND WORMS FOR 20 DAYS

Species	Replicate Number	Control	Commercial Basin
Acanthomysis sculpta (10-day test)	1	18	16
	2	17	19
	3	20	17
	4	20	19
	5	20	15
		95%	86%
Neanthes arenaceodentata (20-day test)	1	23	20
	2	25	21
	3	24	20
	4	15	23
	5	19	25
		84.8%	87.2%
Macoma nasuta (20-day test)	1	20	19
	2	20	20
	3	20	20
	4	20	20
	5	19	20
		99%	99%

TABLE 3. STATISTICAL VALUES FOR DETERMINING SIGNIFICANT MORTALITY IN 100 PERCENT PARTICULATE AND SOLID PHASE TESTS FOR THE COMMERCIAL BASIN BIOASSAY*

Test	Mann-Whitney U-Test Results		
	Calculated	Critical	Decision
100% PARTICULATE PHASE			
Citharichthys stigmaeus	13.0	21.0	NSD
Acanthomysis sculpta	----	21.0	NSD**
Acartia tonsa	15.0	21.0	NSD
SOLID PHASE			
Acanthomysis sculpta	20.5	21.0	NSD
Neanthes arenaceodentata	13.0	21.0	NSD
Macoma nasuta	----	21.0	NSD**

* The Mann Whitney U test was used in all cases. All data were evaluated at the 95 percent confidence level. A statistically significant difference exists for the comparison if the calculated value is greater than the critical.

NSD = No significant difference in mortality.
NSD** = No significant difference in mortality determined by inspection of data.

TABLE 4. THE CONCENTRATIONS (PPM) OF TRACE METALS, PESTICIDES (TICH), PETROLEUM HYDROCARBONS, AND PCBs MEASURED IN THE TISSUES OF CLAMS (*MACOMA NASUTA*) AND WORMS (*NEANTHES ARENACEODENTATA*) AND IN SEAWATER AND SEDIMENTS. DATA ARE EXPRESSED ON WET-WEIGHT BASIS

Sample	Cadmium	Chromium	Copper	Mercury	Silver	PHF	PCBs	TICH
				Macoma nasuta				
C-1	0.150	0.520	2.000	0.050	0.130	<0.100	0.010	<0.0020
C-2	0.150	0.410	1.700	0.060	0.130	<0.100	0.004	<0.0010
C-3	0.120	0.420	2.200	0.030	0.100	<0.100	0.010	<0.0020
C-4	0.130	0.390	2.300	0.020	0.130	<0.100	0.010	<0.0020
C-5	0.140	0.400	1.700	0.040	0.080	<0.150	0.010	<0.0030
BASIN-1	0.120	0.340	2.800	0.060	0.120	<0.100	0.040	<0.0020
BASIN-2	0.090	0.370	4.200	0.060	0.090	<0.100	0.050	<0.0030
BASIN-3	0.130	0.290	2.900	0.040	0.130	<0.200	0.040	<0.0020
BASIN-4	0.120	0.550	4.100	0.050	0.120	<0.100	0.040	<0.0020
BASIN-5	0.070	0.330	2.600	0.060	0.090	<0.100	0.050	<0.0020
				Neanthes arenaceodentata				
C-1	0.030	0.200	8.000	0.100	0.090	<1.500	0.090	<0.0020
C-2	0.020	0.100	6.000	0.100	0.070	<1.400	0.080	<0.0020
C-3	0.040	0.100	7.000	0.100	0.070	<1.400	0.080	<0.0020
C-4	0.050	0.300	5.000	0.200	0.050	<4.000	0.240	<0.0050
C-5	0.030	0.200	6.000	0.200	0.070	<1.800	0.100	<0.0020
BASIN-1	0.040	0.200	17.000	0.100	0.130	<1.800	0.100	<0.0020
BASIN-2	0.030	0.400	20.000	0.100	0.140	<2.000	0.130	<0.0030
BASIN-3	0.030	0.200	10.000	0.100	0.080	<2.000	0.100	<0.0020
BASIN-4	0.030	0.200	18.000	0.100	0.110	<1.600	0.090	<0.0020
BASIN-5	0.030	0.200	18.000	0.100	0.120	<1.700	0.100	<0.0020
				Seawater and Sediment Samples				
WATER	0.001	0.011	0.002	0.0005	0.002	<0.004	0.0002	<0.00005
CONTROL	0.700	15.000	16.000	0.0980	0.800	<0.100	0.006	<0.0010
BASIN	0.900	26.000	210.000	2.7000	0.800	<0.100	0.025	<0.0010

C = Control.
PHF = Petroleum hydrocarbon fraction.
PCB = Polychlorinated biphenyls quantitated as Aroclor 1254.
TICH = Total identifiable chlorinated hydrocarbon pesticides.

TABLE 5. MEAN CONTAMINANT CONCENTRATIONS (PPM) AND STATISTICAL VALUES USED FOR DETERMINING BIOACCUMULATION POTENTIAL IN THE COMMERCIAL BASIN SEDIMENT

Contaminant	Control	Test	Common	Calc	Crit	Decision
			Macoma nasuta			
Cadmium	0.138	0.106	YES	T:2.53	2.306	SD**
Chromium	0.428	0.376	YES	T:1.02	2.306	NSD
Copper	1.980	3.320	YES	T:3.68	2.306	SD
Mercury	0.040	0.054	YES	T:1.72	2.306	NSD
Silver	0.114	0.110	YES	T:0.30	2.306	NSD
Pesticides	0.002	0.002	YES	T:0.53	2.306	NSD
PCB's	0.008	0.044	YES	T:12.90	2.306	SD
Pet. Hydro.	0.110	0.120	YES	T:0.45	2.306	NSD
			Neanthes arenaceodentata			
Cadmium	0.034	0.032	YES	T:0.37	2.306	NSD
Chromium	0.180	0.240	YES	T:1.10	2.306	NSD
Copper	6.400	16.000	NO	U:25.00	21.000	SD
Mercury	0.140	0.100	NO	U:17.50	21.000	NSD
Silver	0.070	0.116	YES	T:3.81	2.306	SD
Pesticides	0.026	0.022	YES	T:0.63	2.306	NSD
PCBs	0.118	0.104	NO	U:17.00	21.000	NSD
Pet. Hydro.	2.020	1.820	NO	U:17.50	21.000	NSD

NSD = No significant difference.
NSD* = No significant difference determined by inspection of data.
SD = Significant difference.
SD** = Significantly lower concentration in test sample.
T = Students t-test.
U = Mann Whitney U-test.

TABLE 6. CONCENTRATION (PPB) OF MONOBUTYLTIN (MBTO), DIBUTYLTIN (DBTO), AND TRIBUTYLTIN (TBTO) MEASURED IN THE SEDIMENTS AND SEAWATER FROM THE COMMERCIAL BASIN BIOASSAY*

	Initial			Final		
	MBTO	DBTO	TBTO	MBTO	DBTO	TBTO
Seawater						
Control seawater	ND	ND	ND	ND	ND	ND
Particulate phase seawater	0.10	0.30	0.49	NS	NS	ND
Solid phase mysid tank water	0.67	2.00	0.20	0.03	0.05	0.25
Sediments						
Control			62.50			72.50
Bulk Commercial Basin			780.00			NS
Solid phase - mysid tanks			318.00			155.00
Solid phase - clam tanks			610.00			267.50
Solid phase - worm tank			NS			545.00
Clam Tissues						
Control			0.26			
Commercial Basin			2.82			

Statistical Analysis of TBTO in Clam Tissue

Contaminant	Control	Test	Common	Calc	Crit	Decision
TBTO	0.26	2.82	NO	U:16	U:15	SD

* TBTO was measured in *Macoma nasuta* tissues and estimates of bioaccumulation made on a wet weight basis of μg Sn/g tissue.

NS = Not sampled.
ND = Not detectable.
SD = Significant different.

survival was 84.8 percent and treatment survival was 87.2 percent. There was no statistically significant difference in survival between controls and treatments for either species.

Bioaccumulation

There was a significant accumulation of copper and silver in the tissues of *N. arenaceodentata* and a significant accumulation of copper, PCBs, and organotin in the tissues of *M. nasuta*. There was no significant accumulation of other measured contaminants in the tissues of *N. arenaceodentata* or *M. nasuta* when controls were compared with treatments for the Commercial Basin bioassay. The significance of the silver accumulation is questionable as the amounts determined in control and treatment sediments were similar (0.800 ppm).

The total amount of organotins accumulated in clam tissues is shown in Table 6. Treatments accumulated significantly more tin than controls. Control clam tissues were measured to contain 0.26 ppm organotin as TBTO. Clams exposed to the Commercial Basin sediment accumulated 2.82 ppm organotin as TBTO. These values were determined on a wet-weight basis and demonstrate that organotins associated with sediments are bioavailable.

Organotin Measurements

The concentration of butyltin species in particulate phase water was measured as: 0.10 ppb monobutyltin, 0.30 ppb dibutyltin, and 0.49 ppb tributyltin. In solid phase water from mysid tanks, the initial concentration of tributyltin was 0.20 ppb, about half that of particulate phase water. However, both monobutyltins and dibutyltins were significantly higher at 0.67 and 2.00 ppb, respectively. At the end of the solid phase mysid test, water from these tanks was measured at 0.03 ppb monobutyltin, 0.05 ppb dibutyltin, and 0.25 ppb tributyltin (Table 6). Organotins were not detectable in the nearshore seawater obtained through the NOSC seawater system.

The concentration of solvent extractable tin (as TBTO) in control sediment was measured to be between 62.50 and 72.50 ppb. The bulk samples of Commercial Basin sediment (from collection chests) was measured to contain 780.0 ppb tin as TBTO. The concentration of tin in Commercial Basin sediment from test tanks was measured to be between 318.0 and 610.0 ppb as TBTO. This confirms that the amount of tin in the treatment sediment was relatively high and in the control sediment relatively low. The amount of tin remaining in the sediment from the solid phase mysid, clam, and worm tanks at the end of the bioassay was 155.0, 267.5, and 545.0 ppb as TBTO, respectively (Table 6). This decrease in TBTO concentration demonstrates that tins had leached off the treatment sediment.

The composition of control sediment is much different than Commercial Basin sediment. In previous measurements of sediments from another control site near the entrance to San Diego Bay, tributyl-tins were below the levels of detection (27). This suggests that most of the tin found in our control sediment is not TBTO, but other organically bound tins. Most of the tin in the Commercial Basin sediment is probably tributylin-tin and its degradation products. It has also been shown that sediment from the Commercial Basin

collection site contains 77 percent tributyltin (27). To simplify the discussion, the concentration of organically extractable tin multiplied by 2.5 will be used as a relative TBTO concentration.

Sediment Characteristics

The mean grain size of the control sediment was 0.0981 mm. The composition was 81.0 percent sand with 10.6 percent silt and 8.4 percent clay. This was classified as fine sand. The mean grain size of Commercial Basin sediment was 0.0302 mm with a composition of 34.3 percent sand, 5.14 percent silt, and 14.3 percent clay. This sediment is classified as sandy-silt (31).

DISCUSSION

A standard dredged material bioassay was conducted on sediment contaminated with organotins collected from Commercial Basin in San Diego Bay. The purpose of this study was to assess the toxicity and bioavailability of organotins associated with sediment and to determine if this organotin-contaminated sediment would qualify for ocean disposal. On the basis of organotin field measurements of Commercial Basin sediment and water, and concentrations known to cause effects on sensitive planktonic organisms in the laboratory, we anticipated this would be the first sediment from San Diego Bay to produce significant mortalities from a standard sediment bioassay. As part of a Navy program to study the fate and effects of organotins in the marine environment, sediments were measured in a number of harbors. Measurements in San Diego Bay showed that Commercial Basin sediment and water had the highest concentrations of organotins. Sediment values ranged from 32 to 560 ppb TBTO. Surface water samples from Commercial Basin were measured between 0.01 and 0.18 ppb TBTO while bottom water samples varied from 0.11 to 0.55 ppb TBTO (30).

A static renewal toxicity test using *A. tonsa* has estimated a 96-hour LC-50 of 1.0 ppb TBTO. In this same study a 144-hour effective concentration, or that concentration causing some effect on 50 percent of the animals after 144 hours, was calculated to be 0.4 ppb TBTO (29). Our previous work on the mysid *Metamysidopsis elongata* has estimated a 96-hour LC-50 between 0.5 and 0.9 ppb TBTO for juvenile animals. For adults the LC-50 was estimated to be about 3.0 ppb TBTO (27). In addition, it was determined that equivalent tributyltin oxide concentrations prepared from stock chemical solutions and leachates from antifouling coatings had the same effect on mysids (20). Recent work at NOSC has estimated a 96-hour LC-50 of 0.6 ppb TBTO for juvenile *A. sculpta* mysids and a 96-hour LC-50 of 1.7 ppb TBTO for adult *A. sculpta* mysids. All of these factors led to the selection of Commercial Basin sediment as an example of organotin contamination that might cause an effect on sensitive bioassay animals. A standard dredged material bioassay was used to predict the effects of organotins on bioassays required for future dredging permits.

The measured concentration of TBTO in Commercial Basin sediment from treatment tanks varied from 318 to 610 ppb extractable butyltin and confirmed that we had collected contaminated sediment. These concentrations are lower than concentrations measured for the bulk sediment from the collection chests (780 ppb TBTO) because treatment sediment was actually collected from test tanks and may have included some control sediment from the bottom layer of the

tank. The concentration of TBTO in treatment sediment is approximately five to ten times higher than in our control sediment. These high TBTO values are similar to our previous measurements of Commercial Basin sediment.

Measurements of the water and sediment in solid phase and particulate phase test tanks indicated that organotins were leached from the test sediment. Although organotins were not detectable in control seawater, initial TBTO concentrations in mysid tanks measured on the first day of the test varied from 0.20 ppb in solid phase water to 0.49 ppb in particulate phase water. The concentration of TBTO in test sediment decreased concomitantly during the test. These measurements confirmed that TBTO from the Commercial Basin sediment was being released into test water. The speciated butyltin measurements demonstrate that the more soluble and less toxic monobutyltin and dibutyltin compounds were also released from the sediment.

Fish, clams, and worms all demonstrated high survival after continuous exposure to TBTO. These results were predictable based on their tolerance to other toxicants and results from previous bioassays. Due to their high sensitivity to organotins, significant mortalities in mysids and copepods exposed to organotin-contaminated sediments were expected, but not observed. Control and treatment mysid survival was 100 percent in the 96-hour particulate phase tests, where TBTO concentrations were initially measured at 0.49 ppb. This was not expected. The estimated 96-hour LC-50 for mysids is between 0.6 and 3.0 ppb TBTO and for copepods approximately 1.0 ppb TBTO. It is quite possible that the initial concentration of TBTO decreased during the 96-hour test. However, mysids and copepods were exposed to concentrations of TBTO that approximated those which have previously produced significant effects. In the 10-day solid phase test with mysids, treatment survival was 86 percent and not significantly different than control survival of 95 percent. The concentration of TBTO varied between 0.20 and 0.25 ppb during the 10-day test. Copepods also did not exhibit an effect after TBTO exposure. Treatment survival was 82 percent for animals exposed to 0.49 ppb TBTO while control survival was 88 percent.

Considering the rationale for replacing chemical analyses with bioassays for the regulation of dredged material disposal, perhaps these results should have been expected. The Corps of Engineers Dredged Material Research Program has shown that bulk metal analyses of test sediments do not correlate with metal bioavailability (16). It has further been suggested that the magnitude of bioaccumulation does not always reflect sediment DDT concentration (15). Bioassay results give a normalized estimate for evaluating the potential for environmental impact after ocean disposal. Further, bioassays eliminate selecting which contaminants should be measured and which should be eliminated. This is particularly important when there is a chance of eliminating the one contaminant which could be the most significant. The bioavailability of contaminants in sediments is highly variable and dependent on a variety of factors. These include the physical properties of the sediments, the chemical state of the contaminants, and the chemical and biological parameters of the water (9).

The bioavailability of contaminants and their actual toxicity in seawater is also highly variable. Although a minimum concentration of certain trace metals is important for phytoplankton growth (8), the form of the chemical state of those metals determines whether or not they are usable by the

organisms (10). Studies on some species of phytoplankton show that growth inhibition and copper uptake are related to cupric ion activity and not to total copper concentration (28). Along with others, we have shown that toxicity to species used in this bioassay is also related to the state of the cupric ion being tested (20). In experiments with mysids and brine shrimp nauplii, tributyltin was far more toxic than its degradation products, dibutyltin and monobutyltin (27). The addition of sediment to test tanks reduces the toxicity of copper to *Neanthes arenaceodentata* (18) while the type of sediment influences copper toxicity and copper bioaccumulation (17).

It follows that the presence of sediment in toxicity tests will probably reduce toxicity and bioavailability of certain toxicants for a variety of animals. Since most organotin toxicity tests have been conducted without sediment, organotin toxicity has probably been overestimated. Toxic concentrations reported from those experiments are real but the abundance of naturally occurring organics in those tests was probably abnormally low. Thus, standard toxicity tests in glass aquaria without sediment do not realistically represent natural conditions. The addition of sediment, particulates, or other organic compounds provides a much better approximation of natural conditions in the marine environment. The expected toxicity of organotin-contaminated sediment for Commercial Basin was not realized because the organotins were probably not in a form readily available to the test animals.

It has been shown that organic chelators are often the most crucial factor in bioavailability of nutrients for phytoplankton growth in seawater (1, 10). It has also been suggested that, except for copper and iron, complexing agents have a relatively minor effect on initial complexation in seawater and that organic compounds adsorbed on the surface of sediment particles control the adsorption of metals (14). Although the actual chemical mechanism is unclear, some organics found in natural sediments can reduce toxicity and bioavailability. This study suggests that organics in Commercial Basin sediment reduce toxicity and bioavailability of organotins in that sediment.

In addition to naturally occurring organic chelators in sediments that reduce toxicity and bioavailability, animals also have the ability to sequester and detoxify contaminants. Since previous measurements of Commercial Basin sediment have shown elevated levels of copper and organotin, it is not surprising that *M. nasuta* bioaccumulated significant amounts of each of these contaminants. The environmental significance of the bioaccumulation estimate is unclear. Treatment clams were found to accumulate organotins to a concentration an order of magnitude above control clams (2.82 ppm TBTO compared to 0.26 ppm TBTO), and a factor of four above Commercial Basin sediment. Since the animals survived the experiment, these internal concentrations are apparently sublethal. Accumulated tins were probably modified by the animals to become biologically unavailable. Accumulated tins could have combined with lipids to reduce toxicity. They could have been bound to metallothioneins (19) or sequestered within cellular organelles (7). In each case, there is a reduced opportunity for toxic expression.

Thus, when estimating the potential toxicity of organotin antifoulant leachates from Navy hulls to marine organisms, more effort should be directed toward duplicating natural conditions. This will prevent overestimating toxicity values and reduce the significance of questionable values currently

appearing in the literature. For our purposes, however, organotin-contaminated sediments from Commercial Basin, San Diego Bay, would qualify for ocean disposal under the present guidelines. Questions could be raised about the significant uptake of copper, the potential accumulation of organotins, or even the validity of the bioassay itself. With all its shortcomings, the bioassay as conducted here is probably the best available estimator of environmental impact after ocean disposal because it accounts for bioavailability. Changes could be made in species selection, test containers, test duration, etc.; but as a regulatory tool, it is probably all that could be expected at this time.

The other problem is that new contaminants like organotins are not well known. As they are studied further, new insight should be gained into improving the required ecological evaluation of dredged material. The Navy research programs to expedite dredging and study the fate and effects of organotins in the marine environment will help guide the assessment of the potential for environmental impact. However, as the use of organotins increases, bioassays should include an assessment of the potential for bioaccumulation of these compounds. Further, much additional work will be required to fully understand the chemistry and bioavailability of organotins and to predict their effect on the marine environment.

CONCLUSIONS

In conclusion, the results of this bioassay suggest that the sediment from Commercial Basin, San Diego Bay, should not have a significant impact on the marine environment if discharged into ocean waters. These results also suggest that organotins found in sediments at relatively high levels are not necessarily toxic to marine life. It is probable that the organotins present were not as bioavailable as originally anticipated and that the approach in testing organotin toxicity should be re-evaluated in terms of environmental significance as well as adsorption on sediment and detoxification by animals.

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CHARACTERIZATION OF SEDIMENT CONTAMINATION UTILIZING BIOASSAY/BIOACCUMULATION TESTING

J. M. Mansky

U.S. Army Corps of Engineers
New York District
26 Federal Plaza
New York, New York 10278-0090

ABSTRACT

The Ocean Dumping Criteria specified biological tests of dredged material to evaluate short-term toxicity of ocean disposal operations (bioassay tests) and the potential for long-term chronic effects through the bioaccumulation of harmful contaminants. A statistical analysis of 150 three-phase bioassay/bioaccumulation tests conducted on sediment from New York Harbor indicated that only a few dredging projects consisted of sediment that would be of ecological concern if disposed in ocean waters. Liquid and suspended particulate phase bioassays showed that disposal of the dredged material would result in insignificant impact to the water column. Based on the solid phase bioassays, 4 percent of the dredging projects indicated environmentally significant mortality potential to benthic life.

Bioaccumulation analyses for petroleum hydrocarbons (PHC), polychlorinated biphenyls (PCBs), DDT, cadmium, and mercury were conducted on organisms surviving the 10-day solid phase bioassay tests. Bioaccumulation of statistically significant levels of PHC was found to be common for all three organisms tested. PCB bioaccumulation was more common in the marine worm (*Nereis virens*). DDT was never found to be bioaccumulated in statistically significant amounts. All three organisms exhibited a very low bioaccumulation potential for cadmium and mercury. Some predictions were possible for the bioaccumulation of PHC and PCB within the same organism and for the same contaminant between organisms; predictions were not possible for cadmium or mercury.

INTRODUCTION

The continued viability of the Port of New York and New Jersey depends upon the dredging of an average of 7.6 million cubic meters of sediment each year. The urbanized nature of the metropolitan area precludes the upland disposal of large quantities of dredged material. Disposal of over 90 percent of the sediment dredged from the Port has taken place at the "Mud Dump Site" located approximately 11 km east of New Jersey and 20 km south of New York (Figure 1).



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

Two previous papers (5,6), which were presented at the Sixth and Seventh Joint U.S./Japan Experts Meeting on the disposal of toxic sediments, explained the background of the sediment contamination problem in the New York Harbor area. Due to the sediment contamination, disposal of dredged material has been of environmental concern since the late 1960's.

The Marine Protection, Research, and Sanctuaries Act of 1972 (Public Law 92-532), more commonly known as the "Ocean Dumping Act," regulates all types of ocean dumping, including dredged material. In accordance with the Ocean Dumping Act, the US Environmental Protection Agency (EPA) in 1977 promulgated regulations and criteria (2) to determine the potential environmental impact of the disposal activity. The Ocean Dumping Criteria (40 CFR Parts 220 through 229) specified biological tests to consider short-term

toxicity of dumping operations (bioassay tests) and the potential for long-term chronic effects through the bioaccumulation of harmful contaminants. A previous paper (1), presented at the Sixth Joint U.S./Japan Experts Meeting on the Disposal of Toxic Sediments, provides additional information on legal requirements of proposed operations involving the transportation of dredged material for dumping into ocean waters.

Since the inception of the requirement of three-phase bioassay/bioaccumulation testing in 1979, approximately 30 tests per year have been conducted on sediment from New York Harbor, which is proposed for ocean disposal. This paper presents a statistical analysis of the 150 tests reviewed by the New York District Corps of Engineers.

BIOASSAY ANALYSIS

Introduction

In accordance with EPA's Ocean Dumping Regulations (2), liquid phase, suspended particulate phase, and solid phase bioassays are required for the proposed dredged material utilizing the test species as discussed below.

The liquid phase is considered to be the centrifuged and 0.45-micron filtered supernatant remaining after 1 hr of undisturbed settling of the mixture resulting from a vigorous 30-min agitation of one part bottom sediment from the dredging site with four parts water collected from either the dredging site or the disposal site, as appropriate for the type of dredging operation. The suspended particulate phase is the supernatant obtained prior to centrifugation and filtration, while the solid phase is considered to be all material settling to the bottom within 1 hr.

As pointed out in the Implementation Manual for the Ecological Evaluation of Proposed Discharge of Dredged Material into Ocean Water (1), "...the liquid phase may be analyzed in either of two ways, as specified in paragraph 227.13 (13)(c)(2) of The Register....The liquid phase may be analyzed chemically and the results evaluated by comparison to water quality criteria...(or) if the water quality approach is not taken, the liquid phase must be evaluated by bioassays...."

The New York District Corps of Engineers has adopted the direct bioassay approach as being more appropriate since the liquid phase may contain major constituents not included in the water quality criteria and since there is reason to be concerned about possible synergistic effects. The liquid phase bioassay is appropriate in evaluating the total net impact of dissolved chemical constituents released from the sediment during disposal operations.

For the liquid and suspended particulate phase bioassays, lethal concentrations (LC50) and effective concentrations (EC50) were used. LC50's are those concentrations of liquid or suspended particulate phase resulting in 50 percent mortality; they were determined for the animal species *Menidia menidia*, *Mysidopsis bahia*, and *Acartia tonsa*. EC50 is that concentration of liquid phase resulting in 50 percent inhibition; it was determined for the phytoplankton species *Skeletonema costatum*. Liquid and suspended particulate phase bioassays are conducted over a 96-hr (4-day) period.

The solid phase bioassay is a 10-day exposure which is run concurrently on three sediments: test, reference, and control. The reference sediment represents existing background conditions in the vicinity of the dump site but away from the influence of any disposal operation, while the control sediment represents a non-toxic substrate utilized to monitor testing conditions while the solid phase bioassay is being conducted. Results of the solid phase bioassay are utilized to predict the impact of the dredged material on benthic organisms.

Analysis

Based upon the LC50 (or EC50) for the most sensitive species for the liquid or suspended particulate phase bioassays, the LPC (Limiting Permissible Concentration) values are determined by applying a safety factor of 100 (LC50 or EC50/100). The sediment can be authorized for ocean disposal only if the concentration of the disposal water in both the liquid and suspended particulate phases is diluted to below that of the LPC after 4 hr of initial mixing.

The mixing and dilution of the proposed dredged material is evaluated by use of the Tetra Tech computer model. A description of the model has been published as Technical Report D-78-47 of the Corps' Dredged Material Research Program, and is entitled "Evaluation and Calibration of the Tetra Tech Dredged Material Disposal Models Based on Field Data" (4). A study was conducted giving examples of dilutions resulting from typical disposal operations using dredged materials found in the New York District; this study is entitled "Discussion of Dredged Material Disposal Models and Their Application for the New York District" (3). Using the example in this study that most closely approximates the disposal operations conducted at the Mud Dump Site, the dilution after 4 hr would be 0.0047 percent. The Tetra Tech Model uses the total initial volume of dredged material and, therefore, the resultant concentration serves for both the liquid phase and the suspended particulate phase.

For the solid phase bioassay, the results are evaluated for biological and statistically significant differences in mortality between test organisms which are subjected to both proposed dredged sediment and to a reference sediment.

The Implementation Manual specifies that differences in survival of the test organisms in the reference and test treatments be evaluated either individually by species or on a pooled community basis. The Manual states that the LPC for the solid phase is exceeded when the difference in mortality between animals in the reference and test sediments is statistically significant and greater than 10 percent.

Results

Figure 2 presents the results of liquid phase bioassays, while Figure 3 presents the results of the suspended particulate phase bioassays. Results are expressed as cumulative percentages plotted on probability paper. It should be noted, as indicated above, that LC50 values of 0.47 percent or less would result in the limiting permissible concentration being exceeded, and therefore unacceptable water column impacts are anticipated. As indicated by these two figures, the LPC has never been exceeded in either the liquid or suspended particulate phase bioassays. In order to determine the relationship

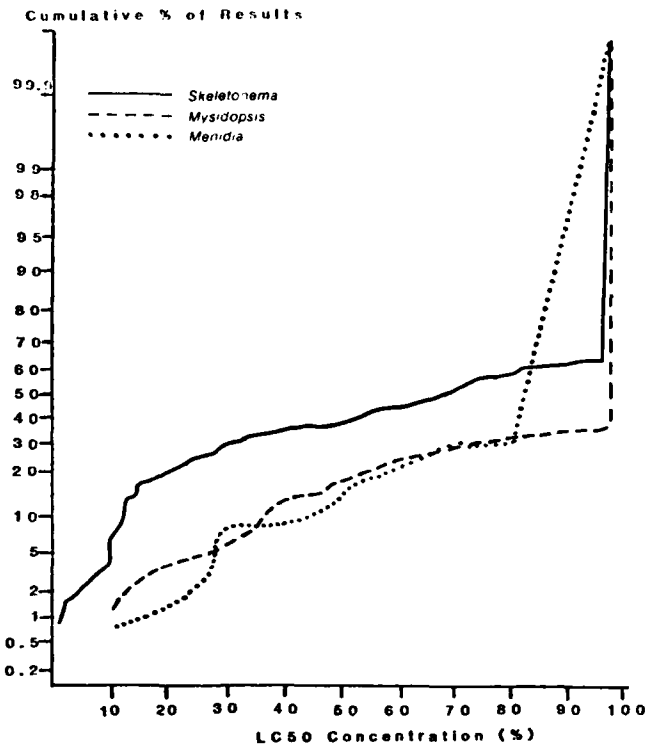


Figure 2. Results of liquid phase bioassay testing expressed as cumulative percentage

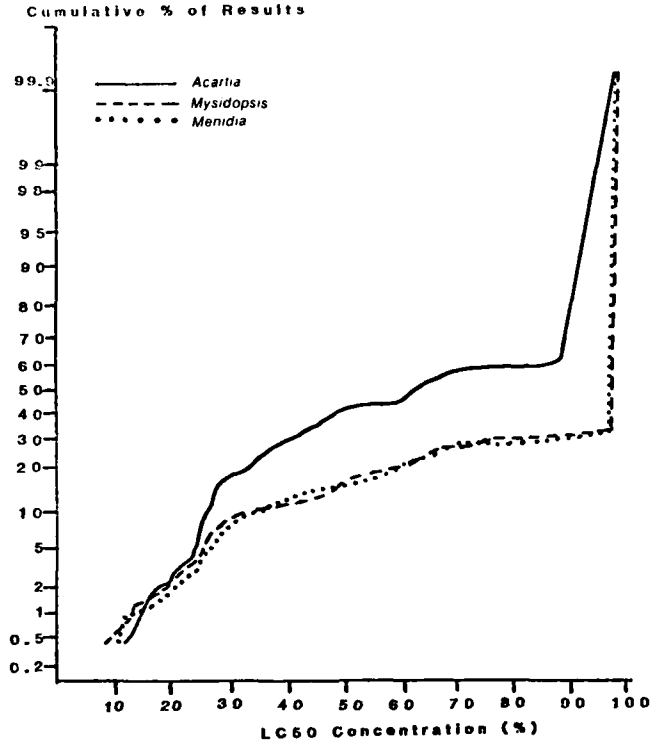


Figure 3. Results of suspended particulate phase bioassay testing expressed as cumulative percentage

between the responses of the organisms in each of the two phases and between organisms in the same phase, the correlation coefficient was calculated. It was determined that the highest correlation coefficient (r = 0.53) was found when comparing the response of *Mysidopsis* in the liquid phase bioassay with *Mysidopsis* in the suspended particulate phase. An r value of 0.53 indicates that there is a moderate positive correlation for this comparison. For all other comparisons, the correlation value was less than 0.30 and it can therefore be stated that there is no relationship between the responses of the organisms in the same phase bioassay or between the same organism in the liquid and suspended particulate phase bioassay.

Species used in the solid phase bioassay are *Palaemonetes* sp. (grass shrimp), *Mercenaria mercenaria* (hard clam), and *Nereis* sp. (polychaete worm). Results of the solid phase bioassay are presented in Table 1. It can be observed that *Palaemonetes* showed the most occurrences (9.2 percent) in which the test had statistically significant greater mortality than the reference. This is followed by 5.9 percent for *Nereis* and 2.0 percent statistically significant for *Mercenaria*. When considering survival differences, which are of ecological significance (i.e. greater than 10 percent), only 3.9 percent of the projects tested had mortality in a test organism that was more than 10 percent higher than the reference.

TABLE 1. SOLID PHASE BIOASSAY RESULTS EXPRESSED AS A PERCENTAGE IN WHICH TEST RESULTS WERE GREATER THAN REFERENCE RESULTS (STATISTICALLY SIGNIFICANT (SS) AT THE 95 PERCENT CONFIDENCE LEVEL)

Species	SS > 10 percent	SS < 10 percent	Not SS
Palaemonetes	2.6	6.6	90.8
Mercenaria	0.0	2.0	98.0
Nereis	1.3	4.6	94.1

BIOACCUMULATION ANALYSIS

Introduction

EPA's Ocean Dumping Regulations (2) require that the potential for bioaccumulation of contaminants from dredged material be evaluated. Field sampling programs are preferred since the animals are exposed over the long term to the conditions of mixing and sediment transport actually occurring at the disposal site. The limitation of field sampling is that it is technically valid only where there exists an historical record for the proposed operation being evaluated. Since this historical information does not exist for the Mud Dump Site, a laboratory assessment of bioaccumulation potential is conducted for New York Harbor sediments that are proposed for ocean disposal.

At the end of the solid phase test, surviving organisms are placed in separate aquaria in clean, sediment-free water to void the digestive tracts of all sediment. Where appropriate, the shell or exoskeleton is removed before chemical analysis is conducted.

Analysis

The London Dumping Convention (7) requires an evaluation of bioaccumulation for sediment which may contain contaminants that cannot be ocean disposed in other than trace amounts. The evaluation is conducted to determine if there will be significant undesirable effects on human health.

For sediment from New York Harbor that is proposed for ocean disposal, bioaccumulation analysis is conducted for total petroleum hydrocarbons (PHC), polychlorinated biphenyls (PCB), DDT, mercury (Hg), and cadmium (Cd). Results of the test phase of the bioaccumulation analysis are statistically compared at the 95 percent confidence level with the results of the reference phase analysis. (Tissue from the control phase is frozen for future analysis, should the need be demonstrated.)

Results

Table 2 presents information on the mean, standard deviation, and maximum values observed for PHC, PCB, Hg, and Cd. Although DDT has been analyzed in each of the 150 tests reported in this paper, it has never been found above the detection limit of 0.02 ppm. Comparing the data, it can be observed that there is a tendency for bioaccumulation to be highest in *Nereis*, with

TABLE 2. BIOACCUMULATION RESULTS EXPRESSED IN PARTS PER MILLION

Variable	PHC	PCB	Hg	Cd
		Palaemonetes		
Mean	6.26	0.06	0.22	0.29
Standard deviation	16.57	0.06	0.10	0.14
Maximum	121.81	0.29	1.30	1.76
		Mercenaria		
Mean	4.84	0.05	0.24	0.28
Standard deviation	17.78	0.02	0.25	0.07
Maximum	184.40	0.19	2.30	0.67
		Nereis		
Mean	7.13	0.09	0.26	0.27
Standard deviation	11.53	0.09	0.24	0.06
Maximum	55.87	0.62	2.90	0.78

bioaccumulation in *Palaemonetes* and *Mercenaria* equivalent for PCB, Cd, and Hg. The low PHC bioaccumulation value for *Mercenaria* may be due to a proportionately lower quantity of body lipid present within the tissue, which is analyzed and reported as PHC bioaccumulation.

Table 3 presents the percentage of times that bioaccumulation of a contaminant in each organism is statistically significant at the 95 percent confidence interval, when compared to bioaccumulation from a reference sediment. Of the contaminants evaluated, PHC has the highest occurrence of statistically significant bioaccumulation. Statistically significant PCB bioaccumulation in *Nereis* is far more common than in the other two species. Bioaccumulation of Cd and Hg in statistically significant quantities is a rare occurrence found in only about 2 percent of the cases analyzed.

TABLE 3. BIOACCUMULATION RESULTS EXPRESSED AS PERCENTAGE IN WHICH TEST RESULTS WERE GREATER THAN REFERENCE RESULTS (STATISTICALLY SIGNIFICANT AT 95 PERCENT CONFIDENCE LEVEL)

Contaminant	*Palaemonetes*	*Mercenaria*	*Nereis*
PHC	25.0	44.1	55.9
PCB	6.6	9.2	25.0
DDT	0.0	0.0	0.0
Cd	2.0	3.3	2.0
Hg	2.0	2.0	0.7

Interpretation of statistically significant quantities of these contaminants was initially a source of technical debate. Once the management of dredged material disposal to prevent unreasonable degradation of existing environmental conditions was accepted as a short-term goal, an interpretive guidance was developed. The details of the development and implementation of the interpretive guidance (matrix) for PCB are presented in Engler et al. (1). Later, guidance values for DDT, mercury, and cadmium were developed, and are implemented utilizing the same approach as for the PCB matrix. To date, no matrix has been developed for PHC bioaccumulation. The matrix, as currently utilized for New York Harbor sediment, is presented in Table 4. The percentage of times in which test results are statistically significant and above the matrix value is presented in Table 5. In only 4 percent or less of the cases was the bioaccumulation value statistically significant and greater than the matrix. *Nereis* exceeded the matrices the least number of times.

TABLE 4. INTERPRETIVE GUIDANCE (MATRIX) VALUES EXPRESSED IN PARTS PER MILLION

Species	PHC	PCB	DDT	Cd	Hg
Palaemonetes	--	0.1	0.04	0.3	0.2
Mercenaria	--	0.1	0.04	0.3	0.2
Nereis	--	0.4	0.04	0.3	0.2

TABLE 5. BIOACCUMULATION RESULTS EXPRESSED AS A PERCENTAGE IN WHICH TEST RESULTS WERE GREATER THAN REFERENCE (STATISTICALLY SIGNIFICANT AT 95 PERCENT CONFIDENCE LEVEL) AND ABOVE MATRIX VALUE

Species	PHC	PCB	DDT	Cd	Hg
Palaemonetes	--	3.9	0.0	2.0	2.0
Mercenaria	--	3.9	0.0	3.3	2.0
Nereis	--	0.7	0.0	2.0	0.7

Analysis was conducted to determine if bioaccumulation of a contaminant in one organism could be used to predict bioaccumulation of that contaminant in one of the other two organisms. For PHC (Table 6), the highest occurrence (35.5 percent) of statistically significant bioaccumulation for the same sediment was found in comparing *Mercenaria* and *Nereis*. The smallest value

TABLE 6. PHC BIOACCUMULATION OCCURRENCE OF STATISTICAL SIGNIFICANCE (SS) IN TWO SPECIES

Species	Percent SS
Palaemonetes and *Nereis*	21.17
Palaemonetes and *Mercenaria*	20.4
Mercenaria and *Nereis*	35.5

(20.4 percent) was *Palaemonetes* and *Mercenaria*. Table 7 presents information on the occurrence of PHC bioaccumulation when comparing two species. Although *Palaemonetes* had the lowest number of statistically significant bioaccumulation cases, it can be observed that when it did occur, it was very likely to

TABLE 7. PERCENT OCCURRENCE OF PHC BIOACCUMULATION MATCHING TWO SPECIES

Species	Number SS Occurrences	Percent Occurrences		
		PAL	MER	NER
Palaemonetes (PAL)	38	--	82	87
Mercenaria (MER)	66	46	--	80
Nereis (NER)	84	39	64	--

occur in *Mercenaria* (82 percent) and *Nereis* (87 percent) as well. The occurrence of statistically significant bioaccumulation of PHC in *Mercenaria* could also be used to predict PHC at a high level (80 percent) in *Nereis*.

Mercenaria and *Nereis* were the two species with the highest (5.3 percent) simultaneous occurrence of statistically significant PCB bioaccumulation (Table 8). The values presented in Table 8 are considerably lower than the corresponding values for PHC found in Table 6. When comparing the occurrence of PCB bioaccumulation in two species (Table 9), the highest values are found for *Palaemonetes* compared to *Mercenaria* (70 percent) and *Nereis* compared to *Palaemonetes* (70 percent).

TABLE 8. PCB BIOACCUMULATION - OCCURRENCE OF STATISTICAL SIGNIFICANCE (SS) IN TWO SPECIES

Species	Percent SS
Palaemonetes and *Nereis*	4.0
Palaemonetes and *Mercenaria*	4.6
Mercenaria and *Nereis*	5.3

TABLE 9. PERCENT OCCURRENCE OF PCB BIOACCUMULATION MATCHING TWO SPECIES

Species	Number SS Occurrences	Percent SS Occurrences		
		PAL	MER	NER
Palaemonetes (PAL)	10	--	70	16
Mercenaria (MER)	14	50	--	21
Nereis (NER)	38	70	58	--

Statistically significant bioaccumulation occurrences for all organisms were so rare for cadmium and mercury that no comparison between species was appropriate. In only two cases out of nine did two species bioaccumulate statistically significant cadmium from the same sediment. For only one case out of six was mercury of statistical significance bioaccumulated in two species from the same sediment.

An analysis was conducted of bioaccumulation of organic contaminants (PHC and PCB) within the same species for the same sediment. The only comparison which indicated a possibility for prediction was for PHC, based upon PCB results (Table 10). Where PCB of statistical significance was bioaccumulated in *Palaemonetes*, 70 percent of the cases also had statistically significant bioaccumulation for PHC.

TABLE 10. BIOACCUMULATION OCCURRENCE OF STATISTICALLY SIGNIFICANT RESULTS FOR PHC BASED UPON PCB RESULTS

Species	Number Occurrences	Percent Occurrences
Palaemonetes	7	70
Mercenaria	7	50
Nereis	3	8

For *Mercenaria*, the comparison value decreased to 50 percent. The occurrence of cadmium and mercury bioaccumulation was so low that a statistical comparison was not valid.

CONCLUSIONS

A statistical analysis of 150 three-phase bioassay/bioaccumulation tests conducted on sediment from New York Harbor indicated that only a few dredging projects consisted of sediment of ecological concern if disposed in ocean waters. Liquid and suspended particulate phase bioassays showed that for the dredged material analyzed, disposal would result in an insignificant impact to the water column. Solid phase bioassays showed that less than 4 percent of the dredging projects indicated environmentally significant mortality potential to benthic life.

Bioaccumulation of statistically significant levels of PHC was found to be common for all three species. PCB bioaccumulation was most common in *Nereis*. All three species exhibited a very low bioaccumulation potential for DDT, cadmium, and mercury. Some prediction was found to be possible for finding PHC or PCB bioaccumulation in two species for the same sediment. In the comparison of finding two of the analyzed contaminants in the same species for the same sediment, only PHC compared to PCB bioaccumulation showed any prediction possibility.

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IDENTIFICATION OF PROMISING CONCEPTS FOR TREATMENT OF CONTAMINATED SEDIMENTS

N. R. Francingues, Jr.

Environmental Laboratory
US Army Engineer Waterways Experiment Station
Vicksburg, Mississippi 39180

ABSTRACT

Because of the nature and extent of contamination associated with sediments along our highly developed waterways and harbors, many of the sediments being dredged in maintenance projects are unsuitable for disposal by conventional methods. In some cases, these sediments are highly contaminated and have been identified as part of Superfund clean-up projects. Examples include Commencement Bay, Washington; Waukegan Harbor, Illinois; and New Bedford Harbor, Massachusetts. In such cases, the dredging operation may require novel approaches to the treatment and disposal of highly contaminated bottom sediments. This paper identifies promising concepts for the management and treatment of contaminated sediments to be disposed in an upland site.

INTRODUCTION

The Corps of Engineers (CE) dredges annually over 290,000,000 cu m in maintenance dredging operations and about 78,000,000 cu m in new work dredging operations. Over 90 percent of the total volume of dredged material is considered uncontaminated. However, the potential presence of contamination has generated concern that dredged material disposal may adversely impact on water quality and aquatic or terrestrial organisms. Since many of the waterways are located in industrial and urban areas, sediments may be contaminated with wastes from these sources. In addition, sediments may be contaminated with chemicals from agricultural practices and uncontrolled chemical releases.

Recently, the CE has proposed a technically feasible and environmentally sound management approach or strategy to the disposal of dredged material (Figure 1) (2). This strategy is based on results of many years of research and dredging experience by the CE and others. The approach presents a number of variations for each major alternative of open-water and confined disposal, each having some influence on the fate of contaminants at disposal sites. The management strategy provides an evaluative framework for assessing and

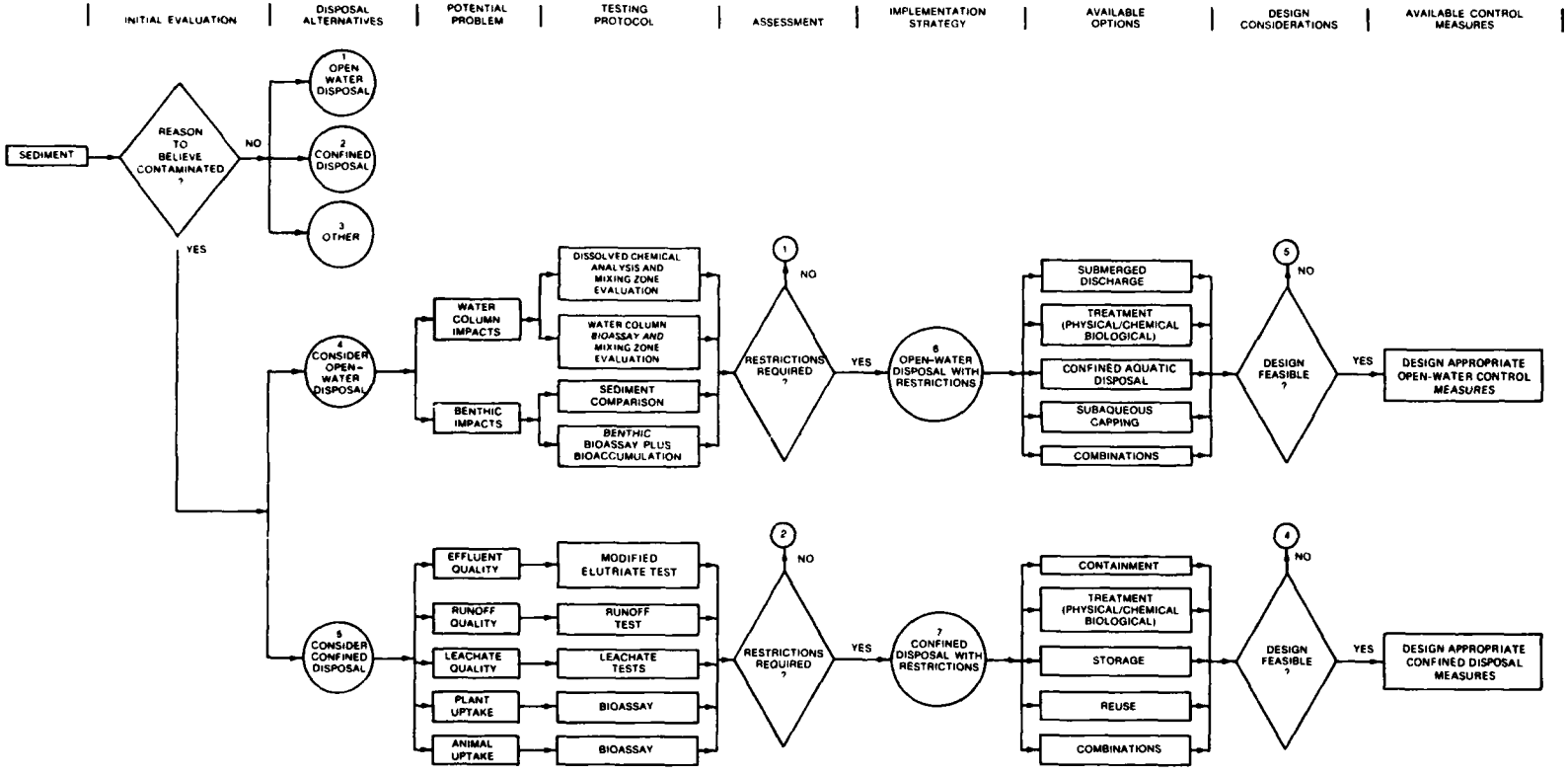


Figure 1. Management strategy flowchart for dredged material disposal

choosing an appropriate alternative for disposal of materials ranging from clean sand to highly contaminated sediments. The detailed technical approach consists of the following:

a. Initial evaluation to assess contamination potential.

b. Selecting a potential disposal alternative.

c. Identifying potential problems associated with that alternative.

d. Testing to evaluate the problems.

e. Assessing the need for disposal restrictions.

f. Selecting an implementation strategy.

g. Identifying available control options.

h. Examining design considerations to evaluate technical and economic feasibility.

i. Choosing appropriate control measures and technologies.

The management strategy (Figure 1) is currently being used and documented at several ongoing CE field demonstration projects. Results of these field activities will be used to further refine the management approach.

As part of the field demonstrations, innovative concepts for the management and treatment of highly contaminated dredged material in confined disposal operations will be evaluated. Systems will be adapted or developed that can be incorporated into existing dredging and disposal operations. Such systems may include techniques for separation and classification of highly contaminated material from that which has low-level contamination, and subsequent treatment of contaminated liquid and solids. The purpose of this paper, therefore, is to identify promising concepts and technologies for treating contaminated dredged material placed in confined upland disposal environments.

CONFINED DISPOSAL OF CONTAMINATED DREDGED MATERIAL

Overview of Confined Disposal

Before we can develop concepts for the treatment of contaminated dredged material confined in upland disposal sites, it is important that we understand the fundamental disposal operations, potential contaminant mobility pathways, and various options for control of migrating contaminants.

Confined upland disposal involves the placement of dredged material in environments not inundated by tidal waters (Figure 2). Upland disposal sites are normally diked, confined areas that retain the dredged solids while allowing the carrier water to be released, and as such are most often associated with hydraulic dredges (pipeline or hopper with pump-out capability).

Upland sites can also accept dredged material that has been dewatered elsewhere and transported in by truck or rail (if hydraulically dredged) or has simply been loaded directly into trucks or railcars by mechanical dredges. Upland disposal sites may be located immediately adjacent to, or removed great distances from, the dredging site.

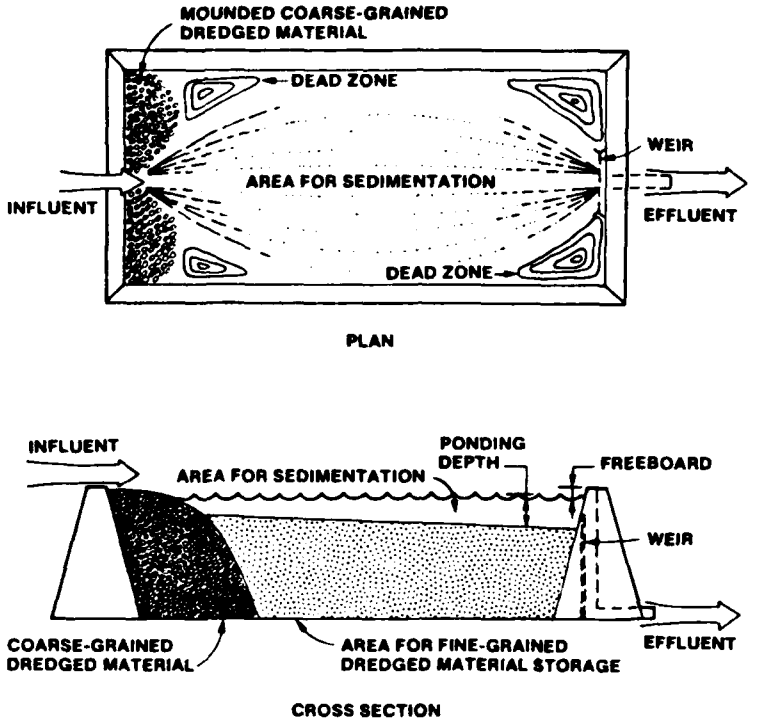


Figure 2. Conceptual diagram of a confined dredged material disposal site

The two objectives inherent in design and operation of containment areas are to provide adequate storage capacity to meet the dredging requirements and to attain the highest possible efficiency in retaining solids during the dredging operation. Basic guidelines for design, operations, and management of containment areas are presented by Palermo et al. (10) and Montgomery et al. (8).

Physicochemical Conditions

When dredged sediments is placed in an upland environment, drastic physicochemical changes occur (Figure 3). As soon as the dredged material is placed in a confinement and allowed to be exposed to the atmosphere, oxidation processes begin. The effluent water initially is dark in color and reduced with little oxygen as it is discharged from a disposal pipe of a hydraulic dredge. Mechanically dredged sediments such as with a clam shell will have sediment pore water that will initially be dark in color and reduced. As effluent water passes across the confined disposal site and approaches the discharge weir, the water becomes oxygenated and will usually become light gray or yellowish light brown. The color change indicates further oxidation of iron complexes in the suspended particulates as they move across the site. Once disposal operations are completed, dredged material densification and consolidation will force pore water up and out of the dredged material and it will drain toward the discharge weir. This drainage water will continue to become oxidized and lighter in color.

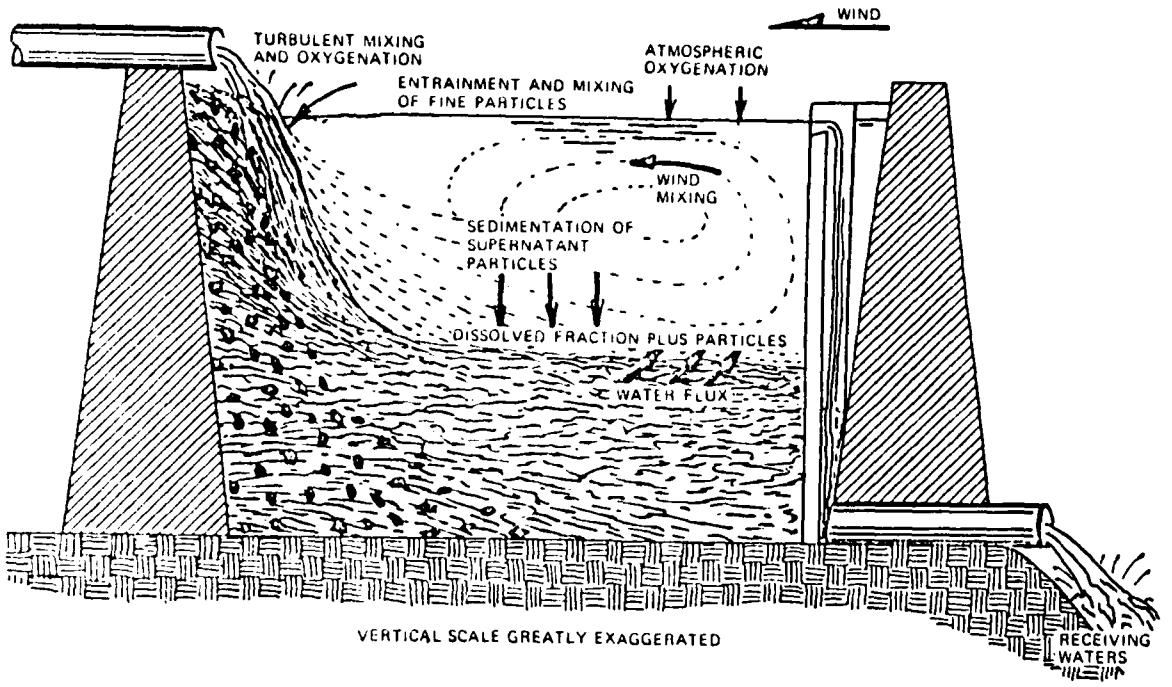


Figure 3. Schematic diagram of ponded water interaction in an active confined disposal site

Once the surfaced pore water has been removed from the confinement, the surface of the dredged material will become oxidized and lighter in color, such as changing from black to light gray. The dredged material will begin to crack as it dries out. Accumulation of salts will develop on the surface of the dredged material and especially on the edge of the cracks. Rainfall events will tend to dissolve and remove these salt accumulations in surface runoff.

Recent research on contaminant mobility from dredged material placed in an upland disposal site indicates that certain metal contaminants can become dissolved in surface runoff as dredged material dries out. During the drying process organic complexes become oxidized and decomposed. Sulfide compounds also become oxidized to sulfate salts. These chemical transformations could release complexed contaminants to surface runoff, soil pore water, and leachate through the material. In addition, plants and animals that colonize the upland site could take up and bioaccumulate these released contaminants. Contaminant mobility will be significantly controlled by the physicochemical changes that occur during drying and oxidation of the sediment.

Considerations for Confinement of Contaminated Material

Confined disposal of contaminated sediments must be planned to contain dredged material within the site and restrict contaminant mobility out of the site in order to control or minimize potential environmental impacts. There are six possible pathways for transport of contaminants from confined disposal sites:

a. Release of contaminants in the effluent during dredging operations.

b. Surface runoff of contaminants in either dissolved or suspended particulate form following disposal.

c. Leaching into ground water.

d. Plant uptake directly from dredged material, followed by indirect animal uptake from feeding on plants.

e. Animal uptake directly from dredged material.

f. Gaseous or volatile emissions during and after disposal.

Options for Contaminant Controls

Depending upon the particular dredging operation, controls may be required for any or all of the contaminant pathways previously described. The proposed management strategy (Figure 1) considers four options as available for confinement of contaminated sediment with restrictions on contaminant mobility. These options include:

a. Containment - dredged material and associated contaminants are confined within the disposal site.

b. Treatment - dredged material is modified physically, chemically, or biologically to reduce toxicity, mobility, etc.

c. Storage and rehandling - dredged material is held for a temporary period at the site and later removed to another site for ultimate disposal.

d. Reuse - dredged material is classified and beneficial uses are made of reclaimed materials.

Obviously, combinations of the above options are available for a particular dredging operation.

Containment of contaminated dredged material can be either in an existing or a new facility. These facilities can be designed or modified to handle a wide variety of contaminants. Most contaminated sediments can be disposed of in an existing site where special controls have been incorporated. In the case of highly contaminated sediments, a more secure disposal facility would be required; and, in all probability, disposal restrictions would dictate the design of a new facility.

The treatment option can be associated with either existing or new facilities. Some form of physical, chemical, or biological treatment would probably be associated with the disposal of highly contaminated dredged material. Treatment may also be combined with other options for disposal of slightly to moderately contaminated dredged material in confined disposal sites.

Of the four available options, storage and rehandling can serve two beneficial functions: continued use of confined sites located close to dredging areas and use as a rehandling facility for contaminated dredged material prior to later disposal offsite.

Finally, the concept of a reuse option would incorporate beneficial uses of materials reclaimed by the classification/separation process. Such materials could include sand and gravel or slightly contaminated construction fill to be used for raising dikes or acceptable offsite uses.

Design Considerations

Contaminated dredged material management includes methods for dewatering, transporting, storing, treating, and disposing of contaminated material. The most technically and economically effective strategy to handle contaminated dredged material will depend on many site-specific variables, which include the following:

a. Method of dredging used--hydraulic vs. mechanical.

b. Method of dredged material transport--pipeline vs. truck or hopper or barge.

c. Physical nature of removed material--consistency (solids/water content) and grain-size distribution.

d. Volume of removed material.

e. Nature and degree of contamination; physical and chemical characteristics of contaminants.

f. Proximity of acceptable treatment, storage, containment, or reuse facilities.

g. Available land area for construction of new or expansion of existing facilities.

Effluent Controls

Effluent controls at conventional confined disposal areas are generally limited to chemical clarification. The clarification system is designed to provide additional removal of suspended solids and associated adsorbed contaminants as described in Schroeder (12). Additional controls can be used to remove fine particulates that will not settle or to remove soluble contaminants from the effluent. Examples of these technologies are filtration, adsorption, selective ion exchange, chemical oxidation, and biological treatment processes. Beyond chemical clarification, only limited data exist for treatment of dredged material (4).

Runoff Controls

Runoff controls at conventional sites consist of measures to prevent the erosion of contaminated dredged material and the dissolution and discharge of oxidized contaminants from the surface. Control measures include maintaining ponded conditions, planting vegetation to stabilize the surface, liming the surface to prevent acidification and to reduce dissolution, covering the surface with synthetic geomembranes, and/or placing a lift of clean material to cover the contaminated dredged material (4).

Leachate Controls

Leachate controls consist of measures to minimize ground-water pollution by preventing mobilization of soluble contaminants. Control measures include proper site selection, dewatering to minimize leachate production, chemical admixing to prevent or retard leaching, lining the bottom and sides to prevent leakage and seepage, capping the surface to minimize infiltration and thereby leachate production vegetation to stabilize contaminants and to increase drying, and leachate collection, treatment, or recycling (4).

Control of Contaminant Uptake

Plant and animal contaminant uptake controls are measures to prevent mobilization of contaminants into the food chain. Control measures include selective vegetation to minimize contaminant uptake, liming or chemical treatment to minimize or prevent release of contaminants from the material to the plants, and capping with clean sediment or excavated material (4).

Other Controls

The control of gaseous emissions that might present human health hazards can consist of physical measures such as covers, vertical barriers, control trench vents, pipe vents, and gas-collection systems. Wind-erosion control of contaminated surface materials is another type of management or operating control to minimize transport of contaminants offsite. Techniques for limiting wind erosion are generally similar to those employed in dust control and include physical, chemical, or vegetative stabilization of surface soils (13).

Summary

Conventional confined disposal methods, described previously, can be modified to accommodate disposal of contaminated sediments in new, existing, and reusable disposal areas. The design or modification of these areas must consider the problems associated with contaminants and their effects on conventional design. One control option, treatment, will play a vital role in the handling and disposal of highly contaminated dredged material. Innovative treatment concepts will need to be integrated with the overall plans for dredging and dredged material disposal. Some of the more promising treatment concepts will be presented in the following discussion.

TREATMENT METHODS

Water Treatment

The water discharged from a confined disposal site will vary in quantity and quality over time and its characteristics will depend on the contaminants included and the type of site controls and pre-treatment applied. Site effluent will be produced in large quantities for hydraulically dredged sediments during the dredging process. This effluent will usually be of lower contaminant concentration than that found in the interstitial water and will almost always be of lower contaminant concentration than that of future water discharges from the site. Runoff water will be produced during site dewatering and periods of precipitation on the site. Runoff will be of concern primarily during the dewatering and prior to placement of a surface cap on the site.

Runoff water may be of higher contaminant concentration than the original site effluent. Leachate water is produced as water moves through the sediments and out the sides and bottom of a disposal site. This water is produced in the smallest quantities but may contain relatively high contaminant concentrations and may persist for a long period of time. Leachate treatment usually requires collection via drains placed under the site or wells for dewatering the dredged sediments.

There are a variety of physical, chemical, and biological processes that have been developed for municipal and industrial water and waste treatment requirements. The major processes may be categorized according to their intended function, capabilities, and limitations (Table 1). Many of these processes have potential in treating contaminated dredged material discharged at confined upland disposal sites. However, few processes have actually been required for or applied to dredged material disposal. Among the processes widely applied in confined disposal are plain sedimentation for solids and sediment-bound contaminant removal, and chemical clarification and filtration for enhanced removal of particulates (suspended solids), sorbed metals, and organics. Use of activated carbon for removal of soluble organics has received some limited application to dredged material. Other processes not previously applied to dredged material include organics oxidation, dissolved solids removal methods, and volatiles stripping.

Table 2 lists various water treatment processes having potential for application and indicates which of these have been applied to dredged material disposal. These treatment processes can be grouped into various levels of treatment, depending upon a particular phase or class of contaminant being removed. Four levels of treatment have been identified and are defined as follows:

a. Level I is the removal of solids and particulate-bound contaminants.

b. Level II is additional treatment to remove soluble metals.

c. Level III is further processing to remove soluble organics.

d. Level IV is the highest degree of water treatment by dissolved solids removal.

The relationships between levels of treatment are illustrated in Figure 4. A comparison of the relative efficiencies of the treatment levels is given in Table 3. Increasing levels of treatment result in increasing percentages of contaminant removal. The qualitative ranges of concentrations remaining after each treatment level and percent removals are based on actual monitoring of disposal sites for Levels I and II (where applicable) and on best available water treatment technology for levels III and IV. It should be noted that the estimates made for soluble organics and soluble metals removals past Level I are mean values and represent a grouping of contaminants with large ranges of solubility and treatability. The data in Table 3 should be reviewed as preliminary for planning purposes only, and, as such, are presented to illustrate potential levels of removals. Actual removal efficiency data on dredged materials would have to be obtained through site-specific testing, evaluations, and demonstrations. The Waterways Experiment Station will be evaluating various water treatment concepts over the next few years as part of several ongoing and proposed field demonstrations.

TABLE 1. TREATMENT PROCESS CATEGORIES AND APPLICABILITY

Category of Treatment	Unit Process	Capabilities	Limitations
Physical	Organic Phase Separation	Removal of organics present at greater than their solubility limit in water, either settling or flotation	Can only remove compounds to their solubility limit in water, presence of a variety of compounds may create organic plus water mixtures which will not float or settle
	Solids Settling	Removal of suspended solids that are more dense than water and therefore settle	Generally rather low suspended solids levels in ground water, not too significant
	Solids Filtration	Removal of suspended solids by entrapment, solids that do not readily settle	Can only remove suspended matter, backwash requirements create another waste stream to handle
Chemical or Physical/ Chemical	pH Adjustment	Neutralizes excess H^+ of OH^- present, can aid performance of other processes that are pH dependent	Simply neutralizes excess acid or alkali, removes nothing
	Chemical Coagulation	Improved removal of suspended solids that are difficult to settle	Suspended solids generally are not too significant, only a precursor to settling or filtration
	Carbon Adsorption	Removal of adsorbable dissolved organics	Suitable only for carbon adsorbable organics, many low molecular weight organics such as alcohols and others are poorly adsorbed, spent carbon must be regenerated or disposed
	Air Stripping	Removal of volatile dissolved organics	Suitable only for low boiling point organics (less than water) with limited solubility, creates an off-gas contamination problem which may require treatment
	Steam Stripping	Removal of volatile dissolved organics	Some of the same limitations as air stripping, creates condensate stream requiring disposal

(Continued)

TABLE 1. (Concluded)

Category of Treatment	Unit Process	Capabilities	Limitations
	Resin Absorption	Removal of absorbable dissolved organics	Suitable only for specific compounds absorbable on the resin, solvents may attack resin, creates a regenerant stream requiring disposal
Biological Treatment	Aerobic Suspended Growth	Removal of biodegradable dissolved organics	Suitable only for biodegradable organics, many volatile, relatively insoluble chlorinated solvents are not amenable to biodegradation, creates biological sludge requiring disposal
	Aerobic Fixed Film	Removal of biodegradable dissolved organics	Same limitations as suspended growth systems, not as susceptible to sludge settleability upsets found with suspended growth systems
	Anaerobic	Removal of biodegradable dissolved organics	Suitable only for anaerobically biodegradable organics, generally more limited than aerobic oxidation, produces CH_4 which may be usable as a resource
Residuals Treatment or Supporting Processes	Sludge Digestion/ Thickening/ Dewatering	Provides sludge suitable for disposal	Only a supporting operation to try to minimize cost of residuals disposal
	Chemical Recovery/ Reuse	Possible reprocessing of solvents for reuse	Mixtures of solvents are difficult to recover, can offset some treatment costs
	Incineration	Ultimate disposal of either sludge or residual organics	Generally suitable only for concentrated waste streams (e.g., solvents or dewatered sludges), generates an ash for disposal

TABLE 2. LISTING OF WATER TREATMENT PROCESSES

Treatment Process	Proven Method	Proven But Not Demonstrated	Applied to Dredged Material	Not Applied to Dredged Material
Suspended Solids				
Plain Sedimentation	X		X	
Chemical Clarification	X		X	
Filtration	X		X	
Soluble Metals				
Precipitation	X		*	
Soluble Organics				
Adsorption	X		X	
Ozonation	X			X
Dissolved Solids				
Distillation	X			X
Reverse Osmosis	X	X		X
Electrodialysis	X			X
Ion Exchange	X			X
Volatiles				
Stripping	X			X
Leachate**				
Biological	X			X
Physical/Chemical	*	X		X

* Limited success on pilot scale.
** Potential for use of existing municipal or industrial process for treatment offsite.

Immobilization Technology

Overview of Available Technology

Immobilization technology as applied to dredged material uses physical and chemical processes to produce chemically stable solids with improved contaminant containment and handling characteristics (Figure 5). These techniques are commonly applied to bottom sediments in Japan, where filling to improve or develop land is desirable. Reports of the results of the Japanese experience with stabilization have been presented in previous U.S./Japan Expert Meetings and are documented in various proceedings entitled "Management of Bottom Sediments Containing Toxic Substances." The primary thrust of the Japanese research has been the improvement of handling and bearing capacities of the sediments, with minor emphasis on efficiency of immobilization of contaminants. In the US practice, these immobilization techniques are used almost exclusively in oil and gas drilling operations and on wastes from power plants, chemical manufacturing, and the nuclear industry. A recent review of 33 cases of contaminated sediments remediation in the United States (11)

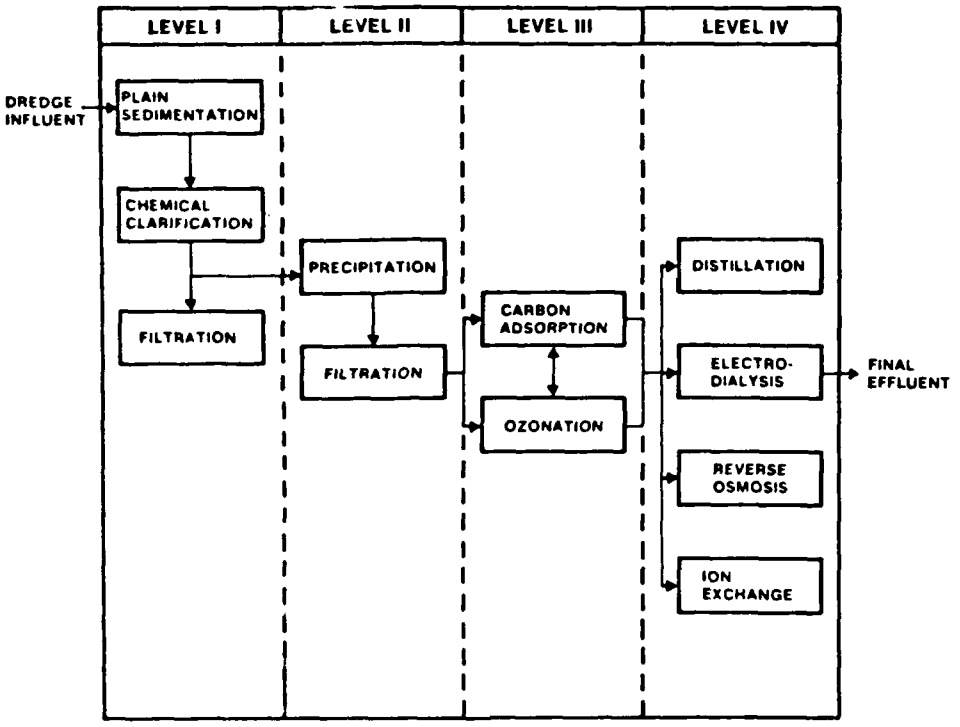


Figure 4. Treatment processes flow diagram

TABLE 3. CONTAMINANT REMOVAL EFFICIENCY OF TREATMENT LEVELS

Level	Class of Contaminant	Percent Removal	Concentration Remaining
I	Solids	99.9+	mg/l
	Metals	80 to 99+	ppb to ppm range*
	Organics	50 to 90+	ppb to ppm range*
II	Metals	99+	ppb range*
	Organics	50 to 90	ppb to ppm range*
III	Metals	99+	ppb range**
	Organics	95+	ppb range**
IV	Metals	99+	Highest quality attainable**
	Organics	99+	Highest quality attainable**

* Concentrations based on Hoeppel et al. (5), and Palermo (9).
** Concentrations based on capability of best available treatment technology.

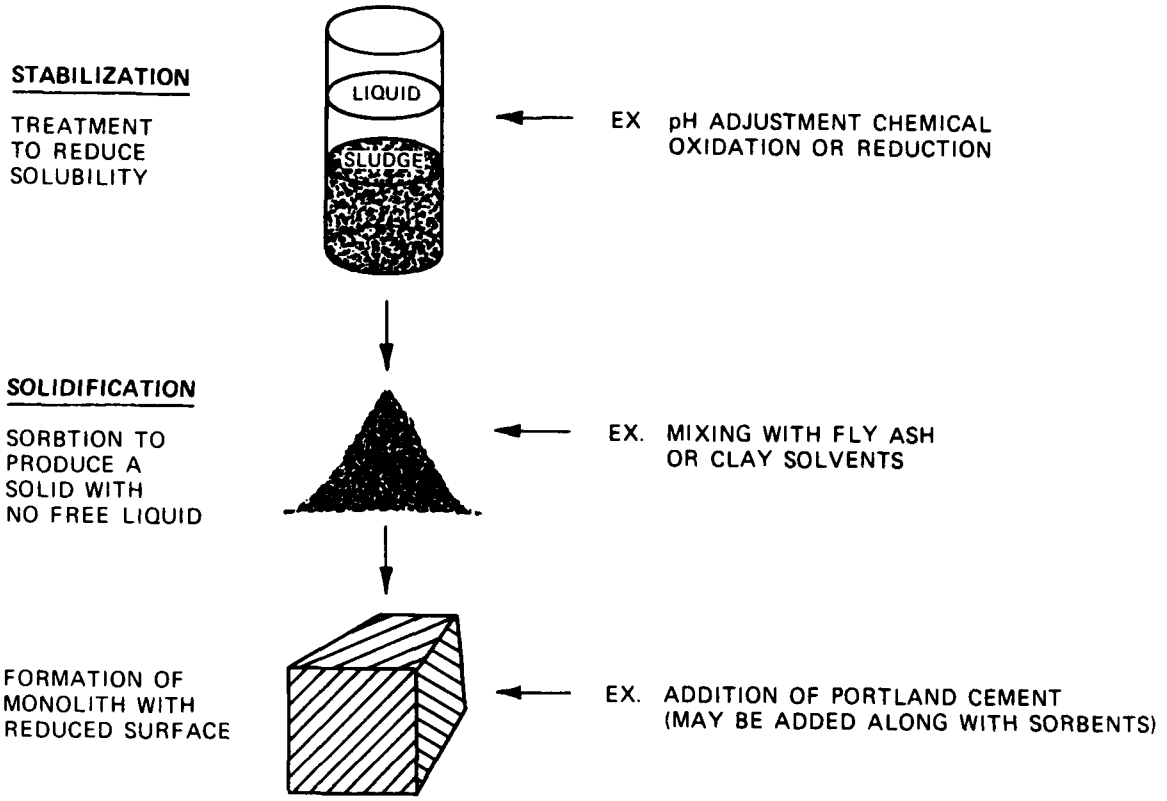


Figure 5. Steps in immobilization of wastes

revealed only two cases, the Upper Hudson River in New York and Waukegan Harbor in Illinois, where stabilization was among the remediation techniques evaluated, but no cases of actual use were identified.

The immobilization techniques, stabilization and solidification, have two different goals. Stabilization systems attempt to reduce the solubility or chemical reactivity of a waste. Solidification systems attempt to convert the waste into an easily handleable solid with reduced hazard from leaching or spillage. Because both processes have the common purpose of improving containment (immobilization) of potential contaminants in treated dredged material, they will be discussed together.

Most immobilization systems being marketed in the United States are proprietary processes involving the addition of absorbents and solidifying agents to a waste. Often the marketed process is changed to accommodate specific wastes. Comprehensive general discussions of waste stabilization/solidification are given in Environmental Laboratory (1), Malone and Jones (7), and Iadevaia and Kitchens (6).

Waste immobilization systems that have potentially useful application in confined disposal activities are:

a. Sorbtion.

b. Lime-flyash pozzolan processes.

c. Pozzolan-portland cement processes.

Other technologies such as thermoplastic microencapsulation, macroencapsulation, and fusing waste to a vitreous mass or using self-cementing material are too specialized or not sufficiently field applicable to be used at present (1).

Sorbtion. In order to prevent the loss of contaminated fluids and improve the handling characteristics of the sediment, a dry, solid absorbent may be added to the dredged material. The sorbent may interact chemically or may be wetted by the water part of the dredged material and retain the liquid as part of the capillary liquid (Figure 6).

The most common sorbents used with waste include soil and waste products such as bottom ash, fly ash, or kiln dust from cement manufacture. In general, selection of sorbent material involves trade-offs between chemical effects, costs, and amounts required to produce a solid product suitable for burial.

Artificial materials have also been advocated for use as sorbents in solidification; however, the relatively high costs of these materials have limited their use. Synthetic materials have generally found use where binding of a specific contaminant in the waste is of paramount importance. Table 4 lists various synthetic sorbent materials that have been developed or tested for use with hazardous waste.

When selecting a sorbent it is important to note the following technical considerations:

a. Quantity needed to eliminate contaminated drainage fluid.

b. Compatibility or reactivity of the contaminants and the sorbent.

c. Level and character of contamination that might be introduced in the sorbent.

d. Chemical binding properties of sorbent for specific contaminants.

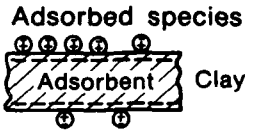
Sorption has been widely used to eliminate free liquids and improve handling. Some sorbents have been used to limit the escape of volatile organic compounds. Sorbents may also be useful in contaminant immobilization when they modify the chemical environment and maintain the pH and redox potential to limit the solubility of the contaminants. Sorbents are widely used in lined landfills in the United States to eliminate or control the pressure head on the liner, but the liner is the major protection for the surrounding environment. Caution must be exercised to ensure that leachable contaminants are not introduced as by-products of the sorbent itself.

Lime-flyash pozzolan processes. Solidification/stabilization of dredged material using lime and pozzolanic materials requires that the material be mixed with a carefully selected, reactive flyash (or other pozzolanic material) to a pasty consistency. Lime (calcium hydroxide) is blended into

NEW CONTAINMENT SYSTEMS

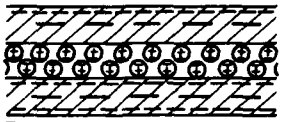
Adsorption

Example: Interaction of single layer of clay mineral with polar groups

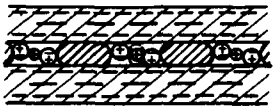


Chemsorption

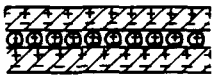
Example: Organic compounds on clays



Polar molecules with montmorillonite



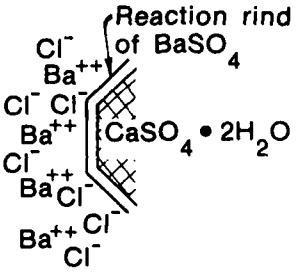
Polar and nonpolar molecules with montmorillonite



Polar molecules with halloysite

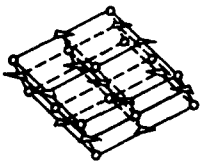
Passivation

Example: Reaction of gypsum with barium solution



Diadochy

Example: Substitution in the calcite crystal lattice



$\bullet Ca^{2+}$ or Mg^{+2}, Fe^{+2}, Mn^{+2}, etc.

$\bullet CO_3^{2-}$

Reprecipitation

Example: *Production of sideronatrite from sodium sulfate*

$$Fe_2(SO_4)_3 + 2Na_2SO_4 + 8H_2O \longrightarrow 2Na_2Fe(SO_4)_2 \cdot 3H_2O + H_2O + H_2SO_4$$

Comparative Solubilities at 25°C

$Na_2SO_4 \cdot 7H_2O$ 524 g/ℓ

$Na_2Fe(SO_4)_2 \cdot 3H_2O$ 0.16g/ℓ

Figure 6. Mechanisms retaining water on and in solid phases

TABLE 4. SYNTHETIC SORBENTS USED WITH HAZARDOUS WASTES

Sorbent	Waste Treated Effectively
Activated Alumina	Sorbs fluoride in neutral wastes
Activated Carbon	Sorbs dissolved organics
Hazorb*	Sorbs water and organics
Locksorb**	Reported effective with oil emulsions
Imbiber Beads†	Reported useful in spills of inert spirits-type liquids (cyclohexane)

* Product of Diamond Shamrock Corp.
** Product of Radecca Corp., Austin, Tex.
† Product of Dow Chemical Co., Midland, Mich.

the sediment-flyash mixture. Typically 20-30% lime is needed to produce a strong pozzolan. The moist soil-like material is then placed in the confinement and roller compacted.

In general, flyash-lime solidified materials are not as durable as pozzolan-portland cement composites. Common problems with lime-pozzolan reactions relate to interference in cementitious reaction that prevents bonding of materials. The bonds in pozzolan reactions depend on the formation of calcium silicate and aluminate hydrates. Several materials to include sodium borate, calcium sulfate, and carbohydrates can interfere with this reaction. Oils and greases can also physically interfere with bonding by coating soil particles. The cementing system is strongly alkaline and can react with certain contaminants to release undesirable materials as gas or in leachates.

Pozzolan-portland cement processes. There are a variety of treatment processes that incorporate portland cement as a binding agent. Pozzolanic products (materials with fine-grained, non-crystalline, reactive silica) are frequently added to portland cement to react with any free calcium hydroxide and thus improve the strength and chemical resistance of the concrete-like product.

Waste solidifying formulations based on portland and pozzolan-portland systems vary widely and a variety of materials have been added to alter performance characteristics. Such materials include soluble silicates (3), hydrated silica gels, and clays such as betonite, illite, or attapulgite.

Cement-based immobilization systems have proved to be highly versatile and flexible methods. The composite materials can be found to have exceptional strength, excellent durability and retain contaminants very effectively. However, these systems have limitations that relate to the effects of the contaminants on the setting (retardation from calcium sulfate, borates, carbohydrates, etc.) and stability of the silicate and aluminates that form

when the portland cement hydrates. The very high alkalinity of hydrating portland cement can cause the evolution of ammonia gas if ammonium ion is present in abundance in the material and the solubilization of some metals (e.g. nickel, lead, zinc) at high pH's.

Compatibility of Contaminants and Treatment Process

The chemical reactivity of the waste component generally controls the selection of immobilization options and its optimization. Table 5 summarizes the major chemical considerations that direct the selection of a particular immobilization system. Most solidification systems will work under adverse circumstances if adaptions are made in the material being treated or the processing train.

Implementation Strategies

Onsite immobilization programs can be classified according to the manner in which the chemical reagents are added to and mixed with the materials being treated. Three basic onsite implementation strategies are considered applicable to confined disposal: in situ mixing, plant mixing, and area mixing.

The selection of an appropriate implementation strategy must be based on an analysis that applies site-specific factors to the attributes of the available strategies. As a result, definitive guidelines for implementation strategy selection based on generalized criteria cannot be developed. However, it is possible to develop some generalized attributes of each strategy as applied to conditions expected at confined disposal sites.

In situ mixing. This method is primarily suitable for dredged slurries that have been initially dewatered. In situ mixing is most applicable for the addition of large volumes of low reactivity solid chemicals. This strategy incorporates the use of construction machinery, typically a backhoe, to accomplish the mixing process. Where large containment areas are being treated, clamshells and/or draglines may be used. Data are not currently available on the mixing efficiency of the in situ process when applied to large field-scale projects.

Major modifications to the in situ mixing strategy include the development of the reagent addition and mixing equipment that allows better control of the process. Equipment specifically designed for in situ stabilization is currently being developed and evaluated (ENRECO 1983, American Resources Corp. 1984). Each system is estimated to provide in situ stabilization of up to 1000 cu yd of sediment per 8-hr working day. The ENRECO equipment is limited to an operating depth of approximately 4 ft. Shallower depths result in blow out or additive communication to the surface. With the American Resources Corp. PF-5 Pond Injection System, depths of up to 11 ft can be treated.

Plant mixing. Plant mixing strategies can be adapted for applications to slurries and solids. Plant mixing is most suitable for application at sites with relatively large quantities of contaminated materials to be treated. In the plant mixing process the materials are physically removed from their location, mechanically mixed with the immobilization reagents, and redeposited in

TABLE 5. COMPATIBILITY OF SELECTED WASTE CATEGORIES WITH DIFFERENT STABILIZATION/SOLIDIFICATION TECHNIQUES

Waste Component	Treatment Type	
	Cement-Based	Pozzolan-Based
Organics		
Organic solvents and oils	Many impede setting, may escape as vapor	Many impede setting, may escape as vapor
Solid organics (e.g., plastics resins, tars)	Good--often increases durability	Good--often increases durability
Inorganics		
Acid wastes	Cement will neutralize acids	Compatible
Oxidizers	Compatible	Compatible
Sulfates	May retard setting and cause spalling unless special cement is used	Compatible
Halides	Easily leached from cement, may retard setting	May retard set, most are easily leached
Heavy metals	Compatible	Compatible
Radioactive materials	Compatible	Compatible

After Environmental Laboratory (1).

a prepared disposal site. Special equipment adaptations have been utilized to handle sludges with high solids contents and contaminated soils. A schematic diagram of a typical plant mixing strategy is illustrated in Figure 7.

Area mixing. This strategy is applicable to those confined disposal sites where high solids content slurries must be treated. The term "area mixing" is used to denote those strategies that use horizontal construction techniques to add and mix the stabilization reagents with the dredged material. Area mixing is land area intensive, requiring a relatively large land area to carry out the process. Area mixing strategies present the greatest possibility for fugitive dust, organic vapor, and odor generation. The typical area mixing strategy will require that the dredged material be sufficiently dewatered to support construction equipment.

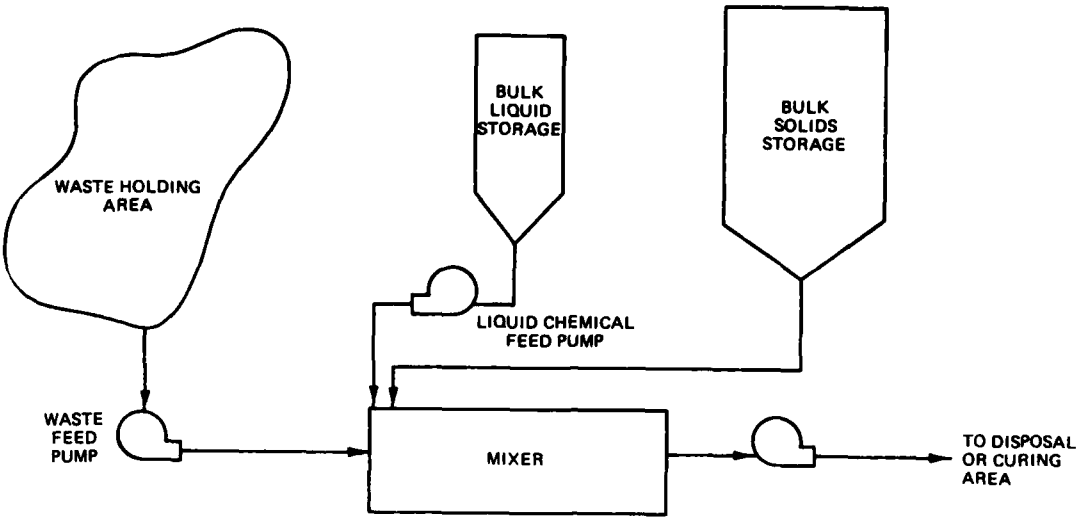


Figure 7. Schematic of plant mixing scenario

The three basic implementation strategies can be incorporated into confined disposal operations in a variety of ways to improve contaminant containment. In most cases, the same immobilization reagents and equipment could be employed.

Application of Immobilization Technology at Confined Disposal Sites

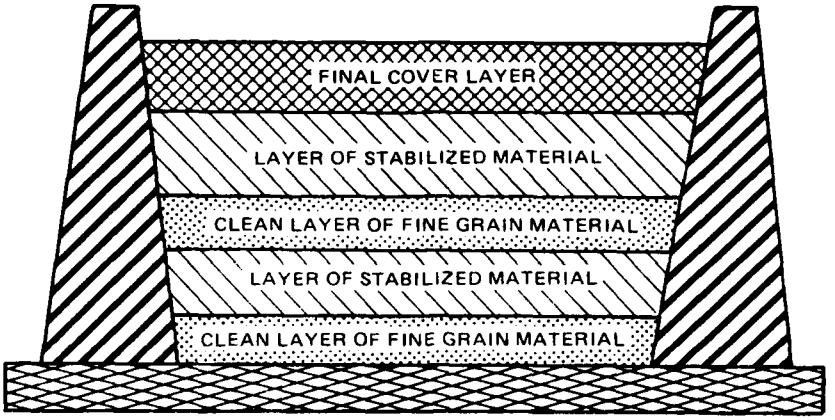
Immobilization technology can potentially be used in a variety of ways to treat contaminated dredged material to provide improved management and disposal operations (Figure 8). The concepts presented in Figure 8 can be incorporated into the overall management and disposal operation beginning with the project planning stage.

Concept "A"

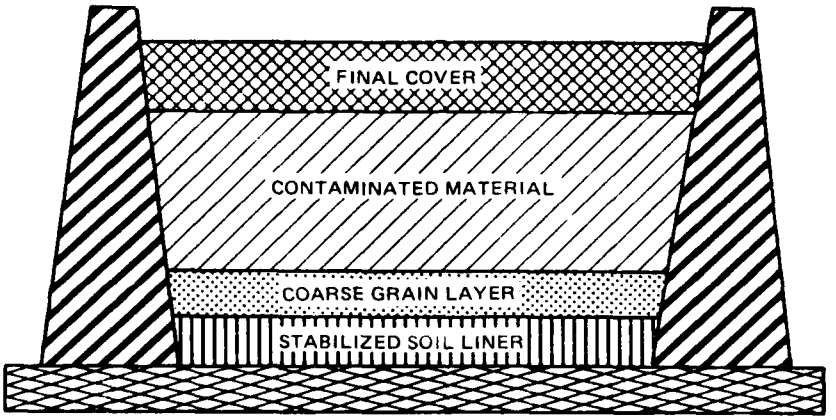
As shown in Figure 8, Concept "A" involves alternating layers (thin lifts) of relatively clean dredged material and contaminated dredged material that is stabilized. The initial lift of clean, fine-grained sediments would be dewatered to promote densification and consolidation to provide a low permeable soil layer or natural liner for the containment area. Once this layer has achieved the desired degree of consolidation and permeability, the contaminated material would be placed on top, dewatered, and stabilized in situ. (Various schemes for in situ stabilization have been discussed previously.) This layering process would continue until the containment area was filled. A final soil cover would be added, shaped, and vegetated.

Concept "B"

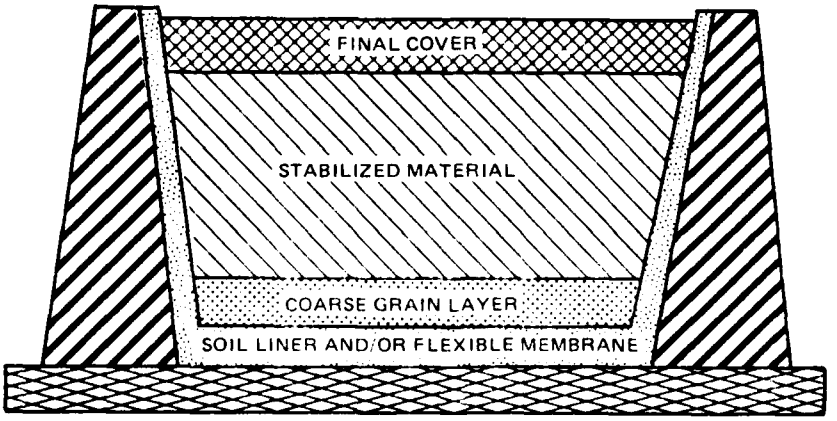
The second disposal method shown in Figure 8 incorporates soil stabilization as a treatment to produce a low permeable liner and is designated Concept "B." The stabilized soil layer is used to contain any leachate generated from the dewatering and long-term disposal of the contaminated dredged material.



a. DISPOSAL CONCEPT "A" ALTERNATING LAYERS OF STABILIZED MATERIAL



b. DISPOSAL CONCEPT "B" STABILIZATION FOR LINER SYSTEM



c. DISPOSAL CONCEPT "C" STABILIZATION IN SECURE FACILITY

Figure 8. Immobilization concepts for confined disposal operations

Appropriate chemical reagents are added and mixed with the disposal site soil using either the in situ or area mixing methods. Then a layer of coarse material is added above the stabilized layer to facilitate dewatering and collection of leachate. The contaminated dredged material is disposed next and dewatered. A clean layer of dredged material is used as final cover. One modification to this concept would be the additional step of stabilizing the contaminated dredged material to further protect against contaminant escape.

Concept "C"

The final concept illustrated in Figure 8 is Concept "C." This disposal method provides the highest degree of environmental protection. A soil liner, or flexible membrane, or both, are used to line the bottom and sides of the disposal site. Then a coarse-grained layer is used to facilitate dewatering and leachate collection. The contaminated sediment is disposed into the lined site, dewatered, and stabilized. An alternative would be first to dewater the contaminated sediments in a temporary processing facility, and then to apply the stabilization process prior to placement of the treated material in the confined disposal site. This stabilization operation would be accomplished using the plant mixing strategy (either mobile or field erected) or the area-wide mixing strategy, stockpiling the stabilized material prior to rehandling, and disposal in the prepared site. The last operation in Concept "C" would be to final cover, shape, and vegetate the site.

Summary

The treatment methods identified as having potential for application to confined disposal of contaminated dredged material can be grouped into water treatment processes and immobilization technologies. Four levels of water treatment were identified with increasing efficiencies for contaminant removals. It should be remembered, however, that the relative degrees of contaminant removals past treatment level II are estimated, and, as such, are useful for preliminary planning purposes only. Final processing trains will be identifiable only on a site-specific basis because of the individual testing and evaluation studies required to develop the detailed design and operation parameters.

Three immobilization systems were identified as having useful application in confined disposal operations: sorbtion, lime-flyash pozzolan processes, and Pozzolan-portland cement systems. Various modifications to the basic systems exist and process applicability is evaluated on a case-by-case basis. Compatibility of the contaminants in the dredged material and the immobilization process will play a key role in the selection of a particular process or system. Strategies for implementing the immobilization processes were grouped according to the manner in which the chemical reagents are added to and mixed with the materials being treated. Three strategies (in situ mixing, plant mixing, and area-wide mixing) were presented and discussed as being applicable to confined disposal operation.

Finally, three confined disposal concepts that incorporate some form of immobilization technology were conceptually presented. These concepts were designated "A," "B," and "C." The feasibility of applying the various treatment concepts presented in this paper at an existing or new confined disposal site will be established under an ongoing U. S. Army Corps of Engineers

project demonstration at Indiana Harbor, Illinois. The potential for extending this evaluation to the "Commencement Bay Washington, Superfund Study" also exists.

COMMENTS

The treatment concepts identified as part of this paper and presented at the 10th U. S./Japan Experts Meeting on "Management of Bottom Sediments Containing Toxic Substances," are based on a requirement to incorporate proven technology with existing U. S. Army Corps of Engineers confined disposal operations. The emphasis necessarily is therefore on an adaptation of existing treatment methods developed for treatment of waste liquids and solids. The treatment of extremely contaminated toxic sediments to include chemical spills clean-up is not addressed as part of this paper. Research on the handling and clean-up of chemical spills is sponsored by the U. S. Environmental Protection Agency (EPA). Our research programs are, however, closely coordinated with the EPA's in order to derive mutually beneficial program technology exchanges.

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THE INFLUENCE OF ACCUMULATING ORGANIC SEDIMENT ON THE MARINE ENVIRONMENT IN OSAKA BAY

M. Kawaguchi, T. Saski, H. Noguchi

The 3rd District Port Construction Bureau
Ministry of Transport, Japan

INTRODUCTION

The coastal area along Osaka Bay is a densely populated region with 12,920,000 people living in approximately 4,800 km^2. Water quality in Osaka Bay has been deteriorating due to an inflow of contaminants discharged from industrial sites along the coast since the rapid economic growth period of the 1960s. Particularly, in the offshore bay area (bay head) where the tidal flow is stagnant, eutrophic pollution is progressing. The accumulating organic sediments which release phosphorus and nitrogen are accelerating the eutrophication of the water. Since 1979, a variety of investigations have been conducted by the 3rd District Port Construction Bureau of the Ministry of Transport to probe measures for improving the offshore sea environment. This report presents our study of the influence of the accumulating sediments on the sea environment and on its clarification plans.

PRESENT CONDITIONS OF WATER POLLUTION IN OSAKA BAY

The deteriorating seawater environment in Osaka Bay is creating problems such as the occurrences of red tide and the appearance of anaerobic water masses. In 1973, the Special Law for Conservation of the Environment of the Seto Inland Sea was enacted, followed by various efforts to enhance the regulatory system such as the enforcement of the Areawide Total Pollution Load Control Concerning COD (Gross COD Control). However, despite the regulation of inflowing loads of COD, no significant improvement in water quality has been attained as was initially expected, leaving water quality in the same state as several years ago (Figure 1, Table 1).

Figure 2 shows the levels satisfying the environmental standards at the standard stations in Osaka Bay in 1982. Of 21 monitoring stations, 14 did not meet the standards.

Moreover, as seen in Table 2, the incidence of red tide is still high, totaling 20 in 1983.

DETERIORATION OF MARINE ENVIRONMENT DUE TO ORGANIC SEDIMENT

Realities of Population in Bottom Sediments

As indexes for the pollution level of the Osaka Bay sediments, COD, ignition loss, sulfide, total phosphorus, and total nitrogen were chosen and investigated. Figure 3 displays the horizontal distribution of the index substances in the bottom sediments (the uppermost 0- to 5-cm layer). The investigation revealed that, in the offshore bay area, COD was not less than

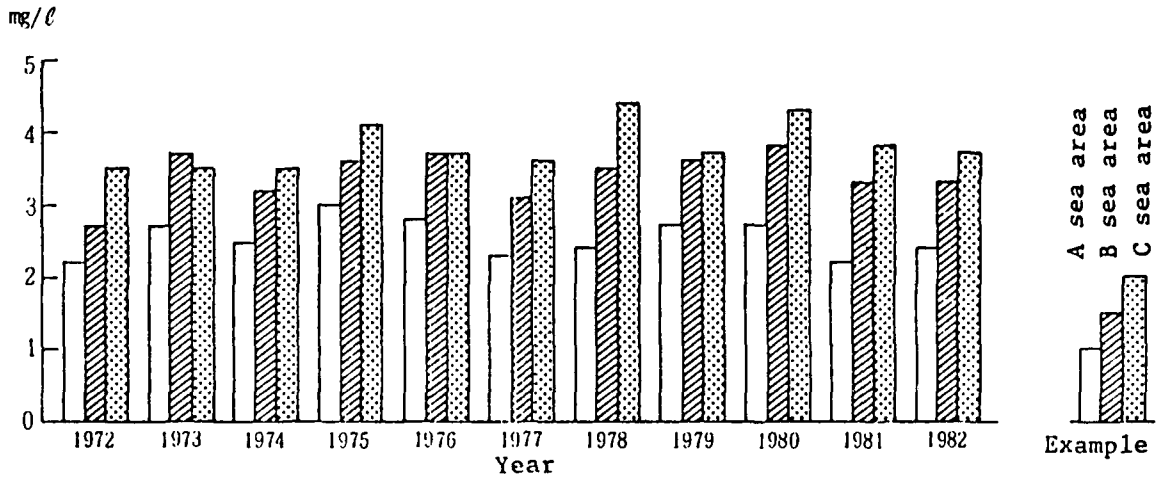


Figure 1. Yearly course of COD change in Osaka Bay (annual mean at the monitoring stations for environmental standard). Prepared from: White Paper on Pollution (Osaka Prefecture) and White Paper on Environment (Hygo Prefecture)

TABLE 1. YEARLY COURSE OF THE CHANGE OF NUMBERS OF DISQUALIFIED AREAS BY COD ENVIRONMENTAL STANDARD IN OSAKA BAY

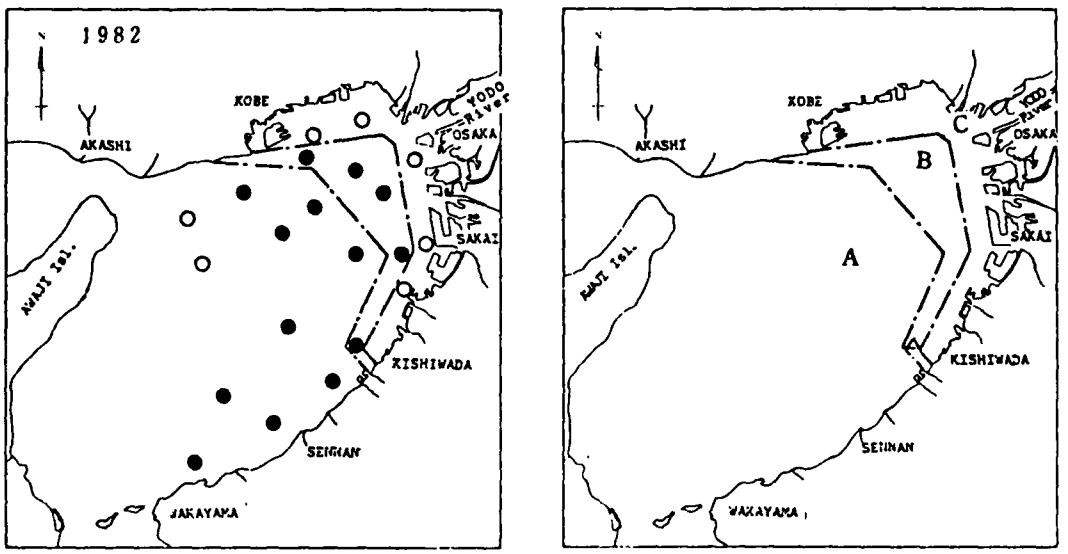
Year / Sea Area	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982
C (5 points)	0	0	0	0	0	0	0	0	0	0	0
B (5 points)	1	3	5	5	5	4	5	5	5	3	5
A (11 points)	9	6	8	8	9	9	8	9	9	6	9
Total (21 points)	10	9	13	13	14	13	13	14	14	9	14

(Number)

TABLE 2. NUMBER OF OCCURRENCES OF RED TIDE IN OSAKA BAY

Year	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983
Osaka Bay	20	29	39	50	49	54	34	38	38	42	38	31	40

From: Red Tide in Seto Inland Sea Compiled by Setonaikai Fishery Adjustment Office, Fishery Agency



○ : Site where the environmental standard is satisfied.

● : Site where the environmental standard is not satisfied.

C: 8 mg/ℓ max.

B: 3 mg/ℓ max.

A: 2 mg/ℓ max.

Figure 2. Level of compatibility to the environmental standard in terms of COD

30 mg/g, sulfide not less than 0.6 mg/g, ignition loss not less than 10 percent, T-N 2.0 mg/g or more, and T-P 0.65 mg/g or more.

Core samples were taken from the bottom sediments by using a box corer (sediment core sampling device) for an assay to obtain the thickness of the layers of the bottom sediments. The results of the assay (Figure 4) show that the bottom sediments with COD $\geq$ 30 mg/g, ignition loss $\geq$ 10 percent, and sulfide $\geq$ 0.6 mg/g tend to increase in thickness in the offshore bay area. However, the accumulated sediments are relatively thin, with the maximum thickness being 40-50 cm.

Correspondence of the Areas of Polluted Sea Bottom Sediments and the Areas Where the Quality of the Bottom Water Layer is Aggravated

Realities of Pollution in the Bottom Layer of Water

As indexes for aggravating water quality at the bottom layer, dissolved oxygen (DO) in water, ammonium nitrogen (NH_4-N), and inorganic phosphorus (PO_4-P) were chosen and determined for their distribution in Osaka Bay (Figure 5).

The investigation showed that the distribution of areas where quality of the bottom water layer deteriorated corresponded to that of the areas shown in

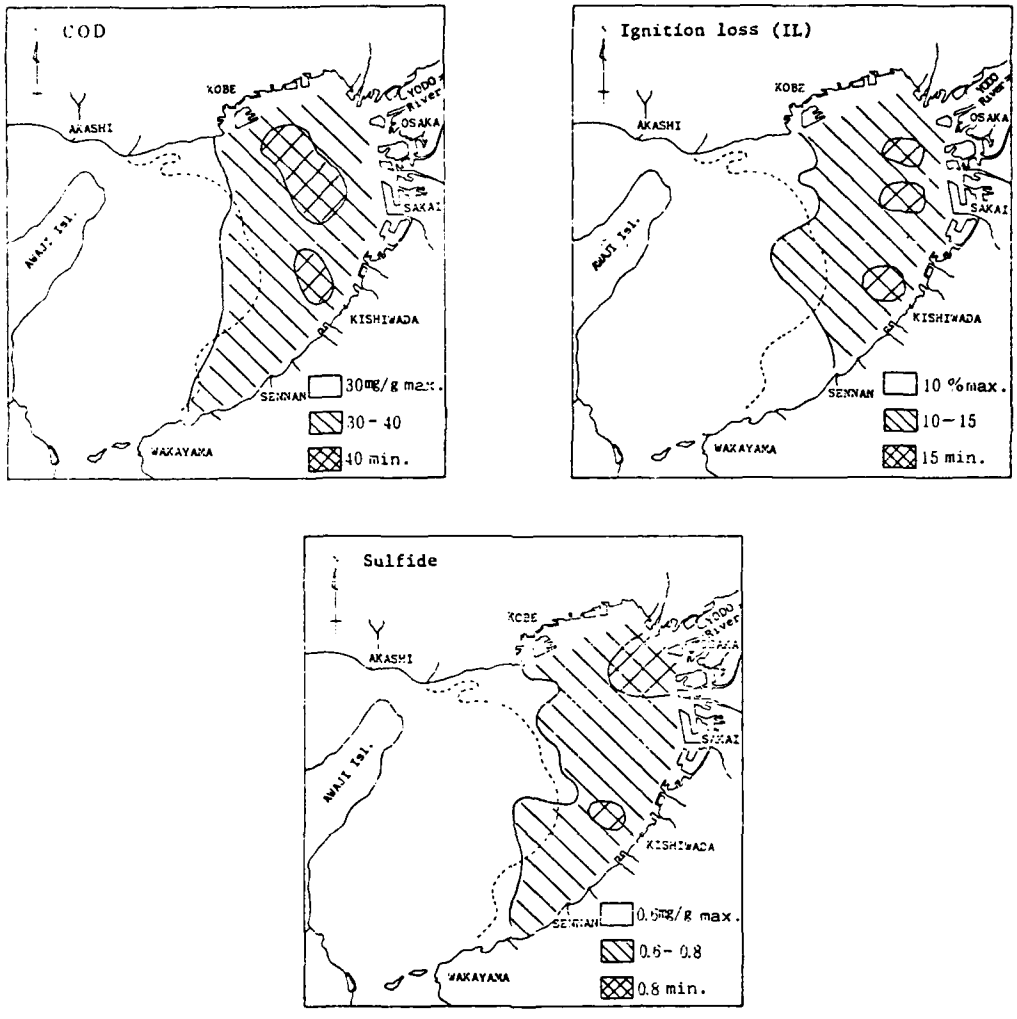


Figure 3. Horizontal distribution of COD, sulfide, T-N, and T-P in bottom sediments (0-5 cm)

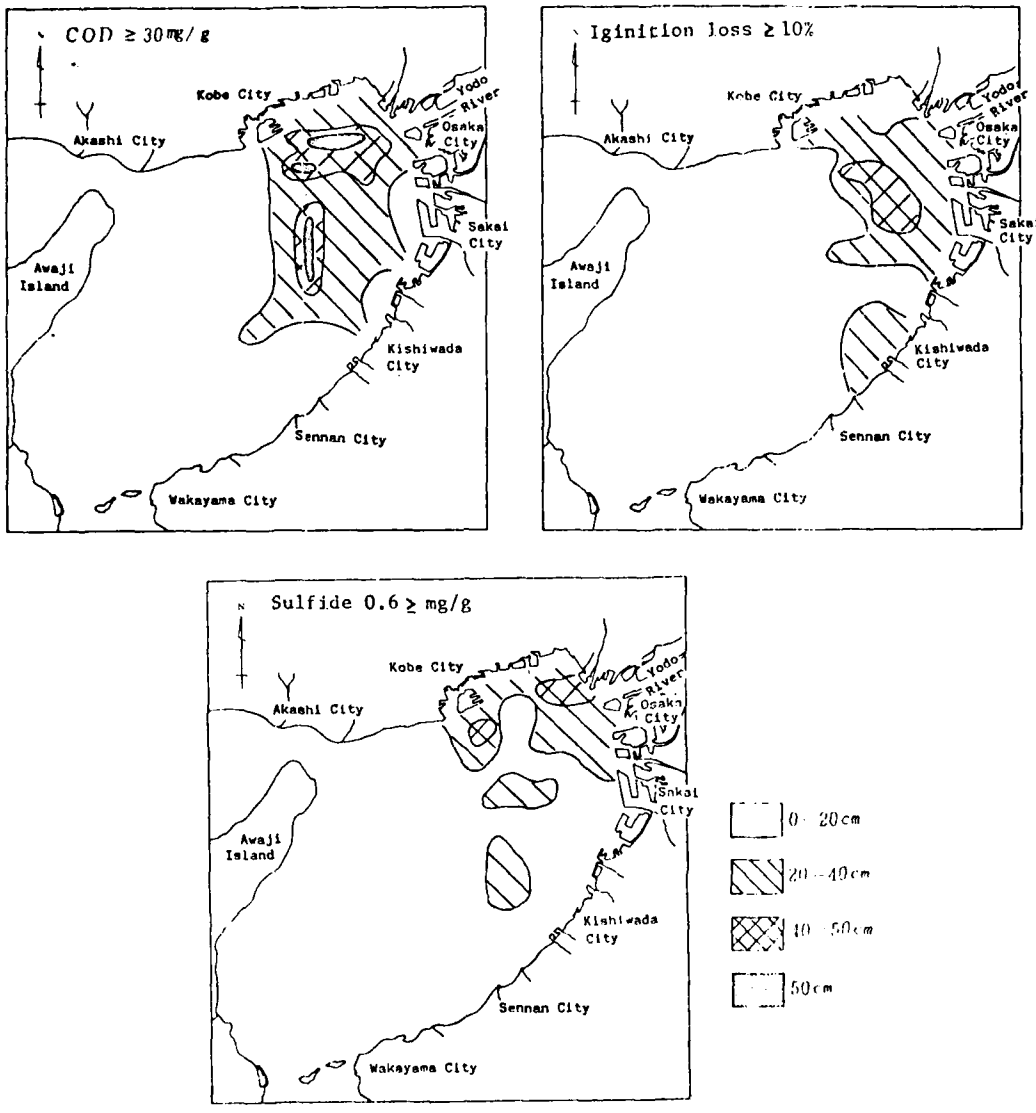


Figure 4. Distribution of the accumulated sediment thickness

Figure 3, i.e., where the bottom sediments were polluted. From the above results, we may assume that the accumulated organic sediments deteriorate water quality and constitute a factor of eutrophic pollution. Pollution is particularly serious in the bottom layer of seawater in the offshore bay area where the DO level is 5 mg/g or less and phosphoric acid phosphorus is 0.02 mg/ℓ or more.

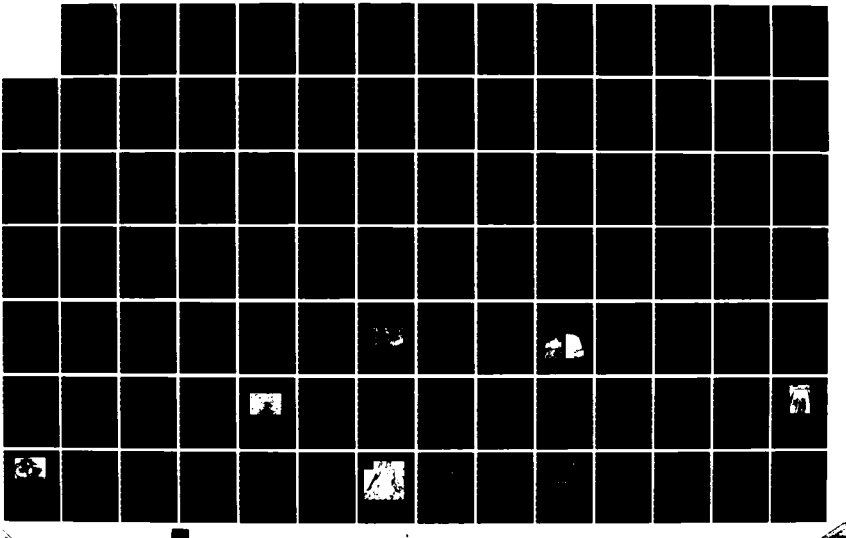
Relationship Between the Quality of the Bottom Water Layer and the Surface Layer of Bottom Sediments

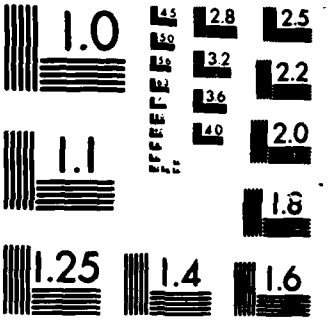
As seen in Figures 3 and 5, the quality of the bottom water layer is

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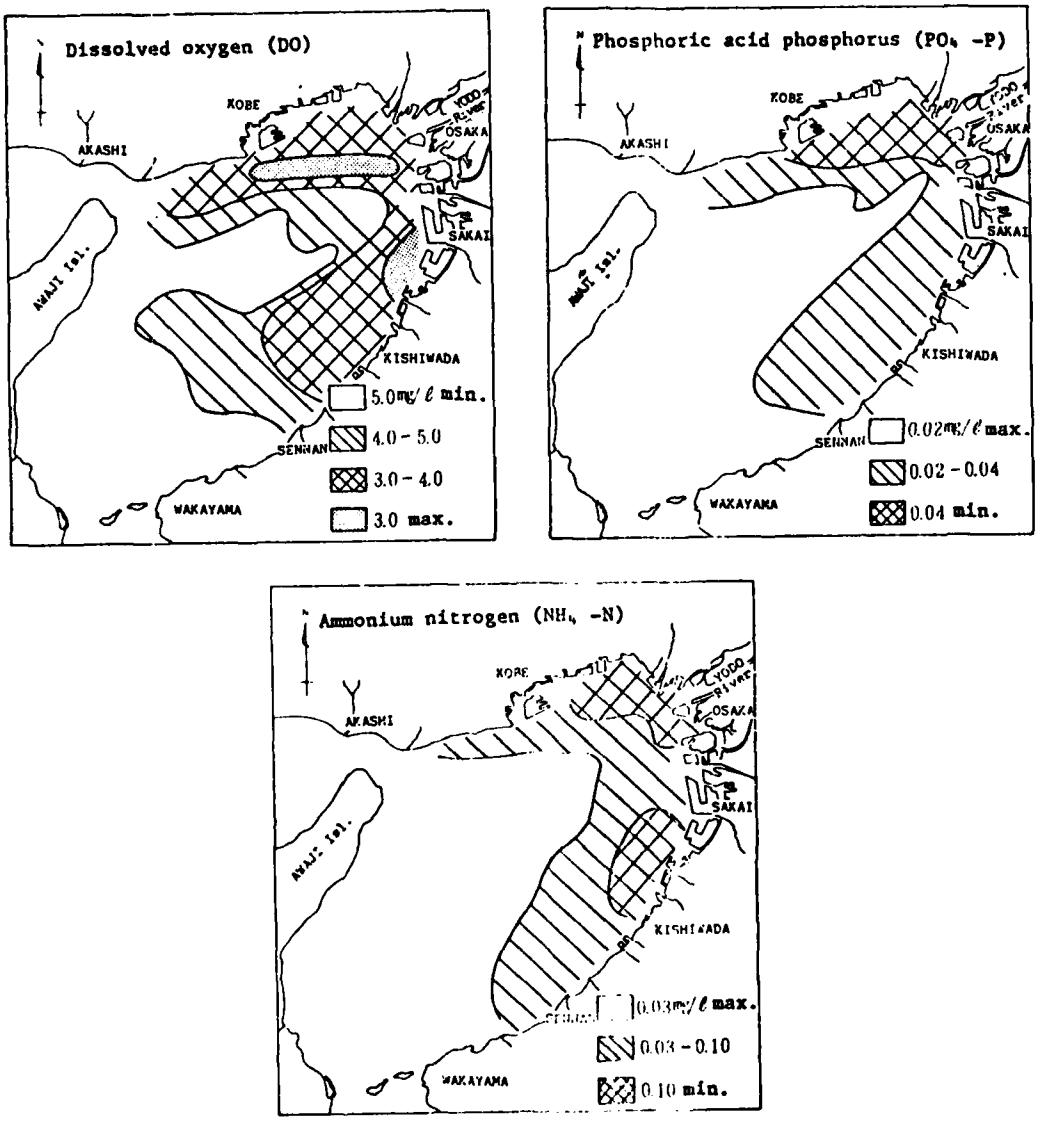
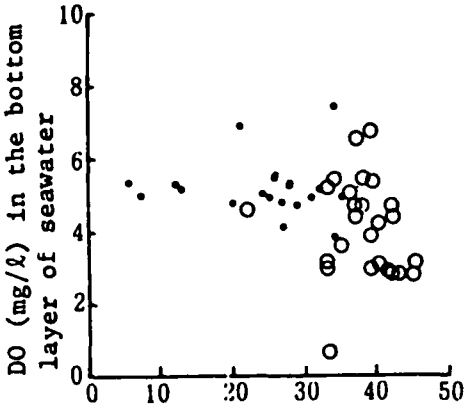
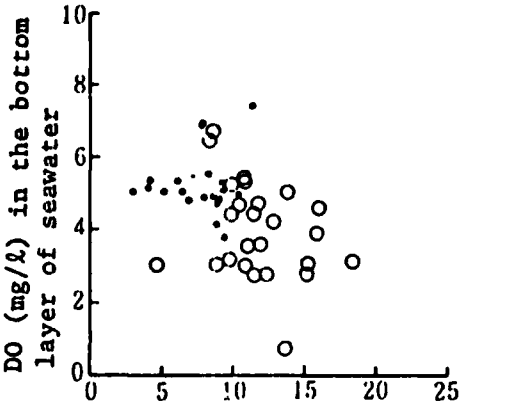
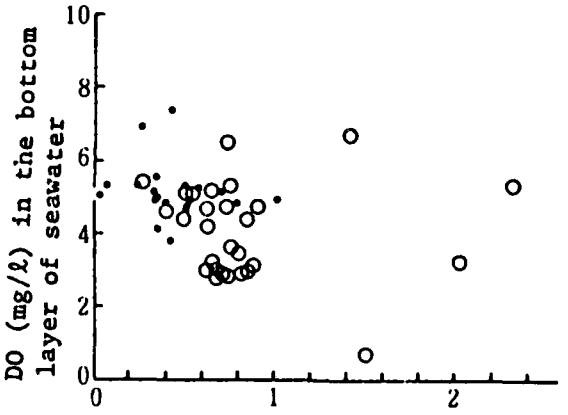


Figure 5. Distributions of the concentration of DO, N, and P in the water of the bottom layer

largely affected by the levels of pollution in the surface sediments. The relationship between these two layers is shown in Figures 6-8 where the abscissa represents pollution index of the bottom water layers at each monitoring station, and the ordinate the pollution index of the bottom sediments. When COD, sulfide, and IL in the bottom sediments are 30 mg/g and more, 0.6 mg/g and more, and 10 percent and more, respectively, the concentration of DO in the bottom water layer tends to decrease, while the concentrations of inorganic phosphorus (PO_4-P) and ammonium nitrogen (NH_4-N) show marked increase. This phenomenon is particularly remarkable in the shallow seawater areas less than 20 m deep.







Example

Bottom layer of seawater :values measured in summer at 2 m above the sea bottom

Surface layer of the bottom sediment: values measured in summer in the uppermost 0 - 5 cm layer

○ Depth of water≤20 m

● Depth of water≥20 m

Figure 6. Relationship between DO in the bottom layer of seawater and surface layer of the bottom sediment

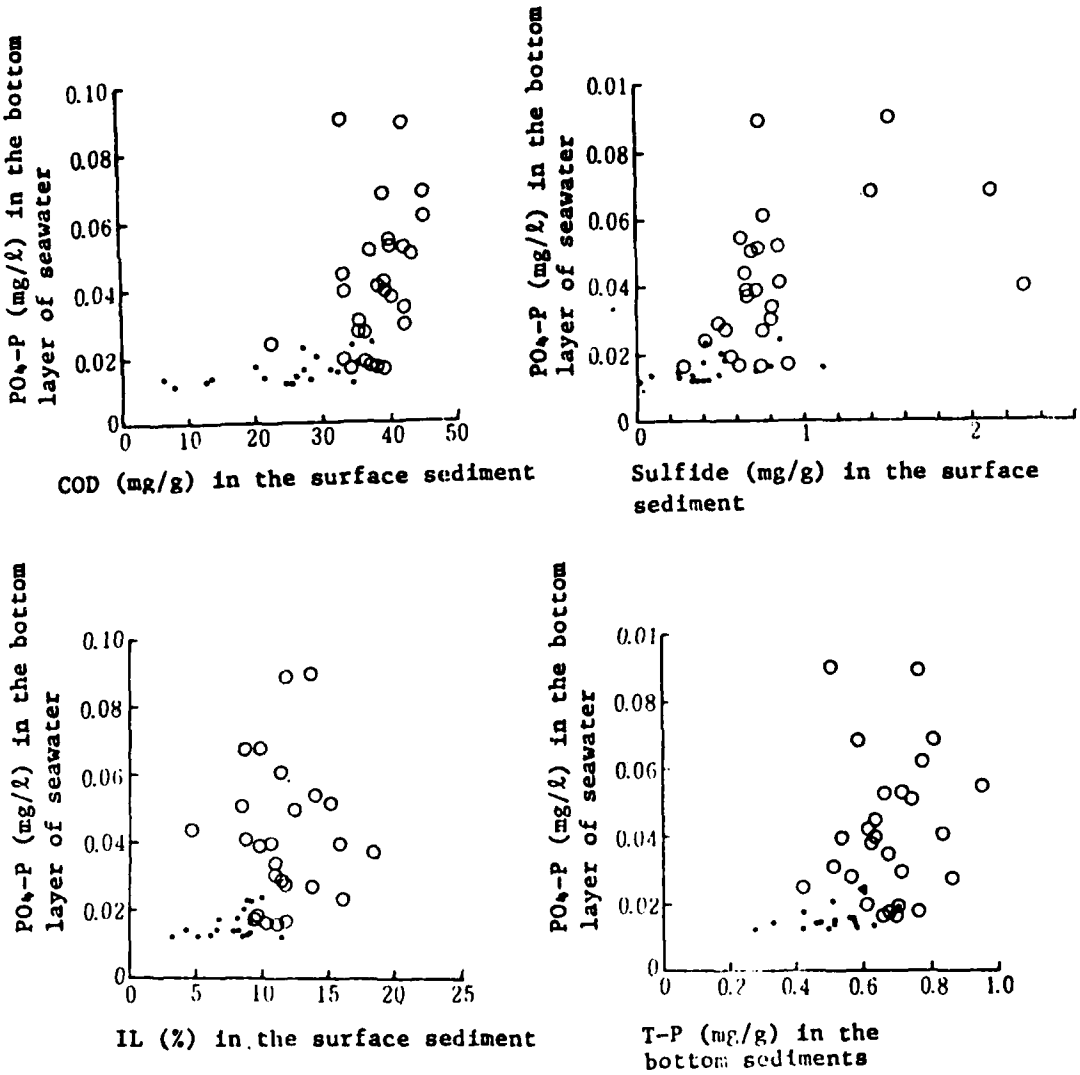
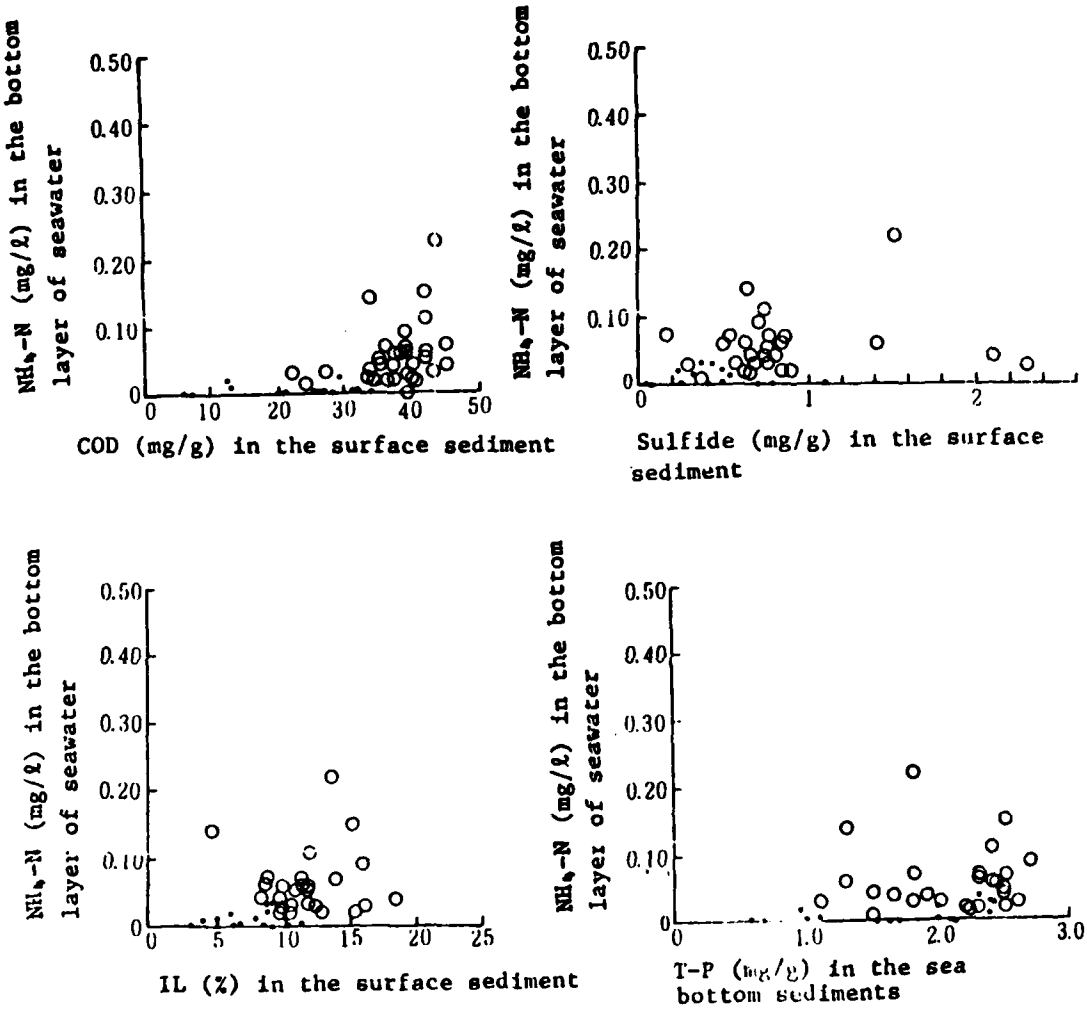


Figure 7. Relationship between phosphoric acid phosphorus (PO_4-P) and the surface layer of the bottom sediment

Release of Index Substances from the Bottom Sediments

In the distribution of the areas of the polluted bottom water layer, as seen in the case of the polluted bottom sediments, the pollution is advanced in the offshore bay area, suggesting a strong influence by the pollution in the bottom sediments. Hence, the release rates of the index substances were determined for their distribution in the bottom water layer by using the



Bottom layer of seawater: values measured in summer at 2 m above the sea bottom

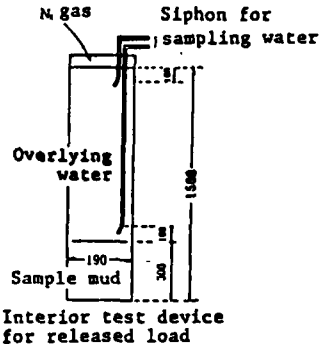
Surface layer of the bottom sediment: values measured in summer in the uppermost 0 – 5 cm layer.

○ Depth of water≤20 m

● Depth of water≥20 m

Figure 8. Relationship between ammonium nitrogen (NH_4-N) in the bottom layer of seawater and the surface sediments

interior released load test device shown in Figure 9. The sediment cores were obtained at each monitoring station from the 0- to 30-cm layer, the 50- to 80-cm layer, and the 100- to 130-cm layer of the bottom sediments. The release rate was calculated by Equation 1 using the values in the time course of the concentrations of the released substances in the overlying water. (Note: Hereafter, the released load of COD and the release rate of COD mean the released load and the released rate of organic substances determined by COD, respectively.)



(Note) The release rate means the released volume per day, and is obtained from the gradient of the straight regression line relating the elapsed days from the start of the test to the released volume calculated by equation (1) using the change of concentrations in the overlying water in the released load test device.

$$M_i = \frac{\sum_{i=1}^{n} V_0 - \sum_{i=1}^{n} V_{i-1} \quad C_i - C_{i-1}}{S} \qquad (1)$$

M_i: Released load (mg/m^2) from the start of the test to the ith sampling day

V_0: Initial volume of the overlying water (ℓ)

V_i: Sampling volume for the ith day (ℓ)

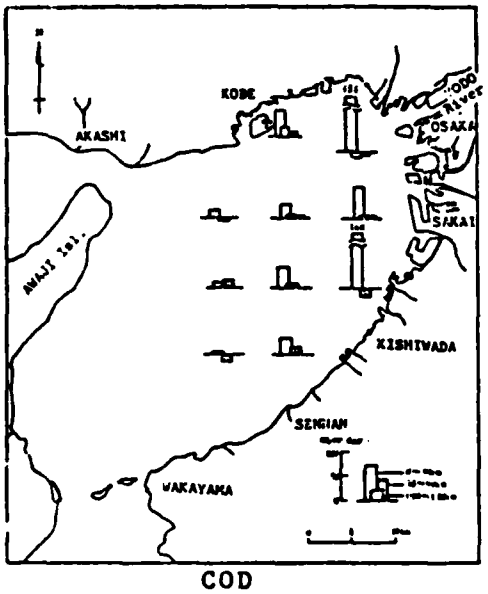
C_i: The concentration of released substances in the overlying water at the ith sampling (mg/ℓ)

S: Surface area of the bottom sediment in the device (m^2)

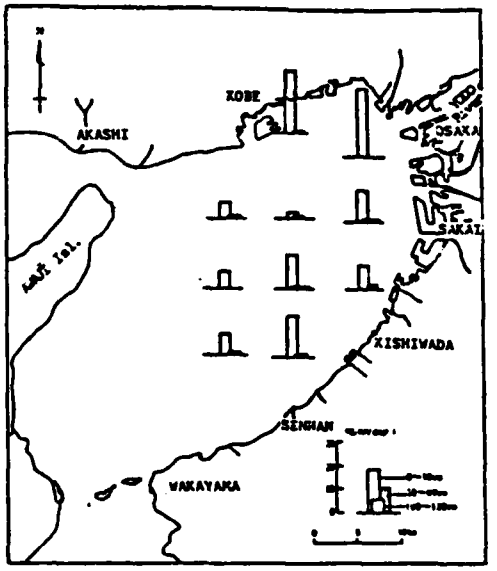
Figure 9. Test device for released load

Figure 10 shows the release rates for COD, T-P, and T-N at 10 monitoring stations at Osaka Bay. The release rates for COD, T-P, and T-N are all higher in the areas of the polluted bottom sediment in the offshore bay area. Their release rates are markedly lower in the 50- to 80-cm and 100- to 130-cm layers from the sediment surface as compared with the values of the uppermost 0- to 30-cm layer. Therefore, it can be assumed that removal of the uppermost 50-cm layer will lead to improvement of the water quality in Osaka Bay.

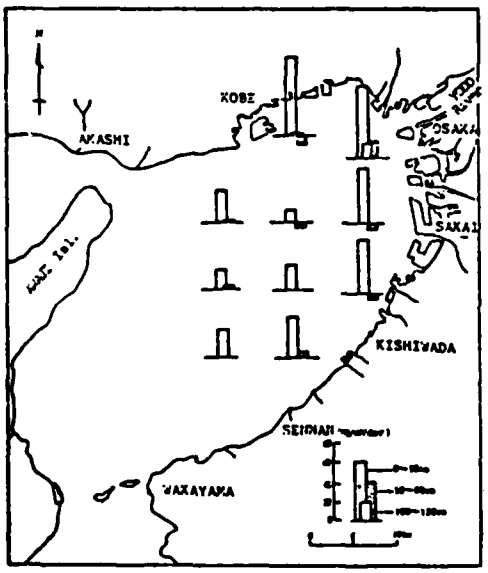
The comparisons between the release rates of T-P and inorganic phosphorus (PO_4-P), and between those of T-N and ammonium nitrogen (NH_4-N) at each monitoring station are shown in Figure 11. The figure shows that the majority of the released phosphorus is inorganic phosphorus (PO_4-P), and that of the released nitrogen is ammonium nitrogen.



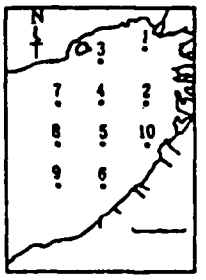
COD



T-P



T-N



Temp.: 25℃

DO: At the first stage

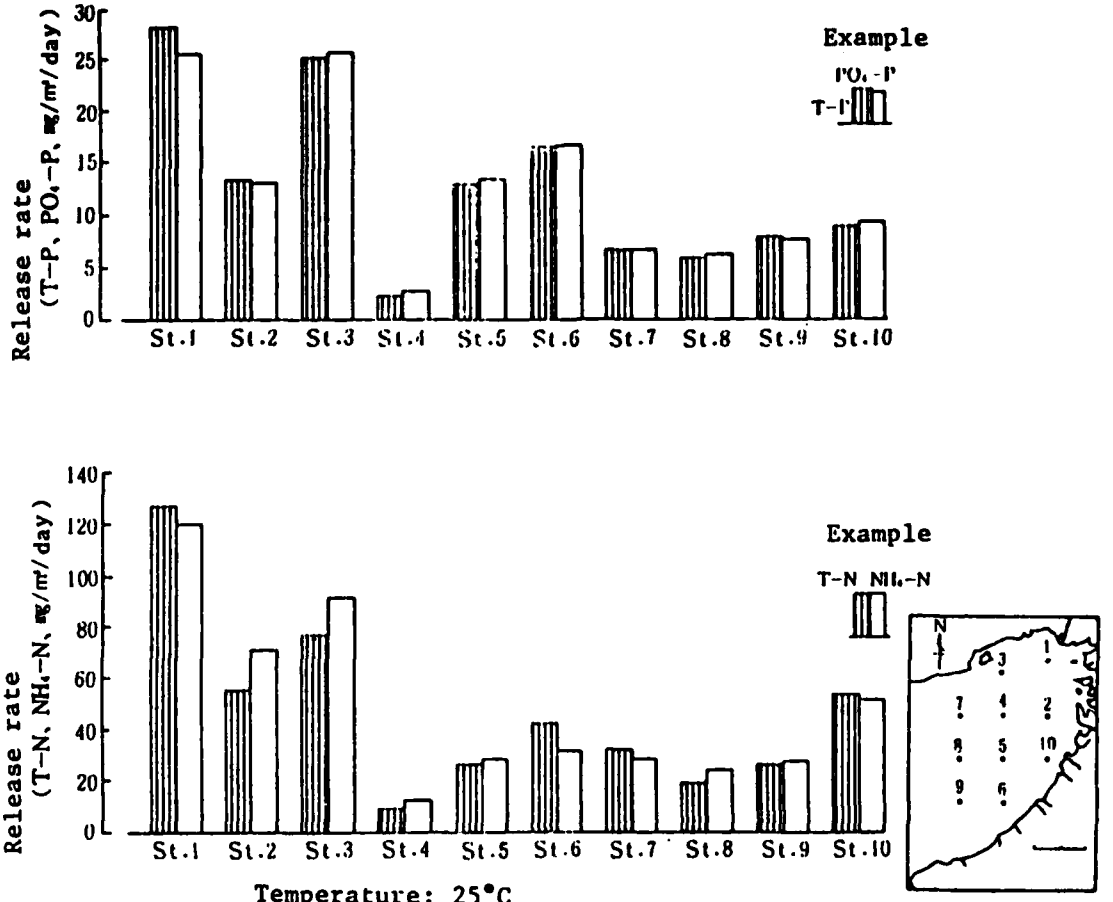
St.1, St.10---1 mg/ℓ

Others--------5 mg/ℓ

(Unit: $mg/m^2/day$)

August 1981

Figure 10. The release rate of COD, T-P, and T-N from bottom sediments



Temperature: 25°C
DO: 1 mg/ℓ in the initial period at St. 1 and St. 10 and 5 mg/ℓ at other stations
Interior test: Sealed airtight and left (August 1981)

Figure 11. Release rate in Osaka Bay

Figure 12 demonstrates the release rates of phosphorus and nitrogen in relation to the bottom sediment.

There is a definite relation between the content of T-N in the bottom sediments and release rate for PO_4-P and between the content of T-N in the bottom sediment and the release rates for NH_4-N. It is also shown that under circumstances where the contents of T-P and T-N are over 0.65 mg/g and 0.2 mg/g, respectively, there is a sharp increase of released PO_4-P and NH_4-N. Figure 12 plots the results of the released load test on the sediments with the uppermost 50-cm layer removed. This figure reveals that removal of this surface layer will lead to improved bottom sediment and a substantial termination of release.

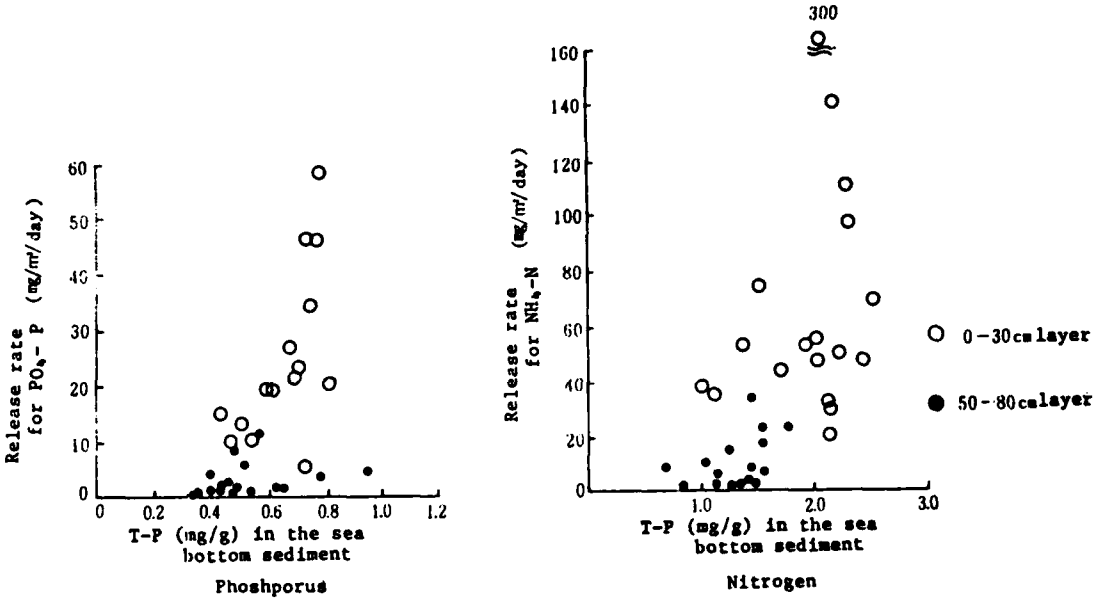


Figure 12. Relationship between release rates and bottom sediments

Relationship Between Pollution of Bottom Sediments and Benthos

The polluted bottom sediments and the deteriorated quality of the bottom layer of water are considered to have an adverse effect upon the benthic biota. Figures 13 and 14 show the number of benthos species and diversity indexes, respectively. As the areas of the polluted bottom sediment spread, the seawater areas of poor biota where only limited ecospecies can survive expand in the offshore bay area. It should be noted that there are areas where the inhabitants' size is not more than 20/0.1 m^2 (Figure 15).

Figure 16 shows the relationship between the surface sediments and the diversity index of the benthos. This figure shows a decreased diversity index in the areas with increased COD, IL, and sulfide contents in the bottom sediments. The grain size of the bottom sediments, the quality of seawater, and seawater movement are generally considered to be the major controlling factors of the ecological distribution of the benthos. Accordingly, discussion of this problem solely from the viewpoint of the pollution levels of the surface sediments will not be sufficient. Nonetheless, the pollution level in the surface sediments seems to be an important factor to control the environment for the benthos.

Mechanism of Pollution

In view of the results of our investigation, the pollution of Osaka Bay seems to be caused by the mechnism as shown in Figure 17.

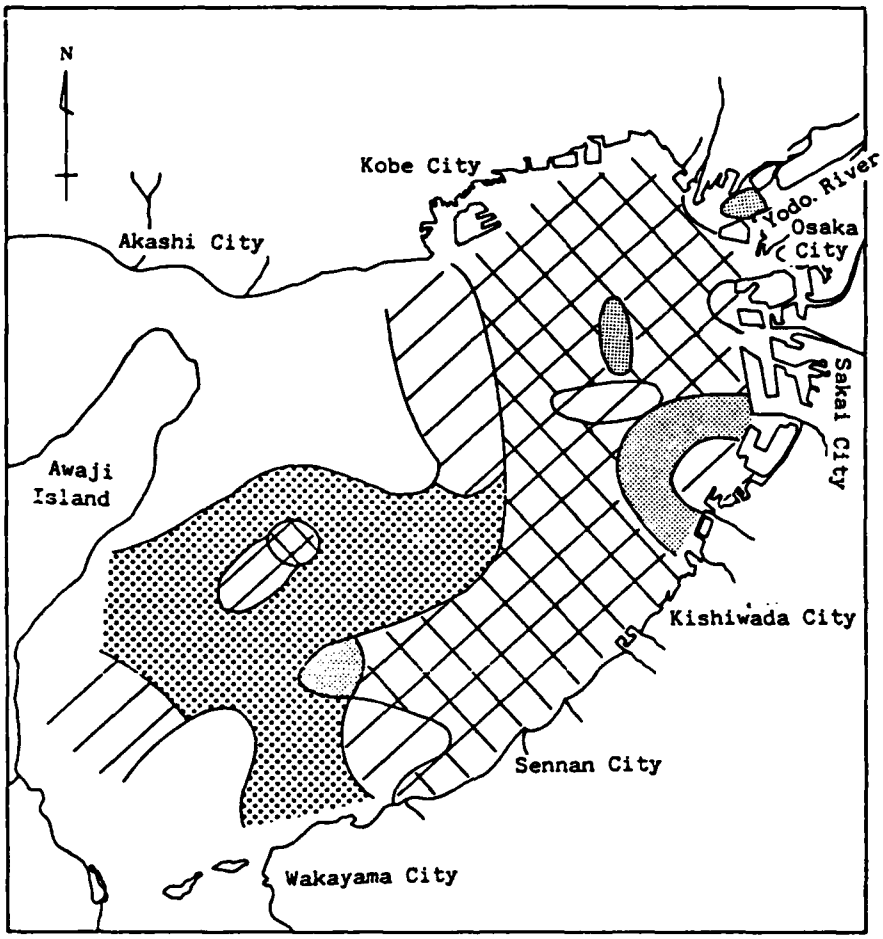
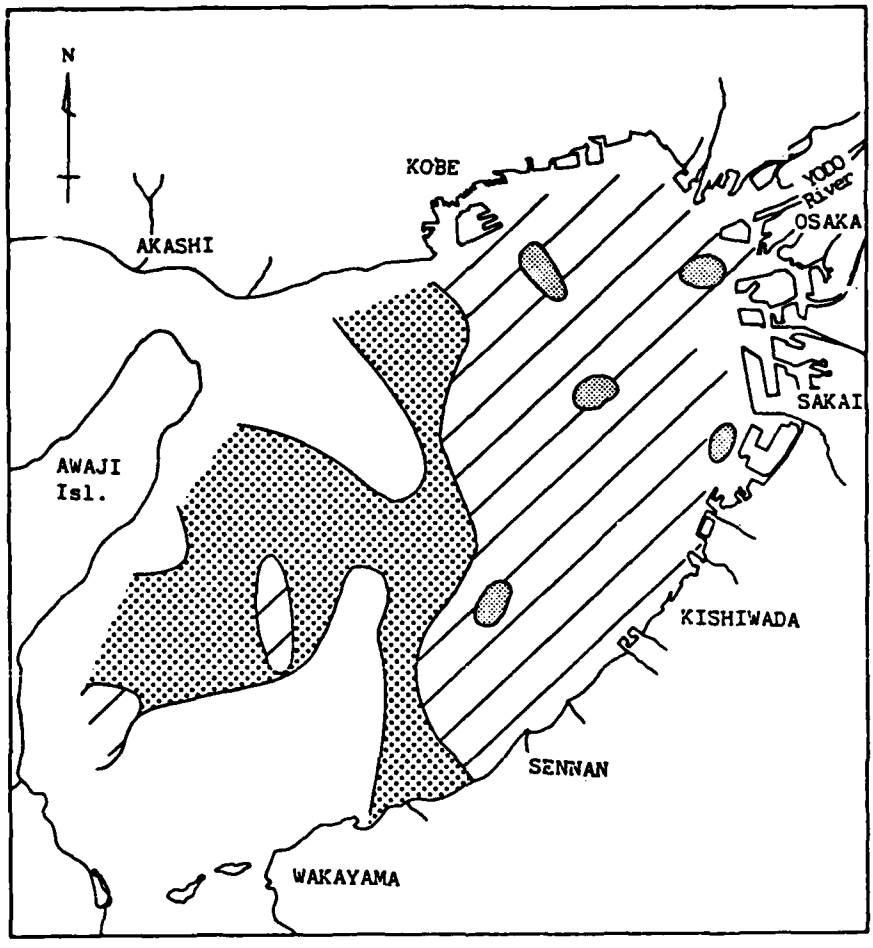


Figure 13. Distribution of the number of species of benthos



1 max
1-2
2-25
25 min.

Diversity index

$$H(S) = -\sum_{i=1} \left| \left(\frac{Ni}{N}\right) \ln\left(\frac{Ni}{N}\right) \right|$$

H(S): Diversity index
Ni : Population of the first species
N : Total population of individual species

Figure 14. Distribution of diversity indexes of benthos

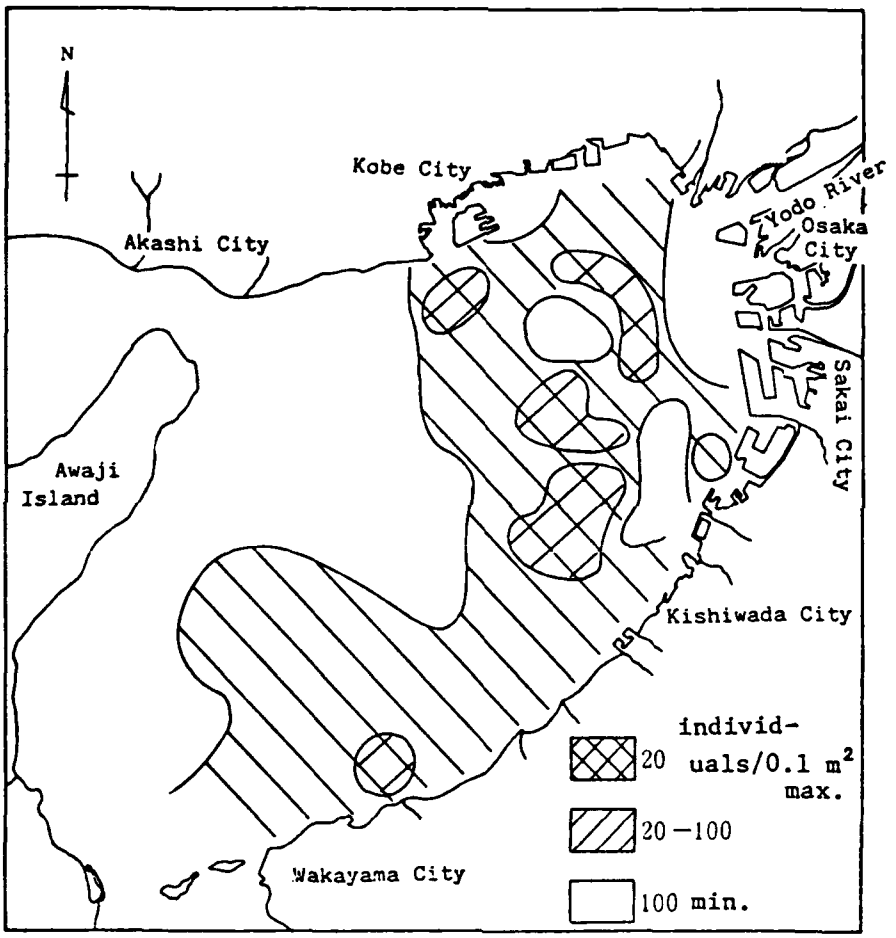


Figure 15. Distribution of population of benthos

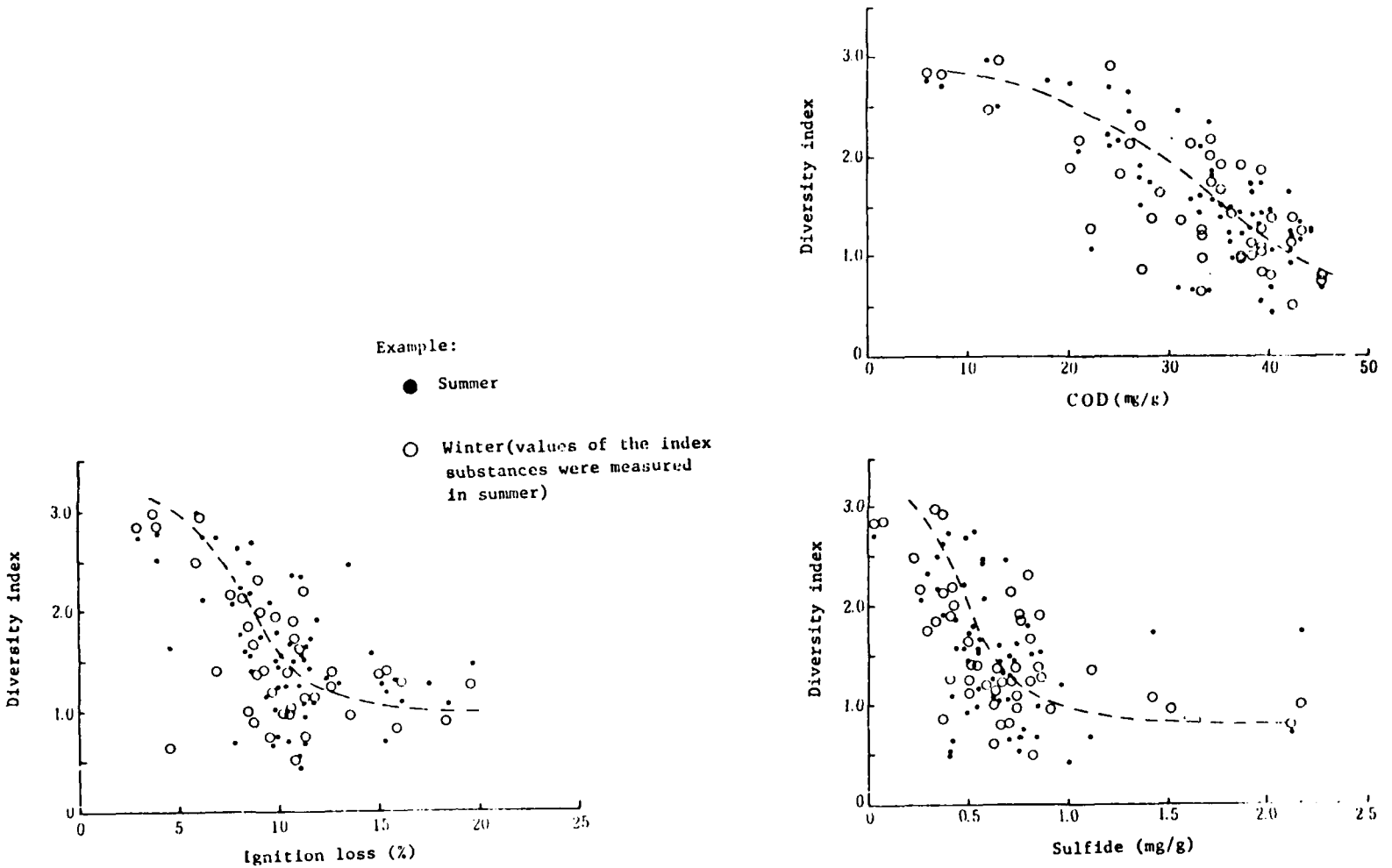


Figure 16. Relationship between the surface sediment and diversity index of benthos

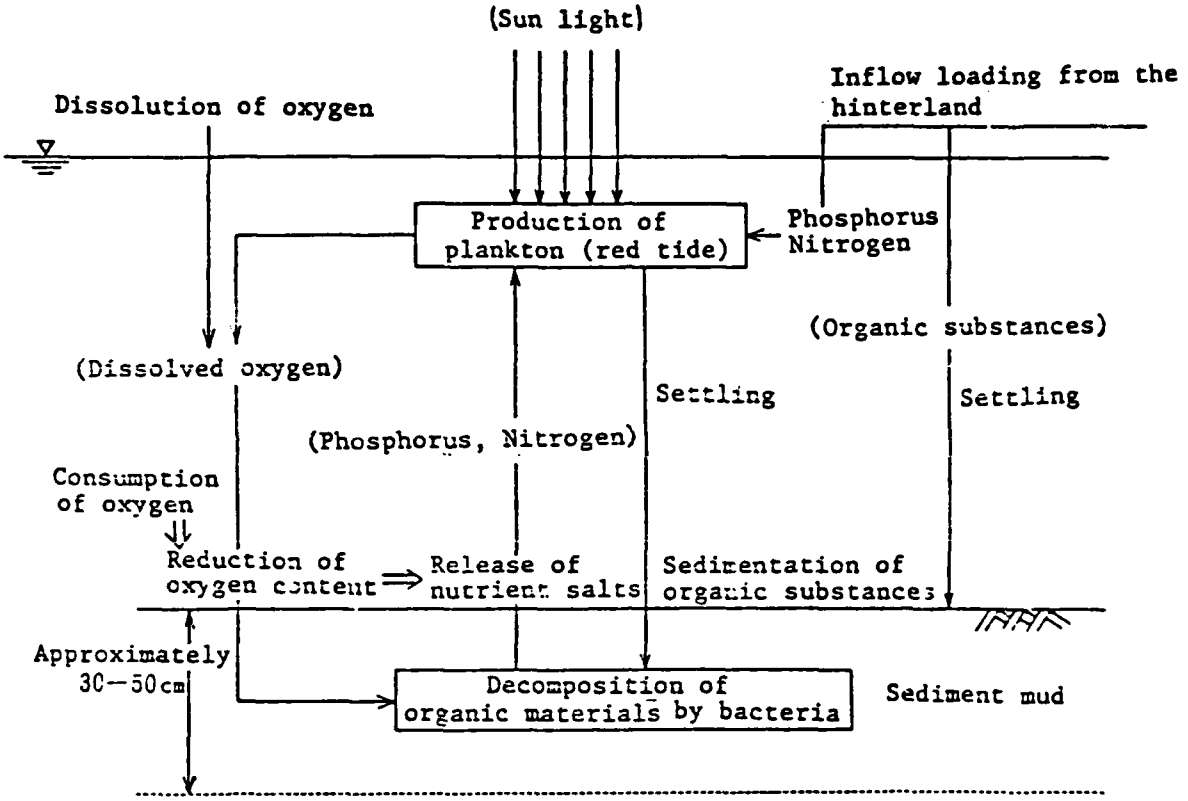


Figure 17. Mechanism of pollution in Osaka Bay

a. Organic materials in the sludge deposited on the sea bottom are decomposed by bacteria.

b. During decomposition, the dissolved oxygen in the bottom water layer is consumed to reduce the oxygen content of the bottom layer of water.

c. As the reduction of oxygen content in the bottom water layer proceeds, the layer is reduced to an anaerobic condition.

d. Under anaerobic circumstances, nutrients (N and P) are released from the bottom sediments.

e. Increased nutrient salts in the seawater caused an increase in the productivity of plankton. Abnormal increase in the productivity may promote occurrences of red tide.

f. The debris of the dead plankton deposit to form organic sediments.

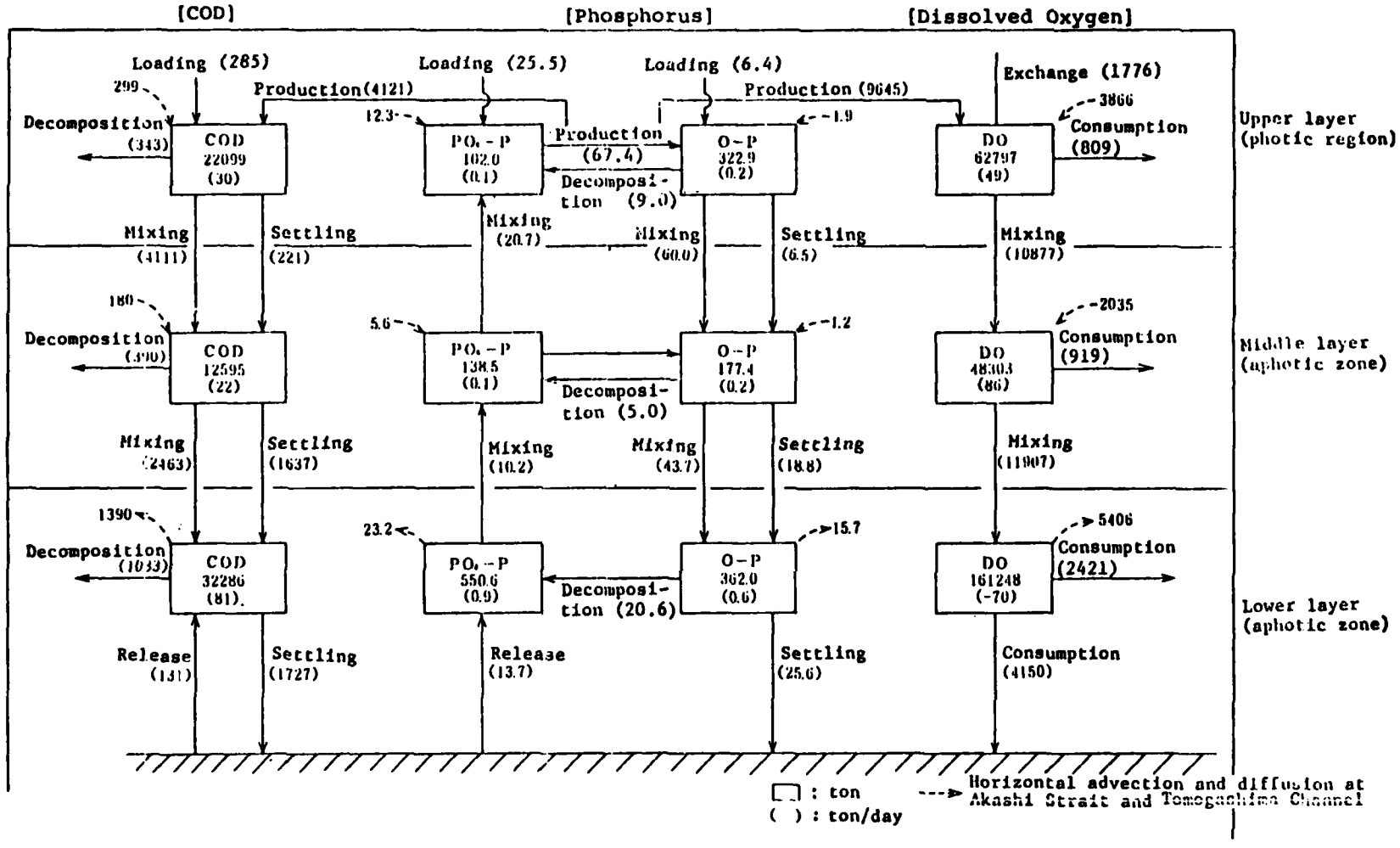


Figure 18. Mass flux of COD, phosphorus, and dissolved oxygen (Osaka Bay, Summer)

This mechanism was simulated using mathematical models. Figure 18 shows the balance of each material calculated using coefficients for summer conditions. The ratio of inflowing load from the coastal area to the released load from the bottom sediment in terms of phosphoric acid phosphorus is 25.5 tons/day against 13.7 tons/day; i.e., it was in the ratio of approximately 2:1. This provides evidence that the released load from the bottom sediments comprises the major factor of water pollution.

BOTTOM SEDIMENT CLARIFICATION PLAN

Removal of Bottom Sediment

The following effects are expected from removing the highly polluted bottom sediment and the resultant improvement of the bottom environment.

a. Removal of the bottom sediments rich in organic matter will reduce the sediment oxygen demand and thereby prevent the appearance of anaerobic water masses in summer.

b. Removal of the bottom sediment high in organic and nutrient concentration, along with the resultant reduction of dissolved oxygen consumption, will lead to decreased nutrient release, which is a factor for eutrophication as seen in the frequent occurrences of red tides.

c. By improving the quality of the bottom sediment and seawater, the present benthos and other species whose variety is decreasing may be improved.

Bottom Sediment Clarification

It is necessary to provide standards for bottom sediment clarification on the basis of the investigation results mentioned earlier, to the effect that efficient improvement of the bay environment may be attained.

The following are example standards for setting up areas to be clarified using the pollution indexes of the bottom sediment.

a. Standard by the organic pollution indexes:

$COD \geq 30$ mg/g (a)

Ignition Loss $\geq 10\%$ (b)

Sulfide ≥ 0.6 mg/g (c)

b. Standard by nutrient release:

$T\text{-}P \geq 0.65$ mg/g (d)

$T\text{-}N \geq 2.0$ mg/g (e)

Figure 19 shows the area where two out of three conditions from (a) to (c) and both conditions of (d) and (e) are satisfied.

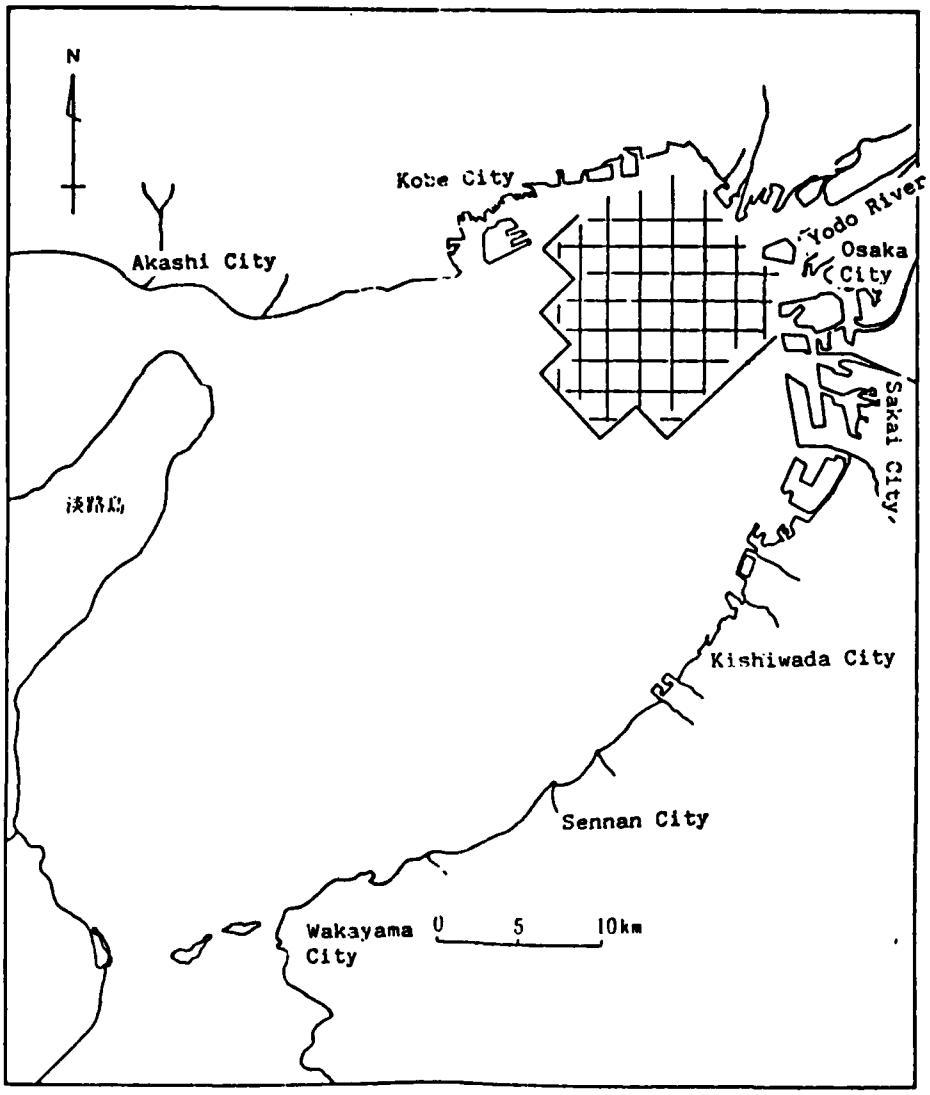


Figure 19. Areas where the bottom sediment is seriously polluted

Methods of Bottom Sediment Clarification

Choice of Clarification Methods

The bottom sediment clarification methods are classified into two groups: (1) the dredging method to remove sediments from the present site, and (2) the field (on-the-spot) treatment method by confining the contaminants at the present site (Figure 20).

The dredging method has a number of successful records in the port pollution prevention works in the past. Furthermore, in such a highly utilized bay as Osaka Baky, this method, when combined with reclamation and beach nourishment, presented in the port development and utilization plan, will make it possible to plan efficient and effective sea clarification works.

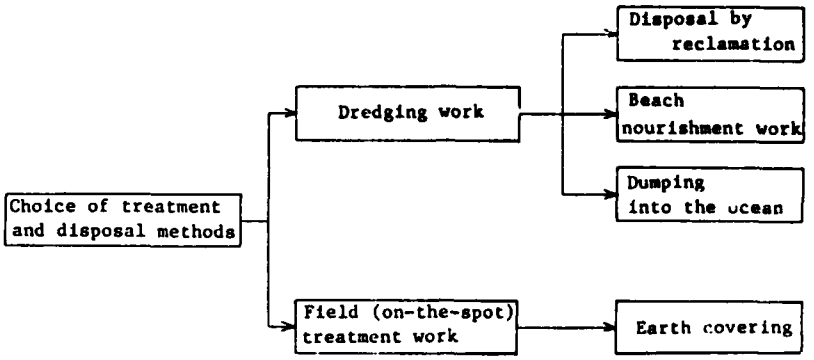


Figure 20. Choice of treatment and disposal methods

There are three ways for disposing sediment dredged by this method: (1) disposal by reclamation, (2) disposal by dumping in the ocean, and (3) disposal into beach nourishment sites (this method is similar to (1)). In the case of reclamation, the reclamation work must be carried out according to the current social demand for land and only when the utilization plan of the reclaimed land is definite.

In the case of dumping in the ocean, although the cost is low, no land can be produced.

In the case of dumping into beach nourishment sites, dredged sediment is utilized as material for forming a beach. This method can meet the social demand for waterside areas.

For the field (on-the-spot) treatment method, earth covering is the primary way to dispose of the sediment. In this method, sand is overlaid on the bottom sediment in the target areas to confine the sediments at the present site within the layers of the bottom.

When considering both the degenerating capacity of nature's self-purification system due to the reduced natural beach areas and tide lands and the shortage of recreation sites for people, it can be concluded that the beach nourishment method meets the social demand.

Figure 21 is a diagram of dredging-beach nourishing work. Boats used exclusively for sediment dredging that generate little turbidity were employed for dredging organic sediments approximately 50 cm thick. The dredged sediment is carried to a disposal site by the boats for transportation and discharge. After the desired volume of sediment is dumped and adequately consolidated, sand is overlaid on the disposed sediment to nourish the beach.

Technological Problems in the Bottom Sediment Clarification Works

The bottom sediment clarification works (by dredging and dumping the dredged sediment into beach nourishment sites) is classified into steps: dredging work, disposing of the dredged sediment, and nourishing the beach.

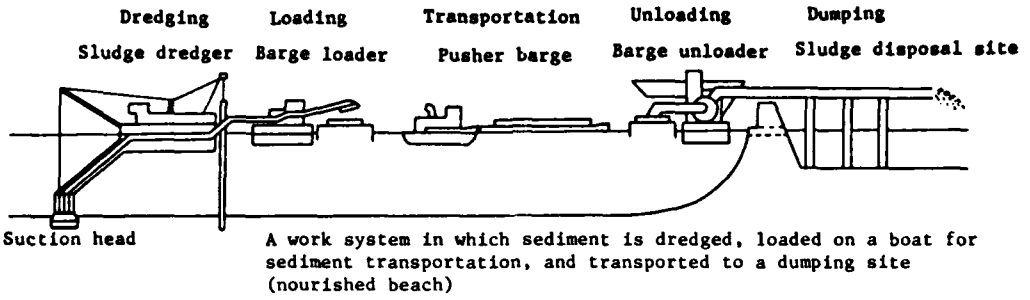


Figure 21. Imaged diagram of work system

Figure 22 is the flowchart summarizing the technological problems drawn by examining the requisites for these works based on the premises of the work.

Dredging. Under the premise that the sediment to be dredged is thinly deposited over a wide area, the technological problem is to establish a technique of dredging the thinly deposited sediment. By this technique, the volume of sediment dumped into disposal sites will be reduced, and the reduction of the size of the disposal sites will have a considerable economic influence. Accordingly, upon the recognition that the problem of developing the thin layer dredging technique is an independent technological theme, we are going to solve this problem by examining the past records concerning this technique and by conducting necessary investigations and experimental work.

Disposal. The next premise is disposal of massive volumes of dredged sediment. The requisites for this premise are (1) stabilization of embankments, (2) efficient disposal, and (3) stabilization of disposed sediment at the disposal site in an early period.

The problem with designing a stabilized embankment (No. 1 above) is how to estimate the pressure of earth and water upon the embankment. Since the sediment to be disposed is super soft with a high water content, it is necessary to devise an efficient disposal technique for this type of sediment (No. 2 above). Requisite No. 3 arises from the necessity to accelerate the dehydration and consolidation of disposed sediment after its disposal for prompt and smooth beach nourishment. Out of requisites 2 and 3, the following five technological problems were extracted: high concentration dredging, separation of solid and fluid phases, high concentration discharge, understanding the characteristics of volume change of disposed sediment with time, and acceleration of consolidation.

Nourishment. The requisites for beach nourishment include: (1) feasibility of earth covering work where the overlaid sand is stabilized, (2) consideration for scouring and sand drift phenomena after beach nourishment is complete (3) methods to prevent immersion of the contaminants from the sediments under the overlaying sand (a requirement especially needed after the completion of beach nourishment), (4) exchange of seawater, and (5) water

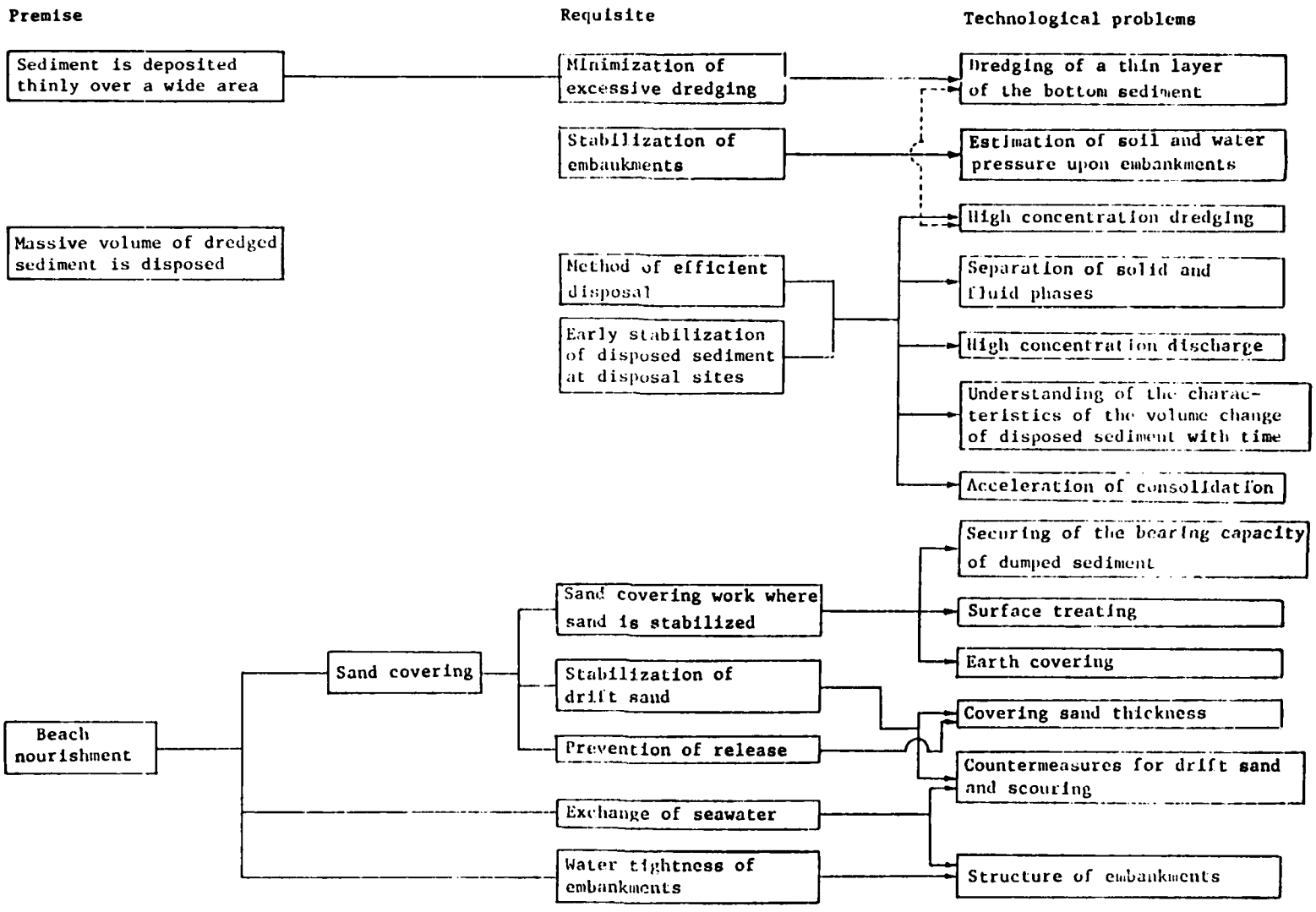


Figure 22. Presentation of the technical problems

tightness of the embankment. Out of these five requisites pertaining to beach nourishment work, the following six items are drawn to be technologically solved: (1) securing the surface-bearing capacity of dumped sediment, (2) choice of surface treatment method, (3) earth covering technique, (4) earth covering thickness, (5) measures against drift sand and scoring, and (6) structure of embankments.

Upon examining the methods mentioned above, 13 technological problems were extracted to be solved. Researchers at the 3rd District Port Construction Bureau are seeking the solution of these 13 technological problems by conducting necessary experiments and investigations.

CONCLUSIONS

This paper reports on the study of the bottom sediment clarification conducted by the 3rd District Port Construction Bureau to improve the marine environment in Osaka Bay with the primary focus set upon the influence of the accumulating sediments on the marine environment.

The followings are the main conclusions:

a. Pollution of the bottom sediment in Osaka Bay is serious in the offshore bay area.

b. There is a positive relationship between the areas of polluted bottom sediment and the areas of degenerated quality of the bottom water layer.

c. Nutrient release is prominent in the offshore bay area.

d. Contaminant release occurs in the uppermost 50-cm layer of the bottom sediment. Consequently, removal of this layer will eliminate the release phenomenon.

e. The clarification method of the bottom sediment in Osaka Bay that can meet the social demand is the dredging-beach nourishment method.

The authors are committed to finding the solution to the technological problems involved in this work.

RESULTS OF SEDIMENTATION AND ELUTRIATE TESTS FROM OSAKA BAY SEDIMENT

H. Kosasa, K. Nikaido, and M. Natori

Japan Sediments Management Association

ABSTRACT

A study on sediment removal as it affects seawater pollution in Osaka Bay was performed by the Japanese government. In relation to this, it has become necessary to characterize dredged material regarding sedimentation and elutriates. Consequently, sedimentation and elutriate tests were conducted in 1983. This paper reports the test results.

SEDIMENT PROPERTIES

The sediment samples were taken from the area indicated in Figure 1. Sediment quantities of 150 ℓ were sampled from four points. The grain-size distribution was obtained by three methods: conventional soil mechanics methods, photo-extinction, and coulter counter (Table 1). The chemical and physical characteristics of the sediment in Osaka Bay are shown in Table 2.

SEDIMENTATION TESTS

The sedimentation tests were carried out principally to learn the effects of coagulants on sedimentation. The results are shown in Figures 2 and 3. From these curves it is readily apparent that the settling velocities decrease with increasing sediment concentration. Therefore, dredging with higher sediment concentration is not always advantageous from the point of view of sedimentation. Figure 4 is a comparison of sediment curves with and without coagulants for the same sediment content.

From the sedimentation curves the void ratio of the settled material at each point can be determined. The initial void ratio of the sediment as it lies on the sea bottom is 6.43 ($w = 240\%$, $\gamma s = 2.76$). The volumes of the solid particles, expressed as percentages of the total volume, were 3.51%, 7.79%, and 13.6%, respectively, for the three sediment concentrations of 10%, 25%, and 40%.

Since the initial concentrations of the dredged material are known, the heights of solid to be settled (Figure 5) corresponding to the initial

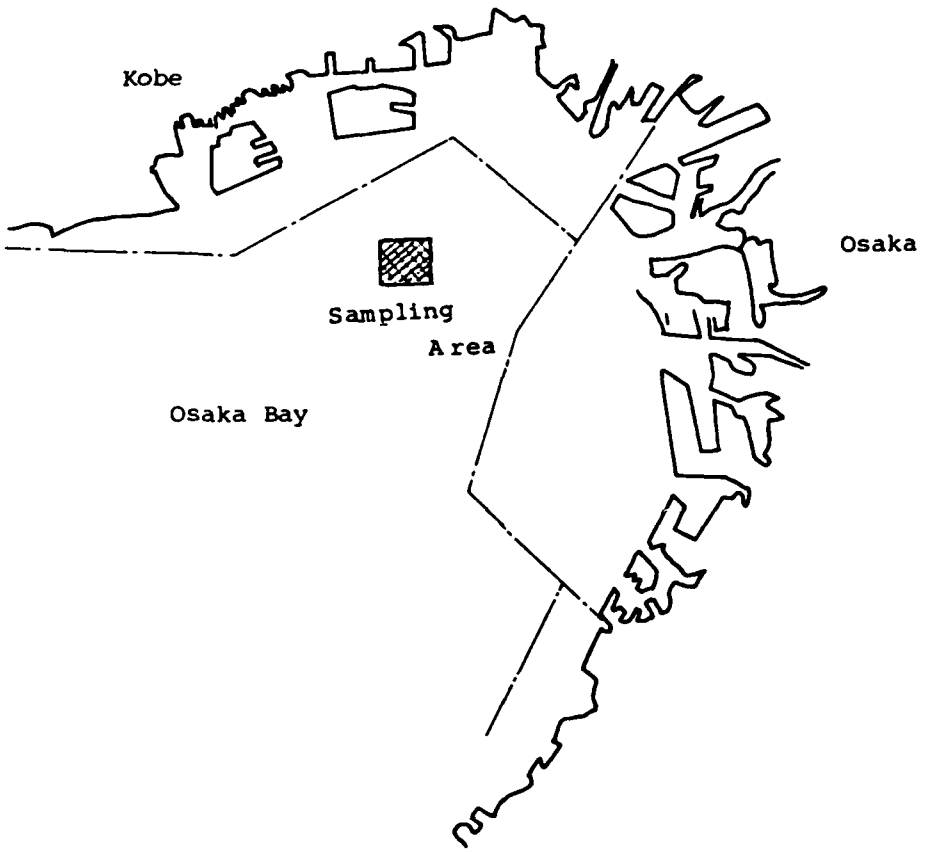


Figure 1. Sampling area

TABLE 1. GRAIN SIZE DATA FOR SEDIMENT SAMPLES

Size	Soil Mechanics Test	Photo-Extinction	Coulter Counter
Sand	0	0	0
Silt	82.5%	46.0%	55.7%
Clay	17.5%	54.0%	39.5%
D_{60}%	74 μm	55 μm	110 μm
D_{30}%	25.5 μm	25 μm	33 μm
D_{10}%	10.7 μm	11.5 μm	10 μm

TABLE 2. CHEMICAL AND PHYSICAL PROPERTIES OF SEDIMENT SAMPLES

Appearance	Silty and clayish	Ignition loss	11.0%
Odor	H_2S	COD	39.0 mg/g
pH	8.4	TOC	3.31%
Wet weight	1.206	T-N	2,900 mg/kg
Dry weight	2.761	NH_4-N	95 mg/kg
Water content	240.0%	T-P	830 mg/kg
Liquid limit	135.3%	PO_4-P	230 mg/kg
Plastic limit	33.8%	ORP	-75 mV

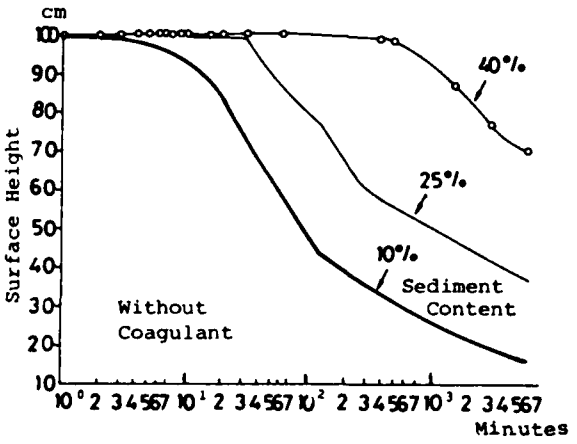


Figure 2. Sedimentation curves without coagulant

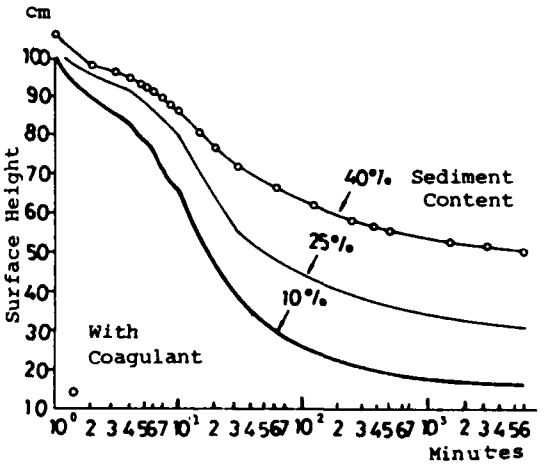


Figure 3. Sedimentation curves with coagulant

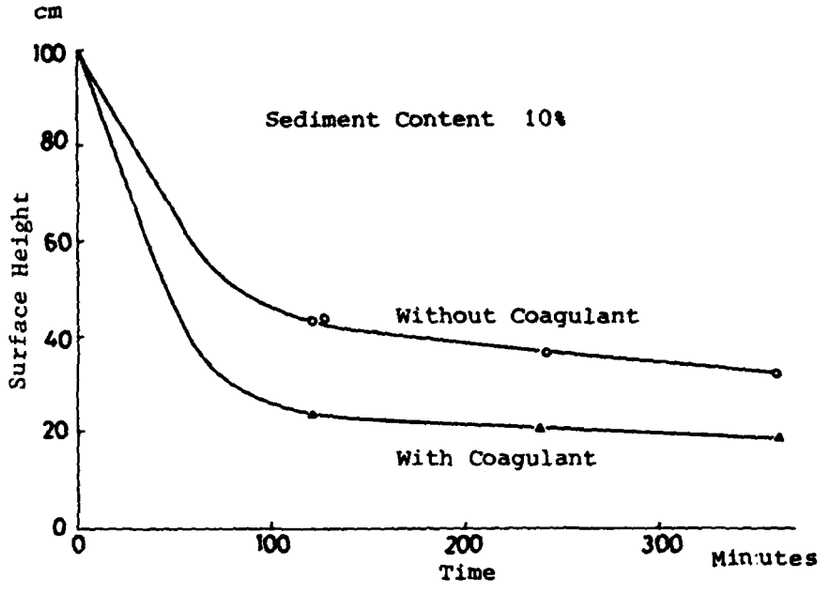


Figure 4. Sedimentation curves with and without coagulant

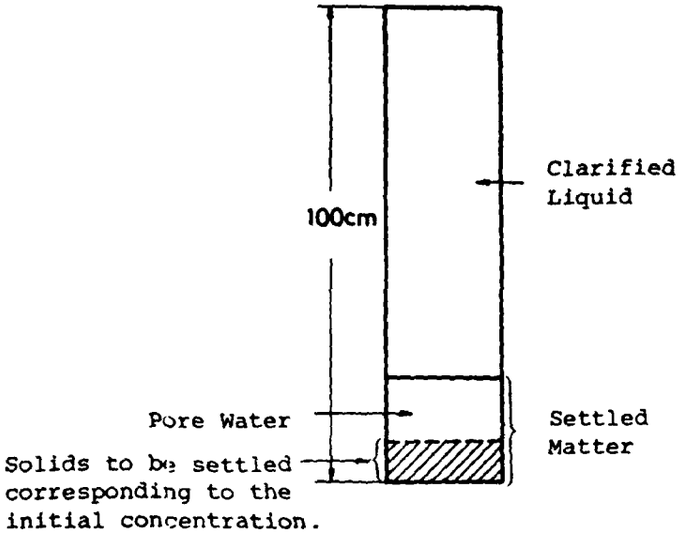


Figure 5. Settled matter in column

concentration are obtained as shown in the tabulation below. However, the actual height of the settled material is much larger because it contains water. The void ratios are calculated as shown in Equation 1. (The concentrations of solids in the clarified liquid were considered negligible.)

$$e = \frac{\text{Volume of voids}}{\text{Volume of solids}} = \frac{b - a}{a} \tag{1}$$

where a and b are as tabulated below.

Sediment Content	a Solids to be Settled	b Settled Matter (t = 480 minutes)	Void Ratio
10%	1.27 cm	31.3 cm	23.61
25%	2.82 cm	55.7 cm	18.73
40%	4.93 cm	98.2 cm	18.73

The ratio of volume occupied in the pond to volume occupied in the channel E is an important factor for determining the receiving capacity of a pond. The variable E is defined as:

$$E = \frac{e + 1}{e_o + 1} \tag{2}$$

where

e = void ratio in the pond

e_o = void ratio in the channel

If E is larger than one, it is the swelling ratio of dredged material. But the value of this ratio varies against time. Figure 6 shows the relationship of E versus time. The value of E lies in a swelling zone for a settling time of over 1440 min.

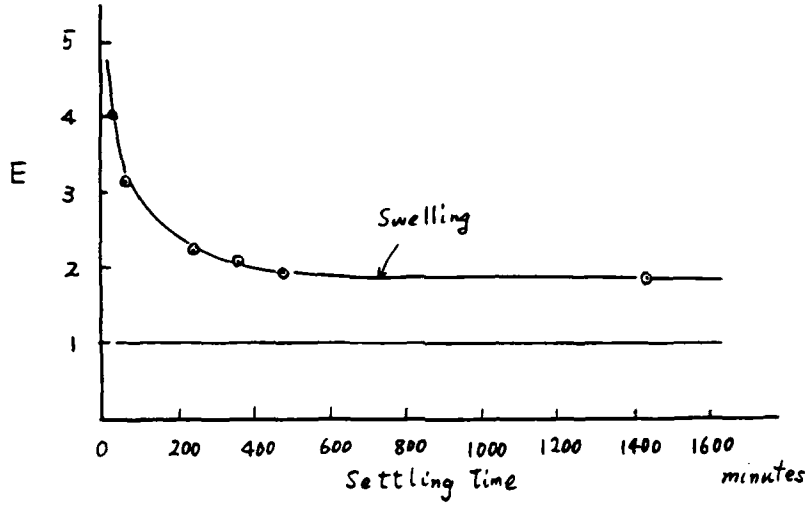


Figure 6. Variation of void ratios for 10% sediment content without coagulant

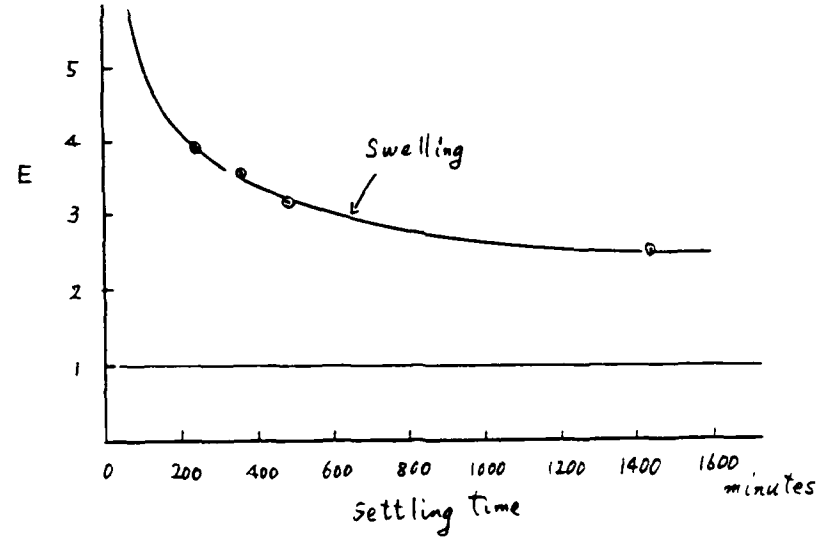


Figure 7. Variation of void ratios for 10% sediment content with coagulant

ELUTRIATE TESTS

Elutriate tests are not standardized in Japan as in the United States. Therefore, the tests were performed in the manner most usually adopted in Japan. Their purpose is to learn the effects of sediment content on elutriates. The tests were conducted under the following conditions:

a. Flusco vessel volume: 3 ℓ

b. Sediment content: 10, 25, 40%

c. Agitation time: 60 min

The test procedure is shown in Figure 8.

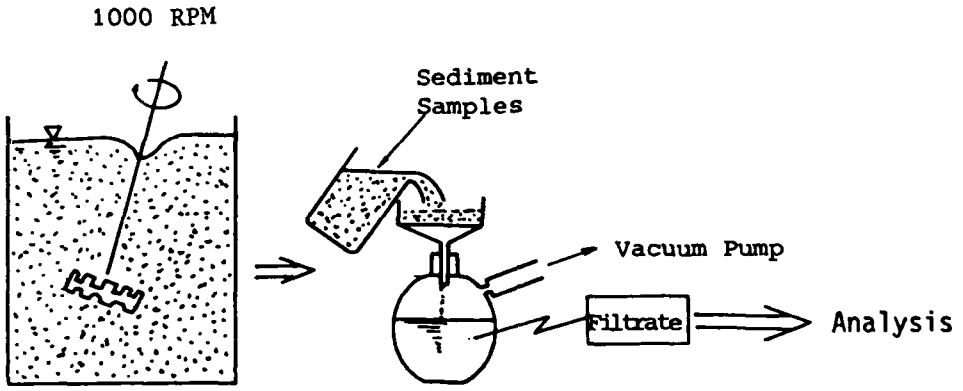
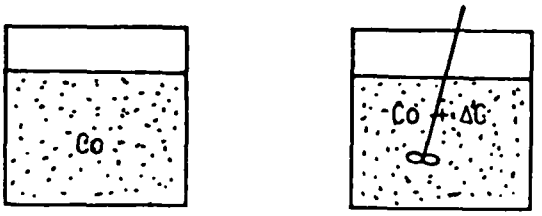


Figure 8. Test procedure

Although the volumetric ratio of sediment to water is 25% in the guideline of elutriate tests in the United States, our tests are conducted with three sediment contents as above.



If we represent the initial concentration of a particular chemical parameter in the sample as C_0 and the corresponding concentration of elutriates as ΔC , we obtain

PO_4-P $\quad \Delta C/C_0 = \frac{0.115}{0.370} \times 100 = 31\%$

T-N $\quad \Delta C/C_0 = \frac{0.79}{3.49} \times 100 = 22.8\%$

NH_4-N $\quad \Delta C/C_0 = \frac{0.67}{3.16} \times 100 = 21.2\%$

(COD) $\quad (\Delta C/C_0) = \frac{1.2}{4.2} \times 100 = 28.6\%$

These results for elutriates are plotted for the three sediment contents as shown in Figure 9.

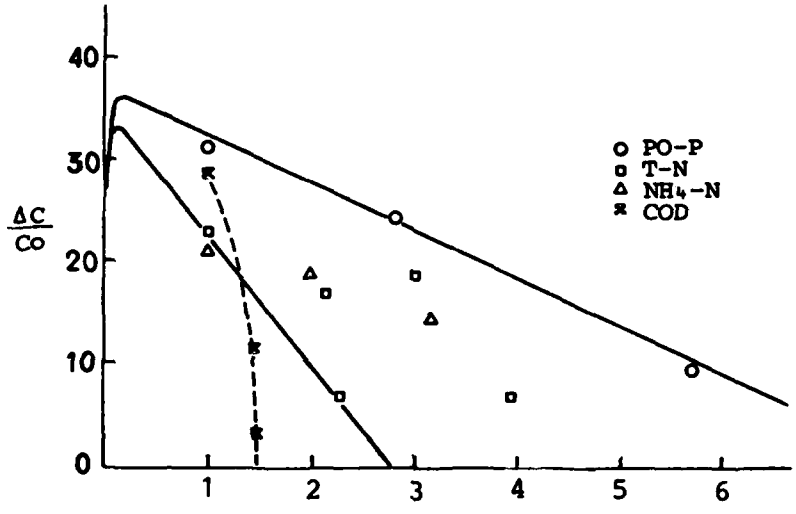


Figure 9. Behavior of $\Delta C/C_0$

Figure 9 indicates that elutriate concentrations decrease according to the increase in SS concentrations in the samples. It is presumed that the disturbance torque does not prevade thoroughly to each individual particle.

The elutriate behavior of phosphorus is different from other materials. As Figure 10 shows, the phosphorus concentration of elutriates reaches a maximum after 1 min and then rapidly decreases. This is caused by the generation of iron phosphate from to oxidation during agitation, which is then precipitated. The mechanism of iron phosphate generation is well known from many studies on phosphorus release behavior.

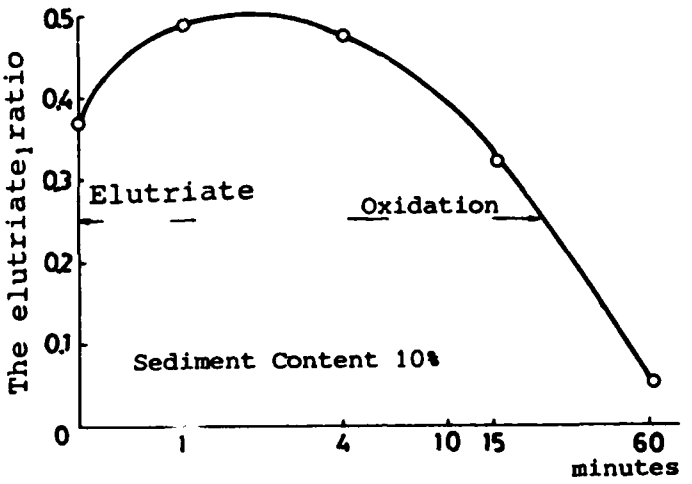


Figure 10. Behavior of phosphorus

Figure 11 shows the mobilization of dissolved orthophosphate (ORP) from the bottom sediment of standing water as a result of oxygen depletion and formation of H_2S at the sediment-water interface. The 'barrier layer' consisting of iron (III) phosphate and iron (III) hydroxide serves as a sink for phosphorus and is destroyed by the reduction process.

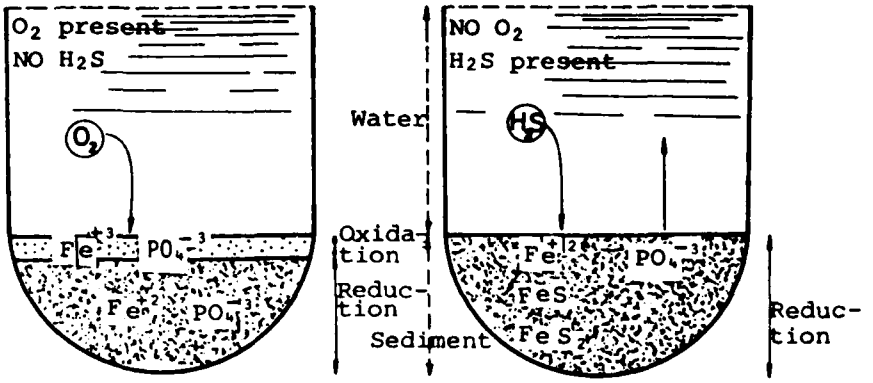


Figure 11. Mobilization of orthophosphate for sediment

CLARIFIED WATER

In order to accelerate the sedimentation of suspended solids, several chemicals such as alum and polymers are used. The effects of these on coagulation are shown in Table 3.

The table clearly shows that the combined addition of alum and polymer produces a much better result than the addition of only a polymer. But the advantage of the combined use diminishes with the higher sediment contents as shown in Figure 12.

TABLE 3. CLARIFIED WATER BY VARIOUS COAGULANT ADDITIONS

Sediment Content	Coagulant, mg/ℓ Alum	Coagulant, mg/ℓ Polymer	Clarified Water mg/ℓ	Combined Only
10%	--	5.0	58.0	1.0
(SS 3.05%)	200	5.0	12.4	0.214
25%	--	25.0	292	1.0
(SS 7.98%)	500	25.0	212	0.726
40%	--	110	142	1.0
(SS 13.0%)	750	110	104	0.732

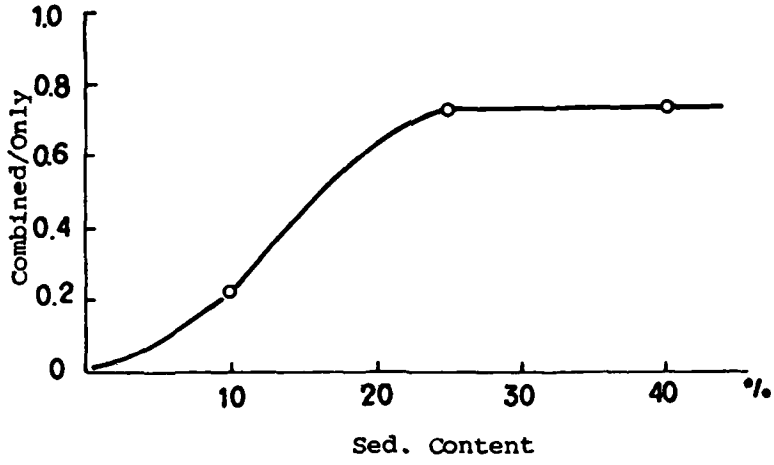


Figure 12. Advantage of combined addition

The ratio of the concentrations at the combined addition to the ones that use only a polymer is very small in the lower sediment content range.

Figure 13 shows the relationship between added quantities of alum and turbidities of clarified water. From the figure it can be seen that the addition of only an inorganic coagulant, alum, has no effect on water clarification for the higher sediment contents. It is readily apparent that dredging with higher sediment content is not advantageous from the viewpoint of sedimentation.

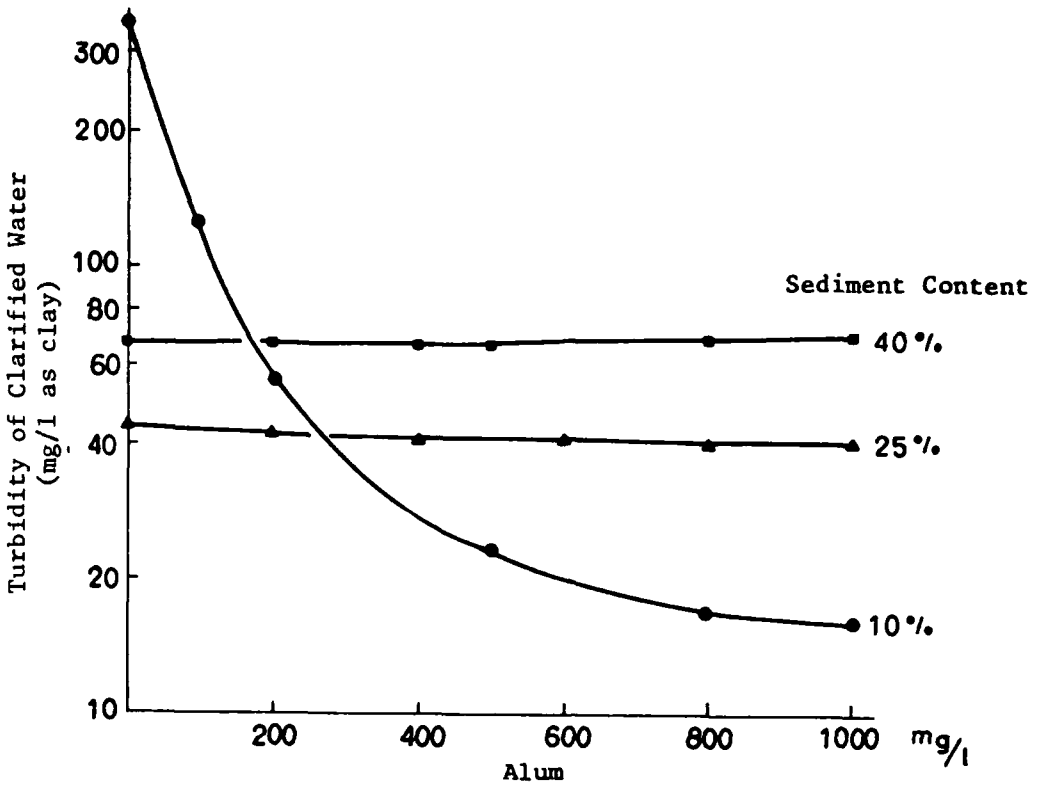


Figure 13. Effects of alum on turbidity

CONCLUSIONS

From the preceding tests it was found that the sediment in Osaka Bay has the unusual characteristic of sedimentation.

From the elutriate tests we have learned that the nutrient concentrations in dredged material elutriate samples are not meager, but do not increase with higher sediment contents.

The tests were conducted in accordance with the sediment removal project promoted by the Ministry of Transport.

PARTIAL REVIEW OF DREDGED MATERIAL DISPOSAL TECHNIQUES

H. Koba
Nishiki Consultant Co., Ltd.
19, Kanda Nishikicho 3 - chome
Chiyoda - ku, Tokyo, Japan

M. Miyake
Toyo Construction Co., Ltd.
17 - 1, Naruohama 3 - chome
Nishinomiya, Hyogo, Japan

ABSTRACT

Coastal disposal areas containing dredged sediments contaminated with hazardous material are not easily reclaimed into stable land areas. The techniques for predicting the period required for disposed material to become stable through sedimentation and self-weight consolidation have progressed, but are not yet satisfactory. This paper views the techniques for predicting sedimentation and self-weight consolidation of disposed sediment in Japan.

INTRODUCTION

In recent years, dredging for industrial area reclamation and environmental improvement and maintenance dredging in waterways and anchorages have been increasing in Japan. With the relatively small amount of available land in Japan, creation of additional areas of land by depositing dredged material along the coast has been the popular method of disposing a large amount of dredged material. The effective disposal of such a large amount of material has become a large problem. Primary geotechnical considerations are the amount of storage capacity that must be provided for a given dredging rate, the amount of subsequent subsidence, and the future land-use capability of the dredged material.

This report reviews the following techniques relating to disposal of dredged material: (a) prediction of surface settlement due to sedimentation and self-weight consolidation, and (b) consolidation tests to obtain the input data for the numerical consolidation analysis. A site application example is also given.

SEDIMENTATION AND SELF-WEIGHT CONSOLIDATION

The first quantitative analysis method in Japan on consolidation in which the self-weight of the fine-grained dredged material was taken into account

was the one-dimensional consolidation equation developed by Mikasa (4) in 1963 in analyzing the ground improvement test for lowering the water level after reclamation in the Port Osaka South reclamation work.

The analogous alluvial clay consolidation process provides the relationship that the consolidation time is proportional to the square of the layer thickness. Upon consolidation of the highly hydrated clay such as the dredged material, however, the coefficient of permeability (k) and the volume compressibility (m_v) are greatly reduced. The variation of the coefficient of consolidation ($c_v = k/m_v \cdot r_w$) may not be ignored due to the dependence on the variation of k and m_v. When consolidation is highly affected by the self-weight and the variation of the layer thickness is great, the square rule stated above is not applicable. Therefore, the following equations are developed.

(1) Correlation between stress and strain stated above . . . Non-linear ideal elasticity

$$m_v = de/dp \text{ : Volume compressibility} \qquad (1)$$

(2) Equilibrium equation

$$\partial p \, \partial z = j + r' \qquad (2)$$

where

j = (seepage pressure) = $i \cdot r_w$

i = (hydraulic gradient)

r_w = unit weight of water

r' = bouyant weight of soil particles

(3) Equation of motion . . . Darcy's law assumed

$$v(\text{flow velocity}) = k \cdot i \qquad (3)$$

(4) Stress components of the soil matrix and pore water (soil particle and pore water are incompressible)

$$P_a \text{ (total stress)} = p + u \qquad (4)$$

where

p = effective stress

u = excess pore water pressure

(5) Equation of continuity (transient)

$$\partial e/\partial t = \partial v/\partial z \qquad (5)$$

When Equations 1 and 2 are substituted into Equation 3, the following equation is obtained:

$$v = c_v(\partial e/\partial z - m_v \cdot r') \tag{6}$$

When Equation 6 is substituted into Equation 5 and if c_v is the function of strain e, the following equation is obtained:

$$\frac{\partial \zeta}{\partial t} = \zeta^2 \left[Cv \frac{\partial^2 \zeta}{\partial z_o^2} + \frac{dCv}{d\zeta} \left(\frac{\partial \zeta}{\partial z_o} \right)^2 - \frac{d}{d\zeta} (Cv\ mv\ r') \frac{\partial \zeta}{\partial z_o} \right] \tag{7}$$

where

ζ (consolidation ratio) = f_o/f

f = volume ratio*

f_o = initial volume ratio

z_o = original coordinate

$\phi(\zeta) = c_v/c_{vo}$

Upon applying Equation 7 to the site test stated above, the linearity, f - log p, is assumed for Equation 1 and a finite difference method is applied. This theory suggests that the clay settles in water initially at a constant velocity of $v = k \cdot r_o'/r_w$; therefore, this velocity is determined by k and r_o', but is independent of the clay layer thickness.

Mikasa and Takada (7) studied the self-weight consolidation of a very soft clay with a uniform water content (single layer filling) using a centrifuge. Briefly, the stress condition of the clay sample with a layer thickness H subjected to the $n \cdot g$ (g = gravitational acceleration) centrifugal acceleration field developed by using the device and container shown in Figure 1 has the same effect as a sample with n times the unit volume weight and is similar to the clay layer with a layer thickness, nH, in a 1-g environment.

Using the relationship between the prototype (p) and model (m) with the following expression:

$$\begin{aligned} \text{settlement} &: S_p = n \cdot S_m \\ \text{consolidation time} &: t_p = n^2 \cdot t_m \end{aligned} \tag{8}$$

the water content was changed to the 10-cm-layer thickness in the 100-g centrifugal acceleration field, which is shown in Figure 2.

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* Editor's note: Japanese researchers utilize the parameter volume ratio, f, $f = V/V_s$ where V is the total volume of a soil mass, and V_s is the volume of soil solids. f equals (1 + void ratio).

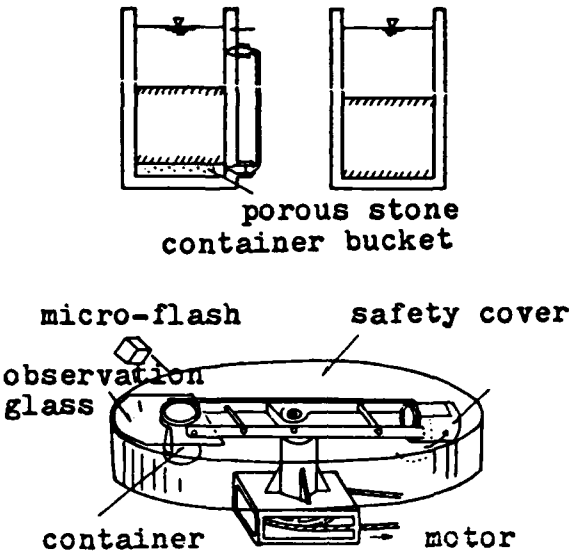


Figure 1. Centrifuge facility

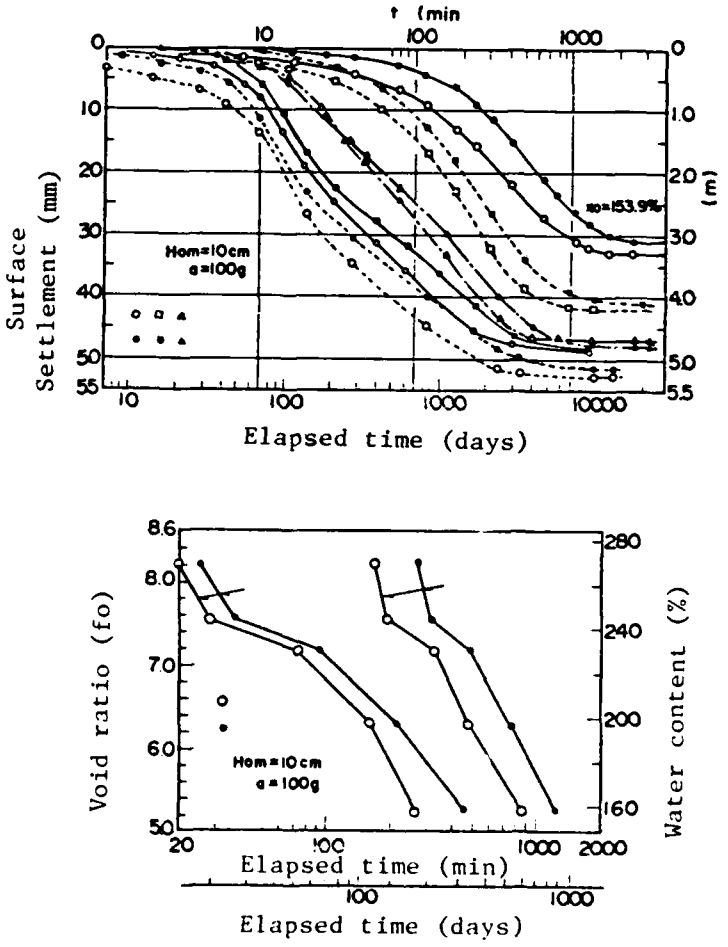


Figure 2. Test results using centrifuge

Thereafter, Mikasa, Takada, and Li (8) in 1976 studied the influence of the coefficient of permeability of clay on the sedimentation velocity during self-weight consolidation. A major objective was to assess feasibility of the projected capacity of the disposal site with the limited reclamation schedule of the dredging and disposal plan for the surface layer of the seabed clay contaminated with hazardous materials. In other words, when the clay in the initial uniform condition is consolidated by its self-weight only, the strain in the clay layer is transmitted from the top to the bottom, and, at the upper portion of the clay layer where the strain in the initial stage of consolidation is not yet transmitted. The surface of the deposited material settles initially at a constant velocity of $v = kr_o'/r_w$. Therefore, this velocity is determined by k and r_o' , but not by the clay layer thickness.

Figure 3 shows the sedimentation condition characteristics of the dredged material, i.e. the variation of the coefficient of permeability and effective stress obtained through the self-weight consolidation test in a 1-g environment and the centrifuge.

In 1978 Takada (10) applied the result of the test described above to a practical problem. As shown in Figure 4, adequacy of the capacity of the dredging disposal pond was studied. In this study, the water content of

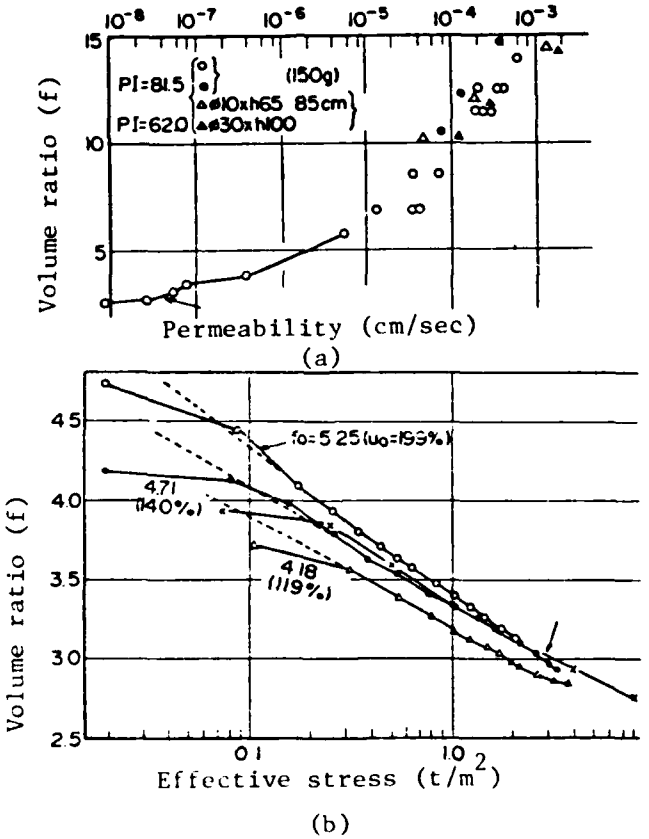


Figure 3. Relationships (a) f - log k and (b) f - log p

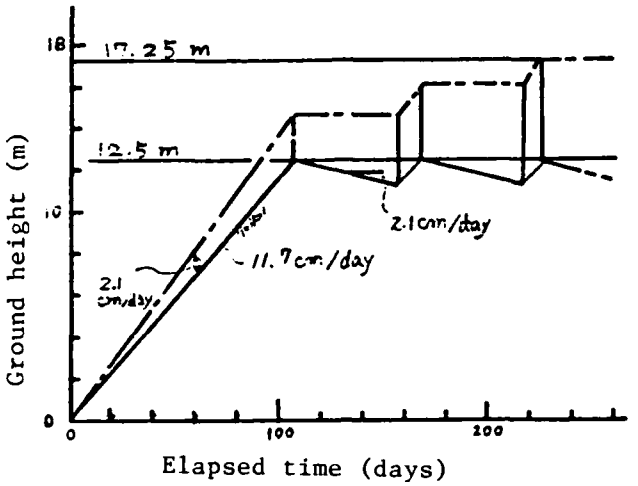


Figure 4. Example of planning dredging and reclamation activities

material hydraulically dredged was estimated to be 450%, and the bulking factor was assumed to be 2.21. The layer thickness of the filling soil was estimated to be 1.8 m, and the overexcavation to be 17.25 m at 25%. Under these conditions, v = 2.1 cm/day was obtained, where $k = 1.9 \times 10^{-4}$ cm/sec, $r_o' = 0.126$ t/m^3, and f = 12.97. The maximum usable ground height of the containment area was determined to be 12.5 m. Since a scheduled material thickness of 13.8 m was planned to be filled in 100 days because of the pumping capacity, a 1.8-cm/day rise in the ground surface was assumed. In addition, a 2.1-cm/day sedimentation rate was estimated along with a 13.8-cm/day rate of filling; therefore, eventually, the rise in the surface level within the containment area was 11.7 cm/day.

If the scheduled thickness of 13.8 m is continuously applied, the ground height becomes 11.7 m, which has an 80-cm allowance with respect to the 12.5-m disposal field capacity. When the 25% overexcavation is filled, the height reaches 12.5 m in 107 days. As shown in the figure, if an interval of 50 days is provided, a 105-cm settlement is generated.

Therefore, filling was conducted for 9 days at a velocity of 1.8 cm/day; then after sedimentation, filling was continued.

Imai (2) performed many sedimentation and self-weight consolidation tests for the dredged clay. He divided the sedimentation process into interface sedimentation and consolidation sedimentation as shown in Figure 5. The effects of the soil particle weight, salt concentration, and initial water content to the water content of this interface are summarized in Figure 6.

Yano and Tsuruya (17) implemented single-layer and multi-layer filling sedimentation tests to determine the sedimentation and accumulation characteristics of slurried fine-grained dredged material. In doing so, the area and capacity of the reclaimed land necessary for the work plan could be determined and the scale of the sedimentation basin (to prevent outflow from the

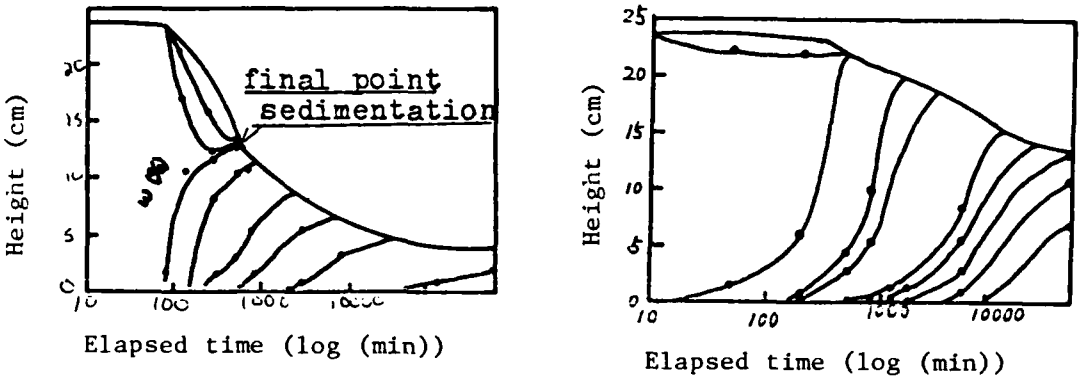


Figure 5. Process of sedimentation and self-weight consolidation

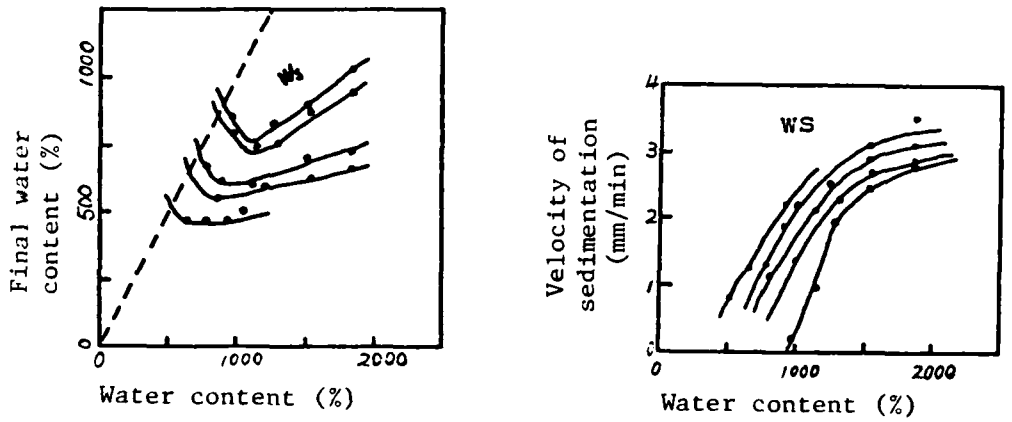


Figure 6. Effect of water content and soil particle weight

pond) could be studied. From the single-layer filling sedimentation test result, the correlation between reclamation layer thickness and soil particle height (H_s) is shown in Figure 7, where the self-weight consolidation starting point is t_o and the self-weight consolidation ending point is t_{100}.

In the multilayer sedimentation test, the correlation between filling velocity and accumulation velocity is summarized in Figure 8. From this correlation, it was reported that the accumulation velocity would be constant if the filling velocity was constant.

Yamanouchi et al. (16) found that the correlation between the practical soil height (H_s) and the accumulated sample height (H), obtained from the contained soil particle amount, becomes a straight line on a log-log graph if the self-weight consolidation degrees are the same. They reported that the practical soil particle height and the accumulated soil layer height in the reclaimed ground could be obtained from this linear relationship.

In a maintenance dredging project (3), a method was shown which demonstrates the correlation between the capacity of the disposal pond and the disposal amount from the sedimentation test result. In other words, using the

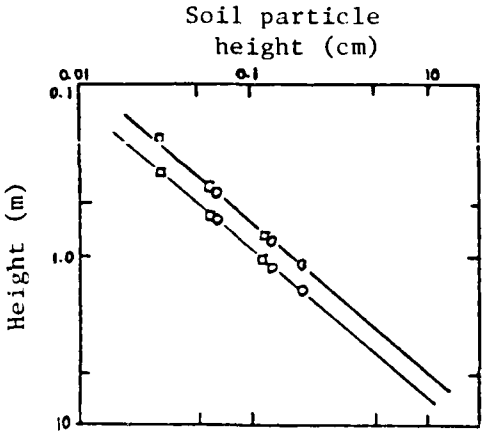


Figure 7. Relationship between reclaimed height and soil particle

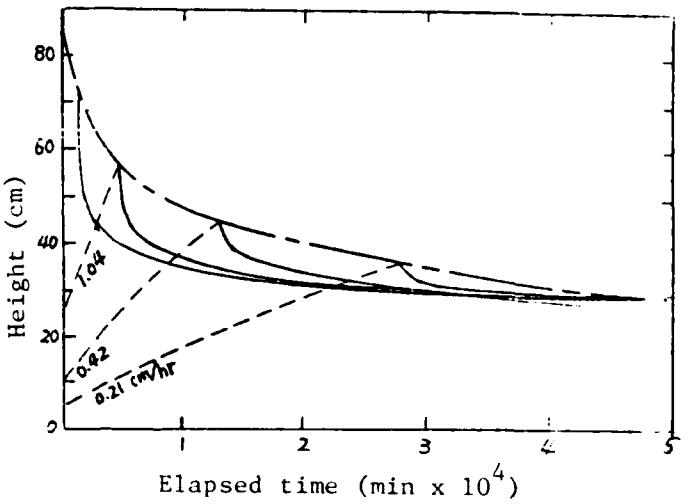


Figure 8. Multi-layer sedimentation curve

results obtained from the single-layer and multi-layer sedimentation tests, the correlations among H_{100} (the layer thickness at the end of self-weight consolidation:cm) H_s (practical soil particle height:cm), filling velocity (cm/day), accumulation velocity (cm/day), elapsed time (days), and accumulated layer thickness (m) were obtained on log-log scales as shown in Figures 9-11.

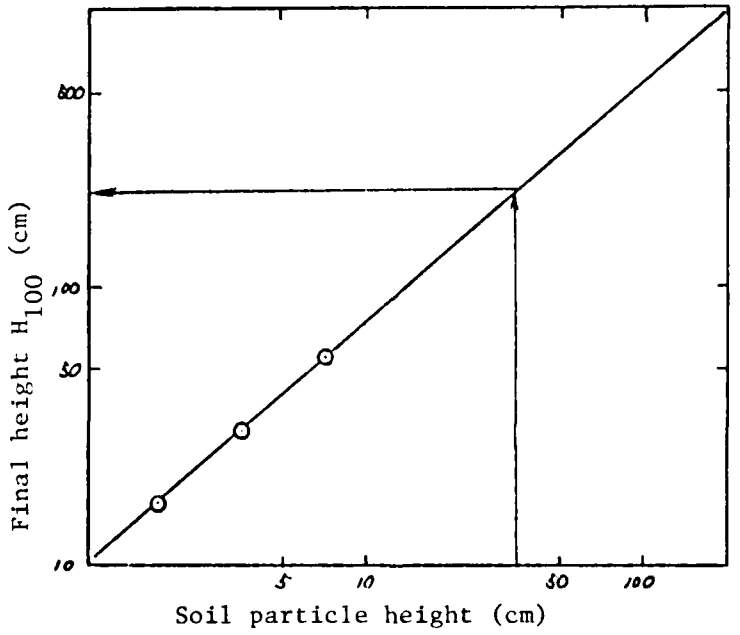


Figure 9. Soil particle height vs. final height

Predicting the ground height using Figure 11 is explained as follows. During the filling phase, the dredged material surface rises at a rate of v = 1.97 cm/day and reaches point B after continuous filling for 200 days.

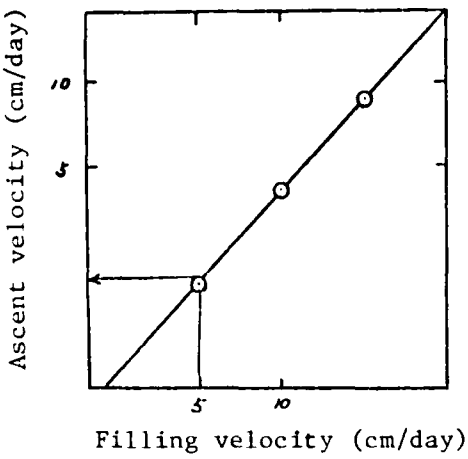


Figure 10. Filling velocity vs. ascent velocity

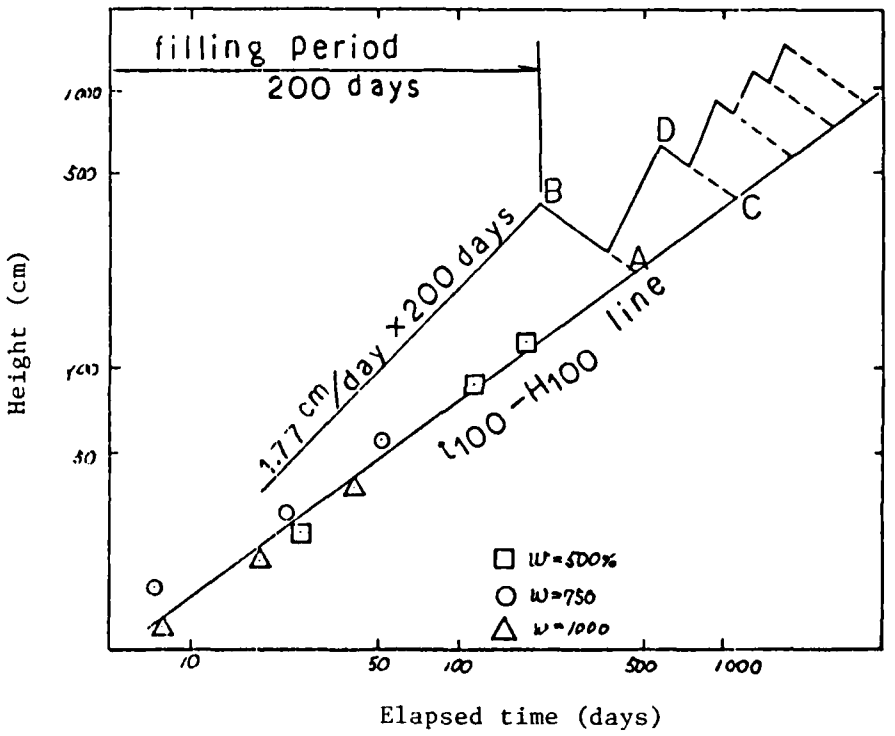


Figure 11. Elapsed time vs. height

The practical soil particle filled during this period is 0.35 m^3/m^2. From Figure 9, it is found that the accumulated layer thickness at the end of self-weight consolidation, H_{100}, is 2.25 m. Accordingly, the dredged material surface that reached to point B in Figure 11 settles to point A. Points D and C are obtained in the same method.

On the other hand, Takada (11-13) presented a numerical calculation example of Equation 7 according to Mikasa's theory in different boundary conditions and supplemented the calculation example in Mikasa's thesis of 1963. Among them, the paper in 1980 (2) dealt with the self-weight

consolidation of the soft clay layer whose layer thickness gradually increases. The clay whose void ratio is relatively small (e_o = 3.2) as dredged soil is reclaimed on the permeability layer in sea. The similarity accuracy was changed to eight steps in case the layer thickness was in proportion to time and the gradual increase of the layer thickness was similarized with the step-by-step increase under the condition that c_v is constant during consolidation, and the time until the final layer thickness was changed to the time factor Tf = 0.025 to 0.2.

In the same project previously introduced (3), Miyake and Akamoto (6) performed numerical calculations for some examples by preparing a program to account for the lowering of the water level and negative pressure effects after filling, covering soil.

CONSOLIDATION CHARACTERISTICS OF DREDGED SOIL

The methods for determining consolidation and sedimentation characteristics and elapsed time for consolidation were previously discussed. The following paragraphs introduce the consolidation constants required in the method (i.e. the correlation f - log p or m_v for the former, or k and m_v or $c_v(=k/m_v \cdot r_w)$ for the latter.

Mikasa (4) implemented a conventional consolidation laboratory test to determine the influence of consolidation characteristics of the very soft clay on the settlement of the reclaimed land at the Osaka South Port. He used an improved apparatus to avoid problems with the fine-grained material seeping through the consolidation ring.

Thereafter, Mikasa, Takada and Lee (8) obtained the f - log k relationship for Tokuyama clay as stated before and the f - log k relationship for Osaka South Port clay using results from the self-weight consolidation test, the conventional consolidation test, and the consolidation test using centrifuge.

Imai (2) thought that consolidation constants in the low-stress range would be necessary for calculation of the self-weight consolidation and therefore obtained the f - log p relationship from the sample soil following the soil sedimentation test. In other words, the vertical profile relationship between f (calculated from the water content after the self-weight consolidation) and the effective stress p at each discrete depth was obtained from the following equation

$$\begin{aligned} f_i &= 1 + Gs \cdot w_i \\ r'_i &= (Gs - Gw)rw/f_i \\ p_i &= \Sigma r'_{i-1}\Delta z_{i-1} + r'_i\Delta z_i/2 \end{aligned} \tag{9}$$

where Δz_i = the representative change in depth for slice i.

The stress range was quite small in the above method, so Imai (1) proposed a consolidation test method using seepage force. The material slurry is put into the transparent container as shown in Figure 12, and the clay is

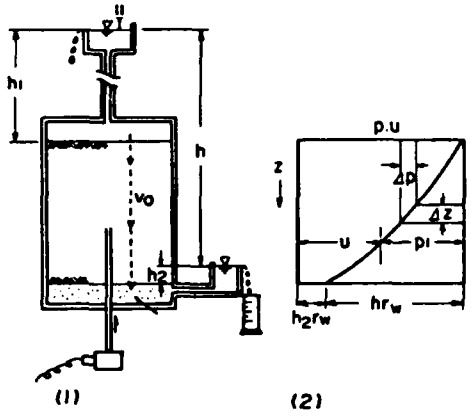


Figure 12. Fundamental ideas of hydraulic consolidation

consolidated by the combined effects of a seepage flow as well as its self-weight. After completion of consolidation, a thin pipe is inserted into the bottom of the clay layer step by step and the distribution of the pore water pressure is obtained. Then, from the void ratio obtained by cutting the clay to proper slices, f_i and r_i' are calculated using Equation 9. The $f - \log p$ relationship is obtained from f_i and the corresponding consolidation pressure p_i . p_i is obtained by deducting the pore water pressure from the total stress obtained by adding r_t to the clay layer upper surface water pressure as shown below.

$$p_i = (h_1 r_w + \Sigma r'_{i-1} \Delta z_{i-1} + r'_i \Delta z_i / 2) - u_i \tag{10}$$

The coefficient of permeability is given by the following expression:

$$k_i = - v_o \left(\frac{\Delta z}{\Delta p}\right) i r_w \tag{11}$$

Umehara and Zen (15) proposed a method using the constant rate of strain consolidation test. In other words, in Equation 7 of Mikasa's theory, the effect of the self-weight is ignored, under the condition that c_v is constant; the relationship is obtained from the equation shown below through differentiation assuming that the displacement velocity is constant.

$$\frac{\partial S}{\partial t} = Cv\zeta^2 \frac{\partial^2 \zeta}{\partial Z_o^2} \tag{12}$$

By applying the curve fitting method to Equation 12 and the test curve, c_v was obtained. The f - log p relationship can be obtained from the actual measurement of volume displaced and effective stress (total load minus pore water pressure).

However, since the method described above is complicated, a method was proposed for obtaining c_v by plotting test values on Figures 13-14 incorporating Equation 12 into $\Delta H_o/H_o$ (average deformation), f_t/f_o (upper end surface consolidation ratio), f_b/f_o (lower end surface consolidation ratio), and F (upper end surface to lower end surface strain ratio = $\log \frac{\sigma - Uh}{\sigma'_o} \Big/ \log \frac{\sigma}{\sigma'_o}$).

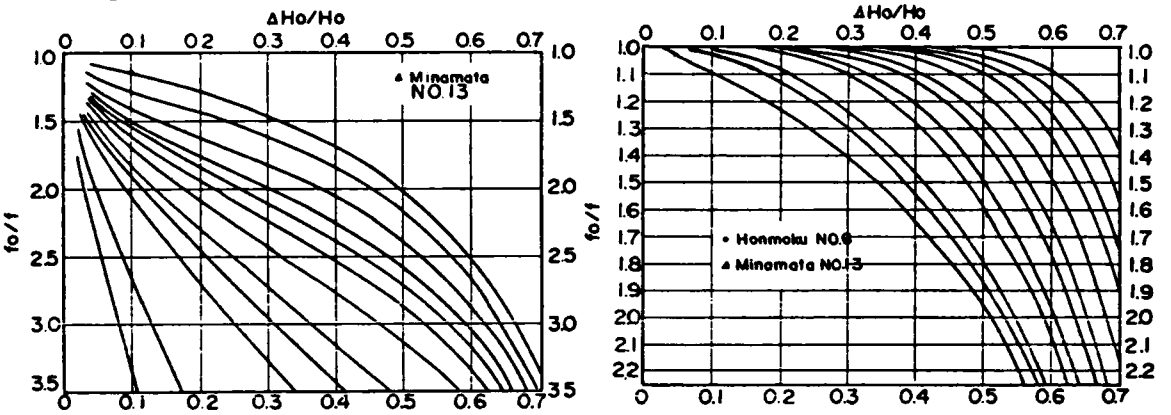


Figure 13. Variations of consolidation ratio at the top and bottom of the specimen for average strain

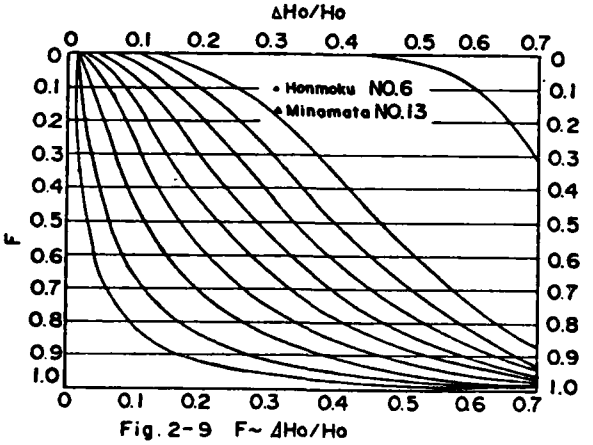


Figure 14. Variations of the ratio of bottom strain to top for average strain

Takada (11-13) obtained the f - log p , f - log k , and f - log c_v relationships of the Osaka South Port fine-grained material over a wide range including the low-stress range obtained with the centrifuge.

Toriyama (14) obtained the consolidation characteristics of the reclaimed soil of the Hikono District Chukai reclamation through the tri-axial compression test. c_v was obtained through the "t method." Also, he implemented the self-weight consolidation test and obtained c_v by curve fitting from the relationship between the settlement and time. He reported that the ordinary stress range result would be applicable to the low-stress range.

SITE APPLICATION

Applications of the previously described laboratory and analyses methods (including the model test using the centrifuge facility) are described in Mikasa and Takada (7, 8), Miyake and Akamoto (6), Umehara and Zen (15), Takada (13), and Toriyama (2).

Mikasa's Osaka South Port reclamation settlement analysis (4) and Tokuyama Port dredged material disposal land capacity feasibility study (8, 10) were used as the site application example.

In addition, Nakamura (9) studied the applicability of the model test using Mikasa's theory and centrifuge facility for the preparation of a reclamation plan standard for the Chukai reclamation Hikona District:

a. He implemented the centrifuge-based self-weight consolidation test (acceleration = 33g, scale = 1/33, model layer thickness = 15 cm, initial water content = 303%) in the centrifuge. The consolidation time was 25 hr (actual time = approximately 3 years), settlement was 5 cm (166 cm at site), and C_c was 1.4 to 1.8.

b. For the initial condition for numerical analysis through Mikasa's theory the initial void ratio was unity near the surface layer after reclamation. After the calculation using w_o = 300%, Gs = 2.65, f_o = 8.95, reclaimed layer thickness H_o = 5 m, single drainage, C_c = 1.4, p_o = 0.00914 tf/m^2 , and c_v = 0.015 m^{2}cm^2/day , the final settlement was 1.246 m.

From these results, he reported that the model test method using Mikasa's theory and centrifuge facility would be effective in applications to the site.

Miyake (5) attempted the application of Mikasa's theory to determine surface settlement times and ultimate ground height in Takuma Port Reclamation Work. Two months after reclamation he sampled the soil using a soil sampler that excluded mud and obtained the water content distribution in the layer 50 to 300 cm. Using Equation 9 from the conventional consolidation test and self-weight consolidation test, he obtained relationships of f - log p and f - log c_v from the actual measured values and calculated value of the water content, and, as a result, obtained the relationship between settlement and elapsed time.

He reported that these would be well explained by Mikasa's theory.

CONCLUSIONS

More effective methods for disposal of bottom sediment containing contaminated material are necessary in Japan. Accordingly, the major analytical technique for this purpose, the consolidation settling equations, needs to be made more accurate.

The author sincerely hopes that this paper will be of some help in realizing these goals.

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LIME-CEMENT HARDENING OF VERY SOFT FRESHWATER CLAY

M. Kamon

Associate Professor
Department of Civil Engineering
Kyoto University
Yoshidahonmachi, Sakyo-ku
Kyoto, Japan, 606

ABSTRACT

Soil stabilization problems for a very soft ground formed during land reclamation by filling in a lake were treated with a lime cement mixture treatment. The usefulness of the hardening treatment was shown in the laboratory using several stabilizing agents. It was clearly shown that fine particles solidified by pozzolanic products were needed for good hardening, although a great amount of fresh water was fixed into the ettringite crystal and the handling properties were improved.

INTRODUCTION

It takes a long time for very fine clay particles to settle in fresh water because their interparticle repulsive forces make them suspend; therefore, after sedimentation a very soft surface with a high water content is formed. Because of the high water content, soil improvement is very difficult. A pollution potential exists during reclamation work in lakes and ponds, from these suspended fine clay particles because they can flow out to open areas without sedimenting inside the reclamation pond.

To prevent this pollution a retaining basin is often built at the part of spillway in a reclaimed area and water treatment is used to accumulate the fine clay particles. A large retaining basin, however, is needed to obtain good accumulation and the basin becomes an obstacle to effective land use because the basin is in a very soft condition at the end of the reclamation work.

An artificial island, which is an example of this kind of reclamation works, was constructed in 1980 south of Biwa Lake in Shiga Prefecture, Japan. The retaining basin in this reclamation was 11 ha which is one seventh of the total area of the reclamation project, 73 ha. It is still too soft for construction machinery. Therefore, soil improvement or hardening is now required.

The purpose of this study was to investigate the possibility of using the hardening treatment, which is introduced by lime and cement mixture, for very soft freshwater clay in the laboratory. Special attention was paid to obtain a suitable stabilization mixture. It was hoped to fix a great amount of fresh

water into the crystal of ettringite and to obtain a stable ground cemented by lime and cement stabilizer.

ENGINEERING PROPERTIES OF VERY SOFT CLAY

The reclaimed soil was dredged from the lake bottom with a soil profile consisting of sandy and gravelly soils, and transported to the reclamation area by a pump dredge. The coarse particles, such as sand and gravel, precipitated rapidly near the end of discharge pipe and thereby formed hard ground. On the other hand, the fine particles, the silts and clays, were thrown out far from the discharge pipe and accumulated around the wasteway of the reclamation pond. Since a retaining basin at the part of the wasteway had been built in order to treat water and to prevent outflow to open waters, the accumulation of very fine clay particles accelerated especially. These particles are still in suspension several years after the end of the reclamation work. The water content of these soils under the slurry condition is 150-200 percent.

The engineering properties of this soft clay soil are listed in Table 1. This is a highly plastic soil. The particle-size distribution is colloid and clay particles as shown in Figure 1. An X-ray diffraction pattern of this clay soil is illustrated in Figure 2. The dominant forming clay minerals are halloysite and kaolinite. Illite and montmorillonite minerals are contained with fine primary minerals such as quartz and feldspar.

A sedimentation test was performed using arranged samples which were set at 250, 500, 1,000, and 2,000 percent of the initial water content. Sedimentation properties were obtained as in Figure 3. The interference sedimentation is observed in 250 and 500 percent of the initial water content samples. It clearly shows the high water content of the soft ground under the in situ slurry state condition.

The bearing capacity of this clay soil in relation to the water contents was examined by the vane shearing test shown in Figure 4. The dewatering of the samples was forced by vacuum suction. The shearing strength at 150 to 200 percent of water content is 0.5 to 3 kPa, thus indicating quite soft ground.

TABLE 1. TESTING CONDITIONS

Factor	Condition
Initial water content (%)	150, 200
Mixture percent (%)	5, 10, 15, 20, 30, 40
Combination ratio (P:L)	9:1, 8:2, 7:3, 6:4
Specimen size (mm)	50 (ϕ), 100 (h)
Curing temperature (°C)	20 ± 2
Curing time (day)	7, 30, 60, 90, 360, 720

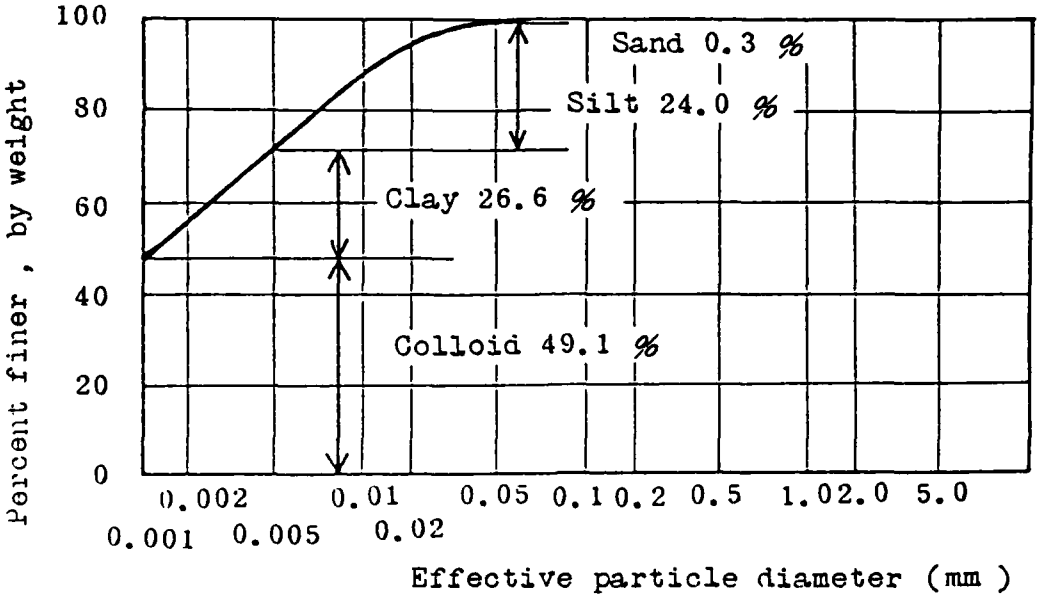


Figure 1. Particle-size distribution

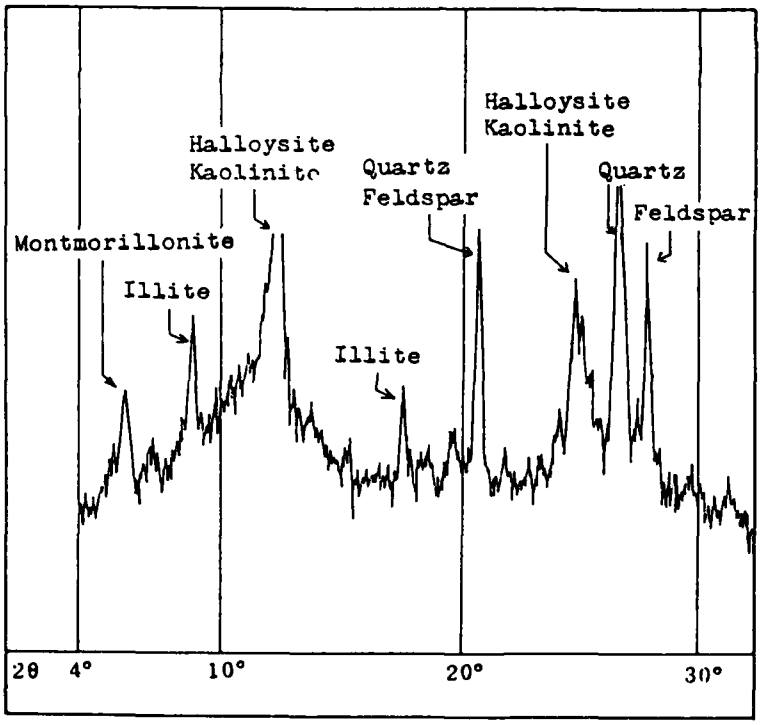


Figure 2. X-ray diffraction pattern (Cu-K_α)

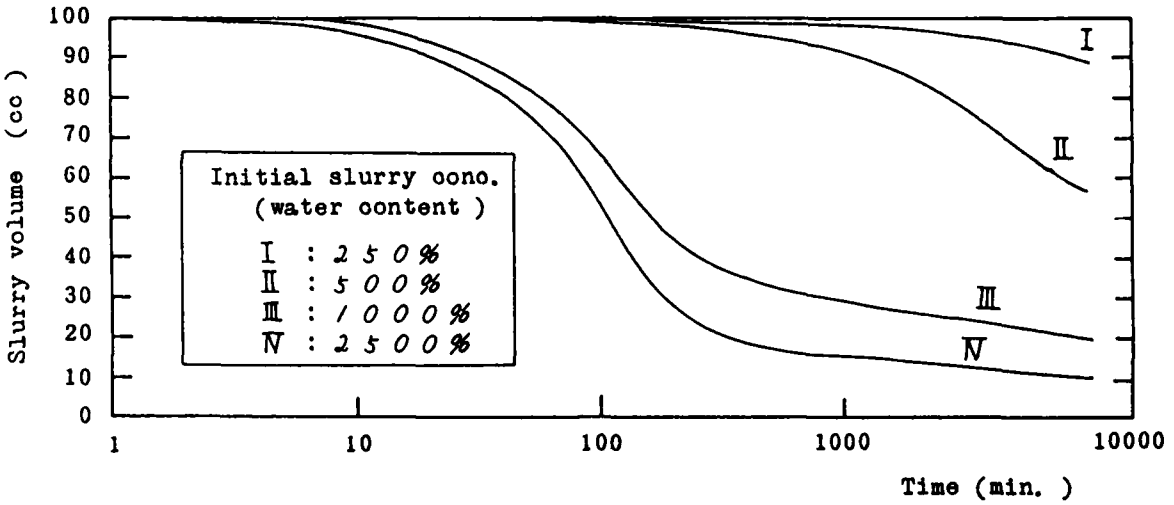


Figure 3. Sedimentation properties

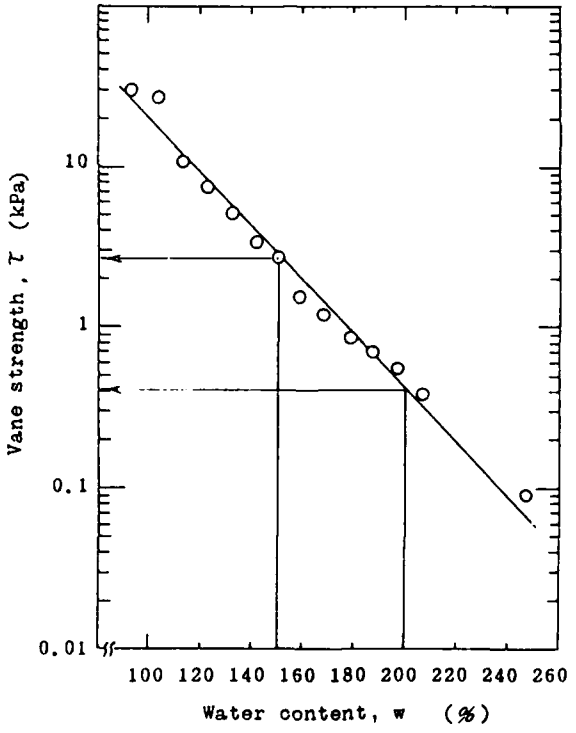


Figure 4. Vane strength

STABILIZING AGENTS AND EXPERIMENTS PERFORMED

Stabilizing agents used

In order to improve the engineering properties of this very soft clay ground, the soil hardening technique with lime or cement addition was chosen. Lime alone or cement alone did not cause any hardening in preliminary tests. Hence, a special stabilizing agent which can reduce the free water by enclosing water into ettringite crystals was introduced. A lime-cement mixture with aluminum sulfate [$Al_2(SO_4)_6$] was used with varying relative proportions of their contents. The proportion of lime to cement was mixed in the weight ratio of 2 to 1. The combination ratio of lime and cement mixture to liquid aluminum sulfate was varied in the weight ratio of 9 to 1, 8 to 2, 7 to 3, and 6 to 4. These stabilizing agents were added into the soil samples at 5, 10, 15, 20, 30, and 40 percent of the soil dry weight.

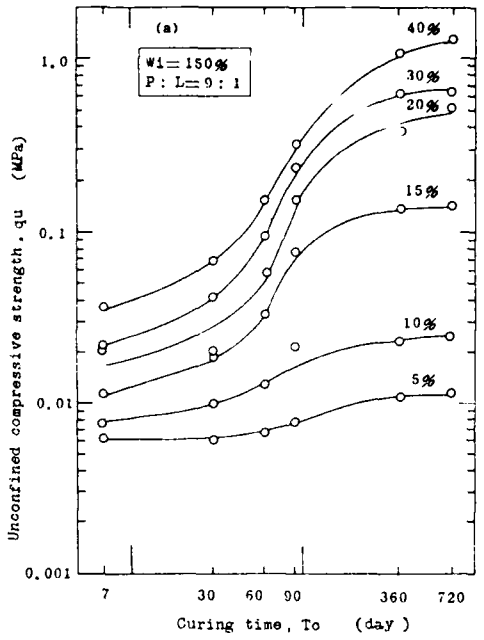
Experimental procedure

The varied water contents of the soil sample were 150 and 200 percent. These correspond to the in situ water content. The effect of the hardening treatment was examined by the unconfined compressive strength of the treated samples. (The testing conditions were listed in Table 1.) P:L in the table means the ratio of the powder mixture of lime and cement to liquid aluminum sulfate. The soil samples were mixed with the stabilizing agent for 10 min. After the samples were put into the cylindrical containers without entrapped air, they were covered by a thin plastic film and then cured at constant temperature.

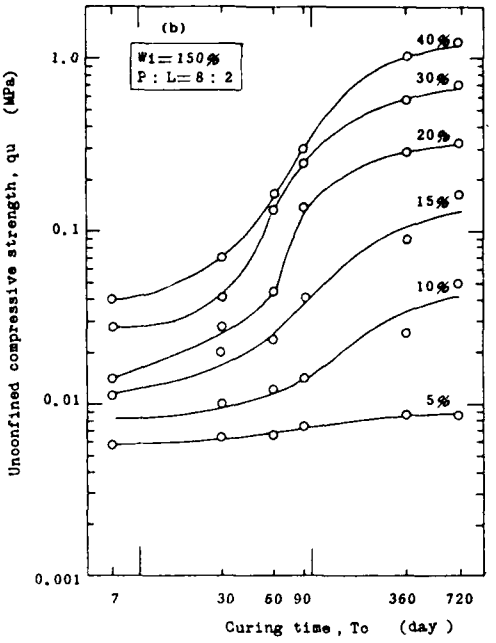
In order to determine the reactive products of the added stabilizing agents, X-ray diffraction analysis was applied for specimens which had been used in the unconfined compression test. It is considered that X-ray intensity of the reactive products indicates their content; namely, I_{ett} is the X-ray intensity of ettringite crystal, $2\theta = 9.08°$ and I_{CSH} is one of CSH-series products and tobermorite. These reactive products were also observed by a scanning electron microscope.

HARDENING CHARACTERISTICS

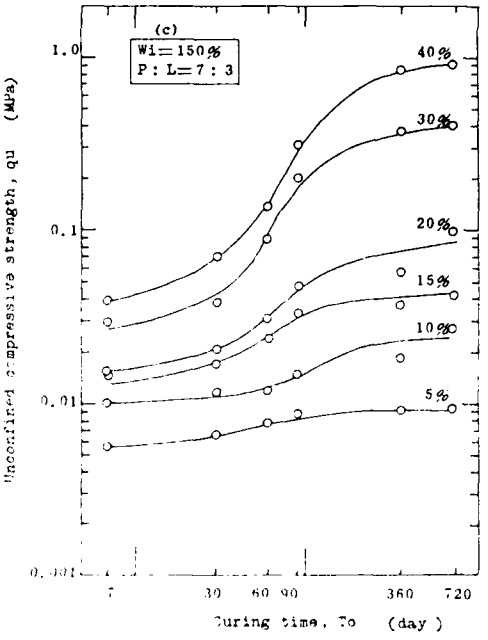
The unconfined compressive strength of treated samples is shown in Figure 5. The results of the samples with an initial water content of 150 percent are shown in Figure 5a-d. Those with an initial water content of 200 percent are shown in Figure 5e-h. These figures show the curing effect in relation to the added volume of stabilizing agents. The compressive strength increased with the duration of curing, and a special increase in the strength at 60 to 90 curing days was observed. The treated samples under 150 percent of initial water content, 9:1 combination ratio, and more than 15 percent adding volume conditions have a great increase in their strength. This, however, tends to decrease for the smaller combination ratio of the treated samples. This means that the enclosed water into ettringite crystal may cause the strength decreases to the treated samples.



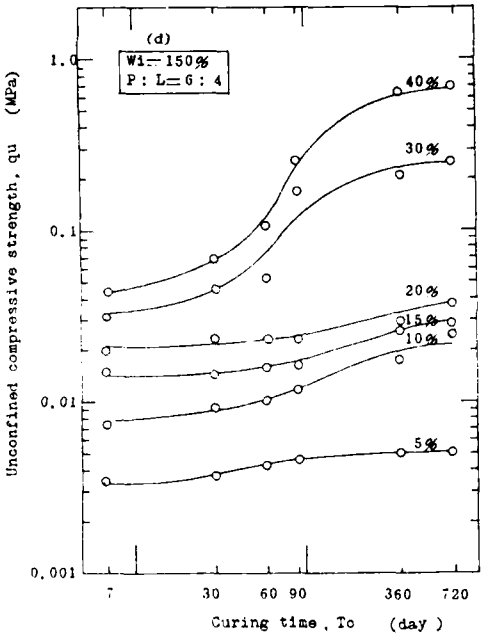
a. Initial water content (Wi) = 150%, P:L = 9:1



b. Wi = 150%, P:L = 8:2

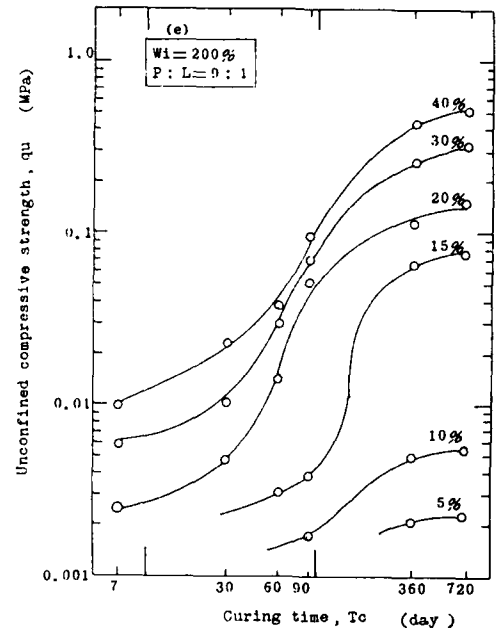


c. Wi = 150%, P:L = 7:3

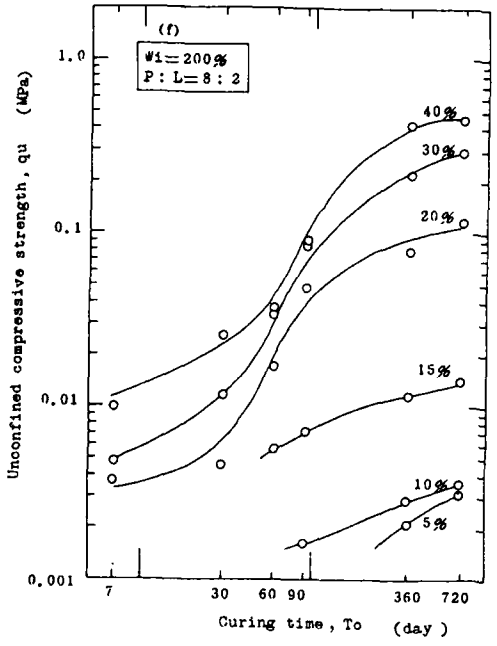


d. Wi = 150%, P:L = 6:4

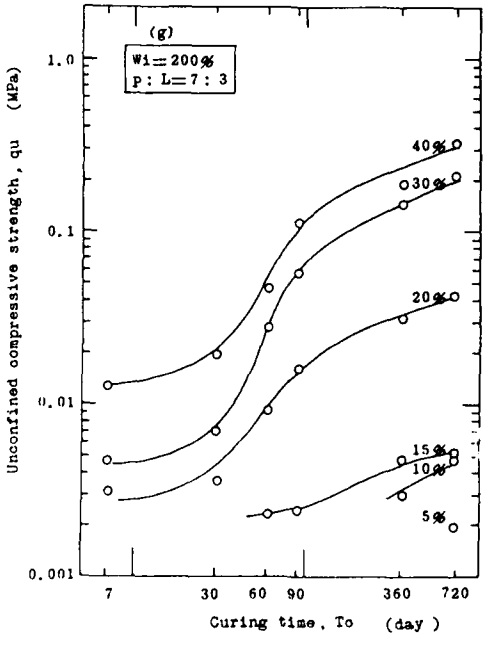
Figure 5. q_u ~ curing time (Continued)



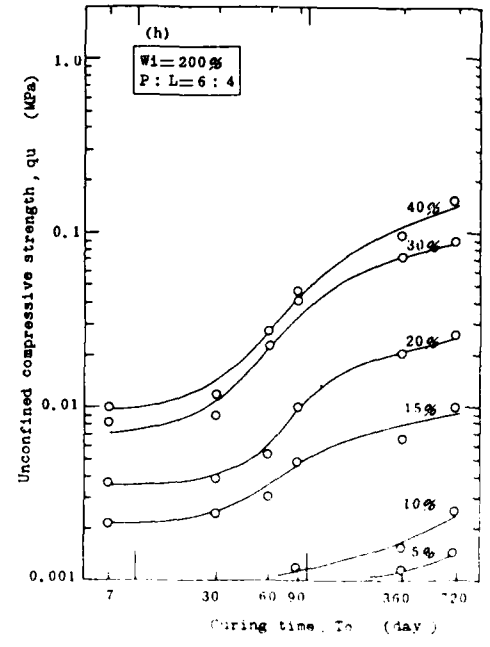
e. Wi = 200%, P:L = 9:1



f. Wi = 200%, P:L = 8:2



g. Wi = 200%, P:L = 7:3



h. Wi = 200%, P:L = 6:4

Figure 5. (Concluded)

The treated samples with 200 percent initial water content had less strength than those with 150 percent initial water content as shown in Figure 5e-h. However, the initial strength (7-day strength) was almost the same at both 150 and 200 percent initial water contents. Their unconfined compressive strengths did not differ at the different combination ratios. On the other hand, the long-term strength (more than 90 days) was quite different between samples. The increased strength at the 6:4 combination ratio was the least of all treated samples.

It is generally said that the bearing capacity to obtain trafficability on soft grounds is more than 30 to 50 kPa in q_u value. In order to obtain this value, the required volume of added stabilizing agents is more than 30 to 40 percent in the 150 percent initial water content sample according to Figure 5. The most reasonable solution for the added volume is about 15 percent at 60 to 90 days of curing periods because the treated samples by higher added volumes (30-40 percent) show excess long-term strengths (700-1500 kPa). Excessively solidified ground which will become an obstacle to construction must be avoided.

The unconfined compressive strength in relation to combination ratio and added volume is shown in Figure 6. The increasing strength in samples of 9:1 and 8:2 combination ratios and at more than 15 percent added volume was clearly illustrated in the slope shape of the figures. Generally speaking, the bigger combination ratio samples (those which produce the pozzolanic reactive products), have the greater improved effect.

HARDENING MECHANISM

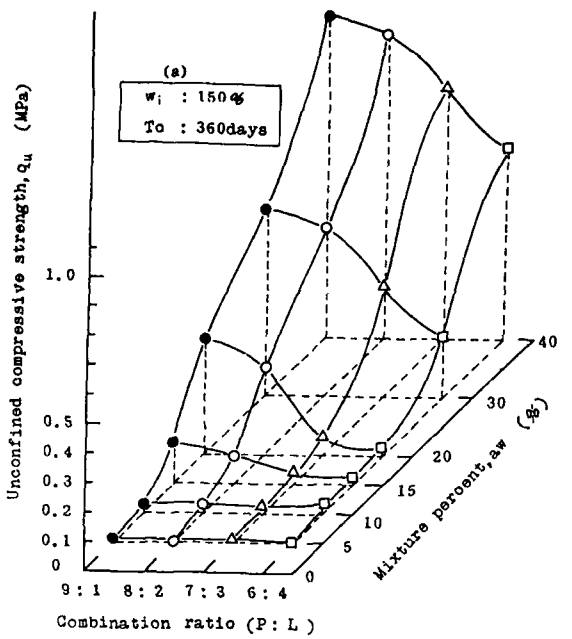
The hardening mechanism of the lime-cement mixture treatment was examined with a scanning electron microscope and X-ray diffraction analysis.

A large number of needlelike crystals of ettringite were observed in treated samples. These crystals weaved each other into clay particles according to the scanning electron microscopic research.

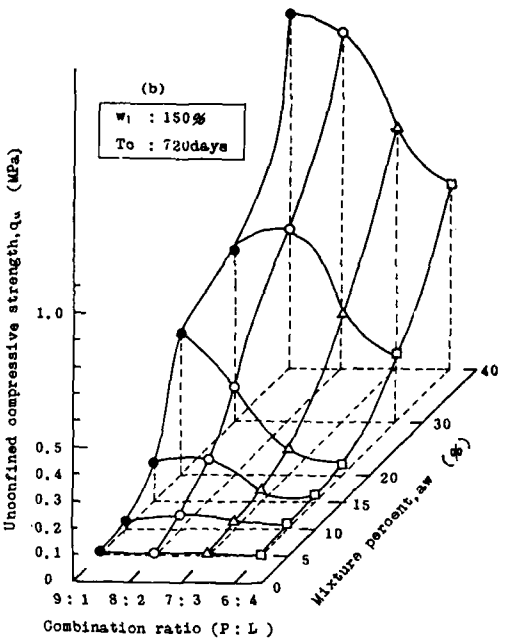
The amount of these reactive products was quantitatively obtained by X-ray diffraction intensities of pozzolanic reaction series and ettringite series shown in Figure 7. In the figure, ● , ○ , △ , and □ represent the combination ratios 9:1, 8:2, 7:3, and 6:4, respectively. The distribution was different from the types of stabilizing agents, and the least combination ratio, 6:4, or the highest content of aluminum sulfate, produced the maximum product of ettringite crystal.

X-ray intensity, I_{CSH}, $2\theta = 29.06°$, which is the main peak of CSH(I), (II) and tobermolite, was clearly observed in the long curing periods of 360 and 720 days. This corresponds to the large increase in the q_u values shown in Figure 5. X-ray intensity of ettringite crystal, I_{ett}, $2\theta = 9.08°$, was also observed in the long curing periods. As the excess free water was crystalized into ettringite as seen by its molecular equation

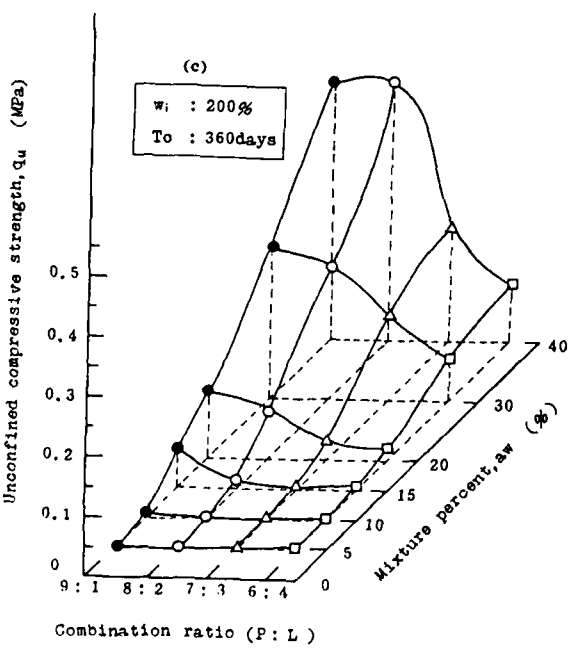
$$3CaO \cdot Al_2O_3 \cdot 3CaSO_4 \cdot 32H_2O$$



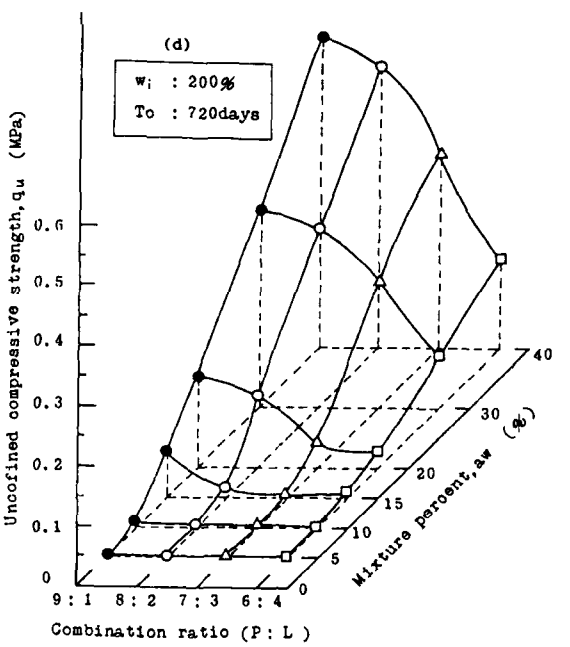
a. Wi = 150%, To = 360 days



b. Wi = 150%, To = 720 days



c. Wi = 200%, To = 360 days



d. Wi = 200%, To = 720 days

Figure 6. $q_u \sim a_w \sim (P:L)$

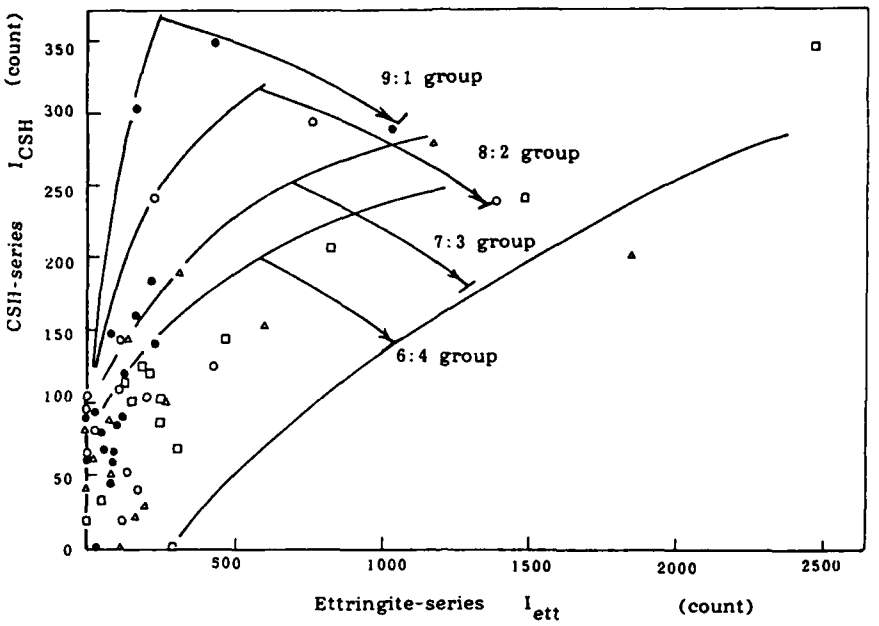


Figure 7. $I_{CSH} \sim I_{ett}$

and the water content of the samples was arranged suitably, it is considered that the pozzolanic reaction was accelerated. These results are shown in Figures 8 and 9 in relation to the amount of reactive products and unconfined compressive strength. The increase in reactive products, I_{CSH}, was directly related to the increase in q_u value shown in Figure 8. The increase in ettringite crystal, I_{ett}, was also connected with the strength increase shown in Figure 9. The decrease in combination ratio 9:1 to 6:4, however, showed the decrease in the strength in spite of the great products of ettringite crystal. This means that reactive products of ettringite do not directly correspond to the strength increase, but are related to it in inverse proportion. If we tried to increase the ettringite product by enclosing free water, a decrease in the strength would occur.

Consequently, the lime-cement mixture treatment is recommended and the optimum combination ratio is 9:1 or 8:2.

CONCLUSIONS

The hardening treatment of freshwater very soft clay soil formed by land reclamation was introduced. The applicability of lime-cement-mixture was investigated in the laboratory. This special stabilizing agent aims at the pozzolanic reaction and ettringite product in order to solidify the very soft clay soils by enclosing free water into its crystal. The hardening mechanism was examined from the physicochemical point of view.

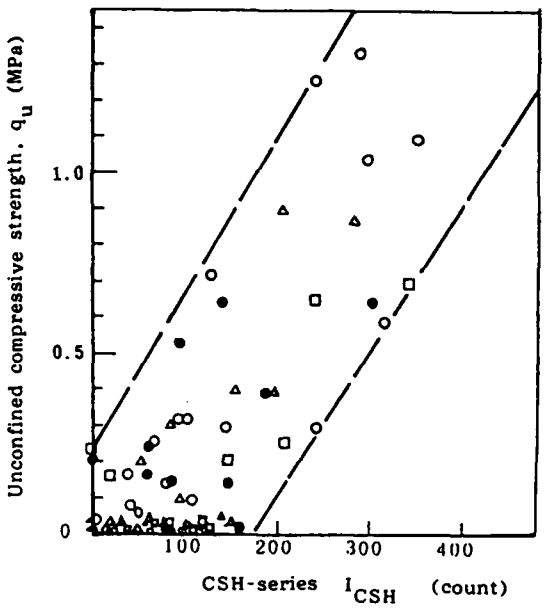


Figure 8. $q_u \sim I_{CSH}$

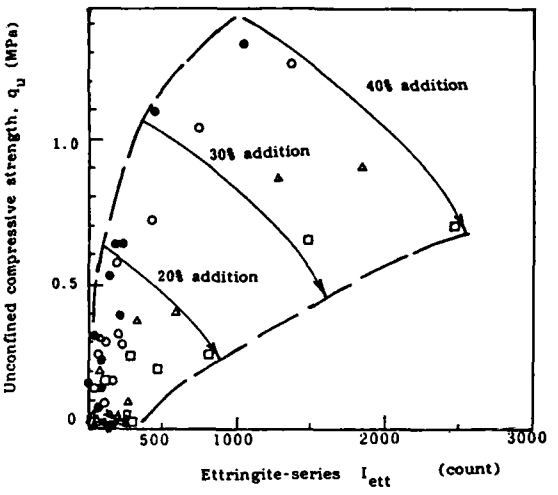


Figure 9. $q_u \sim I_{ett}$

The results obtained are as follows:

a. The freshwater very soft clay soil used is mainly halloysite and kaolinite minerals in suspension for a long time. Thus, the strength due to self-consolidation after sedimentation is very small.

b. The hardening strength increases with the increase in the added volume of stabilizing agents. Trafficability on this soft ground is obtained by adding more than 15 percent of the stabilizing agent.

c. The optimum combination ratio is 9:1 to 8:2. Beyond these ratios, the effect of the ettringite crystal reduces the hardening strength.

d. The hardening mechanism was quantitatively shown in the reactive products by X-ray diffraction analysis and scanning electron microscopic study. It was clearly seen that the pozzolanic reaction products are directly related to the strength increase.

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DEWATERING OF BOTTOM SEDIMENTS

Z. Yoshino, K. Sato, H. Okajima
Science University of Tokyo
Kagurazaka 1-3, Shinjuku-ku
Tokyo, Japan, 162

F. Kodama, S. Taguchi
Kasumigaura Work Office
Ministry of Construction
Tsuchiura, Ibaragi Prefecture
Japan, 311

R. Sudo, M. Okada
Laboratory of Freshwater Environment
National Institute of Environmental Studies
Tsukuba-Gakuen, Ibaragi Prefecture
Japan, 305

ABSTRACT

The properties and dewatering characteristics of bottom sediments that had accumulated in Shinobazu Pond and Lake Kasumigaura were examined. Comparative studies were conducted using dewatering methods available to handle troublesome sludge. The continuous vacuum precoat filter was chosen as the most effective dewatering technique based on theoretical and experimental considerations.

Field investigations at Ueno Shinobazu Pond and Lake Kasumigaura confirmed that the continuous drum vacuum precoat filter method is the best method available for treatment of bottom sediments from both a technical and environmental viewpoint.

INTRODUCTION

In general, bottom sediments in closed bodies of water contain extremely fine particles and colloidal substances which are troublesome to dispose of. As the following equations show, the smaller the particle size x, the slower the dewatering velocity u ($=1/A \cdot dV/dt$).

For a gravitational dewatering process such as sand filter or dewatering by solar drying bed:*

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* See Notation at end of paper.

$$u = (1/A)dV/dt = \varepsilon^3/k'(1 - \varepsilon)^2(1/S)^2\rho g\Delta h/L_s$$

$$= \varepsilon^3/k'(1 - \varepsilon)^4(x/\phi)^2\rho g\Delta h/L_s \quad (1)$$

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

$$u = (1/A)dV/dt = \varepsilon^3/k'(1 - \varepsilon)^2(1/S)^2\Delta p \cdot g_c/\mu \cdot L_s$$

$$= \varepsilon^3/k'(1 - \varepsilon)^4(x/\phi)^2\Delta p \cdot g_c/\mu \cdot L_s \quad (2)$$

Figure 1 shows the particle-size distribution of bottom sediment measured at Shinobazu Pond and Kasumigaura which demonstrates that very small particles predominate with a small percentage of sand particles (Figure 2).

Figure 3 indicates the surface subsidence rate of the dredged bottom sediments deposited in the solar drying bed at Lake Kasumigura from March to July 1984. It can be seen that the dewatering rate is very slow, less than 0.4 cm/day.

Even if the area of solar drying bed is 70,000 m^2, the quantity of water to be removed might be no more than 280 m^3/day. Therefore, the improvement of an effective dewatering method has received considerable attention in the last few years at Lake Kasumigaura and Lake Biwa in Japan.

The theoretical and practical considerations of dewatering methods have been investigated for several years. The findings are reported herein.

FUNDAMENTAL CONSIDERATIONS

In general, the relation between filtrate volume V and dewatering time t, i.e. a filtration curve, may be expressed by a parabola, as shown in Figure 4, in accordance with the following equation:

$$t/V = (1/K)V + (2/K)V_m \quad (3)$$

where

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

$$\alpha = 2A^2\Delta pg(1 - mw)/Kw\mu\rho$$

$$= (1 - \varepsilon)^3/\varepsilon^3(1/\rho_s)k(\phi/x)^2 \quad (4)$$

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

From Equation 3, the following relation can be derived,

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

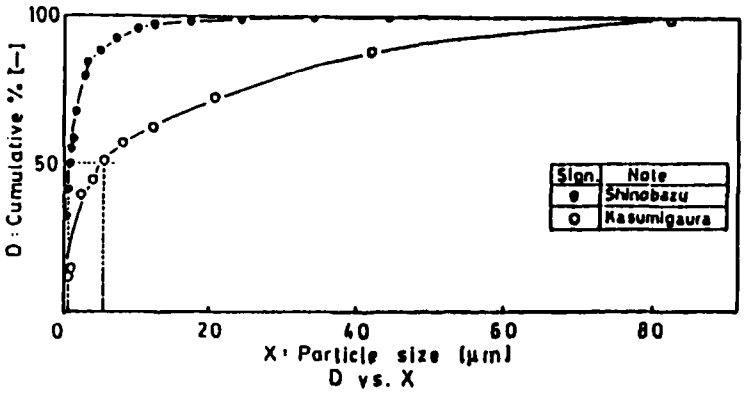
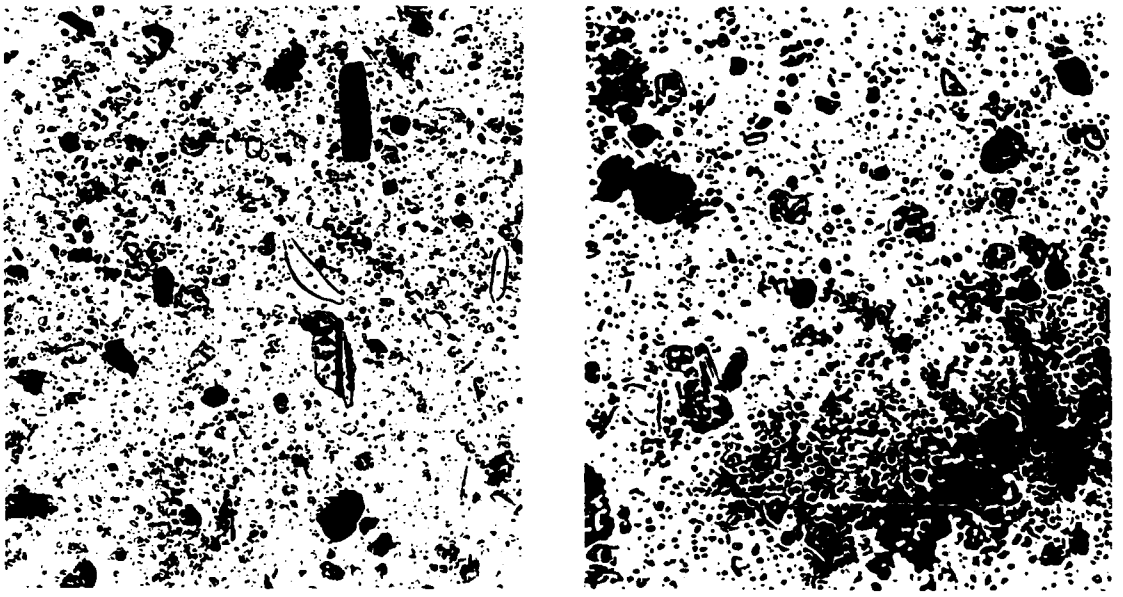


Figure 1. Particle-size distribution of bottom sediment



a. Shinobazu Pond

b. Lake Kasumigaura

Figure 2. Bottom sediment

Usually, the resistance of filter medium R_m is much smaller than the specific cake resistance α. Therefore, the imaginary filtrate volume V_m corresponding to the resistance R_m of filter medium becomes negligible as shown by Equation 6. In this case, Equation 6 becomes the following expression, approximately:

$$V/t = 2A^2\Delta p g_c(1 - mw)/\alpha w\mu\rho \cdot t^{-1/2} = K \cdot t^{-1/2} \qquad (7)$$

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

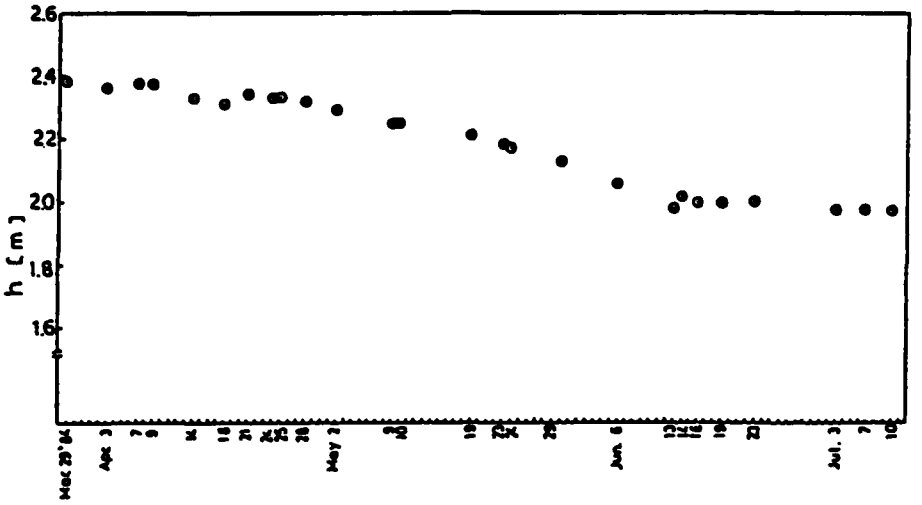


Figure 3. Subsidence of surface of the dredged bottom sediment (Kasumigaura)

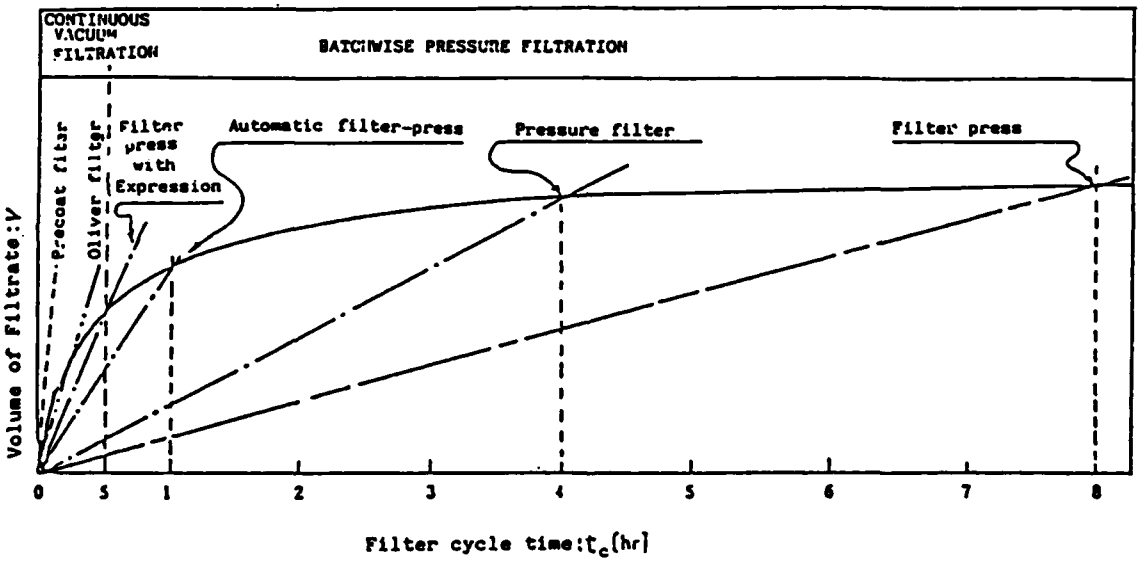


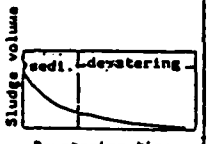
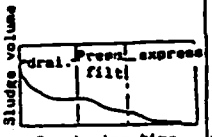
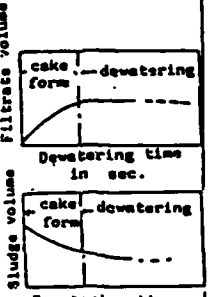
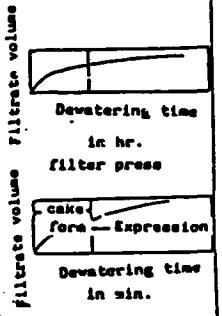
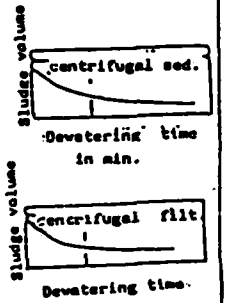
Figure 4. Filtration curve and dewatering apparatus

The slopes of the dotted line in Figure 4 and Table 1 provide a comparison of the filtration rate of conventional dewatering methods; the continuous vacuum drum precoat filter is seen to be the most effective dewatering technique from among the conventional processes.

EXPERIMENTAL APPARATUS AND PROCEDURES

Fundamental experiments and investigations on the dewatering characteristics of various bottom sediments were conducted with (a) the batch testing apparatus using a vacuum filter (Figure 5) and a pressure filter, and (b) the small-scale continuous vacuum filter (Figure 6) at Science University of Tokyo, supported by the Ministry of Education, Japan.

TABLE 1. DEWATERING PROCESS AND CYCLE TIME OF SEVERAL DEWATERING APPARATUS

PROCESS	Gravitational Dewatering	Dewatering by Screening	Vacuum Filtration		Pressure Filtration	Centrifugal Dewatering
PROCESS	Lagoon Sand filter	Flocculation Agglomeration	Batchwise	Continuous	Batchwise	
CYCLE TIME	several days several tens of days	2–15 (min)	2–6 (hr)	30–600 (sec)	1–15 (hr)	
DEWATERING CURVE						

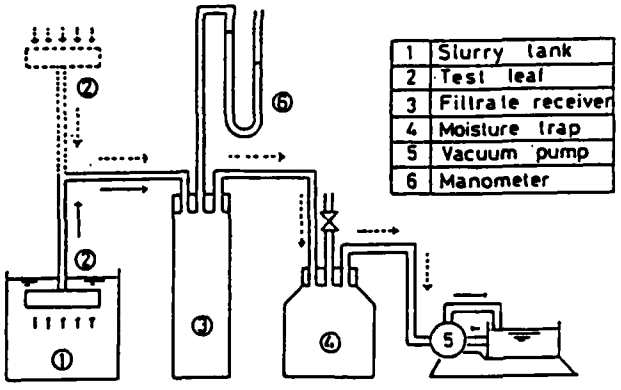


Figure 5. Experimental apparatus (batchmode)

Following these investigations, we carried out field experiments at Shinobazu Pond with the continuous vacuum precoat filter (surface area of 0.5 m^2) (Figure 7) from January to March 1982 and with the continuous vacuum precoat filter (surface area of 2 m^2) (Figure 8) at Lake Kasumigaura from January to July 1984.

Figure 9 illustrates the outline of the experimental plant employed at Kasumigaura, which consisted of a clarifier to remove the sand fraction and the continuous vacuum precoat filter as the sludge dewaterer ⑥. Dredged material is pumped to the clarifier ② by a slurry pump ① and hydroseparated to remove sandy material, and then flows into the filter tank ⑥ in Figure 9.

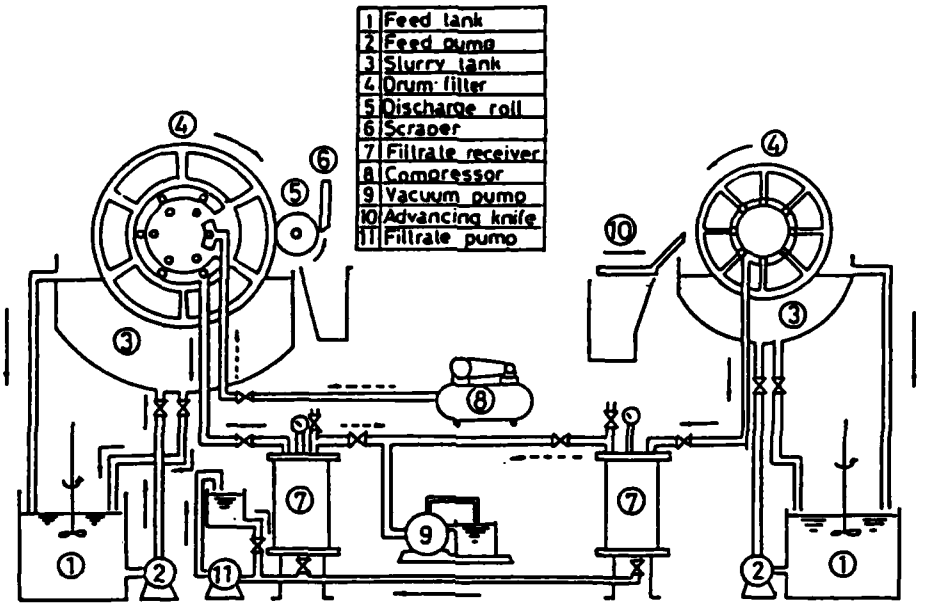


Figure 6. Experimental apparatus (continuous mode)

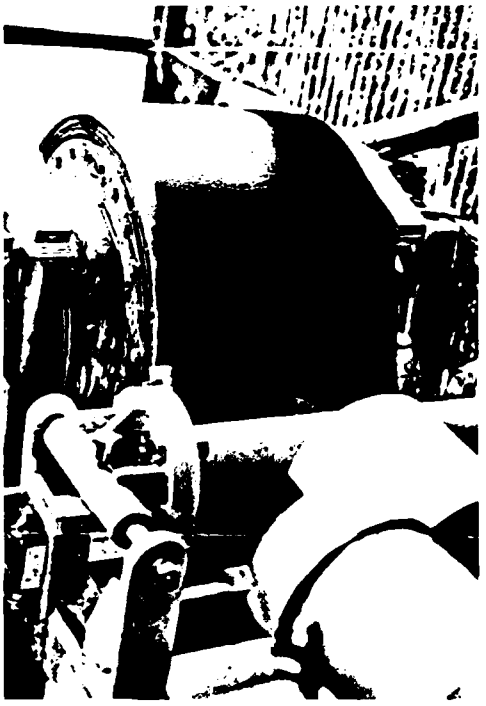


Figure 7. Continuous vacuum precoat-filter dewatering the bottom sediment of Shinobazu Pond, Ueno Tokyo



Figure 8. Continuous vacuum precoat-filter dewatering the bottom sediment of Lake Kasumigaura, Ibaragi Prefecture

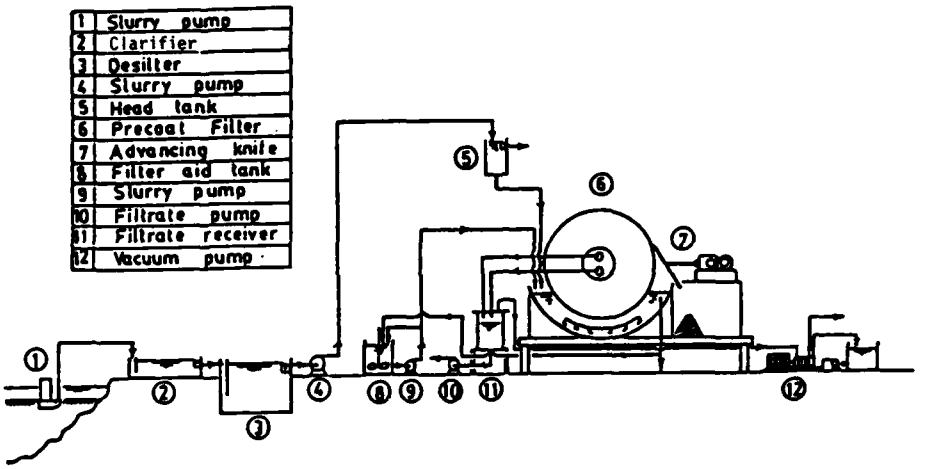


Figure 9. Experimental plant situated on Lake Kasumigaura

The thin cake formed on the precoat cake of filter aid, e.g. diatomite or pearlite, is removed by the advancing knife (7) while the clarified filtrate is pumped back to the lake.

EXPERIMENTAL RESULTS AND CONSIDERATIONS

Dewatering Characteristics

Figure 10 indicates the relation between filtration pressure Δp and specific resistance α of various bottom sediments measured by the Leaf-test mentioned in Figure 6.

It was found that these sediments were very difficult to dewater by conventional dewatering processes such as filter press, belt-press, and/or solar drying bed (Table 1) because of the extremely high specific resistance and excessive compressibility (Figure 10).

Recovery of Sandy Materials

Figure 11 illustrates the particle-size distribution curve of the settled fraction from each stage of the clarification process shown in Figures 9 and 12.

It was confirmed that sandy materials could be easily recovered by the specially designed hydroseparater, with a retention time of about 10 min.

Filter Capacity

Figure 13 indicates the filtrate capacity of the continuous vacuum precoat filter facility. It was found that even troublesome sediments could be efficiently dewatered. Filtrate capacity was approximately 0.7 $m^3/m^2/hr$ for the bottom sediments of 10% solid content, and 0.3 $m^3/m^2/hr$ for sediments of 30% solid content.

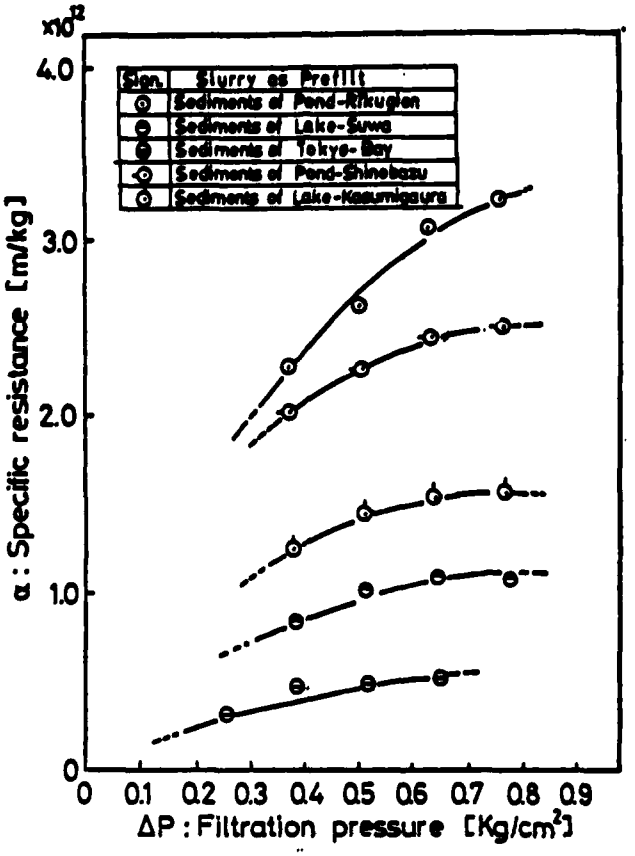


Figure 10. Specific resistance of bottom sediments

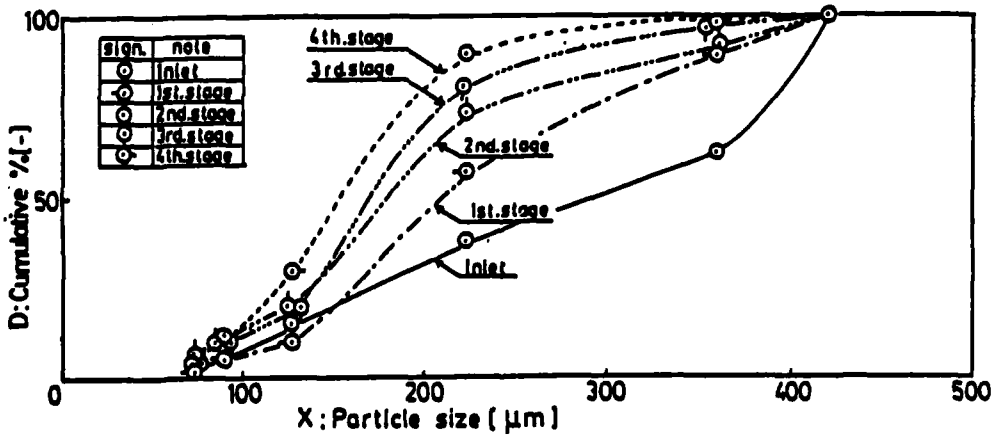


Figure 11. Particle-size distribution of settled solids

Since the slope of the dotted line expressing experimental results is about -1/2, the fundamental equation (7) is applicable to these dewatering phenomena and can be represented by the following equation, corresponding to the relation shown by Figure 14.

$$S = V/(A \cdot t) \fallingdotseq K \cdot w^{-1/2} \cdot t^{-1/2} \qquad (8)$$

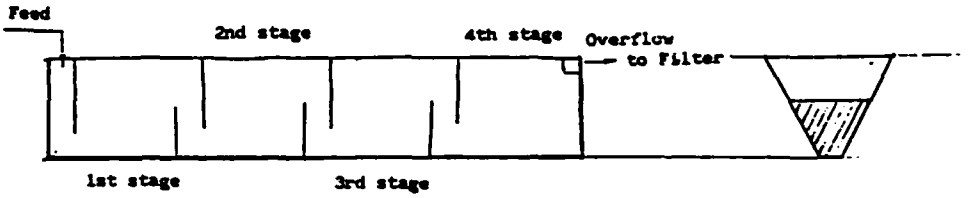


Figure 12. Schematic of clarifier

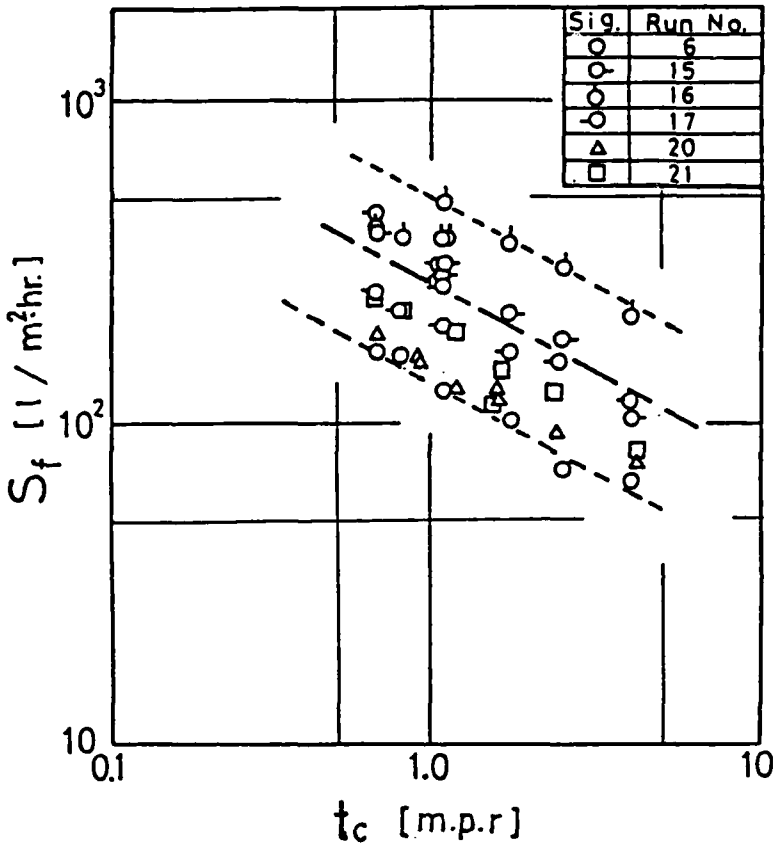


Figure 13. Filtrate capacity of continuous precoat filter for the bottom sediments of Lake Kasumigaura (tc is cycle time, m.p.r. = min)

Filtrate Effluent Concentrations

Filtrate discharged from the continuous vacuum precoat filter was clear enough (10 mg/ℓ suspended solids) to be pumped back to the lake and/or reused as process water.

Moisture Content of Cake

Moisture content of the cake removed from the continuous vacuum precoat filter was less than 50%, and therefore eligible for disposal or reuse as a

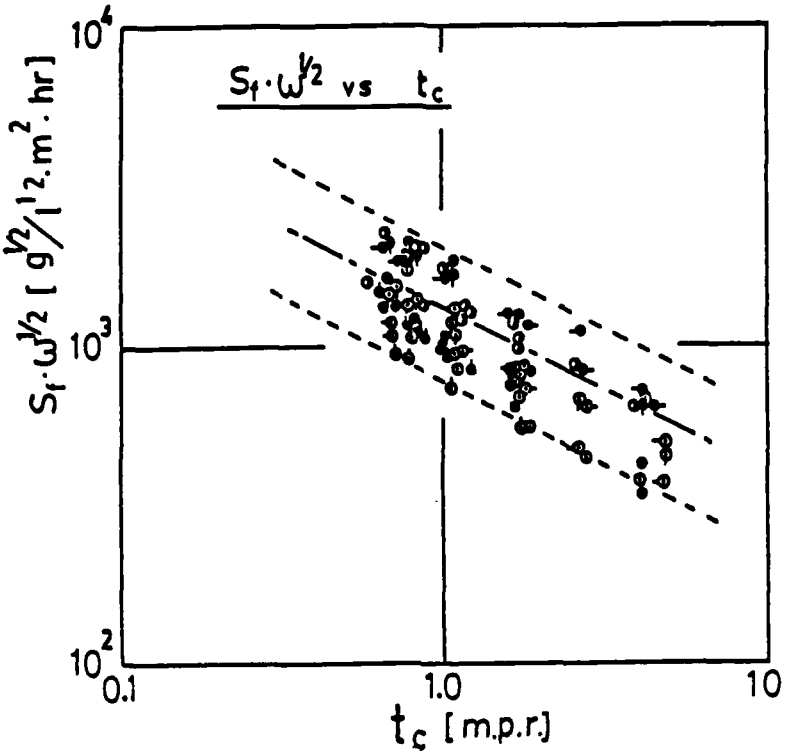


Figure 14. $S_f\ W^{1/2}[g^{1/2}/L^{1/2} \cdot m^2 \cdot hr]$v.s. t_c[m.p.r.]

fertilizer without any difficulty. In addition, since the cake was very thin (less than 1.0 mm), it could dewater until the moisture content was 35% by subsequent solar drying, during 1 or 2 days as shown by Figure 15.

CONCLUSIONS

The following conclusion can be drawn from this study:

a. The properties and dewatering characteristic of various bottom sediments were examined, and it was found that these sediments were very difficult to dewater by conventional methods because of the extremely high specific resistance and the excessive compressibility.

b. The sandy materials contained in the sediments were easily recovered by the hydroseparater.

c. Comparative studies were made on a dewatering process available to such troublesome sludge, and the continuous vacuum precoat filter was chosen as the most effective dewaterer by theoretical and experimental investigations.

d. According to field experimental investigations at Shinobazu Pond and Lake Kasumigaura, the following data have been extracted:

(1) These troublesome sediments could be efficiently dewatered with the continuous vacuum precoat filter.

(2) The filtrate capacity may be expressed by the following equation:

$$S = V/(A \cdot t) \fallingdotseq K \cdot w^{-1/2} \cdot t^{-1/2}$$

(3) Filtrate discharged from the continuous vacuum precoat filter had suspended solids concentrations as low as 10 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 was less than 50%, and therefore eligible for disposal or reuse as a fertilizer.

e. From the theoretical and experimental investigations, the continuous vacuum precoat filter may be the best method available for the treatment of bottom sediments from both a technical and an environmental viewpoint.

ACKNOWLEDGEMENTS

We are very grateful to Mr. Yasushige Kobayashi and Shiro Suimatsu, Tokyo Metropolitan Government. These investigations were largely supported by a Grant of the Ministry of Education, Japan.

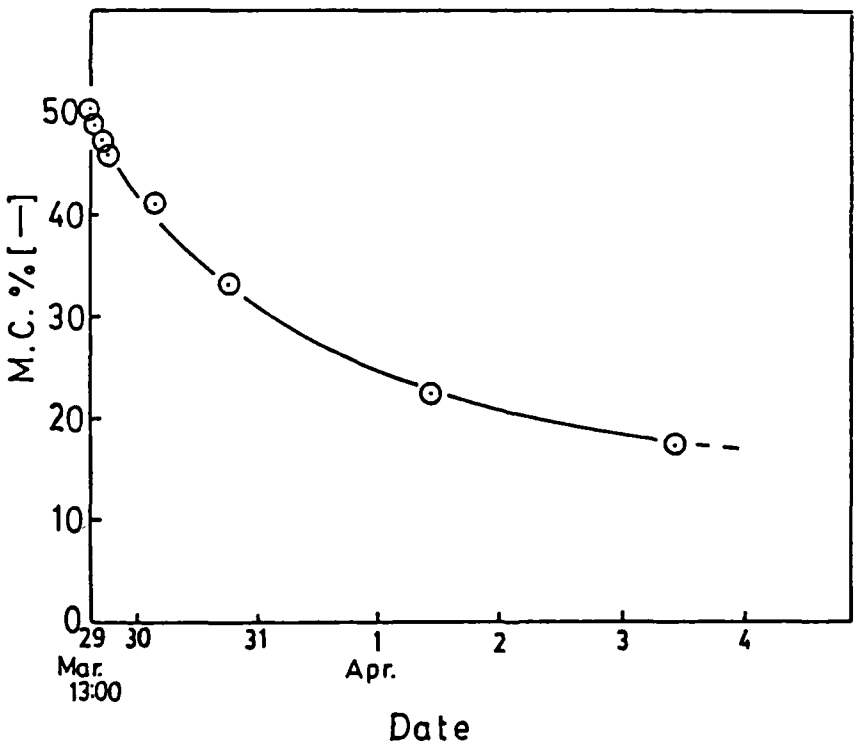


Figure 15. Dewatering curve of the filter cake by solar drying (M.C. = moisture content)

NOTATION

A	Filtration area	cm^2 or m^2
k	Empirical constant	--
K	Filtration constant	m^2/sec
m	Weight ratio of dry cake to wet cake	--
g	Acceleration due to gravity	cm/sec^2
g_c	Conversion factor	$kg \cdot m/kg \cdot sec^2$
L_s	Depth of filter medium	cm or m
S	Specific surface area	cm^2/cm^3
S_f	Filtrate capacity	$\ell/(m^2 \cdot hr)$
t	Filtration time	sec or hr
t_c	Filter cycle time	min or hr
u	Filtration velocity	$cm^3/cm^2 sec$
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	$g/cm \cdot sec$
ρ	Density	g/cm^3
ϕ	Shape factor	--

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FLOW PROCESSES OF A CUTTERHEAD DREDGE*

L. S. Slotta

President, Slotta Engineering Associates, Inc.
PO Box 1376
Corvallis, Oregon 97339

ABSTRACT

Feasibility studies of new dredge intake and cutterhead shielding techniques were conducted to improve hydraulic cutterhead dredge performance. Simple modification of the suction mouth to the intake piping and utilization of both internal as well as external dredge cutter shrouds have been found to significantly improve the production of a cutter suction dredge model.

INTRODUCTION

In sustaining the nation's navigable waterways, the Corps of Engineers annually dredges over 400 million cubic yards in maintenance dredging operations and over 200 million cubic yards in new work dredging at a (1976) cost over $150 million. Much of this work is contracted out competitively to independent dredgers. Surprisingly, dredging work is difficult to obtain as the work is spread seasonally and spatially into small work packages. The result is that about half of the U.S. contractor-owned dredging fleet is idle at any given time. With a bleak economic outlook, the private dredging contractors are unmotivated to upgrade and modernize dredging equipment to operate more efficiently or to reduce environmental impacts caused during dredging operations in rivers and estuaries.

National Science Foundation-Small Business Innovative Research support was received for investigating the economic effect on dredging production of cutterhead dredges through simple design modifications such as adding cutterhead shields, making minor changes to the cutterhead form, and properly placing the suction inlet relative to the cutterhead rotation. The ultimate goal was to provide the dredging industry with cutterhead and suction inlet design concepts that would increase the excavation and production efficiency of hydraulic cutterhead pipeline dredges, and accordingly reduce objectionable turbidity currents frequently created by dredges operating in waterways.

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* Permission to reprint was granted from the American Society of Civil Engineers who first published this paper in the PROCEEDINGS for DREDGING 84, a specialty conference held in Clearwater Beach, Florida, November 14-16, 1984.

Computer model simulations were used to investigate the processes of soil movement and transport by a confused three-dimensional flow field affected by complex boundaries, rotating cutterblades, variable flow rates, and positioning of the intake to the dredge pump. Rotational speed, haul velocity, depth of suction inlet, and cutterhead shielding techniques were verified and tested during hydraulic model simulations. Suction inlet types, internal codes, and external shields utilized in these feasibility studies are shown in Figure 1.

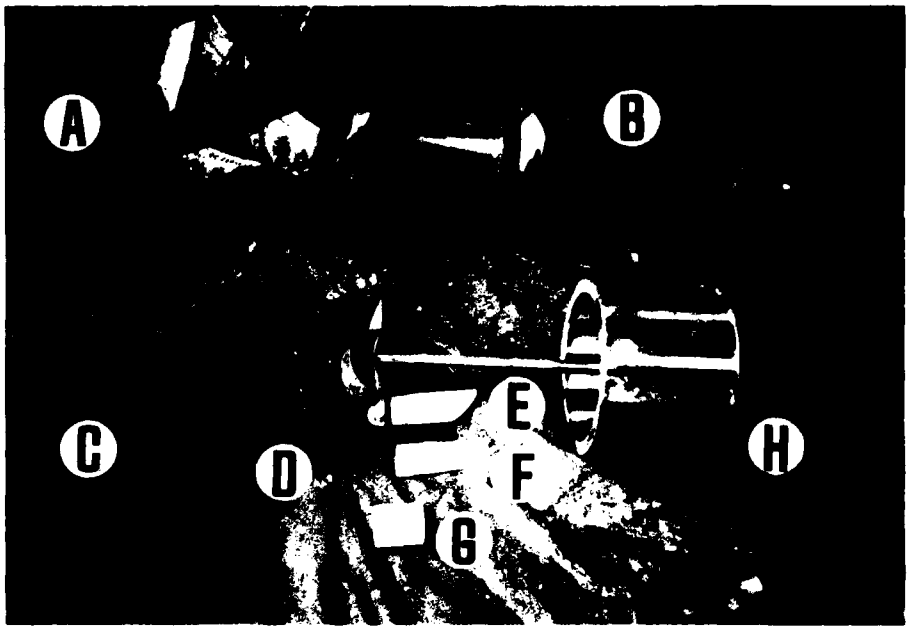


Figure 1. A: crown-shaped cutter (1:6 scale, 19-in. diameter, 16 in. long; B: crown-shaped cutter (1:12 scale, 9-in. diameter, 6 in. long); C: dog-eared cutter shroud; D: half cylinder inner core; E: 45-deg beveled suction mouth; F: long beveled suction mouth; G: small reentrant suction mouth; H: 12 bladed cylindrical cutterhead (1:12 scale, 9-in. diameter, 6 in. long)

For both computer model and hydraulic model simulations, a cylindrical-shaped disc cutter was utilized as a basic model. Evaluations of the cylindrical cutter can easily be extended to basket type cutters. The characteristic feature of the cylindrical shaped disc cutter (Figure 1H) is the potential optimal positioning of the suction mouth which extends into the interior of the cutter towards the rotating blades. High suction velocities to a reentrant suction mouth would reduce the recirculation of the soil cuttings within the cutterhead's crown. This suction mouth can be turned towards the direction of cut to improve the inflow to the pump intake.

COMPUTER CODES

Complex computer codes recently made available by CHAM Ltd. (Concentration, Heat and Momentum Limited, Bakery House 40 High Street, Wimbledon, London SW195AU ENGLAND) and CREARE R&D, Inc. (CREARE R&D, Inc., PO Box 71,

Hanover, New Hampshire 03755), provide three-dimensional results of turbulent two-phase flows past moving boundaries. Sediment transport into the moving blades of rotating dredge cutterhead was simulated using both CHAM Ltd.'s PHOENICS and CREARE R&D Inc.'s FLUENT codes.

CHAM Ltd.

CHAM Ltd. developed a software package designed for the simulation of fluid-flow, heat-transfer, mass-transfer, and chemical-reaction processes. It is called PHOENICS, which stands for Parabolic, Hyperbolic or Elliptic Numerical Integration Code Series. In the parabolic form, effects do not propagate upstream; in the elliptic form, effects do propagate upstream; and in the hyperbolic form effects propagate along "characteristics."

PHOENICS uses a finite difference numerical procedure to solve the fundamental equations governing fluid flow (the Navier-Stokes equations). Built into PHOENICS are the major laws of conservation: the mass of each present phase, the three components of momentum for each phase, the thermal-energy contents of each phase, the mass concentration of each chemical species, the energy and length scale of the turbulence, and the fluxes of radiation. The equations include individual terms expressive of the processes of convection, diffusion, generation, and destruction, and appropriate accounting of the interactions of turbulence with the laminar-flow phenomena.

With CHAM Ltd.'s PHOENICS code a cylindrical model was numerically created to represent a simplified dredge cutter having four blades or arms. Figures 2-5 are representative of these PHOENICS graphics. Figure 2 is a perspective view of a rotating four blade cutter with a velocity field located at one tenth the cutter length. The outflow of water to the suction mouth located at the 6 o'clock backplate position is clearly represented. Figure 3 shows an end view of the cutterhead, orthogonal to the drive axis and cutter blades. The rotational velocity field about the blades accelerating into the suction mouth illustrates some of the flow representations possible with PHOENICS graphics routine. Figure 4 is a perspective composite of the previous information, including: cutterhead shape, blade position, velocity field near the backplate, and the flow paths of particles released at "tenth" positions long the top of the cutterhead. Superimposed on Figure 5 are the flow trajectories of particles cut from the 12 o'clock position.

The PHOENICS code utilizes an Eulerian approach in which the fluid density in a cell of a computational grid is uniform and continuous. However, the density in an adjacent cell can be designated as having a different value, thus allowing for two-phase flow calculations.

The two-phase flow test representation used a submerged vertical suction pipe held above a sand bed. Figure 6 shows the sediment velocity flow field on the right while sediment volume fractions are presented on the left. By examining the break in the sediment volume fractions one observes the zone of sediment pickup into the suction mouth. These results are being analyzed in reference to previous physical model data on the threshold pickup zone of sand taken into a suction intake (see Figures 7A (Reference 4) and 7B (Reference 2)). The results show similarities to potential flow techniques and physical measurements reported by Salzmann (3) and Brahme (1). In Figure 8

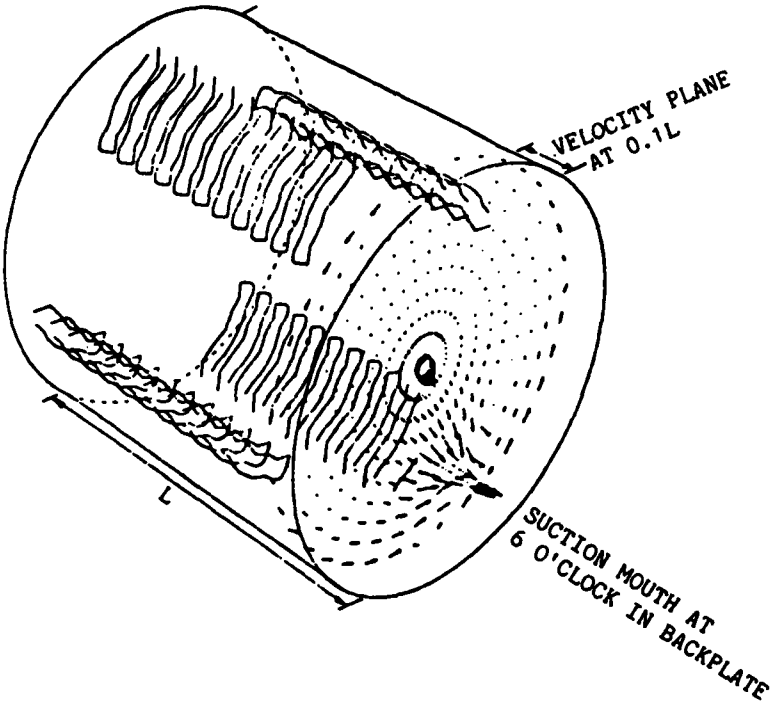


Figure 2. Cylindrical dredge cutterhead flow simulations, perspective view

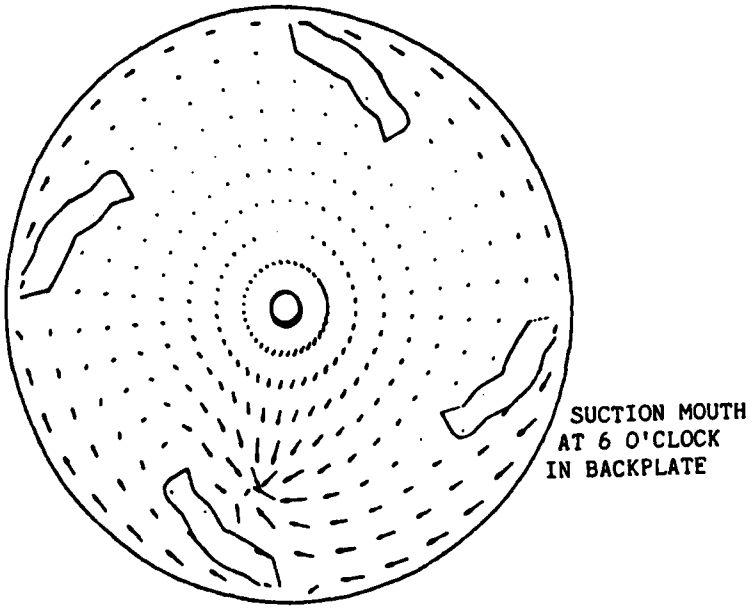


Figure 3. Cylindrical dredge cutterhead flow simulations, end view

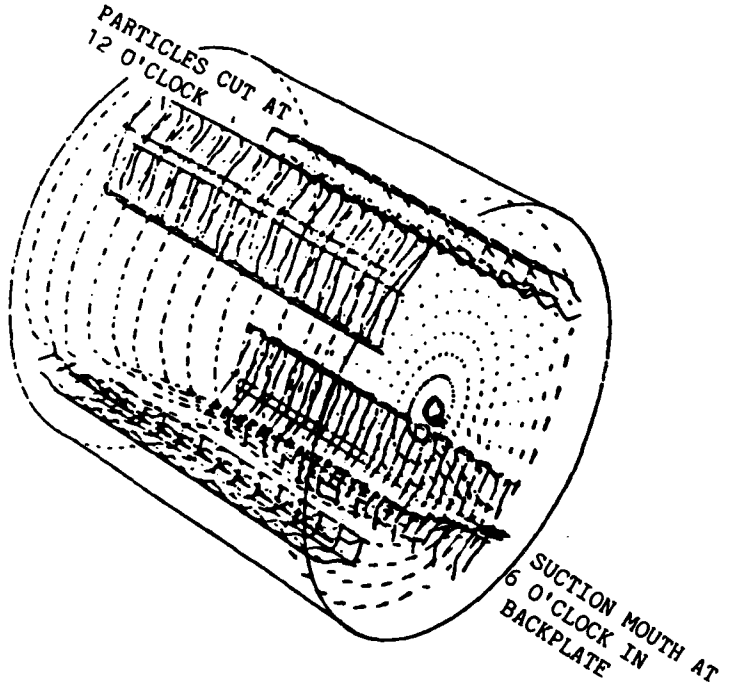


Figure 4. Cylindrical dredge cutterhead flow simulations, composite

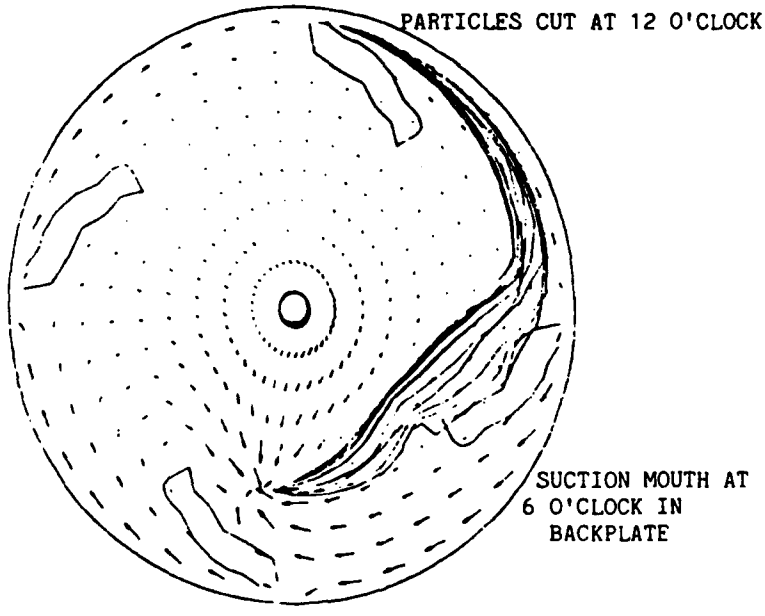


Figure 5. Cylindrical dredge cutterhead flow simulations, flow trajectories

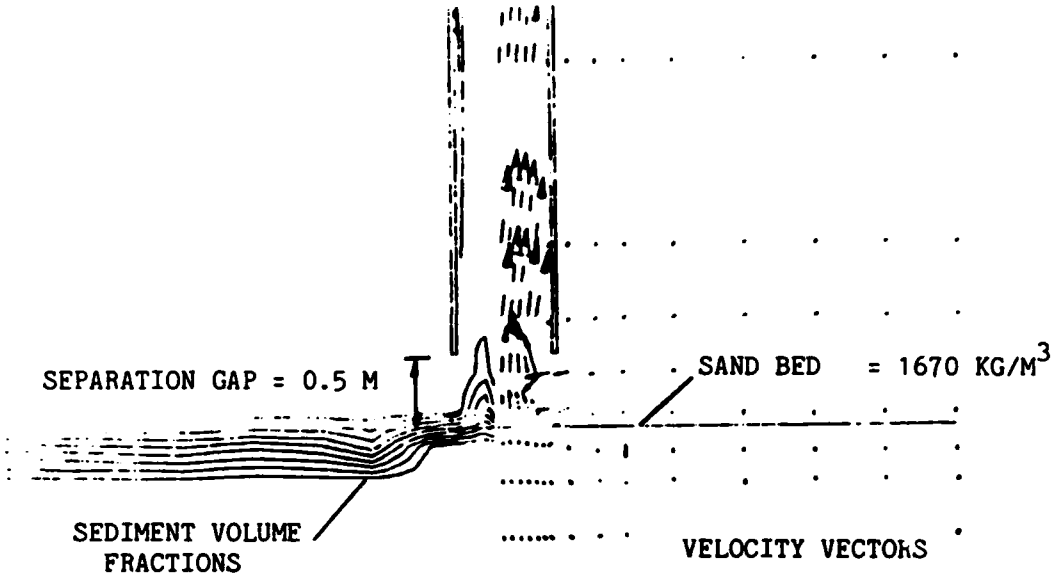


Figure 6. Sediment pickup simulation, sediment volume factions on left and water velocity vectors on right

the velocity and pressure field of flows up a 0.6-m (23.6-in.)-diam intake tube with 0.03-m walls is represented. The flow is created by a negative pressure of one atmosphere within the intake pipe. For the sake of symmetry, water phase velocities are shown on the right while the pressure drops associated with the flow are shown on the left.

CHAM Ltd.'s staff believe PHOENICS can properly simulate sediment transport with the density continuum approach. A sand bed might appropriately be represented by a volume fraction in each grid cell of 0.6 solids and 0.4 fluid. Transient flow simulations using PHOENICS of the loosely packed soil revealed the bed to progressively settle with time under the action of gravity. In actuality, such a bed would remain rigid through granular interactions rather than collapse. As these computational studies are preliminary, plans have been made to continue to evaluate this code for properly representing the physics of sediment transport.

CREARE R&D, Inc.

CREARE R&D, Inc., has developed a competitive, user-friendly, general purpose computer program for modeling fluid flow called FLUENT. FLUENT is made accessible to the user by means of an interactive menu-driven interface for problem definition, computation, and powerful built-in graphics. FLUENT enables the fluids engineer to apply state-of-the-art computer simulation methods to analyze and solve practical design problems without costly, time-consuming programming.

FLUENT uses a finite difference numerical procedure to solve the fundamental equations governing fluid flow (the Navier-Stokes equations). Additional equations are solved for the conservation of the parameters of the k-e turbulence model, chemical species, and enthalpy. Their numerical techniques involve the subdivision of the domain of interest into a finite number of

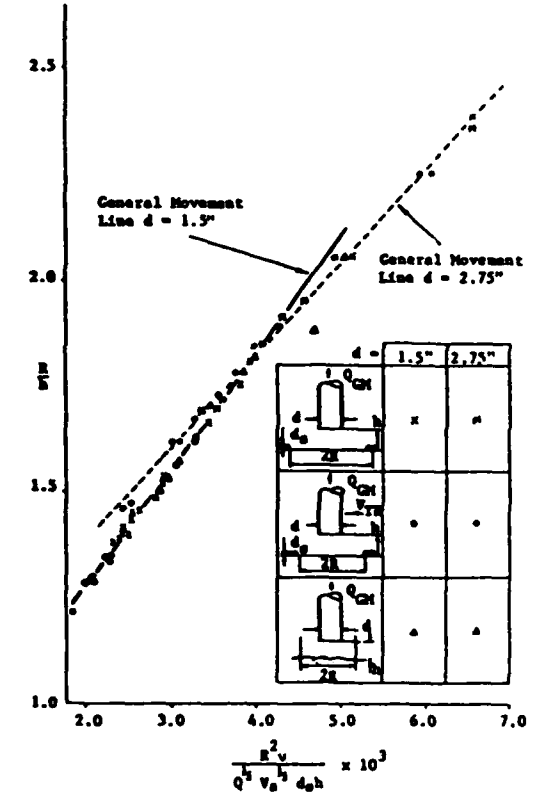


Figure 7A. Slotta's relationship for general movement (4)

$$R/h = f\left(\frac{R^2 v}{Q^{\frac{1}{3}} V_s^{\frac{1}{3}} d_s h}\right)$$

where
R	=	radius of area of scour	(L)
h	=	height of pipe above the bed	(L)
f	=	functional relationship	(-)
Q	=	flow rate	(L^3/t)
V_s	=	sand settling velocity	(L/t)
v	=	fluid kinematic viscosity (μ/ρ)	(L^2/t)
d_s	=	diameter of aggregate particles	(L)

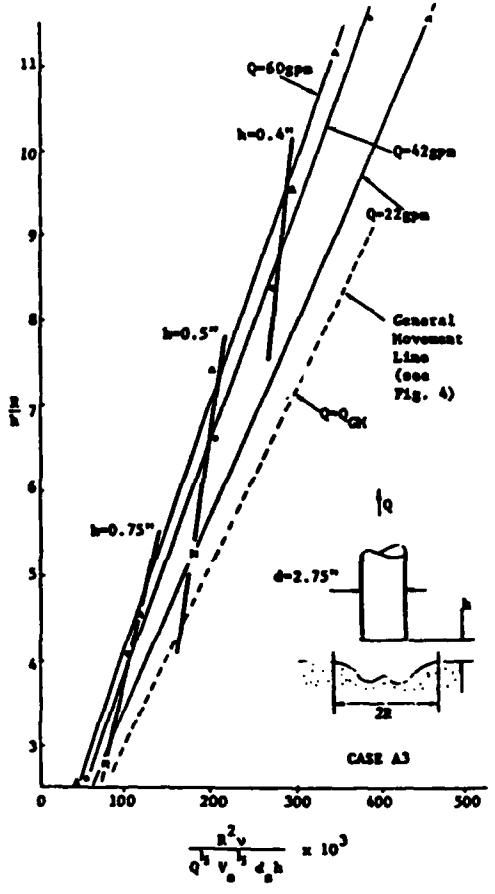


Figure 7B. Equilibrium profile d = 2.75 in. (2)

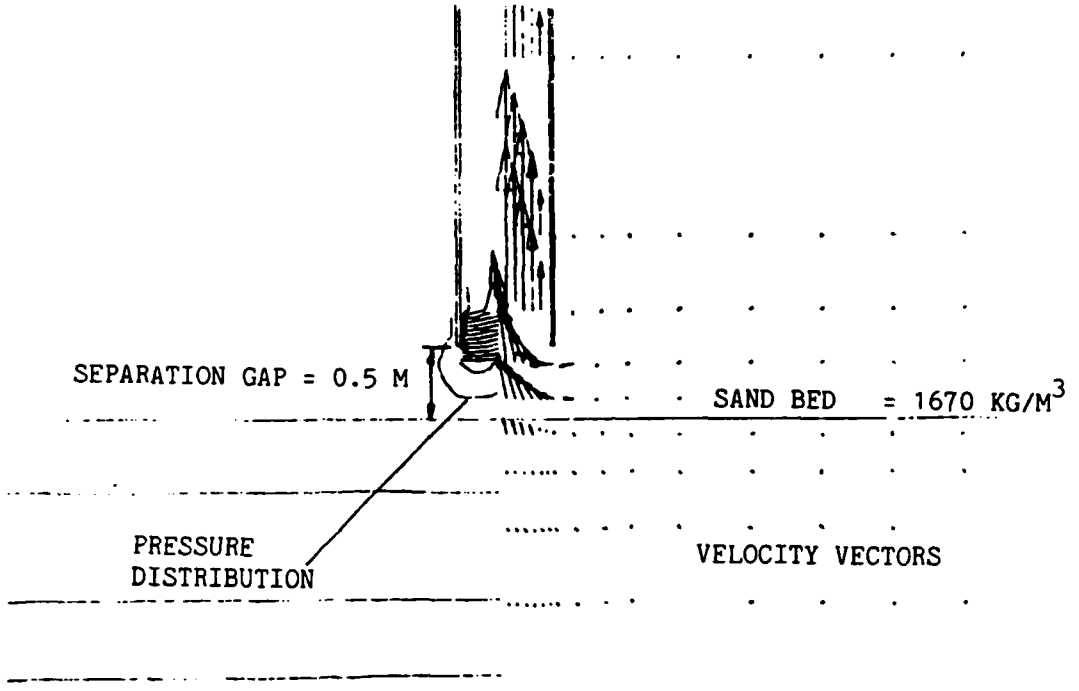


Figure 8. Sediment pickup simulation, pressure distribution

control volumes or cells, the partial differential equations being discretioned over these cells to obtain sets of simultaneous algebraic relations. Presently FLUENT is programmed only for solving steady flow problems. The FLUENT code conveniently incorporates a Lagrangian method for utilizing marker particles of various grain sizes and densities. This is most useful for portraying sediment transport through dredge cutters as shown in Figures 9A to F. A present limitation of the FLUENT scheme is that particles cannot be assembled as a bed, but can only be introduced singly at a point or in a layer through a desired boundary.

FLUENT dredge cutterhead flow simulations are depicted in Figures 9A to F. By considering symmetry, only the left half of the cylindrical cutterhead was plotted and, for clarity, the cutter blades were omitted. A range of soil particle diameters and densities was cut at different positions on the rotating cutterhead and the trajectories were plotted. The flow field vectors, as seen on a vertical plane below the axis of rotation, pass through a segmented gap in the backplate (Figure 9A). Figure 9B shows the trajectories of light particles (ρ = 1670 kg/m^3) released from the 12 o'clock position of the rotating cutterhead with a 45-deg angle; one particle is noted to enter the suction mouth. Figure 9C represents the trajectories of a range of particle diameters having a constant (ρ = 1670 kg/m^3) density, which are released at the two-tenths length position at 12 o'clock; nearly all the grains escape. Figure 9D represents similar conditions but the particle release point is midway along the cutter length; note only two particles escape while the others are trapped to recirculate about the crown of the cutter. Light (Figure 9E) and heavy (ρ = 2670 kg/m^3) (Figure 9F) particle trajectories are shown for release

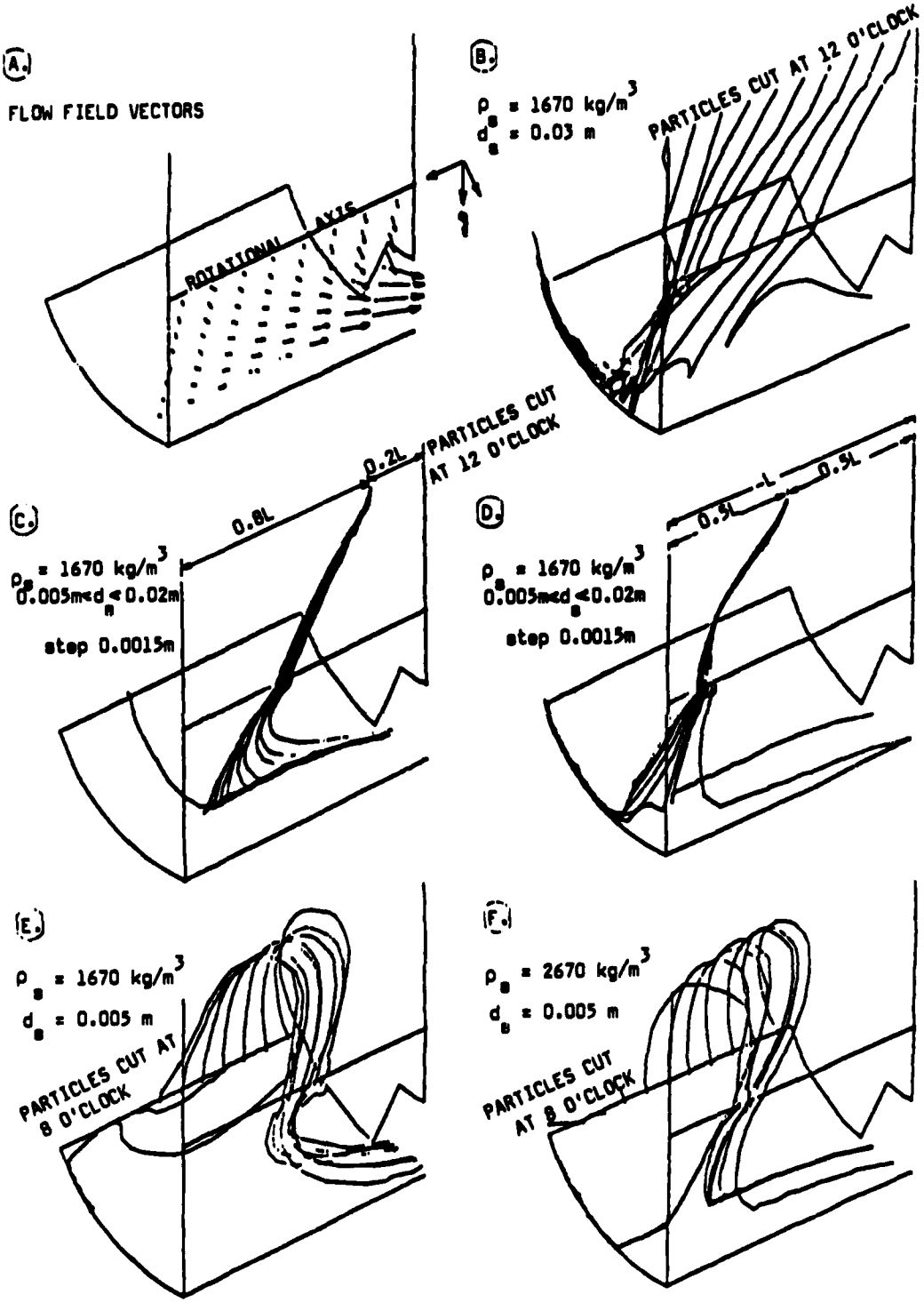


Figure 9. Perspective view of cylindrical dredge cutterhead flow simulations. Suction mouth at 6 o'clock in backplate

points near the 8 o'clock digging position. The light particles are nearly all drawn out the suction mouth whereas the heavy particles are not as successfully drawn into the intake.

Summary

In summary, both the PHOENICS and FLUENT codes have exceptional three-dimensional graphics for representing hydraulic and hydrodynamic phenomena. User-friendly computer programs for solving complex three-dimensional flow fields involving gravity, viscosity, turbulence, and difficult boundary conditions of inflow and outflow are becoming available. Problems, previously intractable by analytical or numerical techniques, are now solvable. The results gained in these dredge studies show that these hydrodynamic codes are at a stage corresponding to the stage where finite element structural dynamics codes were ten years ago.

For these studies, it would have been desirable to represent a submerged static soil bed that did not collapse over time and that was not limited to having only a single layer of particles accessible for transport. Perhaps in the future, such user-friendly hydrodynamic codes will be refined to properly represent sediment erosion and transport.

HYDRAULIC MODELS

A 1:12 hydraulic model test facility (Figures 10 and 11) was utilized as a beginning step to validate the numerical models as well as to examine improved dredge cutter performance using simplex modification to the suction inlet. A 0.23-m (9-in.)-diam cylindrical-shaped disc cutterhead model having a depth of 0.15 m (6 in.) was used to compare basic parameters, including: angular rotational speed, haul velocity, ladder angle, and position of the suction inlet. The model cutterhead was mounted on an adjustable ladder connected to a frame holding a 1/8-hp gearmotor with a speed controller and a 18.3-m^3/hr (0.18-cfs) pump. The rim of a 3.04-m (10-ft) by 0.91-m (3-ft) stock watering tank served as a track for four castor wheels under the carriage (Figure 10).

Masonry sand with a mean grain size diameter of 0.5 mm (0.02 in.) was hardpacked with the intent of representing hard digging of consolidated sediments and to ensure reproducibility of results between runs. To consolidate the sand, a concrete-cement stinger or vibrator was first used for fluidizing the bed. Three 0.03-m (1.25-in.) perforated polyvinyl chloride (PVC) pipe units were interconnected and placed at the bottom of the 0.61-m (2-ft)-deep oval holding tank. Nylon netting was laid down over the pipe to prevent sand intake to the pipes. A consistently hard sand bed was prepared for each run by connecting the dredge pump intake to this network and running the pump for 20 min. Dense compaction of the sand was checked prior to excavation runs.

The cutter production tests were run with haul velocities averaging 0.14 ft/sec (4.19 cm/sec) and average rotational speeds of 48 to 96 RPM. The average velocity in the 32-mm (1.25-in.) suction mouth delivery line was 3.2 m/sec (10.5 ft/sec). The ladder inclination angle was set at 45 deg for all tests.



Figure 10. A 3-m-long cutterhead dredge test facility with water partially drained for viewing tracks of the dredge cuts

The hydraulic laboratory studies confirm that extending the intake pipe within the dredge cutter will significantly increase most hydraulic dredges' production. Tests compared: (a) plain suction mouth, (b) long extended mouth, (c) 45-deg beveled mouth ("looking down"), (d) 45-deg beveled mouth with an internal cowl and a dog-eared external shroud, and (e) 45-deg beveled mouth with a dog-eared external shroud (Figure 11). Overcut and undercut mode results are plotted in Figure 12. Cutter speeds are shown as a percentage of full speed of the drive. Accordingly, 100 percent corresponds to 160 RPM; 50 percent to 80 RPM; 40 percent to 64 RPM, and 30 percent to 48 RPM.

Hydraulic model summary results (Table 1) show that maximum production was observed in the undercut mode at 48 RPM and when the 45-deg beveled mouth and external shroud were used. The 45.5 percent solids by weight concentration corresponds to a 24 percent solids by volume discharge. This production is quite impressive when compared to that achieved by most operational dredges.

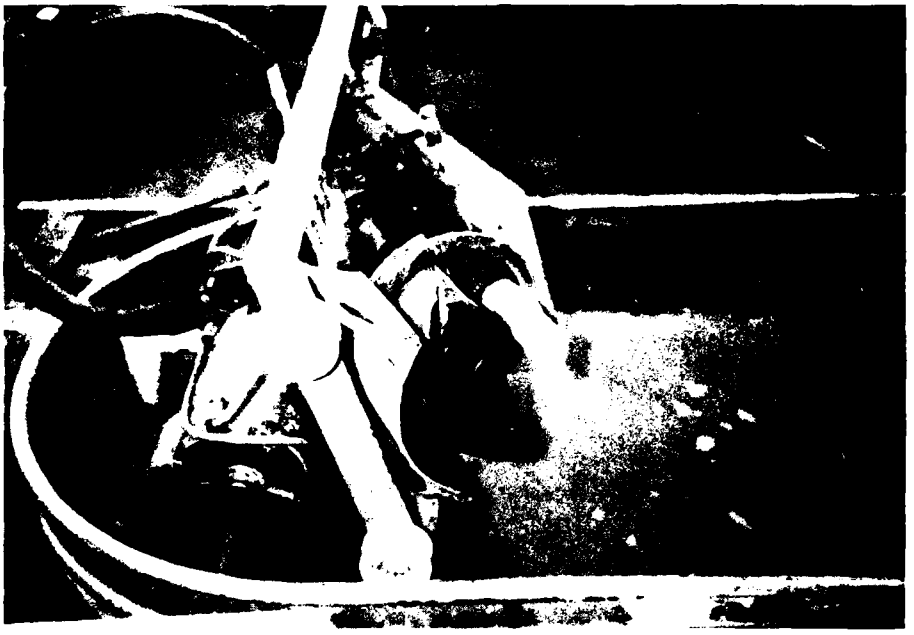


Figure 11. Cutterhead carriage tilted to allow examination of dog-eared cutterhead shroud, half cylinder internal cowl, and extended 45-deg beveled suction inlet

CONCLUSIONS

This research involves both three-dimensional numerical analyses and hydraulic model studies of rotating machinery involved in submerged sediment excavation. These hydrodynamic codes were shown to have the promise of properly simulating complex, three-dimensional, rotating flows conveying water and sediments through dredge cutterheads. Hydraulic modeling showed that simple modifications to the suction intake and the addition of dog-eared external cutter shrouds and internal cowls gave marked increases of dredge production.

The results of more extensive confirming investigations could be used to encourage private contractors to easily and economically adapt their equipment to increase their dredging output, and in the course of operation, to reduce dredge-caused turbidity in the waterway being excavated.

These investigations, particularly those advancing the use of numerical hydrodynamic flow simulations, could also assist in the development of a broad-based methodology for more scientific studies of drilling, tunneling, and dredging equipment than that which has been previously available.

ACKNOWLEDGEMENTS

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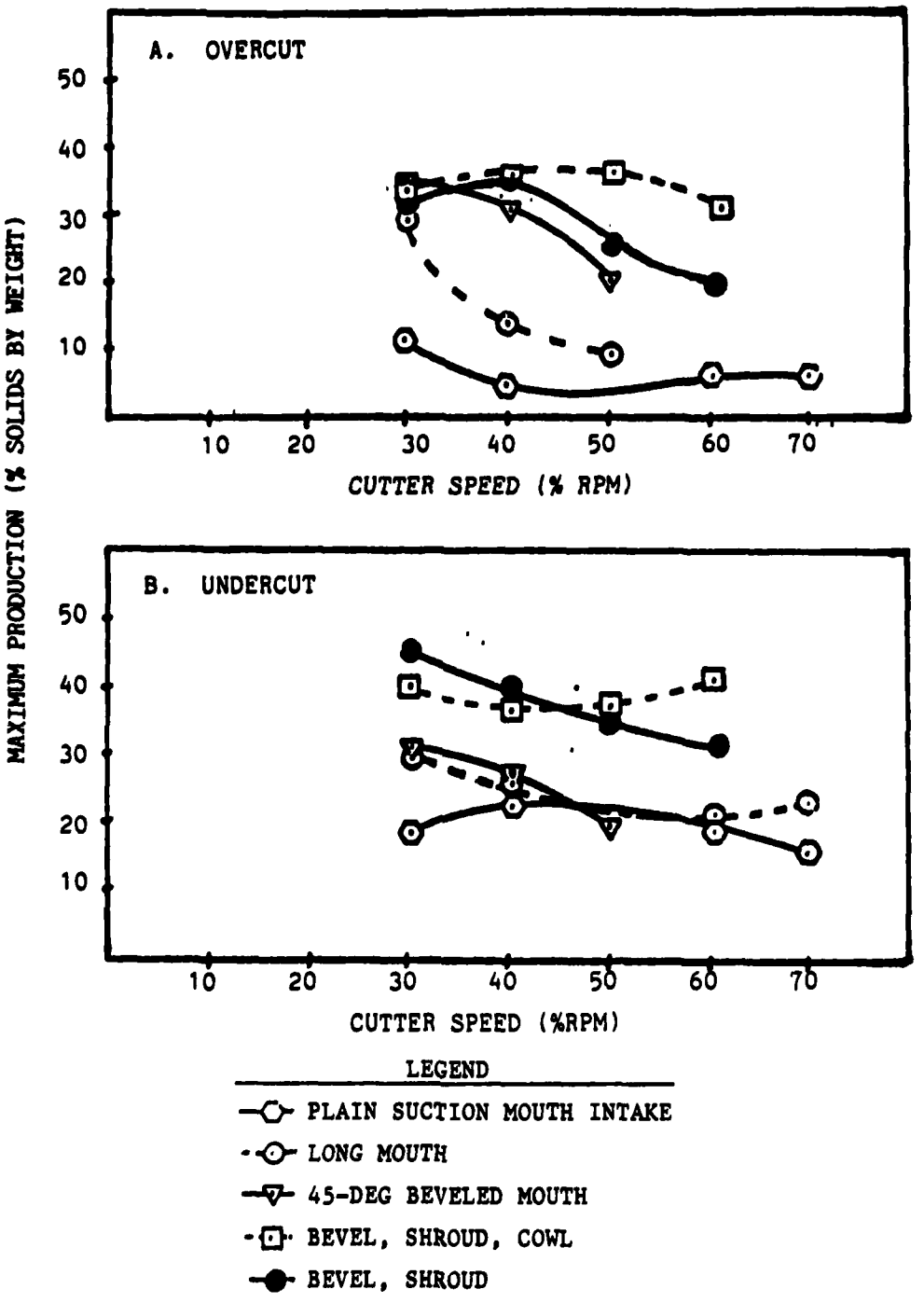


Figure 12. Plots of maximum production (percent solids by weight) and cutter speed for different shroud, cowl, and intake conditions

TABLE 1. HYDRAULIC MODEL RESULTS

Suction Mouth Inlet	Maximum Sediment Production, Percent Solids by Weight	Increase in Production Relative to Plain Suction Mouth, %
Overcut--48 rpm (30%)		
Plain	11	0
Long mouth	29	160
45-deg beveled mouth	36	225
Bevel + shroud	36	225
Bevel + shroud + cowl	41	227
Undercut--48 rpm (30%)		
Plain	30	0
Long mouth	30	0
45-deg beveled mouth	34.5	15
Bevel + shroud	34.5	51.7
Bevel + shroud + cowl	41	36

Foundation. NSF's support is gratefully acknowledged. Interest and communications on this project with Dr. John Weese, Director of NSF's Division of Mechanical Engineering and Applied Mechanics, have been gratifying.

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The staff of SEA (Slotta Engineering Associates) are thanked for their help in seeing this work completed in a professional and timely manner: Ms. J. Milbrat did much to organize the report, Mr. T. Callahan conducted the dredge head lab tests, and Mrs. S. Shribbs did a most commendable job in typing the text of the paper.

Thanks are extended to ESCO Corporation, Riedel International, Inc., and DREDGEMASTERS International, Inc., for their continuing interest in dredging research.

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SELECTION AND MANAGEMENT OF A NEW OCEAN DREDGED MATERIAL DISPOSAL SITE AT COOS BAY, OREGON, USA, TO RECEIVE CONTAMINATED DREDGED MATERIAL

J. R. Reese

Portland District, Corps of Engineers
P.O. Box 2946
Portland, Oregon 97208

ABSTRACT

Ocean disposal of dredged material on the Pacific Northwest coast has, until recently, involved only clean, medium-grain-sized marine sands. Those sediments are dredged by hopper dredge and placed at approved interim ocean disposal sites in the littoral transport zone directly offshore from the estuary. The upper reaches of those coastal river systems are normally maintained by pipeline hydraulic dredges and the material placed at upland disposal sites. The material is a silt clay sediment, and in Coos Bay is considered slightly contaminated by existing Federal criteria. Upland sites are becoming extremely difficult to obtain and at Coos Bay, Oregon, all approved sites are nearly filled to capacity. The only alternative to upland sites is transporting material to the ocean. Portland District determined that upland sites at Coos Bay would be filled by the early 1980's, and in 1977 proceeded with long-term oceanographic studies for evaluating ocean disposal sites to accept fine-grained, slightly contaminated dredged material. The District recognized that the site and sediment must be in compliance with State and Federal regulations, especially the Marine Protection, Research, and Sanctuaries Act. Chemical, physical, and biological studies were conducted for 2 years on nearshore ocean environment and dredged sediments. This led to proposing a site, followed by test disposal operations to verify predictions on transport and ultimate fate of the material. This paper reports on establishment of research, basic assumptions made to locate a new site, and the management decisions necessary for final selection and evaluation of the proposed area.

INTRODUCTION

The Portland District, U.S. Army Corps of Engineers, is responsible for maintaining coastal river channels in the State of Oregon. Approximately 6.7 million cubic meters of sediment is removed from 12 coastal river channel

entrances each year. These channels and harbor entrances are shown in Figure 1. The two coastal projects where the majority of dredging is performed are Columbia River and Coos Bay. The mouth of Columbia River requires removal of approximately 4.6 million cubic meters annually and Coos Bay 1 million cubic meters. Coos Bay dredging and disposal is the topic of discussion in this report.

Coos Bay is a large industrialized estuary, second only to the Columbia River in size and importance to Oregon. The estuary, as shown in Figure 2, is roughly 4047 ha in size and drains a total of 1567 sq km. Approximately 4 million tons of forest products are shipped through Coos Bay navigation channel annually to Far Eastern and European ports. Since 1951, Portland District has been engaged in maintenance and new work dredging to ensure depths suitable for medium-draft (9.14- to 12.2-m) vessels. The channel is 10.36 m deep and 24.14 km long and extends from the Pacific Ocean up the Coos River to the head of navigation in Isthmus Slough. Figure 3 shows the channel and general features of Coos Bay.

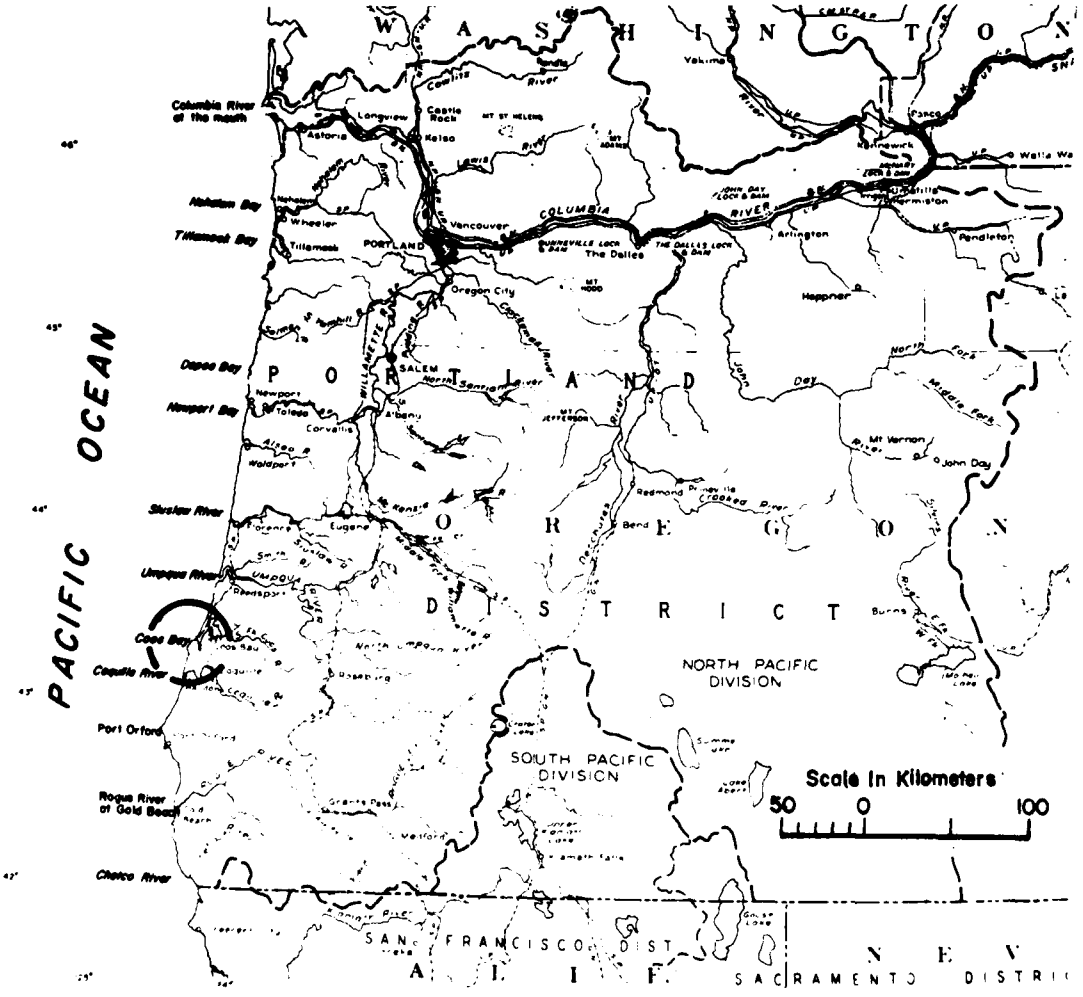


Figure 1. Location map

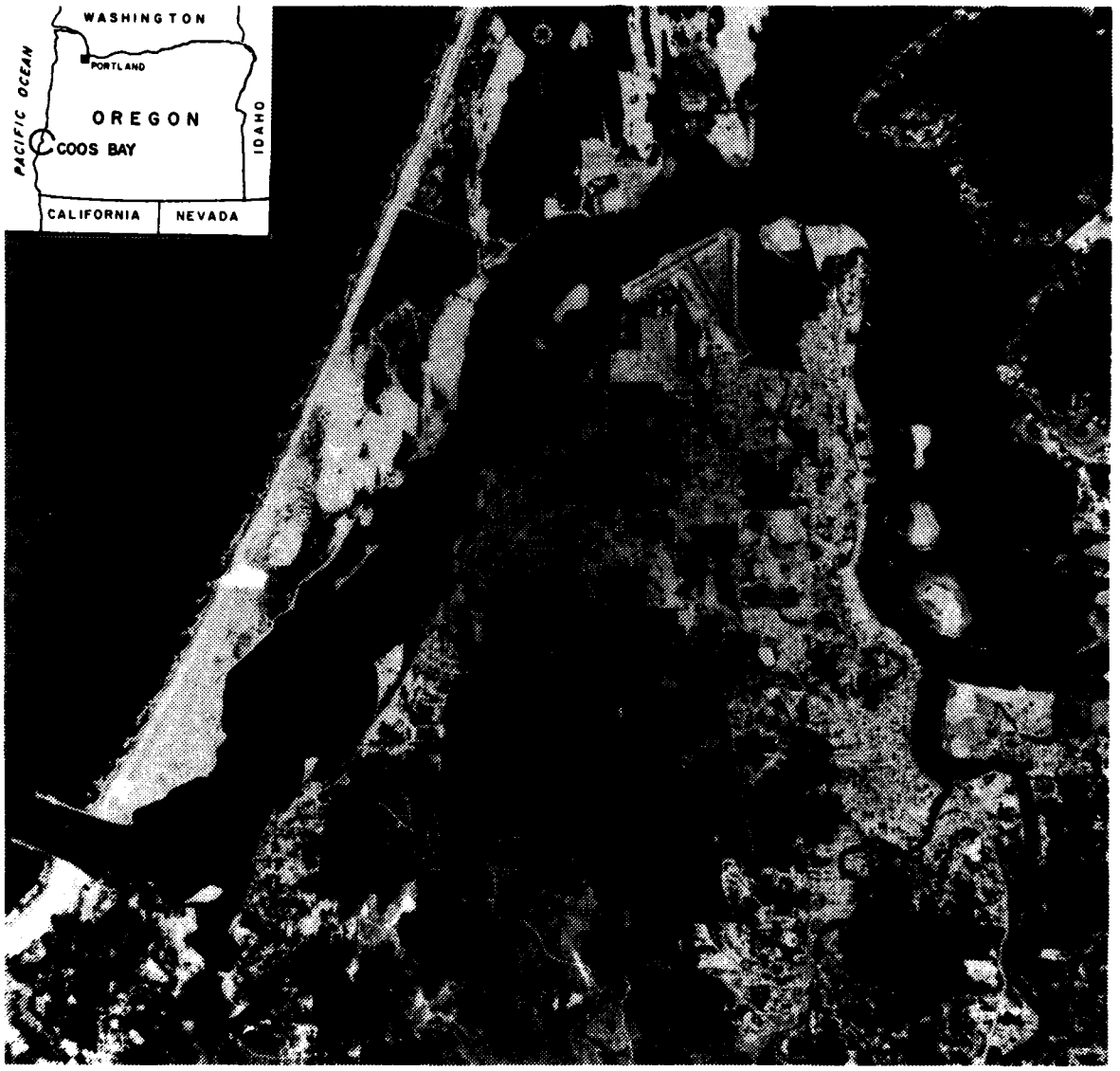


Figure 2. General features of the Coos Bay estuary

Hopper dredges historically have maintained the navigation channel from the entrance to river km 19.3, and pipeline dredges have been used in the upper channel. Hopper dredging accounts for 688,140 cu m annually, and pipeline dredging of river km 19.3 to 24.14 generates 382,300 cu m. The upper bay dredging has traditionally been deposited in upland or in-bay disposal sites. A 1975 Environmental Impact Statement (EIS) for new work dredging at Coos Bay (8) determined that all available upland sites which complied with environmental regulations would be filled in approximately 17 years (Figure 4). Even reducing dredging depths would not help for very long in the upper bay. Figure 5 shows estimated lifetime of suitable sites based on reduced channel depths. It was determined that for the upper bay channel to be kept open, the majority of dredged material would require removal to approved ocean disposal

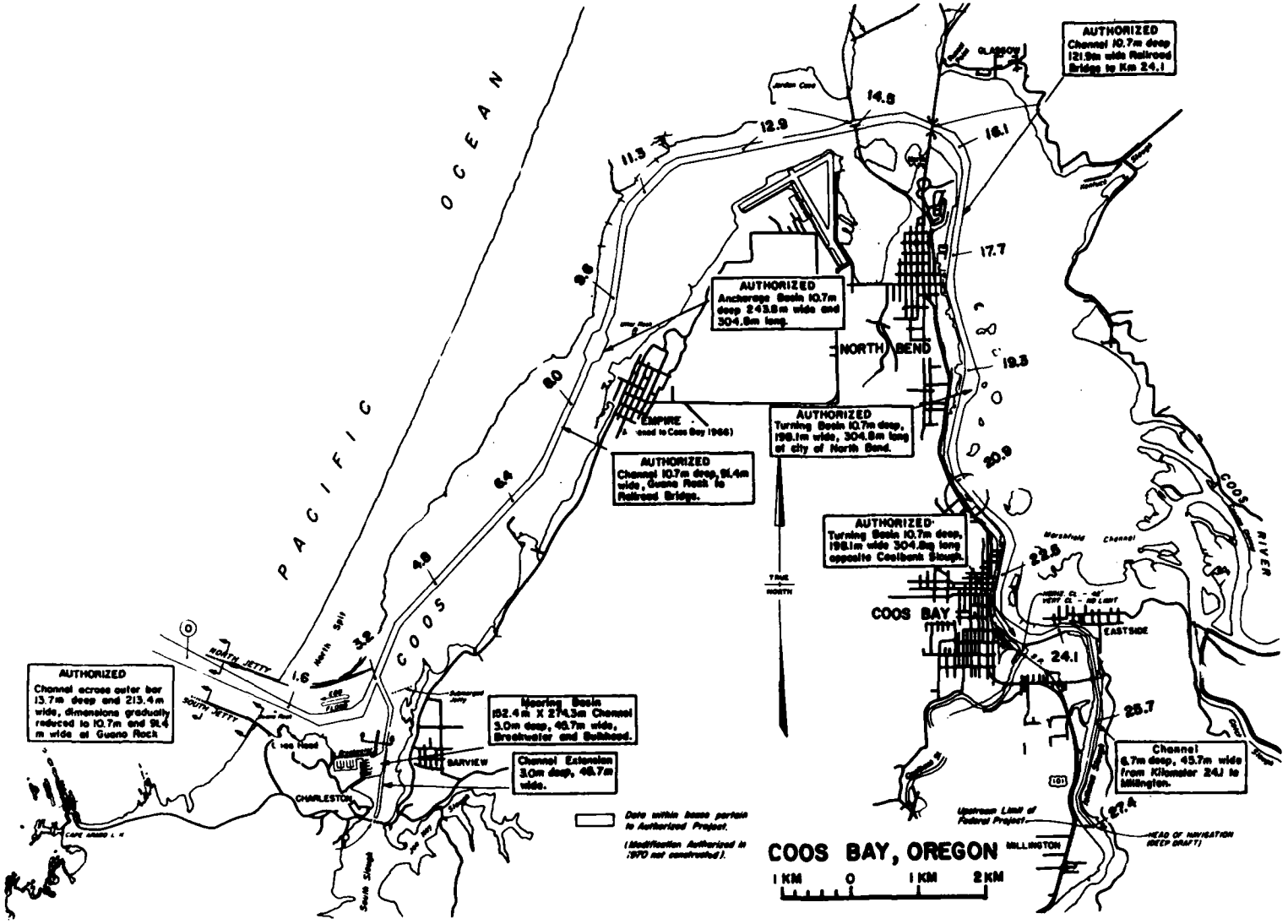


Figure 3. Federally authorized dredging project for Coos Bay, Oregon

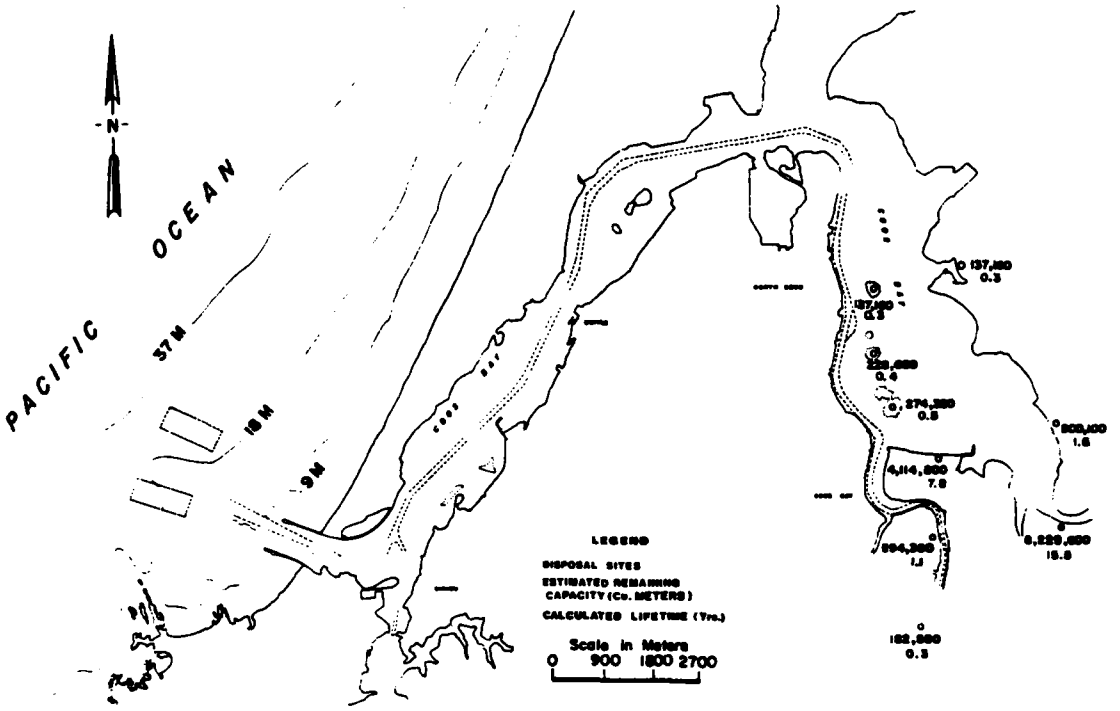


Figure 4. Capacity of and remaining years for all currently approved upland disposal sites at Coos Bay

sites. These predictions were verified in 1980 by the Coos Bay Estuary Management Plan (1) which reported that material must be disposed of in the ocean if the upper bay channel is to be kept open.

This scarcity of economically and environmentally acceptable upland and in-bay sites, and criteria established in Sections 102 and 103 of the Marine Protection, Research, and Sanctuaries Act of 1972 (MPRSA), required evaluations prior to disposal in the ocean. Portland District initiated special studies starting in 1979 to assess the feasibility of utilizing ocean disposal of materials from the upper bay. Criteria established in MPRSA required an evaluation of the material and comparison with the area of the ocean to be used for its disposal because of the high organic levels found in sediments between river km 19.3 and 24.14 due to the industrial and silvaculture inputs to these materials.

PLAN OF STUDY

Objectives

The Coos Bay studies were planned as a 3-year multiphase effort with five major objectives:

1. Determine the best possible location(s) for a permanent ocean disposal site(s) to receive marine sand and/or slightly contaminated fine-grained dredged material.

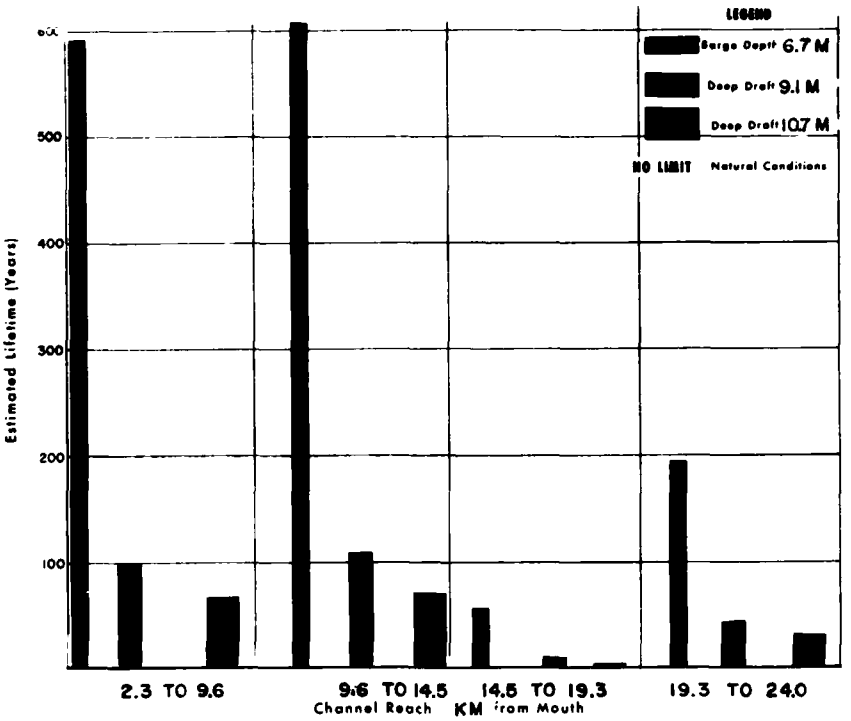


Figure 5. Life expectancy of remaining upland and in-bay disposal sites at reduced dredging depths

2. Determine the suitability of ocean disposal of these materials, as required by MPRSA and ensuing regulations (40 CFR 220-225, 227-229).

3. Determine the fate of dredged material deposited in the established disposal site.

4. Establish regional testing procedures and verify existing testing and modeling procedures required by MPRSA.

5. Utilize the long-term comprehensive studies at Coos Bay for possible determination of probable impacts at other Portland District ocean disposal sites.

These five objectives will be discussed in the following paragraphs.

Objective 1. The interim ocean disposal sites for Coos Bay, designated by the U.S. Environmental Protection Agency in the 1977 Combined Federal Regulations (40 CFR 220-229), were selected utilizing the best engineering judgment and economic information available at the time of selection. Limited environmental information was available. The study plan called for the offshore area, including the existing disposal sites, surrounding the entrance channel to be evaluated for biological, chemical, and hydraulic characteristics, as required in 40 CFR, parts 228.5 and 228.6. These studies would be used to select the best possible location for an ocean dredged material disposal site.

Objective 2. The site or sites located in objective 1 may or may not be suitable for all types of dredged material located in Coos Bay. Objective 2 was to determine the chemical characteristics of channel sediments in Coos Bay and the consequences of at-sea disposal by performing bioassay and bioaccumulation tests.

Objective 3. Dredged material disposed of in the ocean off of Oregon is subjected to extremely high current and wave regimes, and will erode from the deposition area. Objective 3 was established to determine the rate and direction the dredged material would move after deposition. Knowing this, the materials can be placed so that they ultimately are deposited on sediments with similar chemical characteristics.

Objective 4. Both chemical and biological testing requirements of MPRSA and its subsequent legislation required verification and modifications for the Pacific Northwest coast of the United States. Objective 4 was established to adapt the suggested tests for suitability of materials for ocean disposal to Portland District's environmental conditions and regulatory climate.

Objective 5. The fifth objective was established in the hope that additional long-term evaluations and tests could be avoided at other Portland District sites by utilizing the results of the Coos Bay work to reduce the length, costs, and number of tests.

Phases of Study

The five objectives were planned to be accomplished over a five-phase plan of study shown below:

Phase I: Plan formulation and pilot surveys

Phase II: Baseline study

Phase III: Disposal operations monitoring

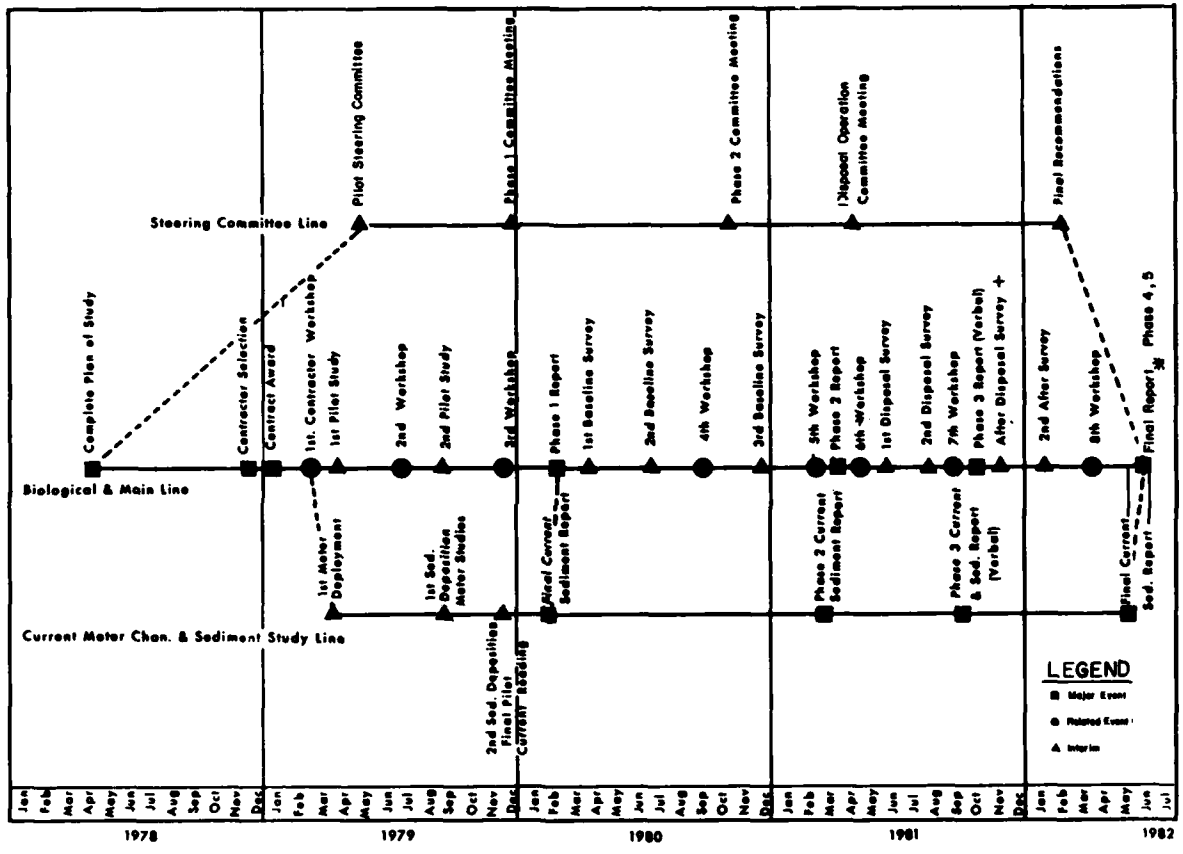
Phase IV: Postdisposal monitoring

Phase V: Analysis and report preparation

The study plan was designed so that each of the first three phases could result in a usable report should the study be cancelled. Figure 6 indicates the 3-year, five-phase plan of study. This plan was interrupted by an 8-month loss of funds, which caused a 2-year slip in the plan due to the seasonality of field sampling requirements. This figure also reflects the scheduling of a steering committee made up of interested Federal and State agencies. The committee was established to review the progress of the scientific work and make recommendations as necessary.

Contractural Arrangements

The five-phase, multiobjective study plan was presented to interested universities and research groups, with requests for proposals. Oregon State University's (OSU) Civil Engineering Department submitted the successful proposal and was awarded the 3-year, multiphase contract.



* - Not finished until July 1984 due to temporary cancellation of funds.
\+ - Delay of 8 months in receiving continuation of funding resulted in a 1-year delay in initiating phase 4.

Figure 6. Plan of study for the Coos Bay offshore disposal site evaluation studies

PURPOSE, SCOPE, AND RESULTS

Phase I

Purpose and Scope

Phase I studies were designed to examine the physical, chemical, and biological characteristics of channel sediments and the general offshore area near the entrance to Coos Bay. The studies were designed to provide data to select candidate ocean disposal sites for detailed examination during subsequent phases of the project. The objectives of Phase I were to:

1. Define the spatial distribution of benthic and demersal floral and faunal assemblages of nearshore Coos Bay area.

2. Define the spatial distribution of sediments in the study area with respect to grain size and volatile solids (organic content).

3. Define the general hydraulic regime of the study area.

4. Define the bathymetry of the study area.

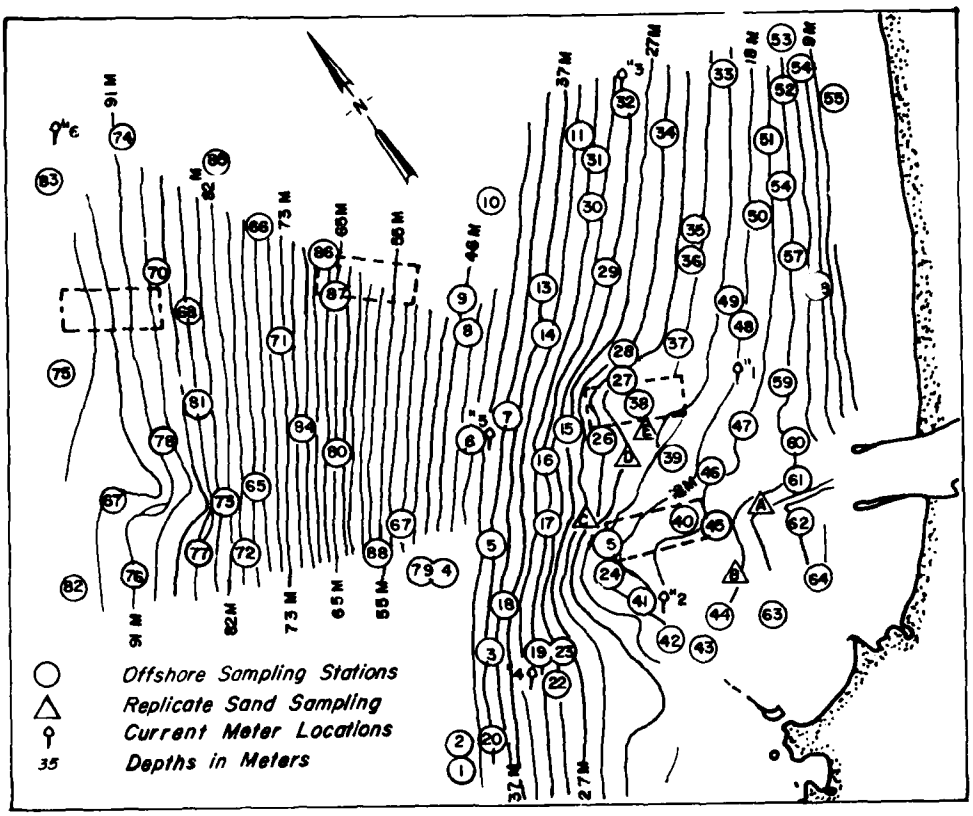
5. Define the physical and chemical makeup of channel sediments that will be dredged for offshore disposal.

In keeping with the desired management strategy, these objectives would minimize the differences between characteristics of channel sediments and the disposal site, avoid impacts on unique biological communities, and minimize the probability of onshore transport of deposited dredged material. Figures 7a and 7b present the sample locations for all offshore and channel sediment stations. Sediment samples were analyzed for sediment texture, organic levels, cohesive properties, bulk chemical and elutriate analyses for heavy metals, chlorinated organic compounds, and other contaminants of concern. Offshore currents were sampled seasonally with acoustic-recording current meters to determine vector average currents and instantaneous local velocities.

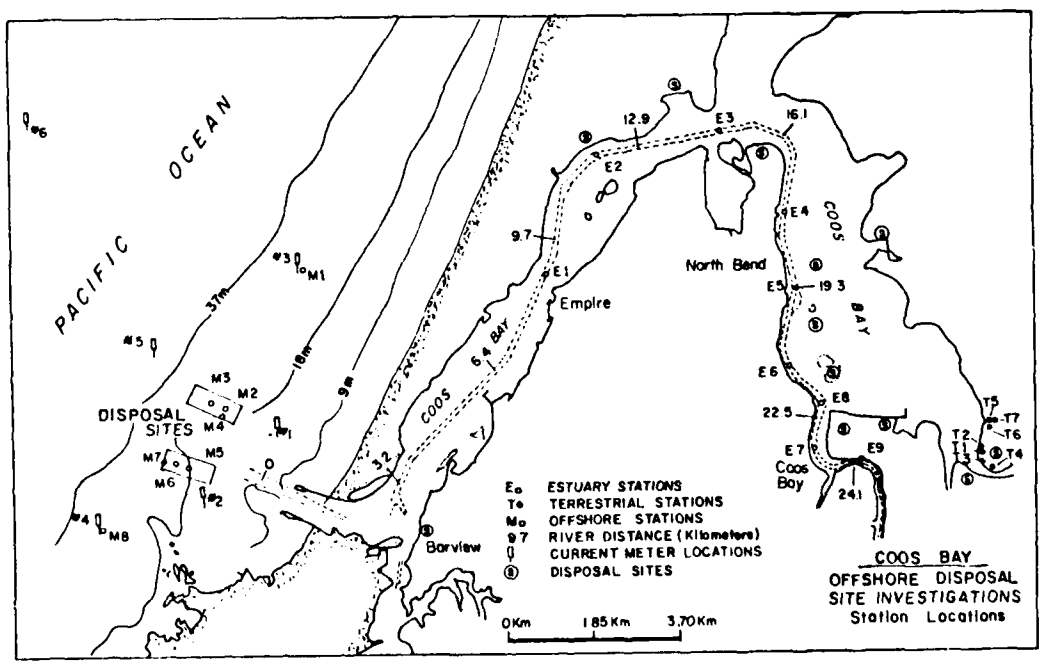
Benthic faunal assemblages were determined by using a stratified random sampling program with a one-tenth square-meter box corer (6). Species composition, relative abundance, and density of organisms were determined in the study area. Figure 8 shows locations of all benthic stations occupied during phase 1. Otter trawls for demersal fauna were carried out at 20-m depth intervals and analyzed for species composition and catch per unit effort.

Results

Study results from Phase I, reported in Coos Bay Offshore Disposal Site Investigation Interim Report, Phase I (2), demonstrated the offshore Coos Bay area is hydraulically very active. Mean currents and wave induced seafloor velocities frequently occur in the range of 30 to 60 cm/sec throughout the year. Average currents tend to follow the bottom contours and are southerly from April through November, reversing to northerly during the winter months. These directional trends are consistent with those previously observed for the Davidson and California currents. Sediments are clean and coarse nearshore



a. Offshore sediment stations

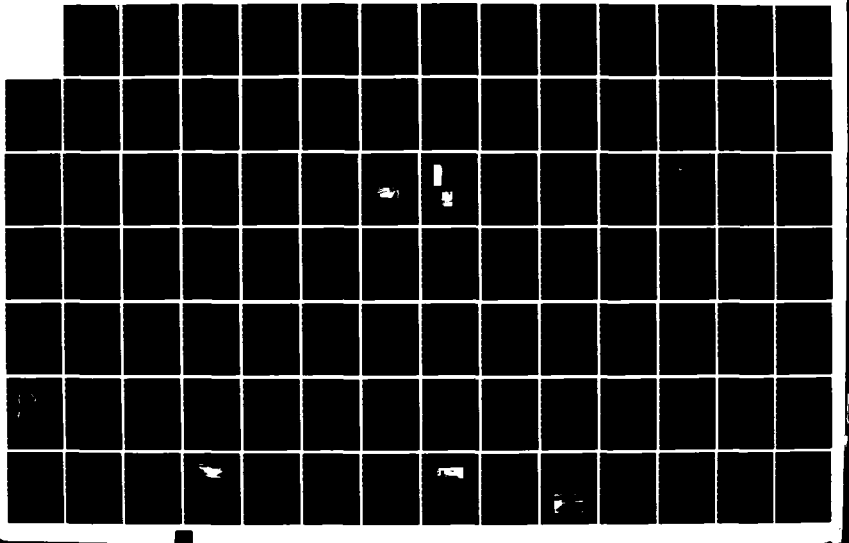


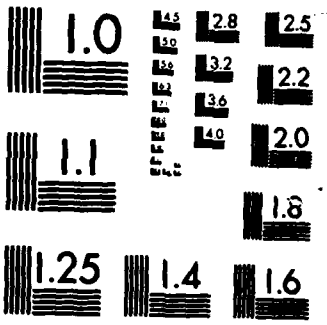
b. Channel sediment stations

Figure 7. Sample locations

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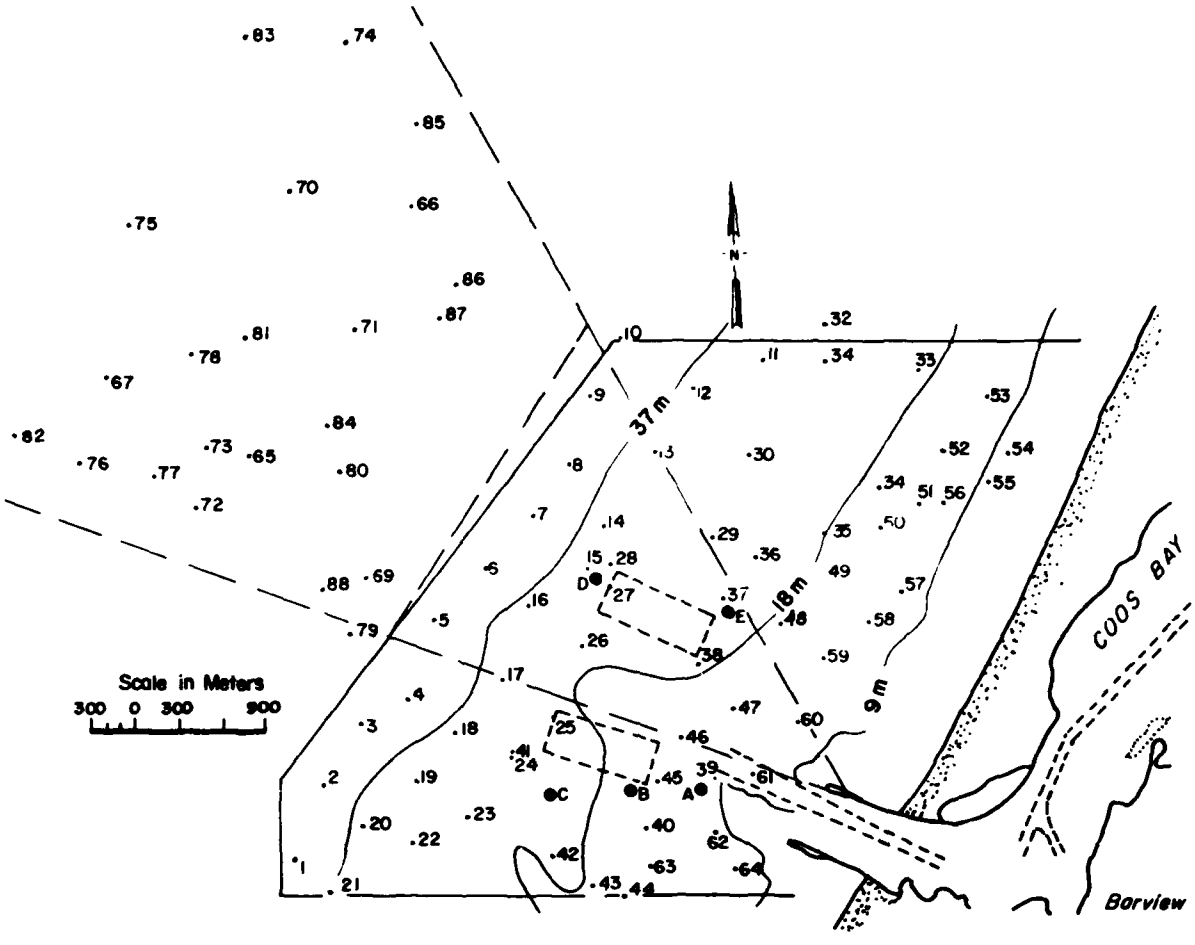


Figure 8. Benthic sampling station locations for first phase of Coos Bay offshore disposal site evaluation studies

(0.5 percent volatile solids, 0.25 mm median diameter) and become more organic and finer with increasing depths offshore (3 percent volatile solids, 0.08 mm median diameter at 90 m). The chemical characteristics of the sediment including total solids, total organics, iron, and oil and grease were found to increase with decreasing grain size offshore (2). The nearshore benthic infaunal community in the depth range of 10 to 45 m was found to be composed of many species typical of a high energy sandy bottom. The offshore region beyond 45 m exhibited even greater species richness and diversity than the inshore area. The shallowest areas offshore tended to have faunal and sediment affinities with the nearshore study area, while the deeper regions tended to be more typical of a marine soft-bottom community.

At all sampled depths in the general offshore Coos Bay area, sediments were found to be less organic and more coarse than dredged material evaluated from Isthmus Slough (2) (less than 10 percent volatile solids, 0.04 mm median diameter) (see Figure 9). Sediment compatibility between the dredged material and in situ seafloor sediments cannot be achieved at any offshore location within the general study area at an economically justified cost. In order to

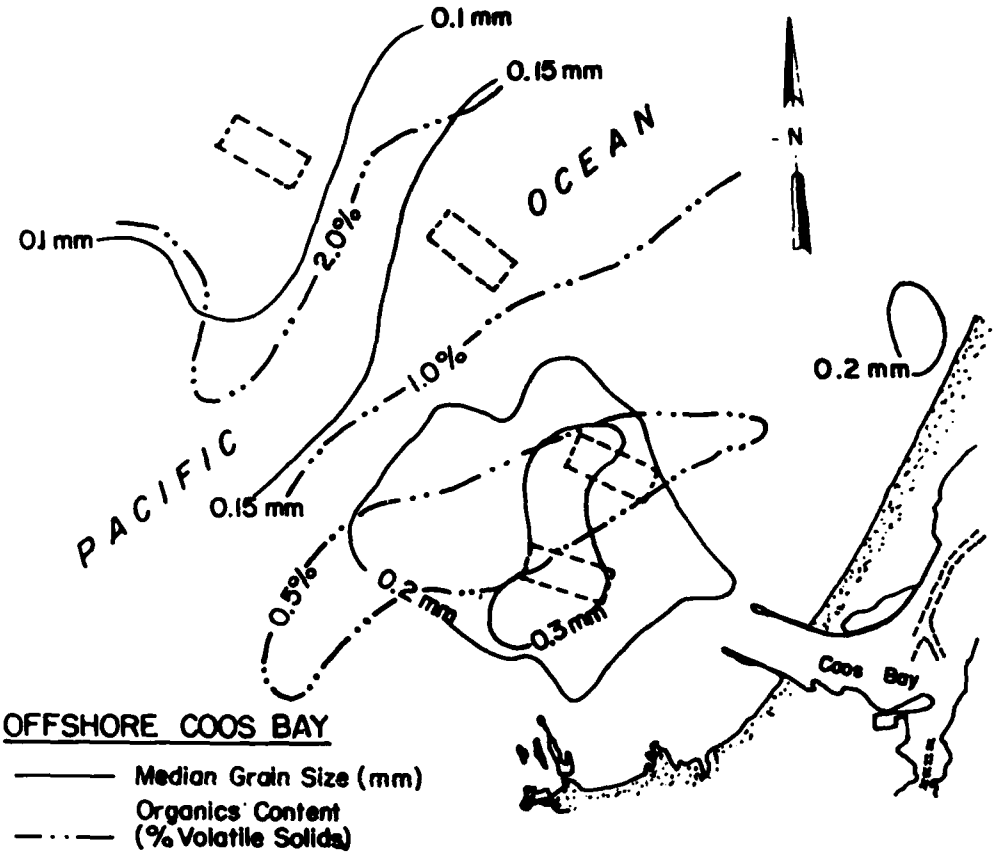


Figure 9. Distribution of sediments by median grain size and percent volatile solids

minimize differences between dredged material from Isthmus Slough and in situ material offshore, and to minimize onshore transport of resuspended materials, the physical-chemical studies indicated the 90-m contour was preferred for dredged material disposal. Biological study results identified a region between 45 m and 65 m where lower abundances appeared in both infauna and epifauna, even though diversity values remained quite high. This appeared to be an area of seasonal transition in both faunal and sediment characteristics. It was reasoned that biological tolerance to varied sediment conditions may be quite high in this region of great temporal change. Consequently, both the 90-m and 60-m contours were established as candidate disposal sites G and H (Figure 10). In addition to these two sites, the north interim disposal area, site F in Figure 10, was selected by Portland District to be included in Phase II, predisposal baseline studies. This was to determine suitability of this site as a sand dredged material disposal area.

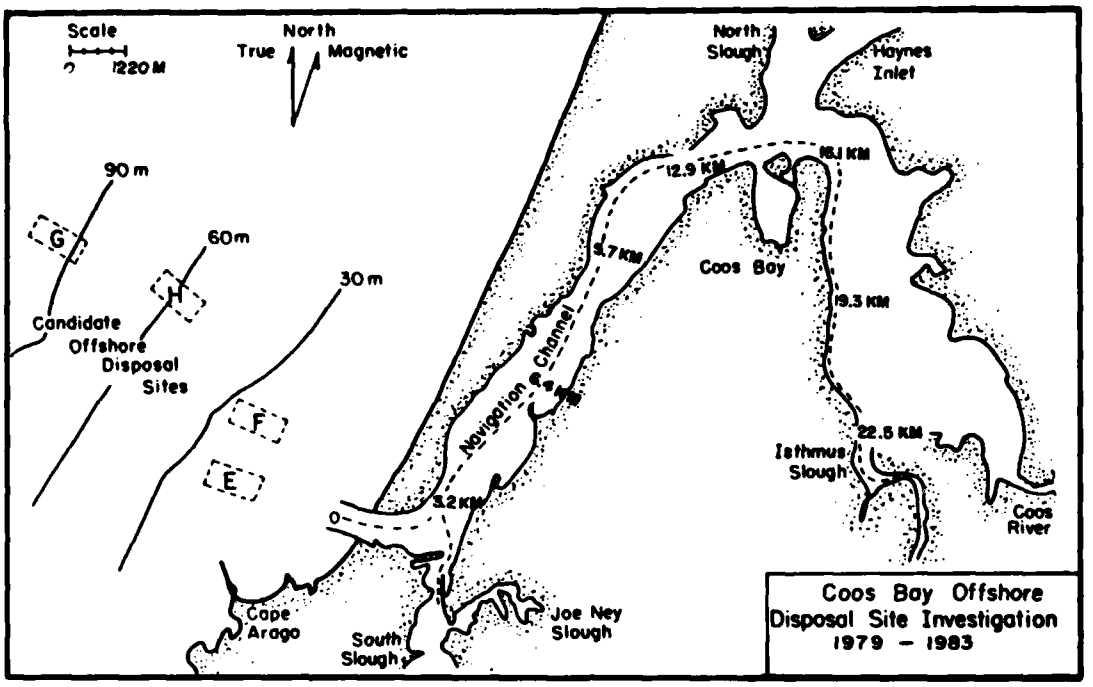


Figure 10. Locations of candidate disposal sites for consideration during Phase II Coos Bay offshore disposal site evaluations

Phase II

Purpose and Scope

Phase II baseline studies provided three seasonal (summer, spring, and fall) physical, chemical, and biological data collections at the three candidate disposal sites between May 1980 and May 1981. Bioassay/bioaccumulation studies evaluated the toxicity of Isthmus Slough sediments to representative marine organisms (9). The results of these studies aided in the selection of a test disposal site to monitor during the Phase III dredged material disposal operations. The test site was selected based upon sediment compatibility, avoidance of impacts on significant biological communities, and minimization of onshore transport of dredged material.

Physical studies were conducted to examine sediment characteristics at the three candidate disposal sites and to evaluate materials during and after disposal operations. Sediment characteristics evaluated in this study included sediment texture and organic levels. Offshore currents were sampled seasonally with digital recording acoustic current meters to determine vector average currents and local instantaneous velocities. Current and sediment texture data were combined to assess sediment erosion capacity.

The Koh-Chang instantaneous injection disposal model (3) was made operational by OSU for the Pacific Northwest coast conditions. The model was operated for conditions which approximate disposal of Isthmus Slough sediments at each of the candidate disposal sites. Estimates of plume concentrations and material accumulation at the three candidate disposal sites were made.

Chemical studies were conducted to establish sediment chemical characteristics and contamination levels in both the Coos Bay navigation channel and the proposed offshore disposal areas. The studies included four sampling times from March 1980 through April 1981. The sampling times were selected to cover approximately a year's duration to identify possible temporal fluctuations in sediment and water column parameters. Total sediment chemical constituency was determined for core samples obtained at five locations for each disposal site.

Figure 11 presents sample locations at the candidate disposal sites for chemical, biological, and sediment texture analyses. Sediment and water column chemical parameters were selected to provide characterization of both major components used in classifying the sample and the minor components useful in identifying sample contamination from pollution sources.

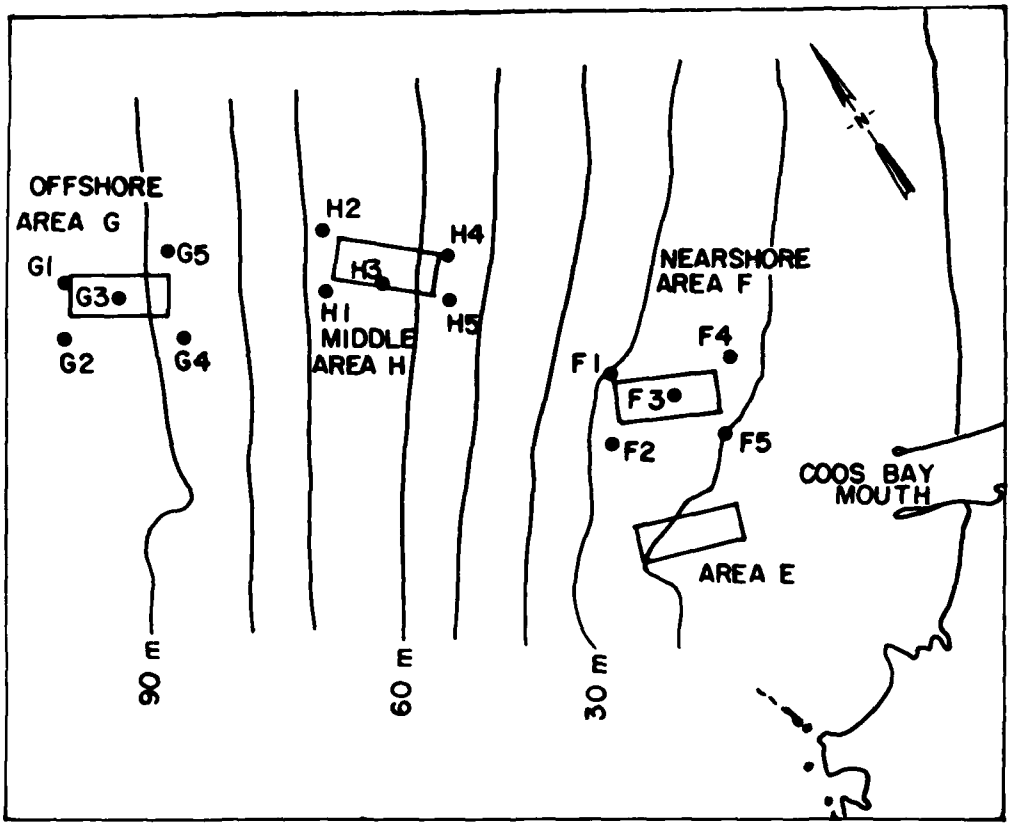


Figure 11. Sample station locations for candidate disposal site chemical, biological, and physical studies

Bioassay and bioaccumulation tests were conducted for sediments from the Isthmus Slough navigation channel. Both solid phase and suspended phase tests were performed using appropriately sensitive organisms indigenous to the offshore disposal site environment.

Elutriate tests on Isthmus Slough sediments indicated very little release of toxic contaminants such as heavy metals or chlorinated organic compounds into the solution phase (2). Thus, only solid and suspended phase toxicity tests were performed. Specific organism selection for testing was based on Phase I biological studies. Organisms were selected that could be obtained in sufficient numbers that represented different feeding types, and that were thought to be appropriately sensitive to exposure from contaminated sediments. Suspended phase tests used two pelecypods (*Macoma inclusa* and *Acila castrensis*), a crustacean (*Cancer magister*), a sand dollar (*Dendraster excentricus*), and a polychaete (*Abarenicola pacifica*). Solid phase tests used *Macoma inclusa*, the crustacean *Crangon nigrica*; the sand dab *Citharicthys sordidus*, the polychaete *Abarenicola pacifica*, and the amphipod *Rhepoxynius epistomus*.

Algal toxicity and bioaccumulation of trace metals and chlorinated hydrocarbons by a polychaete worm were also assessed in separate tests. The algae bioassay utilized a specific marine phytoplankton organism, *Dunialiella tertiofecta*, and was intended to assess the toxicity of soluble phase constituents released by elutriation. To assess compounds from the sediments, a deposit-feeding organism (*Abarenicola pacifica*) was selected that was known to bioaccumulate sediment-bound contaminants from previous studies (5).

Biological studies were designed to quantitatively determine the differences in the benthic fauna between the three candidate disposal sites and within each candidate disposal site, and to assess the seasonal differences exhibited over 1 year at a single site selected for experimental test disposal of dredged material.

To determine the infaunal similarities, five permanent stations were established within each candidate site. Each of the three candidate disposal sites was then sampled with a box-corer and a 1-m beam trawl during May 1980. To increase the sampling precision, five replicate box-corer samples were collected from each of the five permanent stations at each site, and two replicate trawls were made at each site. The abundance and distribution of the infaunal species were statistically compared using multivariate statistical techniques to determine the between-station differences of benthic infauna at each site. The three candidate sites were then compared against each other on the basis of species composition and abundance.

Results

The Phase II and III Coos Bay Offshore Disposal Site Investigation Interim Report (4) selected site H as the optimum location for dredged material disposal monitoring studies in Phase III.

Site G was shown to minimize the difference between in situ and dredged materials. However, an emerging scallop fishery was discovered in the vicinity of this site. Site G was eliminated from further consideration to avoid interference with a commercial fishery.

Site F was proved to be hydraulically quite active. Any material deposited at this site would be eroded very quickly. Prevailing wave and current directions would likely transport eroded sediments to the estuary mouth and to

adjacent beaches. As a result, site F was eliminated from further consideration to avoid impacting the shoreline and estuarine environment. However, it was retained for consideration as a disposal site to receive sand dredged material that would act as beach nourishment and/or a littoral transport site.

The biota at site H continued to exhibit desirable stress tolerance characteristics during the Phase II studies. The sediments at site H are approximately an order of magnitude more coarse and less organic than the dredged material from Isthmus Slough. This dissimilarity is not desirable in terms of sediment compatibility but did facilitate tracing the ultimate fate of the submerged disposal mound during Phase III. Prevailing currents at site H are parallel to the shoreline; no significant onshore transport of eroded dredged material would be anticipated. Also, bioassay and bioaccumulation study results suggest that no adverse impacts on the biota would be anticipated at site H. Based on biota stress tolerance, dredged material tracing ease, longshore transport direction, and lack of fisheries impact, site H was selected for Phase III disposal operation monitoring studies and the Phase IV postdisposal impacts monitoring studies.

Phases III, IV, V

Purpose and Scope

The Phase III disposal monitoring investigations were field studies designed to evaluate the fate and impact of slightly contaminated sediments dredged from Isthmus Slough and deposits at offshore disposal site H in 60 m of water. Two hopper dredges, the YAQUINA and BIDDLE, were used to deposit approximately 45,111 cu m at site H in August 1981. Water column measurements were made to quantify currents; disposal plume location, geometry, and density; water quality; and epibenthic biota impacts. Sediments were sampled before and after dredged material disposal to determine the dimensions of the disposal mound, to quantify changes in sediment physical and chemical characteristics, and to assess impacts on benthic communities.

Phase IV postdisposal monitoring investigations included a laboratory analysis of Phase III data, as well as field studies to assess the persistence of the disposal mound and the related impacts on biological communities. Plume measurements from Phase III were interpreted to examine the utility of the Koh-Chang dredged material disposal model. The final report was prepared as part of the Phase V effort. A funding delay caused a 1-year delay in returning for postdisposal field monitoring and prevented determination of short changes at the candidate site.

Results

Sediment results summarized below are taken from Coos Bay Offshore Disposal Site Investigation Final Report (7).

Disposal site H was sampled 1 day after completion of disposal operations. The samples displayed characteristics of dredged material. Layering was apparent in many samples. Dredged material was identified as having median grain size near 0.05 mm, as opposed to in situ material that is greater than 0.1 mm, and volatile solid levels above 4 percent; in situ material is 2 percent or less. A dredged material disposal mound was readily identified

and determined to have an area of approximately 0.46 by 0.46 km. After 1 year, some dredged material could be distinguished in five samples; however, most of the material was found in lenses beneath the surface of native material. It was hypothesized by OSU that winter storms had transported native materials over the disposal mound, armoring it from further erosion. However, the funding delay kept this from being verified. After 19 months (and the limited amount of dredged material disposed of) the detected mounds were mixed beyond recognition with native sediments.

Current Measurements

Current meter studies indicated that both wave-induced seafloor velocities and mean currents were sufficient to erode native marine sands, as well as the deposited dredged material during a significant fraction of each season. Prevailing currents in the depth of water at site H would tend to transport eroding material parallel to the shoreline or offshore, causing the material to be dispersed without affecting beaches or estuary entrances. Figure 12 depicts fall and winter bottom currents and induced surface currents for current meter stations occupied during 1977 and 1980, and is intended as an example of average conditions encountered during the current meter studies.

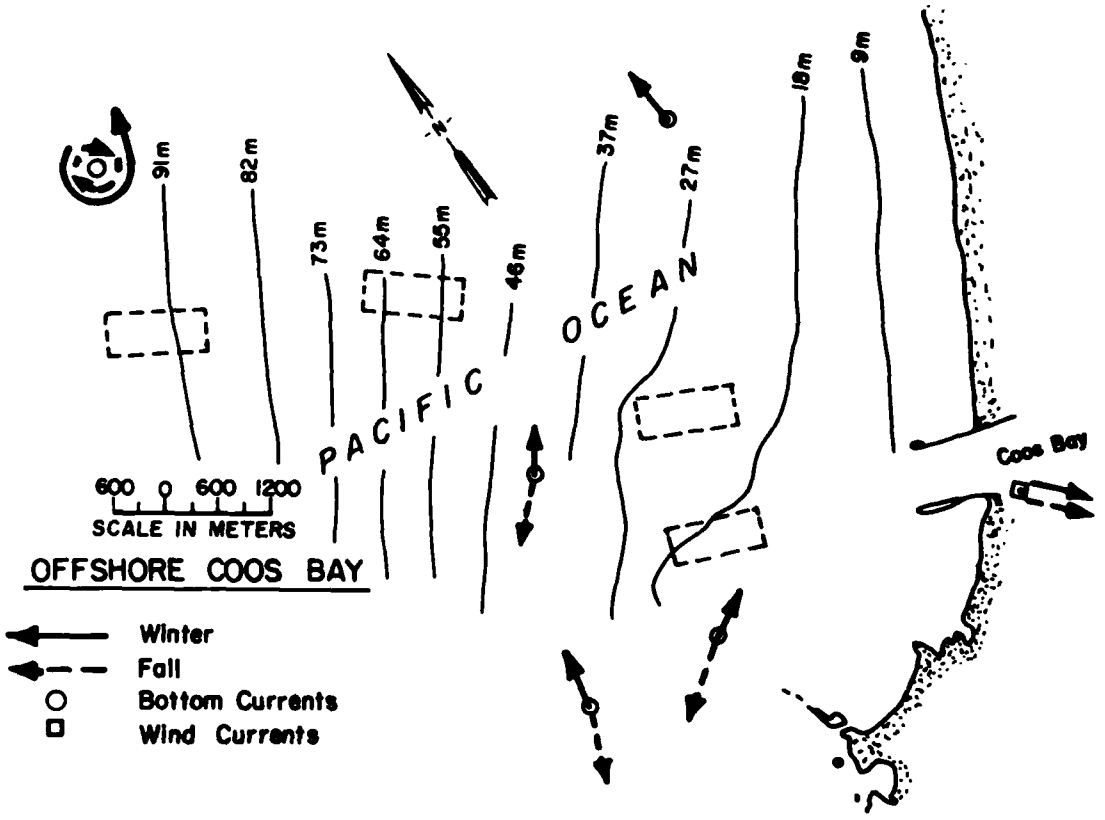


Figure 12. Average winter and fall bottom current directions and wind-induced surface wave direction

Model Studies

Field studies of the resultant dredged material solids plume were conducted. The primary cloud collapsed on the seafloor before the dredge vacated the disposal site. As a result, the properties could not be quantified during descent and collapse with surface-deployed equipment from the research vessel. However, the secondary cloud of suspended solids was monitored and found to persist for 1 hr. The Koh-Chang dredged material dispersion model, modified and calibrated for instantaneous discharge, proved satisfactory for predicting both the primary could accumulation on the seafloor and the secondary clouds if the silt clay size parameter in the models was reduced to a porosity of 96 percent.

Chemical Studies

Water column monitoring of the plume revealed no acute conditions during disposal operations. Ammonia-nitrogen, copper, and manganese showed elevated concentrations for a short duration. Sediment samples showed a return to background conditions after 1-1/2 years of sampling. Chemical dissimilarity between dredged material and native sediments was evident immediately after disposal operations. Concentrations, however, were lower than for sediments analyzed from the channel, indicating mixture with native sediments.

Biological Studies

Biological conditions were examined for each of the five stations at the study site by comparing differences in benthic infauna among stations and among seasons. Two levels of comparison were made, comparison of major taxonomic groups (annelids, molluscs, arthropods, echinoderms, and miscellaneous) and comparison of selected individual species.

Abundance and densities of benthic infauna within the major taxonomic groups varied widely. Variations in major taxonomic groups were viewed over time. Polychaete annelids and echinoderms showed significant variation among the five stations at the disposal site during all sampling periods. Crustaceans showed significant variation only between the period May 1981 and February 1983. Molluscs varied significantly during all periods.

When the major taxonomic groups were examined as to variation between stations, polychaetes showed significant variation, while molluscs varied significantly during some seasons and not others. The crustaceans were usually very consistent. Even with large variations observed between stations, each site appeared to have some species whose pattern of variation would be suitable for detecting changes.

Polychaetes were the most abundant group at the disposal site. Abundances exceeded 1200 per square meter both prior to and after disposal. Polychaete abundances continuously increased throughout the study with a maximum abundance exceeding 3300 sq m in February 1983. Crustaceans and molluscs were nearly equal in abundance during most sampling periods, with abundance levels only slightly reduced after disposal.

Principal component analysis and MANOVA programs were used to evaluate the polychaete, crustacean, and mollusc species data between stations and

between seasons. The species data showed two groups, those with little or no difference between the stations (i.e. depth, sediment, or volatile solids), and those showing differences.

Species within the major taxonomic groups exhibited three types of distributional patterns: species with peak abundances at the shallower water stations (60-m contour); those with peak abundances occurring at the deeper water stations (80-m contour), many of these species having intermediate abundance levels at the intermediate depth (70-m contour); and a third group of species that seemed to be rather evenly distributed across the five stations.

Using the same statistical procedures, the variability of each individual station over the period May 1980 to May 1981 was analyzed to determine if there were seasonal differences at each station. While some species varied significantly at an individual station, they did not vary widely at all stations. The individual species populations fluctuated throughout the study, with little consistent pattern and few species if any taxonomic group exhibited significant postdisposal changes. Several explanations for this are discussed later, but the large natural variations observed in the nearshore coastal zone off Oregon clearly demonstrated the need for evaluation at the species level, if naturally occurring perturbations are to be distinguished from man-induced changes.

The relationship between volatile solids and the abundance of major taxonomic groups was plotted for the faunal data for each of the five stations at the disposal site. Polychaete and mollusc abundances showed a positive correlation with increased volatile solids, with the highest abundance occurring between 2 and 4 percent. Crustacean abundance levels appeared relatively constant and showed no correlation with volatile solids. Temporal patterns displayed by the fauna at the site appeared to be characteristic of the fauna in a moderate to high energy environment. The data set identified the differences which occurred both between stations and sampling periods. The large variations observed in data at site H suggested that these organisms had a past evolutionary history of frequent disturbance, and were well adapted to environmental disturbance levels of the magnitude encountered in dredged material disposal.

Impacts from the test disposal operations were probably reduced, however, because of the small quantity of material (45,111 cu m) disposed of and the fact that the material was not deposited directly over established benthic stations.

CONCLUSIONS

The multiple-year evaluation study conducted by Oregon State University consisting of general survey, baseline, disposal monitoring, and postdisposal monitoring studies concluded that the general area around the 64-m contour was a transitional region for both the benthic community and the sediments. While this area had sediments coarser and less organic than Isthmus Slough sediments, it was the closest approximation to the dredged sediments in the entire economically feasible study area. It was reasoned that this area could tolerate the stresses of exposure to dissimilar sediments, as the biota

exhibited characteristics between those of the dynamic nearshore and static offshore environments. Current and wave studies from this area also assured researchers that no onshore transport of clay/silt-sized material would occur.

The above conclusions, plus the fact that no effects on the biota were shown by the bioassay and bioaccumulation studies, resulted in OSU suggesting the general area around the 64-m contour as the most likely region to establish an ocean dredged material disposal site for fine-grained materials.

SITE DESIGNATION

Site Evaluation

Portland District accepted OSU's report in June 1984. The studies had fulfilled all originally outlined objectives and had collected all information required in 40 CFR, 228.5 and 228.6. These regulations promulgate requirements of the Marine Protection, Research, and Sanctuaries Act, which requires basic environmental information to be obtained before designating a site. OSU's two interim reports and their final report were utilized by Portland District as the site evaluation studies required in the above cited CFR.

Site Designation

The evaluation studies were used to prepare a combination document formatted as an Environmental Impact Statement, which complied with the National Environmental Policy Act (NEPA), and also responded to the five general and eleven specific criteria for site designation in 40 CFR 228.5 and 228.6, respectively. The draft document (11) considered all areas studied by OSU as alternative sitings for a disposal site, a deeper continental slope site required in MPRSA, and no action required by NEPA.

The environmental impact statement considered three basic types of sediments to be disposed of in the ocean from the Coos Bay navigation channel:

Type 1. Predominantly clean sand of marine origin typical of sediments from below Coos Bay river km 19.3.

Type 2. Finer grained sand and silt containing some volatile solids typical of sediments from between Coos Bay river km 19.3 and 22.5.

Type 3. Highly organic fine material (6 to 20 percent volatile solids) typical of sediments from above Coos Bay river km 22.5.

Various disposal sites and combinations of these materials were proposed and a best alternative suggested.

The area studied by OSU as site H, centered on the 55-m contour, was shown after completion of the studies to be in the vicinity of a recently exhausted commercial scallop bed. The western edge of the site encroaches into the southern boundary of the bed. This site was adjusted to the 46-m contour which preserved the physical attributes on site H, but caused the biota to become more similar to the inshore sites.

Portland District proposed, as the preferred alternative, a combination of the two existing inshore sites (E and F), and adjusted site H (Figure 13). The adjusted site H could receive type 2 and 3 materials, up to 382,300 cu m per year, and sites E and F would receive 688,140 cu m per year of type 1 sediments. The placement of type 2 and 3 materials at adjusted site H would keep fine-grained material in the long shore current regime and off of beaches and estuaries. Placement of sand size and larger textured material at sites E and F would ensure that this material was available in the littoral transport system. Upon approval of the Environmental Impact Statement, these sites will be designated by the U.S. Environmental Protection Agency as final ocean dredged material disposal sites, available to receive approved materials tested under 40 CFR 227.

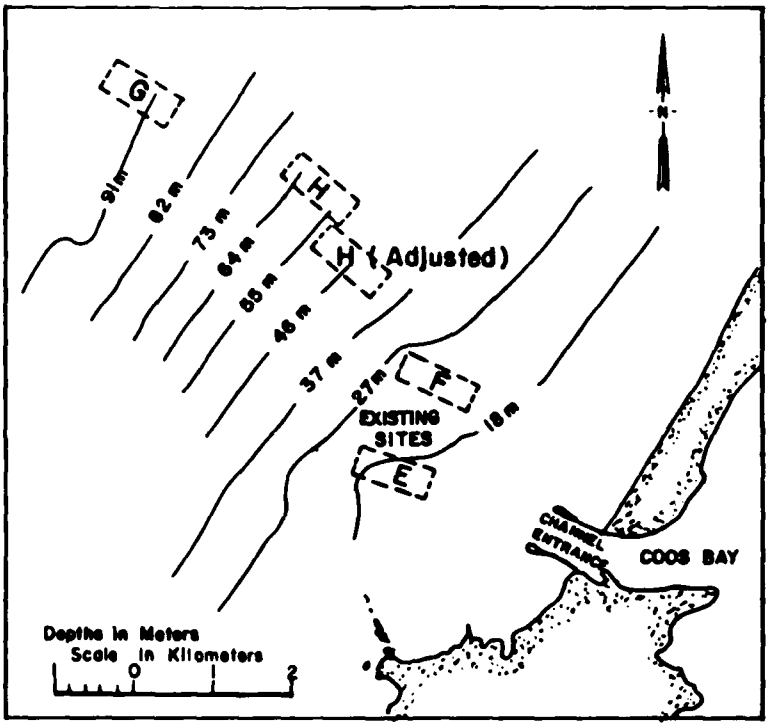


Figure 13. Proposed ocean dredged material disposal site locations

Site Management

A management plan is in the process of being prepared. The plan will include detailed descriptions of locations of types and volumes of materials to be placed at both remaining upland sites and at the ocean disposal sites, times of the year that various options should be used for disposal of materials, and what types of materials should be placed at each site. Alternative methods of dredging the materials and economics of each method will be examined and coupled with the appropriate disposal option. An important part of the plan will be a monitoring program established for the ocean disposal sites. The program would be designed to ensure that no dredged material is transported to, nor affects known ocean resources. The plan would be prepared

utilizing the evaluation studies. It is foreseen that this management scheme will ensure that dredged materials are placed in the most environmentally compatible and economically feasible disposal areas.

SUMMARY

The evaluation studies conducted for Coos Bay ocean dredged material disposal sites have ensured full compliance with all existing International, Federal, and State environmental regulations. More importantly, the long-term nature of studies and the full range of tests conducted and modified for the Pacific Northwest Coast conditions have allowed for shorter term and less expensive tests to be conducted at the remaining Portland District ocean disposal sites. Evaluation studies are under way for eight remaining Portland District ocean disposal sites with comparisons made with the Coos Bay data. This has allowed for the eight remaining sites to be evaluated for less cost and time than the cost for the Coos Bay studies alone.

These studies are using a new evaluation technique developed jointly by the Corps of Engineers and EPA (10) which requires using existing information to the extent possible. New data are only gathered when they are specifically missing from the literature for the area being considered for site designation. Even when data gaps are encountered, utilizing the Coos Bay data base allows for much shorter and less intense sampling efforts to obtain the missing data. This is done by determining that the smaller field sampling effort is similar to the longer term data base and by inference showing whether they behave similarly or differently. If a difference is inferred, additional field studies could be required prior to designation.

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WASTEWATER TREATMENT BY SOIL

T. Tanimoto

Director, Seacoast Division
River Bureau, Ministry of Construction

INTRODUCTION

Japan has enjoyed a remarkably rapid economic growth since 1950. Population concentration in the urban areas, economic growth, and industrial development have increased the amount of pollution all over the country, especially in the urban areas. There was no pollution problem so long as the pollution load remained far smaller than the environmentally permissible limit. However, water quality in public water bodies such as rivers and lakes deteriorated rapidly once the pollution load exceeded the permissible limit and this polluted water resulted in nutrient-rich sediments gathering at the bottom of lentic water bodies, such as lakes and marshes. It is very costly to remove these nutrient-rich sediments caused by the polluted water; therefore, it is cost-effective to purify polluted water before it reaches natural water bodies. It is especially important to remove nitrogen and phosphorus. To treat pollution load at low cost, the methods are: first, not to dissolve; second, if it is dissolved, keep dilution as low as possible; and third, treat the pollution at its source.

Though the activated sludge method is the most popular method in the world, there is a wastewater treatment technology newly developed in Japan that will be introduced in this paper. This is the most economical and energy saving method since it makes best use of natural characteristics, and treatment within individual households is possible if there is a small piece of land.

SELF-PURIFICATION AND SEDIMENT SUBSTANCES IN RIVERS

Natural self-purification will not be entirely expected once the pollution load exceeds the environmentally permissible limit. This theory will be applied to the water and air pollution. In the case of rivers, though self-purification is expected even in deteriorated rivers, it will hardly function if the river bottom is included.

Organic substances discharged into the water are transformed into simple inorganic substances in the form of H_2O, CO_2, NO_3, SO_4, PO_4, etc., through the oxidation function of various aerobic bacteria. The dissolved oxygen in the water will be decreased in volume due to oxidation. The more the water is polluted, the more oxygen is consumed; fish therefore cannot live in heavily polluted water. Some of the dissolved substances will be released to the air, some will be dissolved in the water, and others together with undissolved substances will settle to the bottom. As organic substances decompose, plants such as duckweed will take up CO_2, N, and P as nutrients. These nutrients will be released upon anaerobic decomposition. In this step, foul odor gas including hydrogen such as CH_4, NH_3, and H_2S will be produced. This cycle

will repeat until the sediments at the bottom of the river are moved downstream into the stagnated water body.

The nutrient sediments accumulated in the stagnated lakes and seas which have rivers in their catchments consist of these substances. Substances which normally undergo self-purification in rivers are causing many problems in stagnated water bodies due to overloading. It is far more economical and environmentally acceptable to treat the wastewater at the source rather than to treat the sediments.

WATER POLLUTION IN THE STAGNATED WATER BODY IN JAPAN

Lake Biwa, Lake Kasumigaura, Lake Suwa, and Seto Inland Sea (Figure 1) are representative of stagnated water bodies in which eutrophication is distinctly observed. For the lakes, the population in the drainage basin, chemical oxygen demand (COD) concentration, sewerage served population ratio, and number of piggeries are shown in Table 1. Table 2 shows the COD, total nitrogen (T-N), and total phosphorus (T-P) for the Seto Inland Sea. The frequency of red tide is illustrated in Figure 2. Pig raising is very prosperous around Lake Kasumigaura, and the COD concentration shows the highest rate in 1979 due to the increase in the pig raising since 1977. Water quality in Lake Biwa and Lake Suwa is gradually being improved after the worst situation in 1973. The water quality of the Seto Inland Sea is also being improved since 1974 and the frequency of red tide has improved since 1977.

Water pollution has improved as a result of the enactment of numerous laws on water pollution prevention in 1970. Through these laws, the central government, prefectural governments, and the people endeavored to mitigate the pollution load.

WATER POLLUTION PREVENTION ADMINISTRATION IN JAPAN

Basic Measures

The administration of water pollution prevention in Japan is being comprehensively carried out by the following four measures.

a. Setting up environment standards showing the target of various measures.

b. Effluent regulation according to effluent standards to achieve a.

c. Carrying out public works such as sewerage, dredging, and water conveyance for purification.

d. Public cooperation such as self-control of usage of detergent containing phosphorus compounds.

Effluent Control

In order to prevent water pollution, the Water Pollution Prevention Act was enacted in 1971. The objective of this act is to "prevent the water pollution in the public water bodies, hereby, to protect people's health and preserve the living environment through controlling the effluents which are

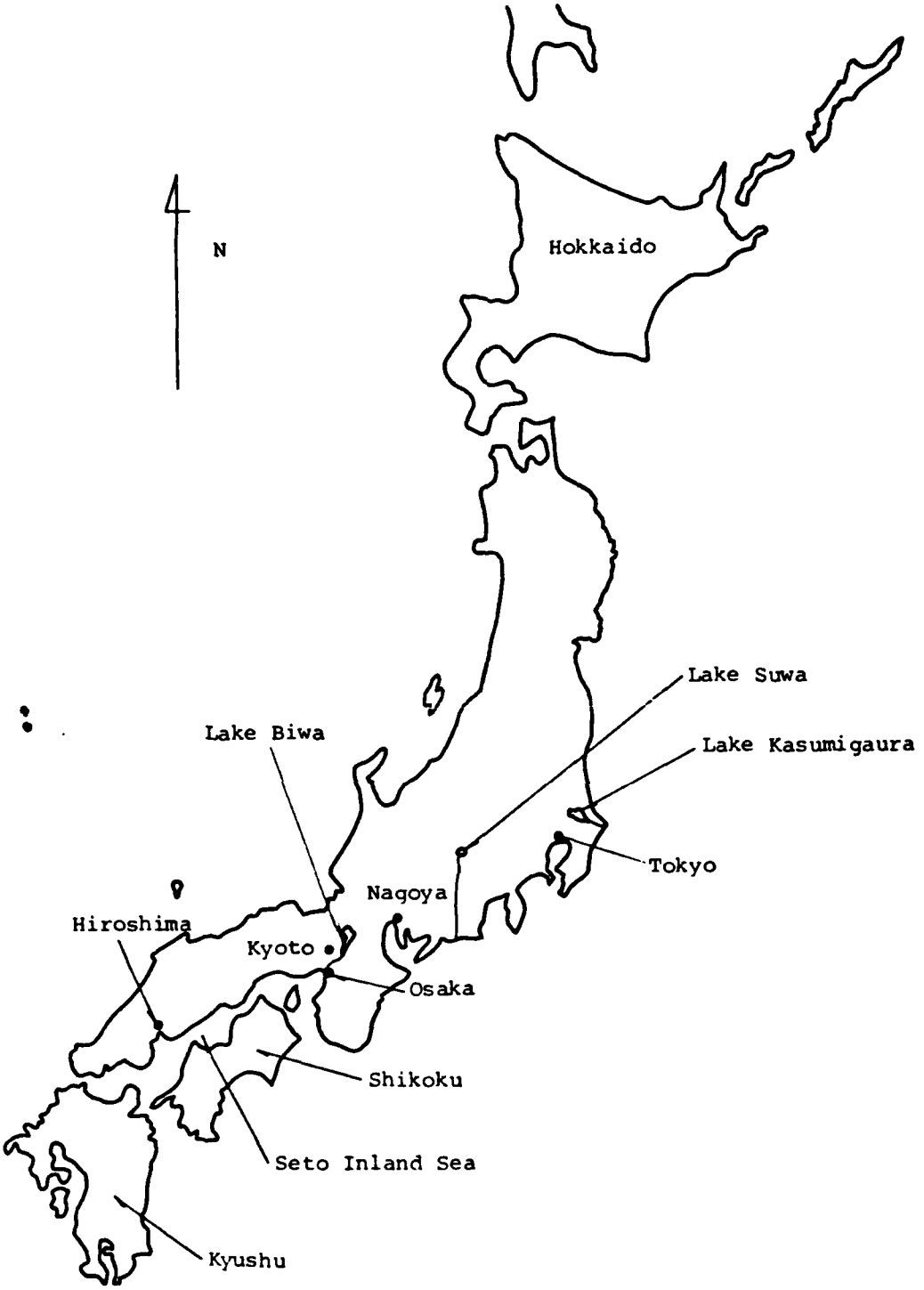


Figure 1. Location of representative stagnated water bodies

TABLE 1. COD, T-N, SEWERAGE SERVED POPULATION RATIO, AND NUMBER OF PIG RAISING ENTERPRISES

Item *	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983
Lake Biwa																
Population (10^3)	806	815	833	845	862	880	900	919	929	947	964	981	998	1,017	1,026	1,039
SSPR (%)	-	1.4	1.7	2.2	2.6	2.8	3.0	3.2	3.6	3.8	4.1	4.3	4.6	4.8	7.8	8.9
COD North (ppm)	0.8	0.8	0.7	0.9	2.7	2.4	2.2	1.7	1.4	2.0	2.2	2.3	2.2	2.2	2.1	2.0
COD South (ppm)	1.8	1.5	1.4	1.8	4.0	4.2	3.4	2.6	2.7	3.2	3.5	3.3	3.1	3.1	3.0	2.8
Lake Kasumigaura																
Population (10^3)	744	755	770	787	801	818	839	862	878	893	904	927	953	974	992	1,008
SSPR (%)									1.1	1.7	3.7	5.5	8.5	8.8	10.1	12.1
COD (ppm)	4.0	4.0	5.7	5.5	7.1	7.9	6.8	7.2	6.2	7.1	9.1	12.1	9.4	8.1	8.6	9.6
NPR (10^3)	304	278	273	260	251	296	303	307	267	301	-	385	344	-	336	
Lake Suwa																
Population (10^3)	188	190	191	193	195	196	197	198	198	200	201	202	203	204	205	205
SSPR (%)												7	17	24	37	36
COD (ppm)	-	3.7	-	-	9.2	13.0	5.7	6.3	6.9	9.2	10.0	5.9	6.4	5.0	4.9	5.5
NPR (10^3)	8	9	10	10	8	7	7	5	6	7	6	6	5	5	4	-

* SSPR: Sewerage served population ratio
NPR : Number of pig raising enterprises

TABLE 2. TRANSITION OF SEAWATER POLLUTION IN SETO INLAND SEA

Year	C O D (ppm)	Transparancy (m)	T-N (ppm)	T-P (ppm)
1982	1.8	5.0	-	-
1983	1.7	6.2	0.275	0.023
1984	2.1	6.5	0.292	0.035
1985	1.6	6.6	0.300	0.028
1986	1.6	6.9	0.398	0.030
1987	1.7	5.8	0.277	0.029
1988	1.5	6.6	0.243	0.023
1989	1.3	6.4	0.257	0.026
1990	1.5	5.7	0.265	0.026
1991	1.6	6.5	0.246	0.024

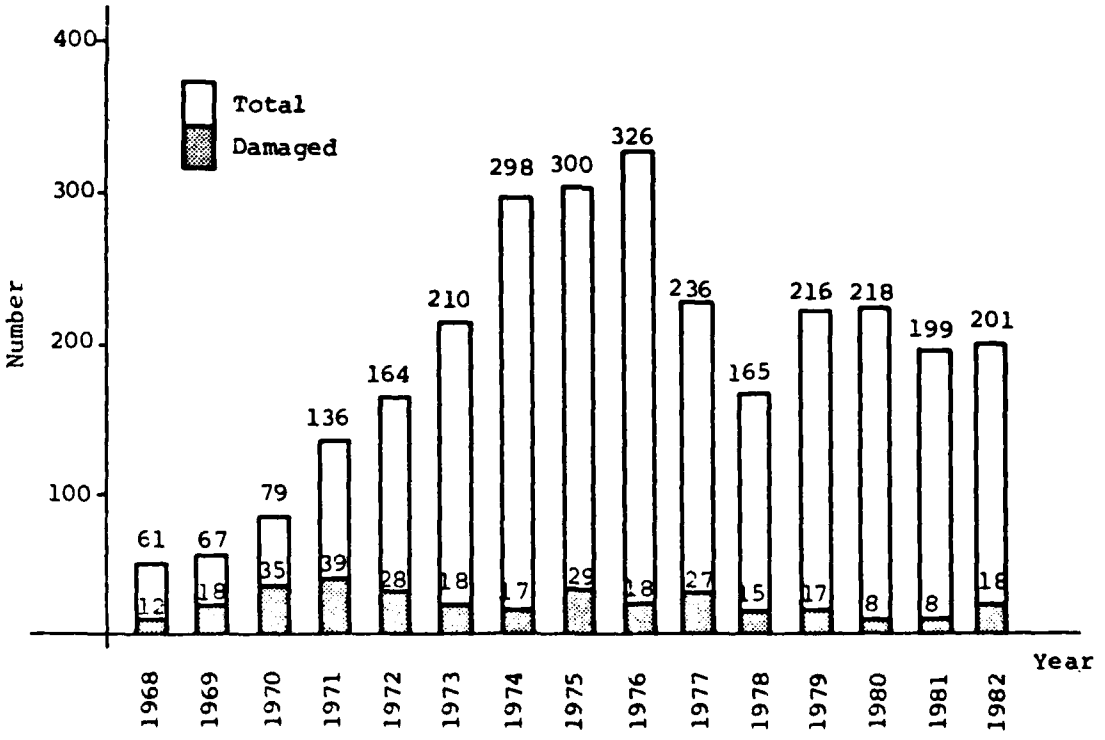


Figure 2. Frequency of red tide into Seto Inland Sea

discharged from factories and buildings." This act contains no clause to control effluent containing phosphorus compounds at present, but such a clause will soon be added by Cabinet Order.

The Special Measures Act on Environment Preservation for the Seto Inland Sea was enacted in 1973. The objective of this act is to "take special measures on control of setting up special facilities, prevention of damage occurrence due to the eutrophication, preservation of natural seacoast, etc., as well as to decide the necessary matters on planning preparation in environment preservation of the Seto Inland Sea in order to execute the effective measures for the preservation of the environment of the Seto Inland Sea." This act contains a clause giving official guidelines to control the effluents and substances containing phosphorus compounds.

On the prefectural level, Shiga and Ibaragi prefectural governments which govern the drainage of Lakes Biwa and Kasumigaura, respectively, prepared their own ordinances to mitigate eutrophication. It is widely recognized that the regulations through these ordinances were very effective.

In July 1984, the Special Measures Act on Water Quality preservation in Lakes and Marshes was enacted in order to take measures more comprehensively and drastically. The objective of this act is to "prepare for the planning on measures to be carried out to preserve the water quality in the lakes and marshes of which the environment standards of water pollution are urgently attained, and to take special measures to control the facilities which discharge effluents or substances which may cause the water pollution in large scale as well as to carry out the public works for the purification of water and also to stimulate the people's cooperation." This act is a recent result of the comprehensive water environment policy.

ADVANTAGES AND DISADVANTAGES OF VARIOUS WASTEWATER TREATMENT

The methods for wastewater treatment which have been used historically are:

a. Underground infiltration through pits.

b. Oxidation ponds.

c. Biofilms.

d. Trickling filters.

e. Activated sludge.

The advantages and disadvantages of these methods are discussed below. The removal ratio of BOD load and suspended solid (SS) mentioned below apply to wastewater from households, i.e. the BOD concentration is about 200 ppm.

Underground Infiltration Through Pits

Since clogging by the SS on the soil surface at the bottom of the pit is inevitable, only a small volume of wastewater should be allowed. The method can purify wastewater well with regard to its BOD, nitrogen, and phosphorus,

and the cost is zero, but this method has the fatal disadvantages of foul odor and sanitation problems.

Oxidation Ponds

This method removes SS and reduces BOD concentration to some extent and is easy to maintain. The problems with this method are that nitrogen and phosphorus are removed only marginally and that a large land area is required.

Biofilms

This method purifies wastewater by using aerobic bacteria which exist at the surface of gravel, etc., through repeating contact, release, and exposure to the air. This system can treat about 1 cu m of wastewater per day per 1 cu m of contact material, and can remove 80 to 90 percent of SS and 65 to 85 percent of BOD load. The defects are that the contact material will be clogged and that the treatment volume is not great in comparison with the trickling filter.

Trickling Filters

This method was developed for the purpose of mass treatment. The improvement is made using biofilm and sprinkling wastewater intermittently over the contact bed. This can treat about 2 cu m of wastewater per day per square meter, and can remove 75 to 90 percent of BOD load. The advantages are:

a. Flexible to the fluctuation of waste load.

b. Easy operation.

c. Less sludge production compared with activated sludge method.

d. Valid in nitrification.

The disadvantages are the occurrence of foul odor and contact flies, and that a large amount of land is required compared with the activated sludge method.

Activated Sludge

This method was also developed for mass treatment and can treat 10 to 20 cu m per day per square meter. The principle is to use the aerobic bacteria by blowing air into the wastewater and stirring. This system can remove about 90 percent of the BOD load, but is not so available with regard to the removal of nitrogen and phosphorus.

The advantages are that it is suitable for mass treatment, has less of a foul odor, no occurrence of flies, and a low cost compared with trickling filters. The disadvantages are that it is not appropriate for load fluctuations, produces a large amount of sludge, has less removal of nitrogen and phosphorus, has less nitrification, and has high maintenance cost.

PURIFICATION METHOD BY SOIL

It has been recognized that soil can purify wastewater to a great extent. Fairly long ago in Japan, the river water quality near rural communities was very good, even though the wastewater from the households infiltrated to the soil through pits. However, the river water quality rapidly deteriorated once the wastewater was piped directly to the river because soil purification was unable to solve the problems of clogging, sanitation, and foul odor.

Soil has many characteristics. These are as follows:

a. The number of bacteria in the soil is thousands of times the amount in water.

b. There are many aerobic bacteria in the surface layer and these bacteria can transform NH_4-N into NO_3-N by oxidation.

c. Anaerobic bacteria are in the deeper layers and these bacteria can transform NO_3-N into N_2 gas.

d. Soil itself has an almost infinite phosphorus absorption capacity.

e. There is no clogging if the underground is used because soil has capillary action.

f. Soil has a strong removal capacity of foul odor.

g. There are many microorganisms such as earthworms.

The newly developed wastewater purification method using soil simultaneously solved the problems of clogging, sanitation, and foul odor, which were not solved in the past by the pit method, though treating the wastewater beneath the ground succeeded as well in the removal of nitrogen and phosphorus which are very difficult by other means. The cross-sectional profile of the concept is illustrated in Figure 3, and the treatment system consists of four steps, settling tank, circulating contact aeration tank, final sedimentation tank through biofilm, and trench. The settling tank is expected to remove about 30 percent of the BOD load through stagnating wastewater for more than 16 hr. The circulating contact aeration tank is expected to remove about 30 percent of the BOD load of inflow water and is designed to cope with 0.3 kg of the BOD load per cubic meter. This method is intermediate between biofilm and activated sludge. The final sedimentation tank through biofilm is set up primarily to remove SS and improve the water quality in order to prevent clogging in the trench, which is the final step. The final trench is primarily to remove N, P, and BOD load. This facility can display a remarkably high capacity of phosphorus removal and may be said to be a super high-class treatment system. The final trench is omitted if the land is limited. In this case since the removal of phosphorus will be degraded and BOD concentration will be 40 to 50 ppm, it is recommended that an additional (second) circulating contact aeration tank be provided following the circulating contact aeration tank. The second circulating contact aeration tank would be designed to cope with 0.2 kg and 0.1 kg of BOD load per one cubic meter if the target of BOD concentration were 20 ppm and 10 ppm, respectively.

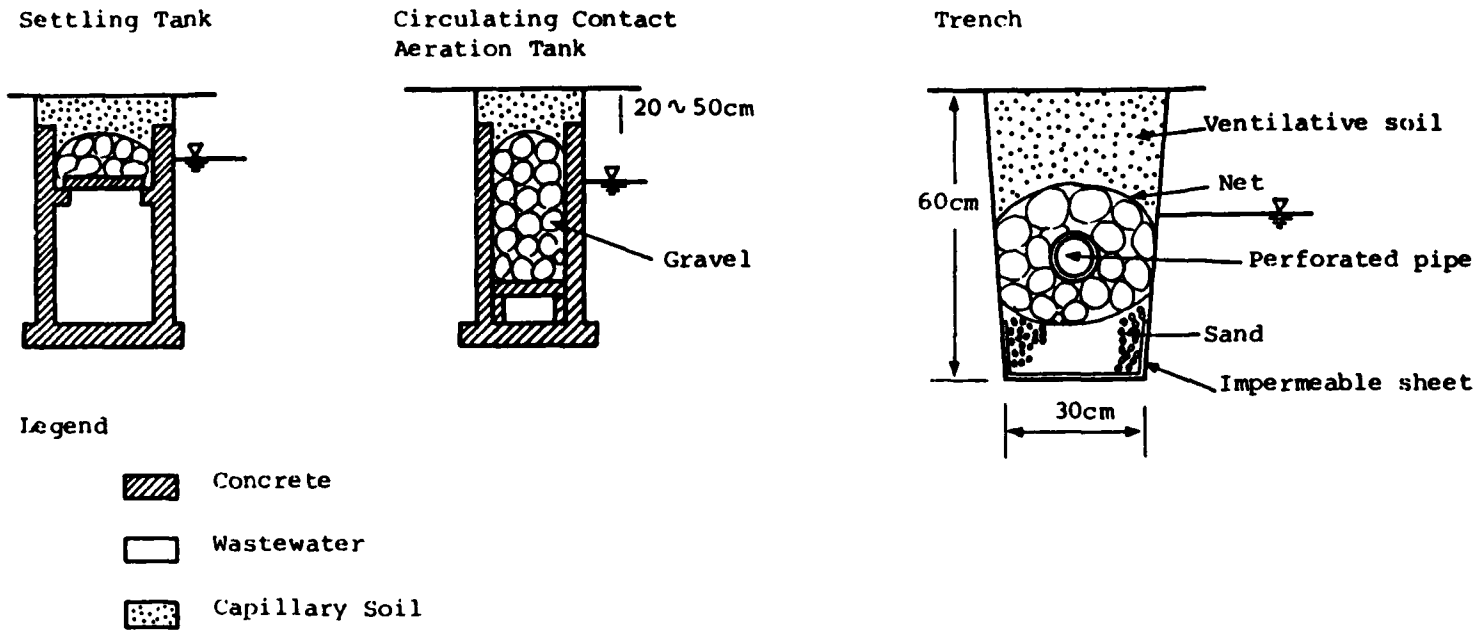


Figure 3. Cross-sectional profile of purification by soil

The most specific characteristics of this system are that the surface of all facilities is covered by ventilative soil; therefore, a very small amount of sludge will be produced due to the activities of microorganisms such as earthworms. The problems of clogging, sanitation, and foul odor are simultaneously solved, and N and P which are very difficult to remove by any other means are easily removed. The construction cost of this system is far more economical compared with public sewerage system when constructed in the garden of individual households; the standard cost per capita is approximate US$500, but pipeline cost will be additionally required when the wastewater is treated by community.

RESEARCHES IN JAPAN

There are more than 2,000 soil wastewater treatment facilities in Japan, most constructed in schools and hotels where the waste load fluctuation is large, and sewerage systems in the rural community where there is sufficient land space. This system was also adopted for the public sewerage system in Shizuoka Prefecture in 1983. In 1978, the Ministry of Construction (MOC) initiated experiments to purify wastewater from pig raising by soil in Iwate Prefecture, Tohoku District, from the aspect of eutrophication prevention, especially in dams. The Environment and Health Center, Okayama Prefecture, started the experiment on soil wastewater treatment in 1982. Both experiments will be briefly described.

Experiment by MOC

Iwate Prefecture is located 500 km north of Tokyo. The average temperature is 22.8°C in summer and -2.5°C in winter. The treatment facilities were designed to purify the wastewater from piggeries having 100 pigs, after the solids and liquids are separated and the liquid is aerated. The treatment system is the same as mentioned in the preceding section and is shown in Figure 4. The expected wastewater volume was 3 cu m/day. Cultivated soil near the test site was mixed with pearlite in order to obtain the ventilative soil. The physical characteristics of the soil are shown in Table 3. Four trenches 25 m long were prepared; three of the trenches were in constant use and the fourth was prepared for emergency use such as clogging. However, no clogging in the trench was observed. The trenches were 50 cm wide, but the design width to purify wastewater for each trench was 1.5 m. One square meter of trench was therefore charged with purifying 26.7 ℓ of wastewater per day. The operation began in January 1978 and the accumulated sludge was extracted three times:

a. January 1980 from settling tank and final sedimentation tank.

b. October 1982 from circulating contact aeration tank and sedimentation tank.

c. February 1984 from settling tank and final sedimentation tank.

The volume extracted in 1984 was 8 cu m and this volume was returned to the activated sludge tank for preliminary treatment.

The wastewater was sampled and the water quality analyzed monthly. Four years of data (April 1978 to March 1982) were used for this analysis because

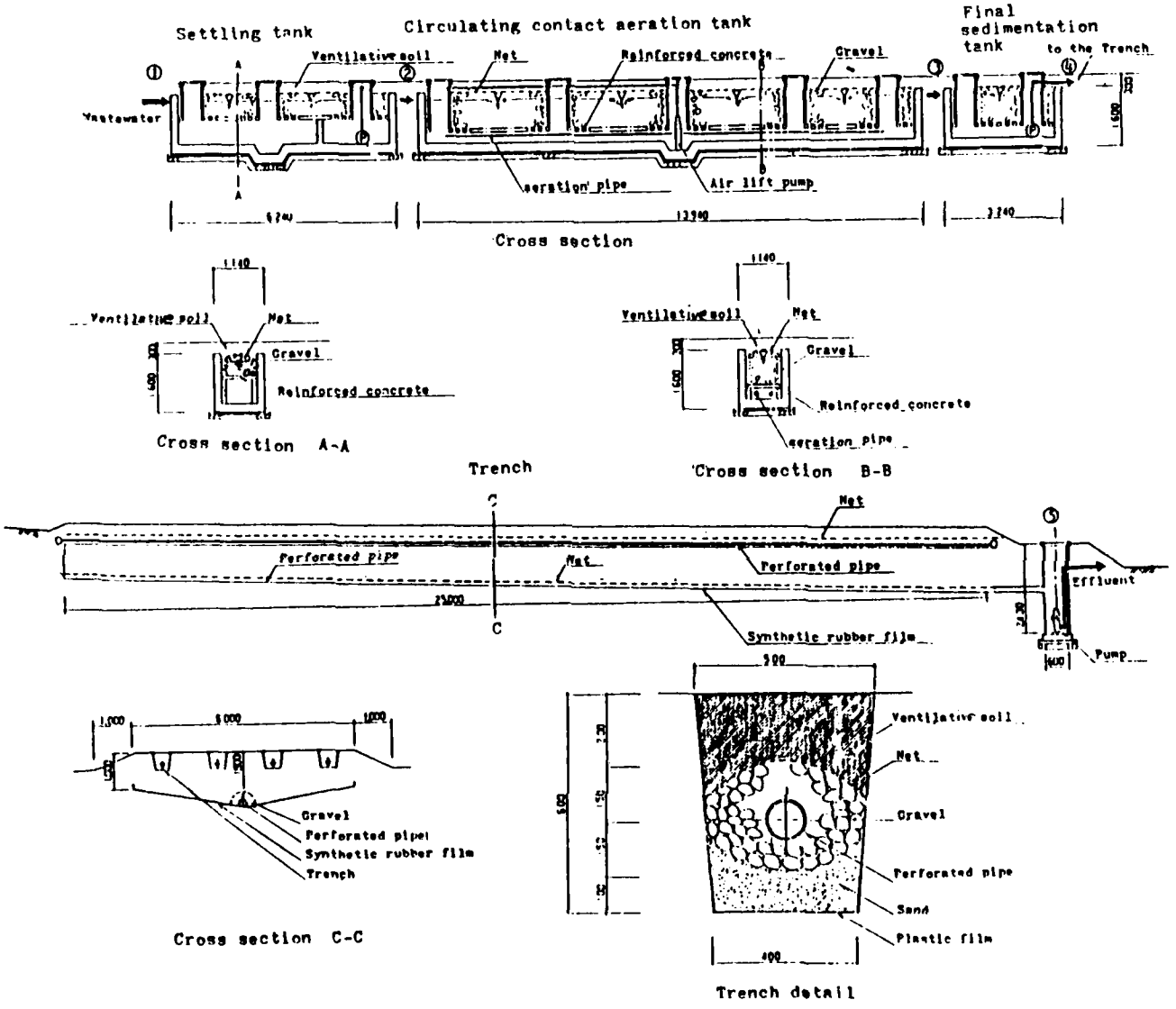


Figure 4. Wastewater treatment system by soil (MOC) (dimensions in millimeters)

TABLE 3. CHARACTERISTICS OF COVERED SOIL

Item	Settling Pond	Circulating Contact	Trench No. 1
Unit weight (dry) (g/cm^3)	1.31	1.47	1.39
Unit weight (wet) (g/cm^3)	0.72	0.80	0.72
Porosity ratio	2.52	2.19	2.55
Specific weight	2.52	2.54	2.56
Size distribution (%)			
> 2 mm	0	0	0
0.2 - 2	35	37	35
0.02 - 0.2	51	47	44
0.002 - 0.02	11	11	16
< 0.002	3	5	5
pH	6.6	6.7	6.7
T-N (mg/kg)	4,280	1,660	3,030
T-P (mg/kg)	3,090	3,540	1,810
K (mg/kg)	1,210	1,480	1,110

other data were not available for the frequency and items analyzed. Fluctuations in the data exist to some extent, but these may have been caused by fluctuations in the number of pigs and by the water sampling method used. The transition of water quality in each step of the process was obtained from mathematical means for 4 years and is shown in Figure 5 and Table 4. Seasonal fluctuations are shown in Figure 6.

Results can be summarized briefly as follows:

a. T-N and NH_4-N in the settling tank are higher than those of wastewater. This may be caused by the method of water sampling.

b. Seventy percent of the T-N is removed--half of it is removed in the circulating contact aeration tank and the other half in the trench.

c. Nitrification is observed in the circulating contact aeration tank and the trench.

d. Phosphorus is almost totally removed in the trench.

e. Ninety-eight percent of the BOD load is removed; most of it is removed in the circulating contact aeration tank and the trench.

f. Only the settling tank and the trench remove SS.

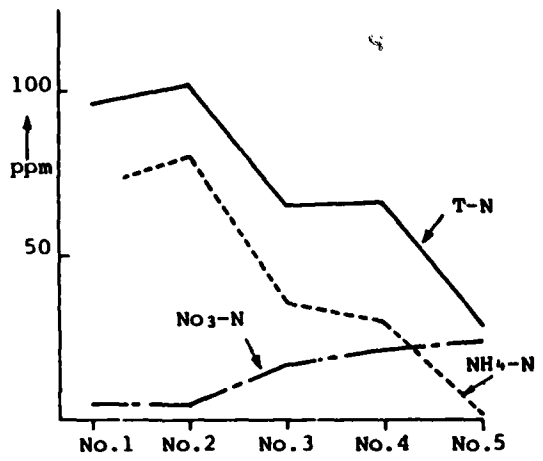
g. In seasonal fluctuation, T-N and NH_4-N show a high rate in winter and summer, but no large fluctuation is observed after the trench step.

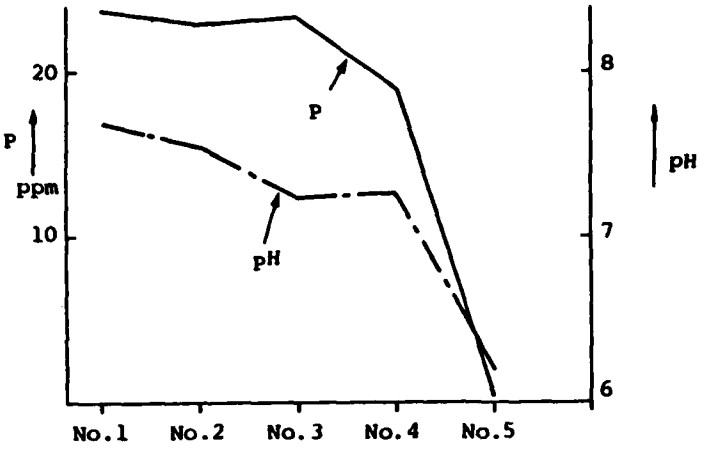
h. The removal ratio of phosphorus decreases in August and September.

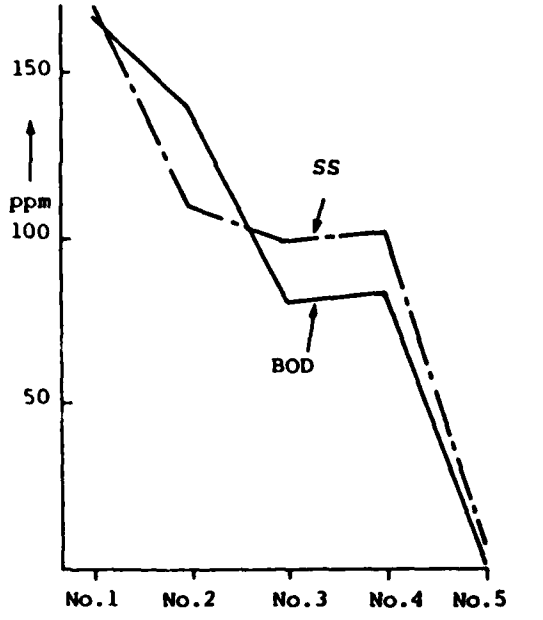
Experiment by Okayama

The experiment facilities were constructed inside the Environment and Health Center to purify all wastewater from the office. The treatment system consisted of a settling tank, circulating contact aeration tank, final sedimentation tank through biofilm, and trench, the same as MOC. The trench was 40 cm wide and 4 m long in order to treat 1 cu m of wastewater per day. The operation began in April 1982. The resulting treated volume was far below the initial design and was 41.4 ℓ/day/sq m (averaged over a year) assuming that a 2.5-m-wide trench is valid to purify wastewater. Common weathered granite was used. The data were analyzed once a month (Table 5) in the form of mathematical means.

The wastewater per square meter of land is bigger than the one of MOC, but 99.5 percent of BOD load, 48 percent T-N, and 95 percent of T-P were removed. Compared with the result of MOC, though the removal ratio of BOD is better, that of T-N is worse. The difference might have been caused by the soil used. The reason nitrogen was not removed might be because the anaerobic bacteria were not as active due to the lack of carbon as nutrients for anaerobic bacteria because the BOD load was improved very well. Generally, coliforms will almost always be killed by various kinds of bacteria and protozoan when they infiltrate the soil 1 m and by sieving function through soil porosity. In this experiment coliforms were rarely observed after the trench step.







Remarks: No.1 Wastewater
No.2 At the end of settling tank
No.3 At the end of circulating contact aeration tank
No.4 At the end of final sedimentation tank through biofilm
No.5 At the end of trench

Figure 5. Water quality in each step of the process (MOC)

TABLE 4. WATER QUALITY BY STEP (MOC)

Item	No. 1	No. 2	No. 3	No. 4	No. 5	Removal Ratio %
pH	7.7	7.6	7.2	7.3	6.2	--
BOD, ppm	169	139	81.1	83.8	2.83	98.3
COD, ppm	112	89.0	62.2	66.1	3.58	96.8
SS, ppm	171	110	100	102	8.7	94.9
NH_4-N, ppm	73.0	79.8	35.7	29.5	3.13	95.7
NO_2-N, ppm	1.90	2.28	4.56	5.70	1.20	--
NO_3-N, ppm	4.42	4.42	17.7	22.11	23.79	--
T-N, ppm	95.4	102.5	66.0	66.7	28.25	70.4
T-P, ppm	23.8	22.90	22.42	19.25	0.58	95.9

Remarks: No. 1 = Wastewater.
No. 2 = At the end of settling tank.
No. 3 = At the end of circulating contact aeration tank.
No. 4 = At the end of final sedimentation tank through biofilm.
No. 5 = At the end of trench.

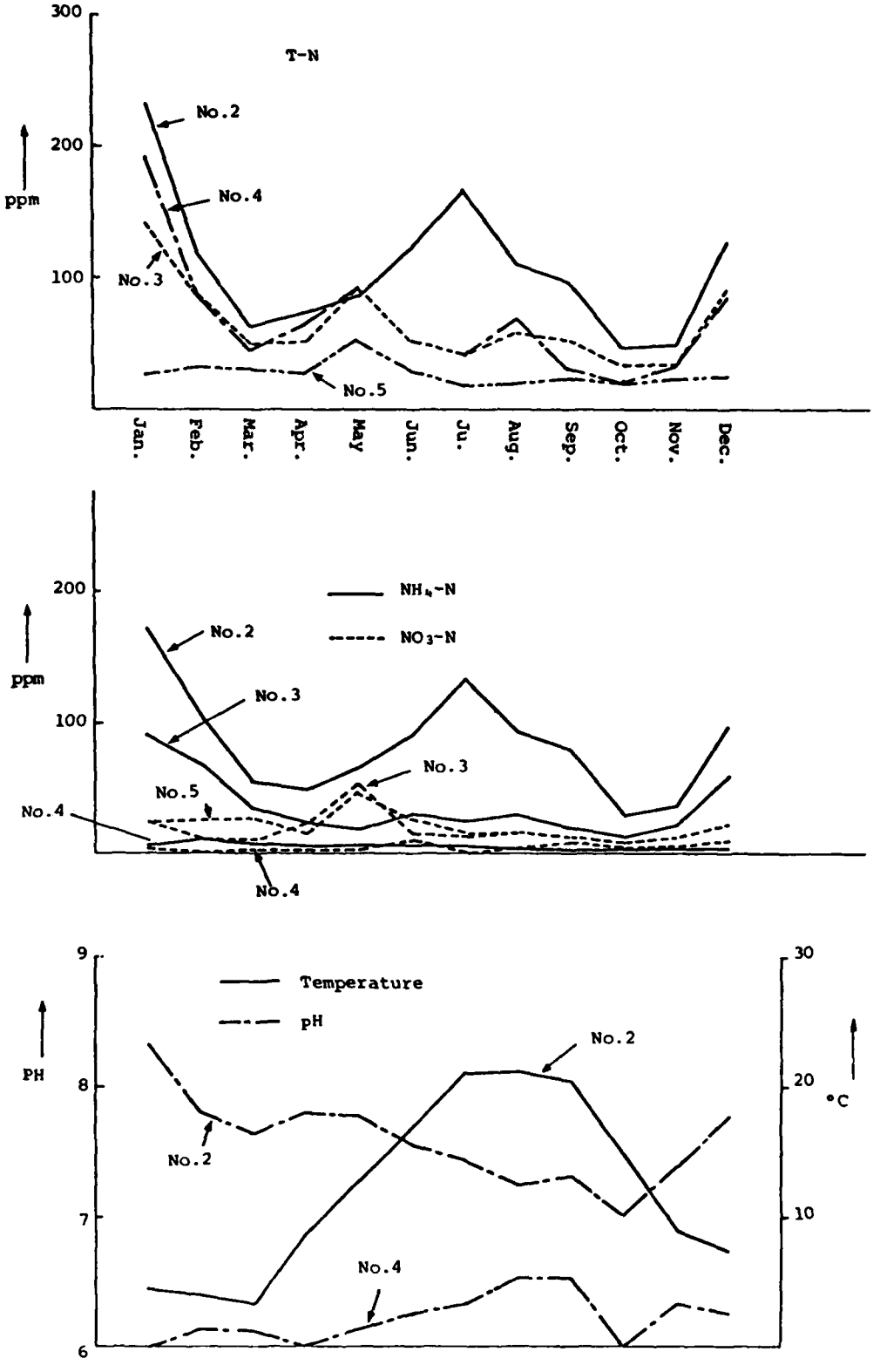


Figure 6. Seasonal water quality

TABLE 5. WATER QUALITY BY STEP (OKAYAMA)

Item \ Place	No. 1	No. 2	No. 3	No. 5	Removal Ratio %
pH	7.9	7.1	5.9	6.7	--
BOD, ppm	162	47.1	9.8	0.8	99.5
COD, ppm	109	37.9	17.2	3.4	96.9
SS, ppm	231	15.5	6.7	1.7	99.6
NH_4-N, ppm	38.9	35.3	10.9	0.03	99.9
NO_2-N, ppm	0.23	0.02	0.14	0.03	87.0
NO_3-N, ppm	0.3	0.1	22.3	24.9	--
T-N, ppm	54.4	44.0	37.3	28.3	48.0
T-P, ppm	5.16	3.94	3.06	0.26	95.0

Remarks: These data were released by Environment and Health Center, Okayama Prefecture.

COMPREHENSIVE CONSIDERATION ON WASTEWATER TREATMENT BY SOIL

There are some problems in these experiments, i.e., the period of data is short and the kind of soil was not fully examined in the Okayama experiment, and coliform and other germs were not observed in the MOC experiment. However, the advantages and disadvantages of wastewater treatment by soil are generally as follows.

The advantages are:

a. Removes phosphorus completely when the soil is selected appropriately.

b. Removes nitrogen far better than activated sludge method.

c. Small amount of sludge produced.

d. Less energy consumption.

e. Removes coliform and other germs.

f. Low construction cost.

g. Treatment in each household is possible if there is a piece of land available.

h. Easy to maintain.

i. No sanitation or foul odor problems.

The disadvantages are:

a. Unable to treat massively.

b. Limited by the land space.

c. Treated water quality may fluctuate due to the soil condition.

d. Treated water quality is not as good when the biofilm at the surface of contact materials is naturally extracted.

e. Not appropriate when the ground-water level is high.

The problems yet to be solved are studying the relationship between nitrogen removal ratio and soil condition and between nutrient supply method and anaerobic bacteria, and obtaining more data on many items and cases in various places using various soils.

IN SITU STREAM WATER CLARIFYING TESTS

M. Tanimoto and T. Yoshida

Japan Sediments Management Association

ABSTRACT

The advance of river pollution has slowed somewhat since legal regulations on various effluents were enacted. However, the situation is still serious in our country. To date many tests and considerable research have been conducted on river restoration. However, due to the difficult conditions inherent in rivers, there are few examples of actual application.

Since lake pollution results from polluted inflowing rivers, river restoration is indispensable to lake restoration. A new method has been developed to clarify stream water in situ. This paper describes the test results and the method's feasibility.

DCF METHOD

This method of clarifying river water in situ employs a boxlike apparatus made from steel divided into several chambers with an inclined fibrous plate in each (Figure 1).

The stream flowing inside the apparatus is subjected to downward flow by the sloped plate. At the bottom edge of the plate the suspended solids in the stream are densified. Then the stream line is changed to an upward flow, where the stream water is clarified because some of the densified solids remain at the river bottom. An unwoven fibre is spread on the plate which

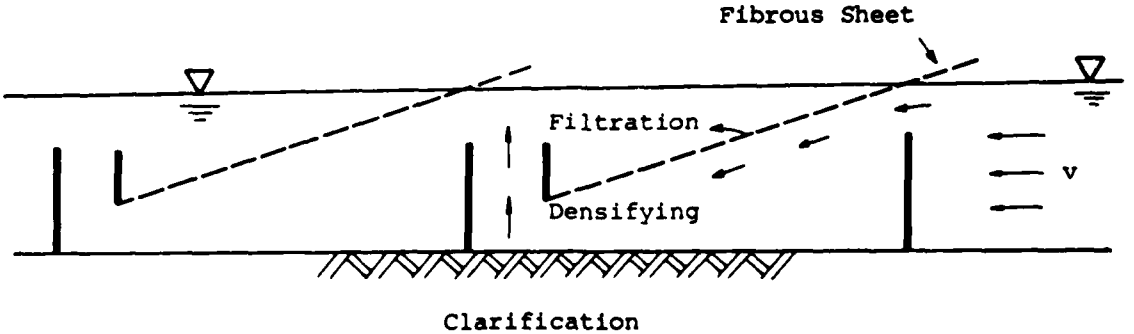


Figure 1. Principle of a direct clarifying method

acts as a filter. Thus, three procedures, densification, clarification, and filtration, are performed inside the apparatus. Therefore, we call it the DCF method.

FIELD TESTS

Field tests using a test apparatus were conducted three times in Shakujii River (Tokyo) and Sanno River flowing into Kasumiga Lake in 1981-1984.

Sanno River

The test apparatus used in Sanno River had five chambers, a total length of 12.5 m, a width of 3 m, and a height of 1 m (Figure 2).

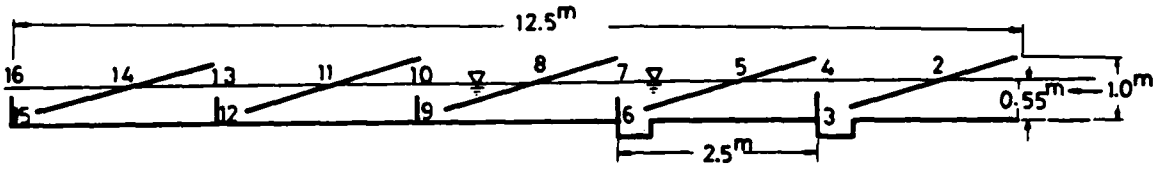


Figure 2. Test apparatus

Figure 3 shows a photograph of the apparatus placed in Sanno River.

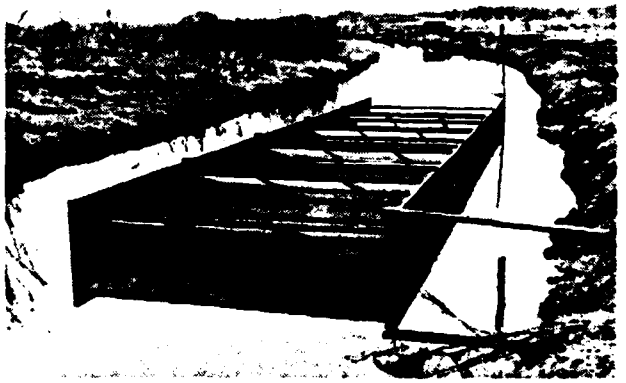


Figure 3. Apparatus placed in Sanno River

Figure 4 indicates clearly that the suspended solids settle quite well on the bottom plate. Figure 5 also shows good sedimentation of solids at the bottom groove.

The water quality of Sanno River is shown in Table 1. The SS concentrations of the water are 11.6 - 22 mg/ℓ averaging 17.2 mg/ℓ and the BOD concentrations, 4.3 - 13 mg/ℓ, average 8.0 mg/ℓ. Nutrient concentrations are not at a lower level. The value of N/P is about 10. The reduction of SS concentrations is significant as Figures 6 and 7 show. The average reduction ratio is about 40%. Figure 8 shows the relationship between inflowing and outflowing concentrations of SS.

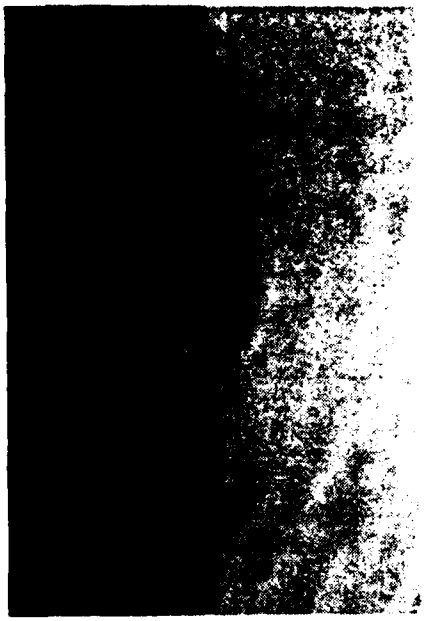
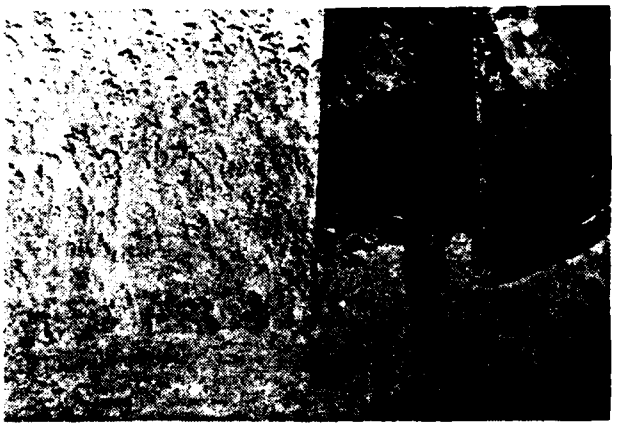


Figure 4. Settled solids



Apparatus erected
on the river shore

Figure 5. Settled solids at bottom groove

TABLE 1. WATER QUALITY OF SANNO RIVER

mg/ℓ without pH

Date	Time	pH	SS	COD	BOD	n-Hex	T-N	T-P
3/7	17:30	7.0	20.0	14.0	9.9	1.5	4.75	0.52
3/9	10:00	7.0	17.0	8.4	6.1	2.8	3.72	0.34
3/9	14:00	7.2	22.2	14.5	13.0	3.1	4.31	0.46
3/12	10:00	7.0	11.6	8.0	7.6	2.2	4.46	0.34
3/12	14:00	7.0	14.0	9.3	8.3	3.7	4.05	0.38
3/17	10:00	7.1	17.2	7.2	4.3	1.2	3.83	0.30
3/17	14:00	7.1	18.7	8.3	6.7	1.6	4.20	0.38
Average		7.06	17.2	10.0	8.0	2.3	4.2	0.39

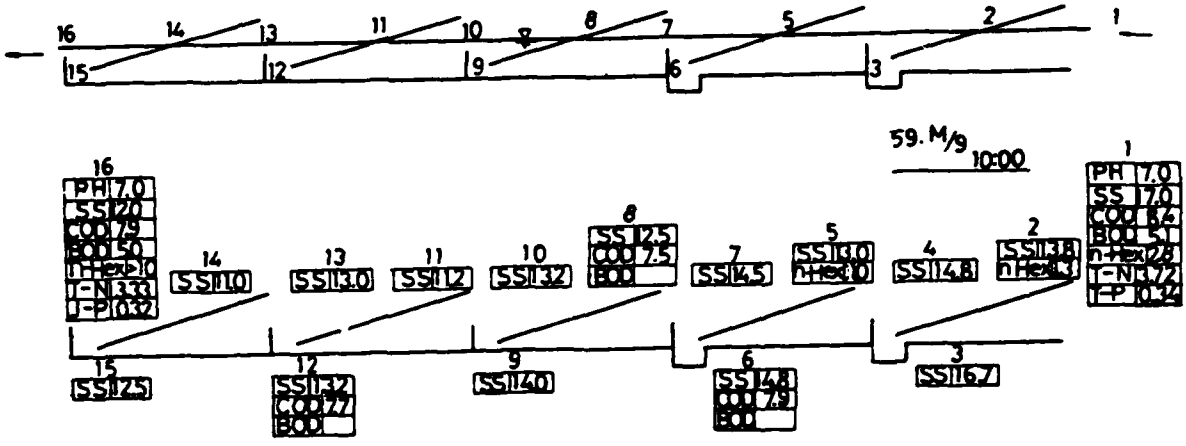


Figure 6. Example of measurements

The reductions of other constituents are not as large. The reduction is especially small for nutrients.

Factor	Reduction Ratio
SS	40%
BOD	19%
T-P	11%
T-N	5%
n-Hexane	47%

This suggests that most of the nutrient concentrations are soluble and only slightly influenced by the apparatus.

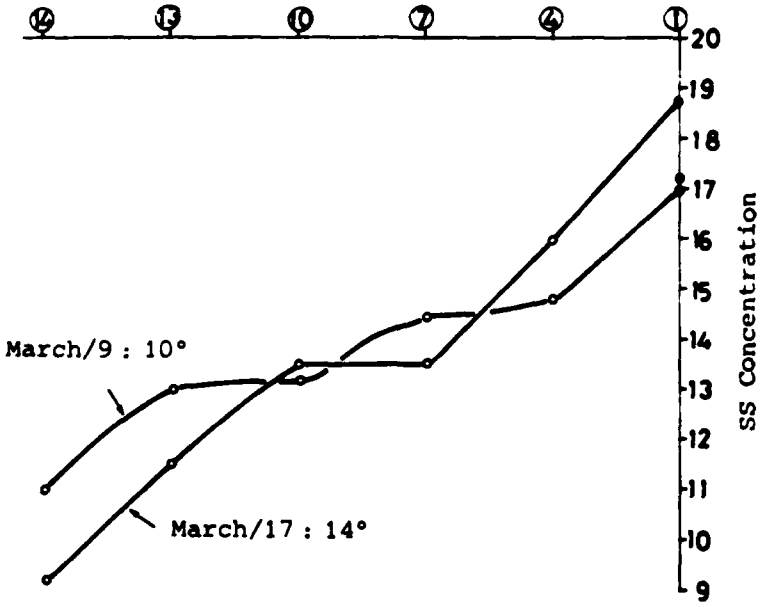


Figure 7. Reduction of SS concentrations

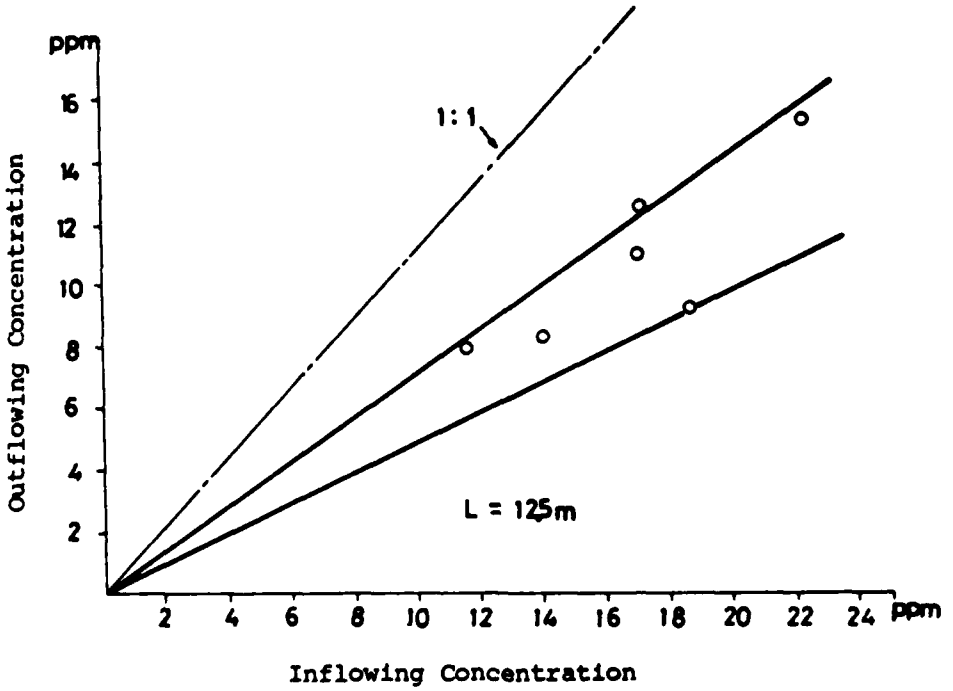


Figure 8. Relationship between inflowing and outflowing SS concentrations

DENSIFYING AND CLARIFYING RATIOS

The reduction in SS concentrations clearly results from the downward flow velocity which accelerates the sedimentation of SS particles. Figure 9 shows the densifying and clarifying ratios.

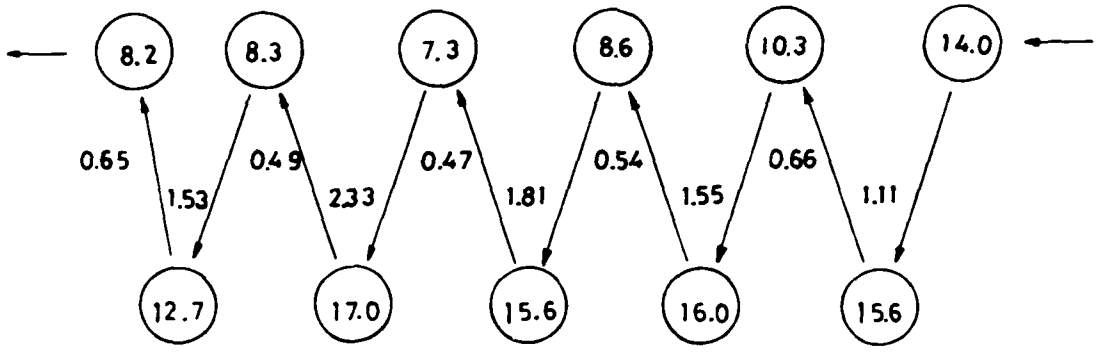


Figure 9. Concentrating and clarifying ratios (March/12, 14°)

The mean densifying ratio is 1.67. Note that the sloped plates functioned effectively for the concentration of solid particles instead of the lower concentrations of SS in the water.

The mean clarifying ratio is 0.56, ranging from 0.47 to 0.66. This means that 44% of the solids reaching the bottom remain there. The mechanism of the concentration by sloped plate can be simulated by a hydrological model (Figure 10). The change of SS concentration under the sloped plate is represented as follows:

$$Ux = \frac{\partial C}{\partial x} + (Uz + w)\frac{\partial C}{\partial z} = Kx\frac{\partial^2 C}{\partial x^2} + Kz\frac{\partial^2 C}{\partial z^2} \qquad (1)$$

Ux = horizontal velocity component

Uz = vertical velocity component

w = mean settling velocity of solid particles

Kx = diffusion coefficient in the horizontal direction

Kz = diffusion coefficient in the vertical direction

C = SS concentration

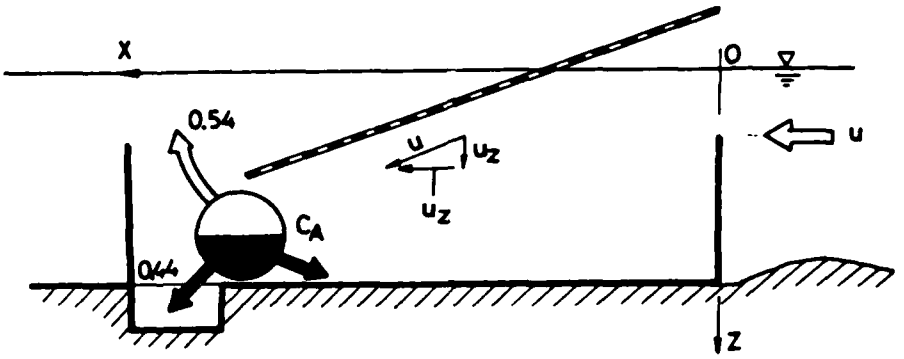


Figure 10. Hydrological condition of stream inside apparatus

We can obtain the exact solution of Equation 1 as follows. The proper value of λ is determined from the following function:

$$\frac{\sin\theta}{\sqrt{\lambda+1}}\sqrt{\frac{Kx}{Kz}}\tan^{-1}h\frac{2a\sqrt{\lambda^2+1}}{1+\lambda^2-3a^2}-\frac{1}{\lambda}\tan^{-1}\frac{(b_1-3b)\lambda}{\lambda^2+3bb_1}=\sqrt{\frac{Kx}{Kz}}\ell\sin\theta-h \qquad (2)$$

$$a = Ux\sqrt{\frac{1}{U_x^2+(U_z+w)^2\cdot Kx/Kz}}$$

$$b = (Uz+w)\sqrt{\frac{1}{U_x^2\cdot\frac{Kz}{Kx}+(Uz+w)^2}} \qquad (3)$$

$$b_1 = b(1-2k)$$

$$h = \frac{H}{2Kz}\sqrt{U_x^2\cdot\frac{Kz}{Kx}+(Uz+w)^2}$$

$$\ell = \frac{L}{2Kx}\sqrt{U_x^2+(Uz+w)^2\cdot\frac{Kx}{Kz}}$$

If the SS concentration at the lower edge of the plate is C_A, we can obtain the concentration ratio as follows:

$$C_A/C_o = \sum_{n=1}^{\infty} An \frac{\lambda_n\sqrt{\lambda_n^2 + 1}\cdot e^{a\ell + bh}}{(\lambda_n \cos\lambda_n h + b_1 \sin\lambda_n h)\{\sqrt{\lambda_n^2 + 1}\cdot\cos h\sqrt{\lambda_n^2 + 1}\cdot\ell + a\sinh\sqrt{\lambda_n^2 + 1}\cdot\ell\}} \quad (4)$$

$$A_n = \frac{4\lambda_n\left\{e^{-bh}\lambda_n(b_1 - b) + (b - b_1)\lambda_n\cos\lambda_n h + (\lambda_n^2 + bb_1)\sin\lambda_n h\right\}}{2\lambda_n(\lambda_n^2 + b^2)\{(\lambda_n^2 + b_1^2) + (\lambda_n^2 + b^2)\}\{(\lambda_n^2 - b_1^2)\sin 2\lambda_n h + 4b_1\lambda_n(\lambda_n^2 + b^2)\sin^2\lambda_n h\}} \quad (5)$$

Shakujii River

Figure 11 shows the test conditions at Shakujii River (conducted in 1981).

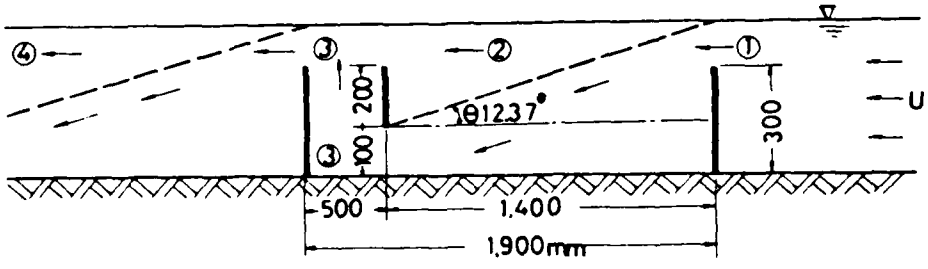


Figure 11. Test condition at Shakujii River

At this time, the mean densifying ratio was 1.3. Figure 12 indicates its computed curve.

From the simulation it was found that the diffusion coefficient in the vertical direction (Kz) had a high value of about 200 cm^2/sec. This is apparently a major cause for the promotion of the sedimentation.

CONCLUSIONS

The DCF method has the following attributes:

a. No energy.

b. No labor.

c. Very simple construction.

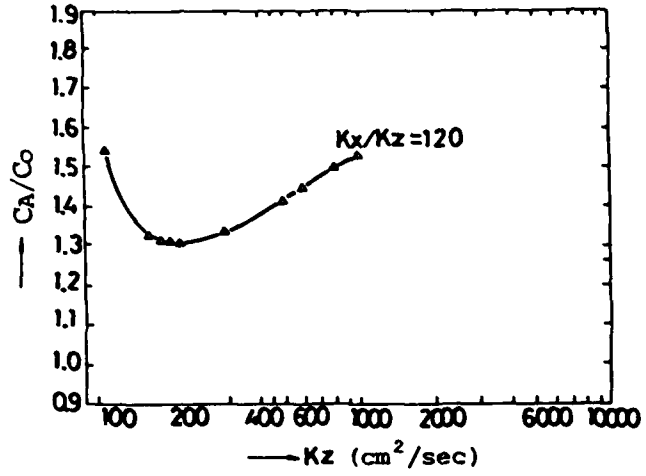
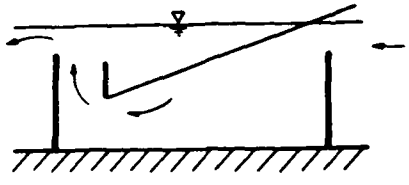


Figure 12. Computed curve of densifying ratio

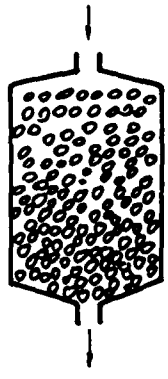
d. Very cheap investment.

e. Very high clarifying capability.

In general industrial water treatment, a filtration procedure is applied to the clarification of water having less than 20 mg/ℓ SS concentrations. Clarification by the DCF apparatus lies in this range of industrial filtration. It is reasonable to note that we attained good results when comparing industrial filtration techniques with the simple device of the DCF method. Consequently, the feasibility of the DCF method seems to hold promise.

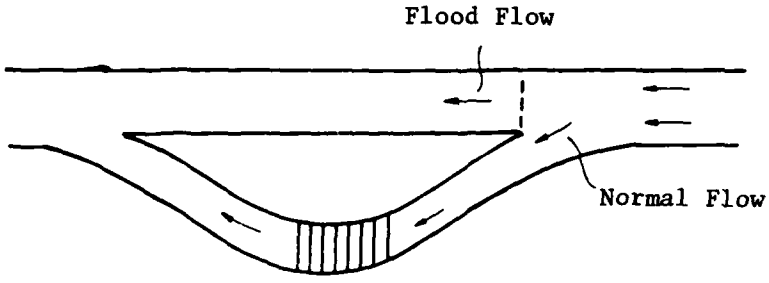


DCF



Sand Filtration

In the application of the DCF method to actual rivers, the most difficult problem may be countermeasures for flood flows. Preparing a by-pass flow is one safe method.



DCF Apparatus

The preceding tests were conducted as the first step in a major project of the Ministry of Construction regarding lake and river restoration.

PURIFICATION OF NHO RIVER BY THE GRAVEL CONTACT METHOD

M. Kobayashi

Director
River Adjustment Division, River Department
Kanto Regional Construction Bureau
Ministry of Construction
1-3-1 Ootemachi, Chiyoda, Tokyo, Japan 100

T. Kiriku

Director
Keihin Work Office, Kanto Regional Construction Bureau
Ministry of Construction
2-18-1 Tsurumi-chuo, Tsurumi, Yokohama, Japan 230

INTRODUCTION

The Tama River originates from the Kasatori Mountains in Yamanashi Prefecture and joins with its tributaries: the Nippara, the Aki, the Asa, the Nho, and the Hirase. It then runs along the border between Tokyo and Kanagawa Prefecture and flows out into Tokyo Bay (Figure 1). This river is a typical urban river in Japan. From the earliest days, the Tama River has played an important role in flood control and irrigation in the socioeconomics and culture of the metropolitan area.

As urban areas and industries continue to develop in the middle and lower basin, it becomes necessary to note the importance of preservation of its natural environment, not to mention flood control and utilization of the water. Therefore, the Master Plan of Environmental Control of the Tama River was presented in March 1980. In the plan, efficient use of the river terrace and good conservation of the natural environment are coordinated.

One of the main characteristics of this plan is the restoration of abundant and clear streams. In reality, pollution of the Tama River began in the latter 1950's and has progressed with urbanization of the river basin. The main cause of river pollution is the discharge of domestic and industrial wastewater from the middle and lower basin area.

The Nho River, a tributary of the Tama River, is polluted heavily by such wastewater and is causing deterioration of the water quality of the main river. The Nho River Purification Facility intends to help improve the Tama River water quality by purifying the Nho River and discharging the purified water to the Tama River. The purification method called "Gravel Contact Purification" (herein-after referred to as GCP method) is used to artificially raise the natural purification potential. Since the Tama River has a wide river terrace and an abundant gravel resource, this method is applicable to the purification facility at Nho River.

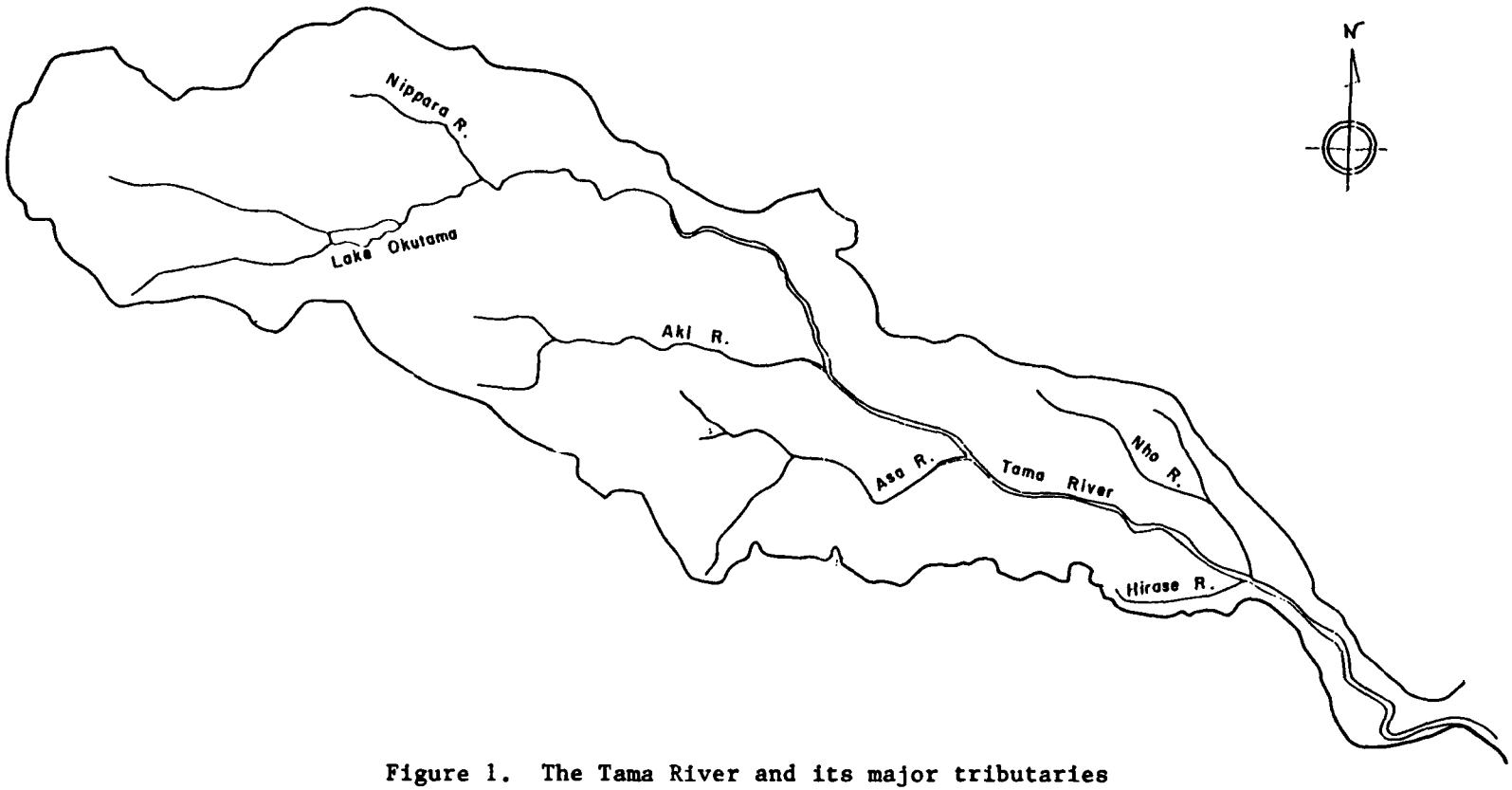


Figure 1. The Tama River and its major tributaries

Application of the GCP method on the Nho River is the first practical use of the method in Japan. The polluted influent water from the Nho River will flow through the gravel laid under the terrace of the Tama River, and, as a result, organic matter in the water will be removed.

The construction of the Nho River Purification Facility, located at off-Kamata, Setagaya District of Tokyo, upstream of Shin-Futako Bridge, began in November 1981 and was completed 2 years later in July 1983 at a cost of 640 million yen.

BACKGROUND OF THE PURIFICATION FACILITY'S CONSTRUCTION

For many years, the Tama River was always clean and was treasured by the people. However, after World War II, in the middle and lower basin areas, rapid urbanization caused a sizable discharge of the domestic and industrial wastewater to the river, and, simultaneously, the upstream clean flow was removed at the Hamura Intake Dam. Consequently, the water quality in the middle and lower basin areas began a rapid deterioration in the first half of 1960's, augmented by the increasing portion of domestic wastewater (Figure 2). In the following years, dredging of the settled sludge on the river bottom, a work for purification, was carried out and gradually improved the water quality.

In the middle basin areas, however, the improvement program including construction of the sewerage system could not cope with the increasing domestic wastewater discharge. The worst water quality was recorded in 1970 and, although the situation has improved slightly since then, the water quality is still less than the environmental standards tolerance. The river's quality ranks in the worst 10 of 109 major (First Class Category) rivers in the country.

The improvement of river water quality can be achieved, in principle, by reduction of the inflowing pollution loads. In the case of the Tama River, pressed by the request for improvement, the reduction of pollution loads has been envisaged to promote sewerage systems construction and strong enforcement of industrial wastewater control. As a result, the increase in pollution has slowed in some areas, but in most areas the effects have not yet been realized.

The Tokyo Metropolitan Office and the municipalities located along the river are primarily promoting secondary treatment sewerage systems. In a river like the Tama where the wastewater flow is relatively high proportional to the river's base flow, improvement of the water quality by dilution is not very effective, because the total pollution load in effluent after the secondary treatment is still substantial, even when the treatment efficiency of the sewerage system has been raised. That is, the discharged effluent becomes a large source of pollution.

Therefore, introduction of tertiary treatment will make improvement of discharged effluent possible, but many years and huge expense is incurred before completion. Early execution of the tertiary treatment cannot be expected.

In order to improve the water in Tama River under such a situation, it is necessary to devise a countermeasure, along with the construction of sewerage systems. The river control authority, the stimulus for concentrated

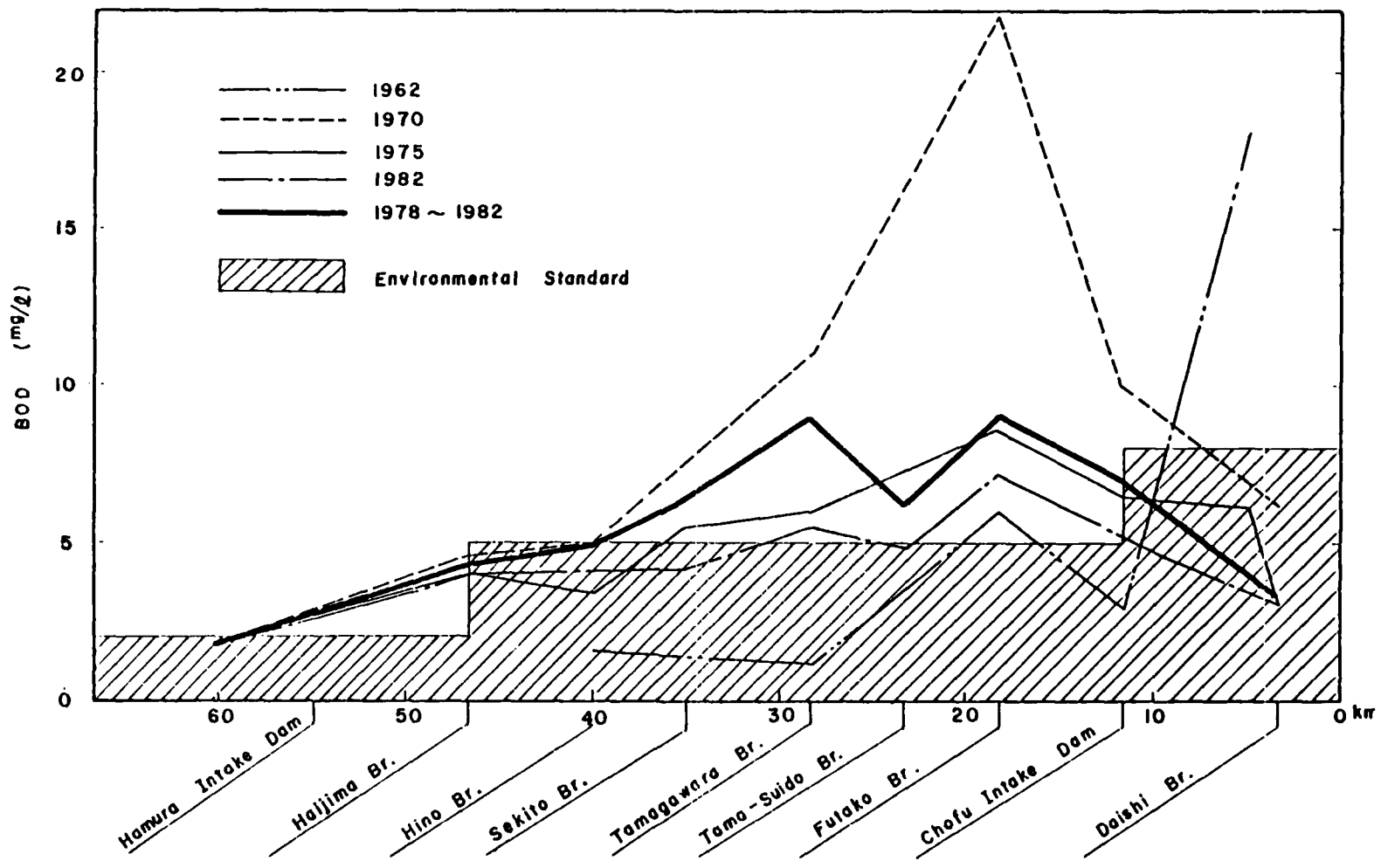


Figure 2. Longitudinal change of water quality of the Tama River

improvement efforts, has judged that those efforts would achieve sufficient technical effects. Besides, the existence of a river like the Tama in an urbanized area is valuable for the inhabitants living there. Making the river as clean as possible, regardless of the environmental standards as a sole criterion, shall be pursued.

With this historical background, the Nho River Purification Facility is a practical installation that employs the direct purification process in which the pollution load flowing in a river is reduced within the river site.

THE GRAVEL CONTACT PURIFICATION METHOD

The GCP method is a process which artificially improves the natural purification potential of a river. Accordingly, the mechanism will be explained in relation to the natural purification potential.

Mechanism of Purification

A river is polluted by receiving inflowing pollutants. However, as the river water flows downstream, it is gradually purified by the various inherent functions, called the river's "natural purification potential." This natural purification potential is broken down into the following four operations:

a. Dilution.

b. Settling and adsorption.

c. Filtration.

d. Intake and decomposition by organisms.

As the river current runs downstream, the pollutants settle and are adsorbed to the river bed, or organisms living in the river bed take in the pollutants, and the river is gradually purified. In this type of purification, the main characters are the living organisms and the river bed's gravel, especially the film covering the river bed's gravel. This slimelike film is called "biofilm" and plays an important role in removing pollutants from the river.

Method

Natural purification of a river requires a long distance and a long time, in principle, to be effective, and when a river is highly polluted, for instance beyond its own purification potential, the pollution will not diminish even as the river flows downstream. For such cases, some means to raise the purification potential must be found.

Increasing the volume of gravel covered with living organisms in the river bed may be one solution. As shown in Figure 3, if a river bed 1 sq m in area is counted as an effective unit, then another bed 5 sq m in area, composed of five layers of 1 sq m in a vertical pile, will be effective as five units. A layer 1 cu m in volume of gravel packed with 5-cm-diam particles presents about 100 sq m of surface area. If such a large surface can act effectively in settling/adsorbing the pollutants and attaching the biofilm, the purification potential will increase markedly.

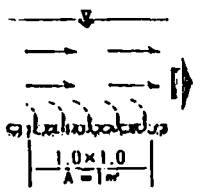
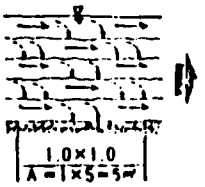
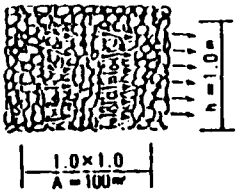
A RIVER	ACCUMULATED RIVER BED	THE GCP METHOD
		
Pollutants are removed by the natural purification capability on the surface of the river bed.	A wider surface area increases the purification capability in the same distance of flow.	Contacting surface area is increased by flowing water into gravel bed so that the purification capability is accelerated.

Figure 3. Accelerating purification capability of the GCP method

A purification method in which an artificially increased contact area removes the pollutants by settling/adsorption to the surface and by intake/decomposition of the living organisms attached to the surface is called "contact oxidation." Various methods using different contact materials to increase the contact area have been devised and practiced. The GCP method is one which uses gravel as the contact material.

Experiment of GCP method

Beginning in fiscal year 1973, the experiment at Tama River was conducted to study onsite purification methods to improve the river's water quality by utilizing the wide river terrace and abundant gravel resource which characterize the river for purifying the polluted water. Figure 4 shows the experimental installation.

The installation was packed with gravel collected at the river terrace, and different effluents of secondary treatment, honey-comb treatment, and sedimentation treatment of sewage water were tested for (a) effectiveness in removing BOD and achieving denitrification, (b) the clogging effect of gravel voids, and (c) the sludge settlement.

The results were as follows:

a. Removal rate. Expected removal rates of 60 to 90% for BOD and 80 to 95% for SS were confirmed. The denitrification rate was small.

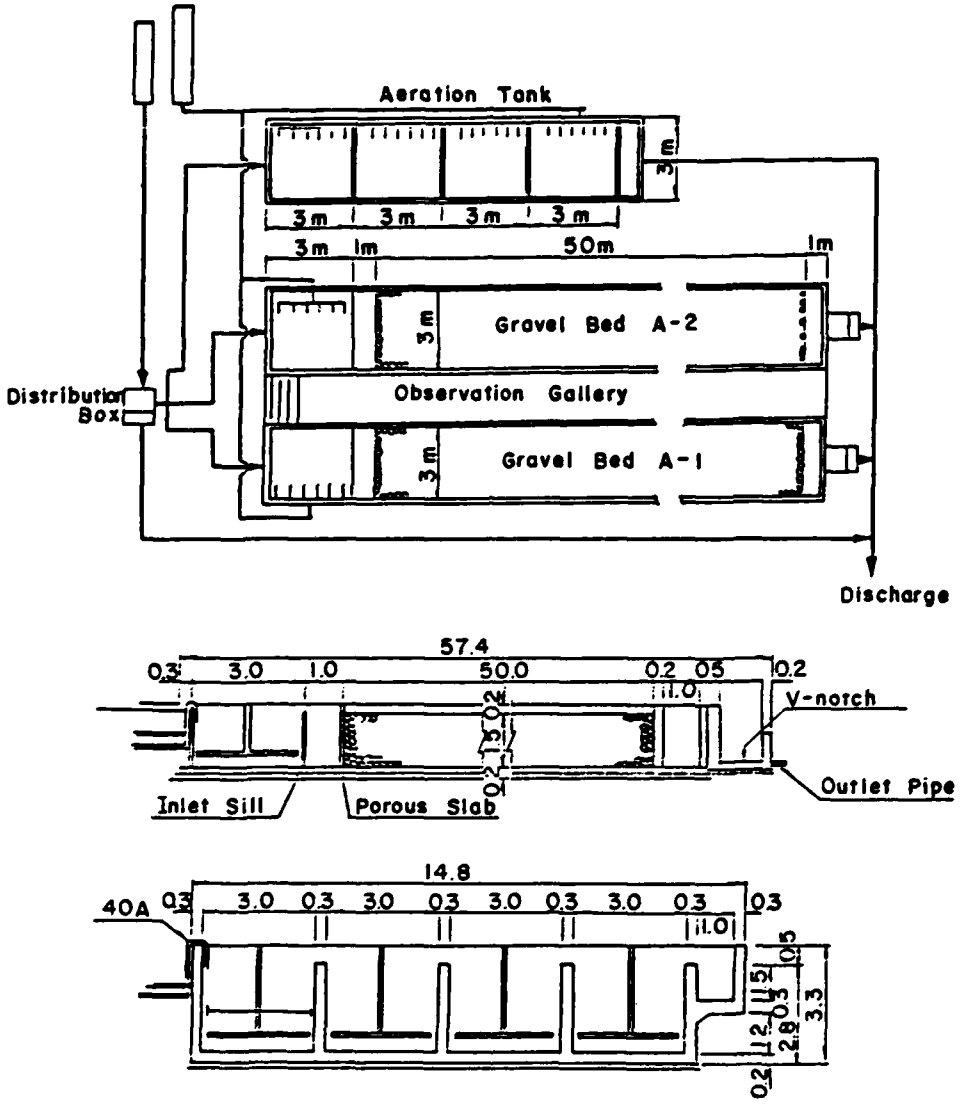


Figure 4. Experimental design for the GCP method

b. *Requirements affecting effectiveness.*

(1) Retention time--The removal rates of organic substances like BOD and SS change depending on the retention time. For instance, 1 hr for 70% removal of BOD and 1.25 hr for 75% were found to be necessary (Figure 5).

(2) Flow length in gravel bed--To attain a steady effect of purification, 15 to 20 m of flow length was found to be necessary (Figure 6).

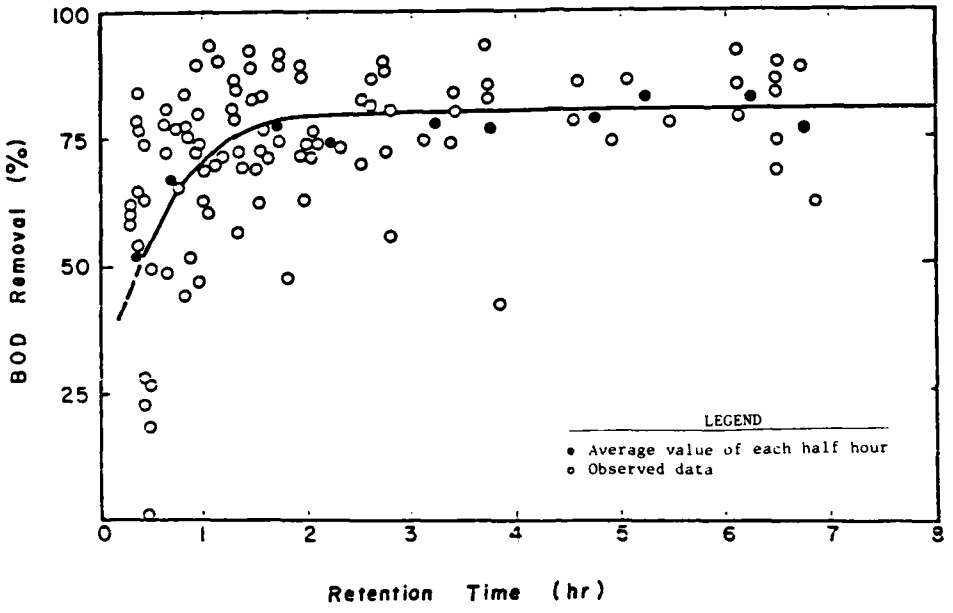


Figure 5. Relationship between BOD removal rate and retention time in a gravel bed

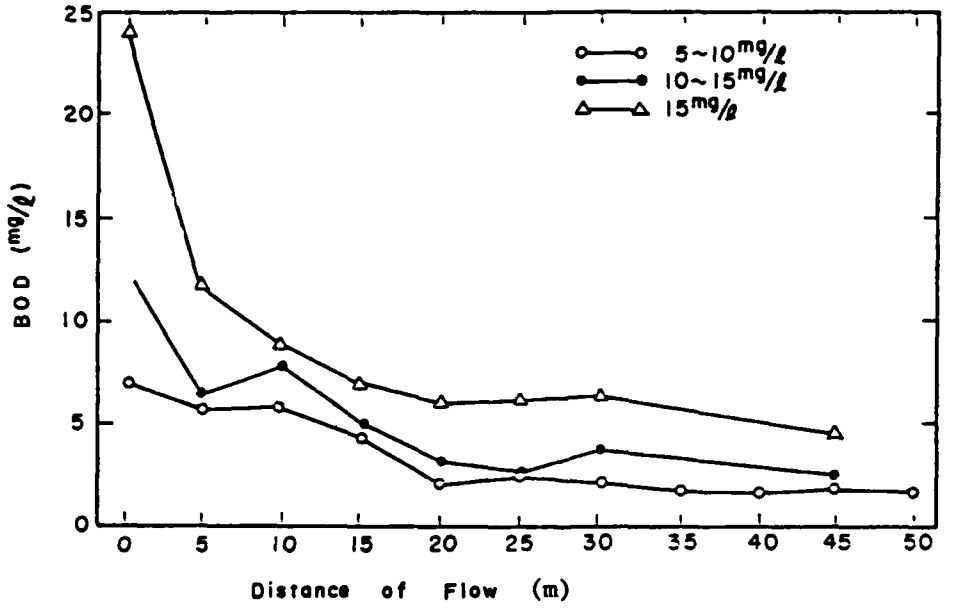


Figure 6. BOD change by distance of flow in a gravel bed

OUTLINE OF NHO RIVER PURIFICATION FACILITY

Design Criteria

Considering the economical feasibility, the retention time of 1.25 hr and the flow length of 17.5 m for the target of 75% removal of BOD were selected for the design (Tables 1 and 2).

TABLE 1. DESIGN VALUES

Item	Design Values	Remarks
Basic flow of Nho River	1.15 m^3/sec	Low flow of Nho River
Treatment flow	1.0 m^3/sec	
Fishway flow	0.15 m^3/sec	
BOD of Nho River	13 mg/ℓ	
SS of Nho River	16 mg/ℓ	
Removal rate of BOD	75%	1.25-hr retention time
Removal rate of SS	85%	1.25-hr retention time
Sludge storage time	5 years	

Description of Flow

The facility is composed of two parts, the intake dam and the gravel bed. The gravel bed is constructed 1.7 m underground in the river terrace of the Tama, not to be submerged by flooding. The intake is an easy-handling inflatable dam made of rubber-coated fabrics selected for the flood's uninterrupted flow.

The water of Nho River raised by the dam forces flows through the inlet pipe into the gravel beds (92 m wide, 148 m long, 1.6 m deep, 8 beds) and spreads by the perforated distribution pipes. The water, flowing in the bed by gravitation, is purified and collected by the perforated collection pipes before being discharged (Figures 7-9).

Advantage of Facility

The advantages of the purification facility are as follows:

a. Saves resources. The main material, 21,400 cu m of gravel, was available at the construction site of Tama River.

TABLE 2. OUTLINE OF THE GCP PLANT

Item	Values	Remarks
	Gravel Beds	
Effective depth	1.5 m	1.6-m bed thickness
Shape of bed	92 m wide × 1.6 m thick × 17.5 m long	8 beds
Size of gravel	20-150 mm	
Void of gravel	Approximately 35%	
Area of beds	1.36 ha (92 m wide × 148 m long)	
Total volume	21,400 m^3	
	Intake Dam (Movable Rubber Dam)	
Dam height	2.45 m	
Width of dam	26.3 m of top	
	16.5 m of bottom	
Material of dam	Ply-rubber (Chloroprene)	12.5 mm thick
Operation system	Automatic deflation by sensing water level	
Attached devices	Water level meter	Monitor and record
	Alarm device	

b. Saves energy. The difference of levels at the intake and discharge points is utilized for the gravitational flow and no energy is input to the system.

c. Maintenance free. With ordinary purification systems, the maintenance problems including the disposal of generated sludge are troublesome, while with this facility the sludge is decomposed and deposited naturally. This makes the system maintenance free for several years and less sensitive to a drop in efficiency.

d. Utilization of surface area. When the gravel bed is covered by earth, the surface can be used for recreational purposes such as a playground and plaza for the people; this is a marked advantage.

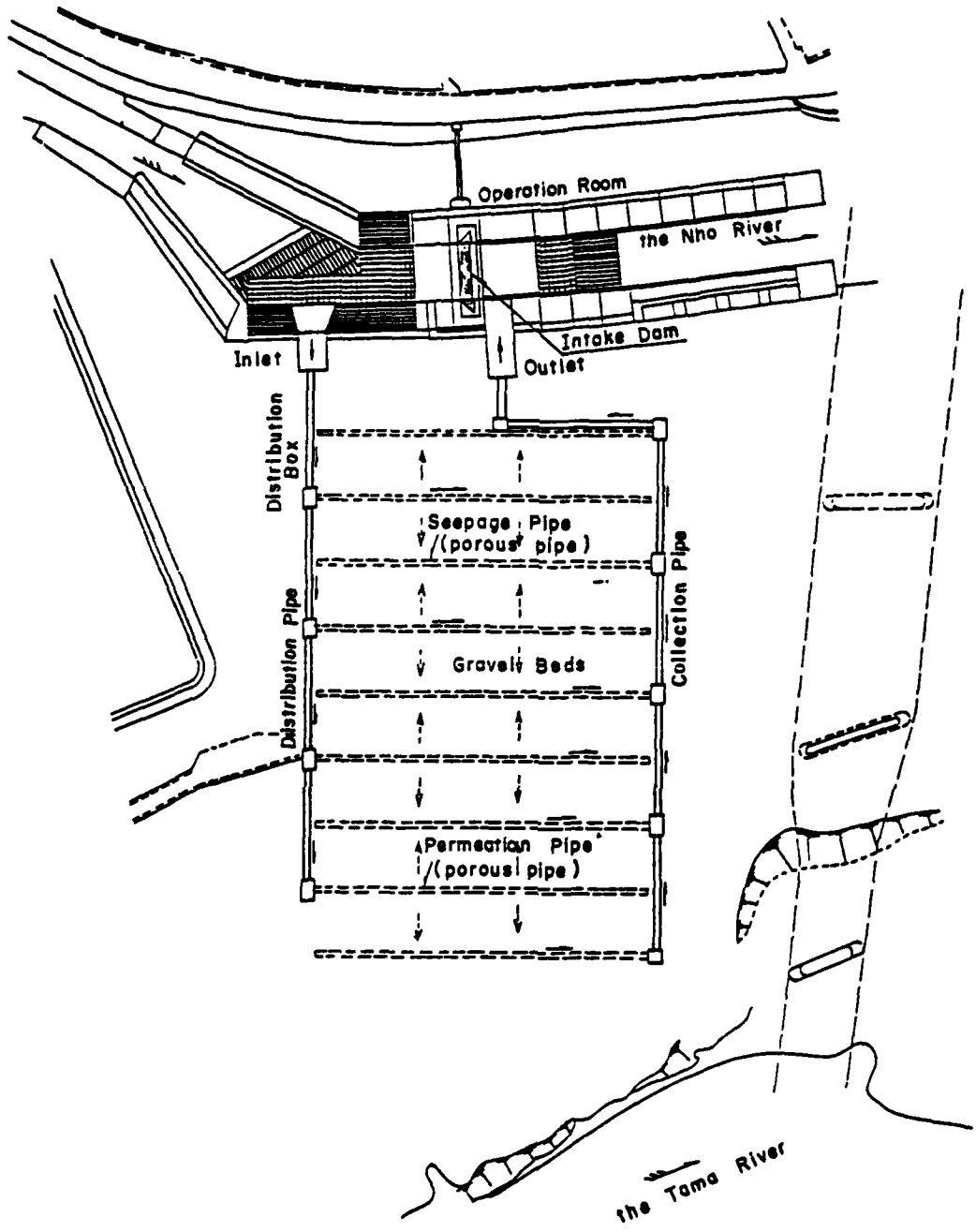


Figure 7. Schematic of the GCP plant

PURIFICATION IN NHO RIVER FACILITY

Intake Dam

The rubber-coated inflatable dam raises the river's level by about 2.45 m. Large suspended solids (SS) are removed by settling in the pond. When inorganic SS are largely contained in the runoff from rainfall and the water is

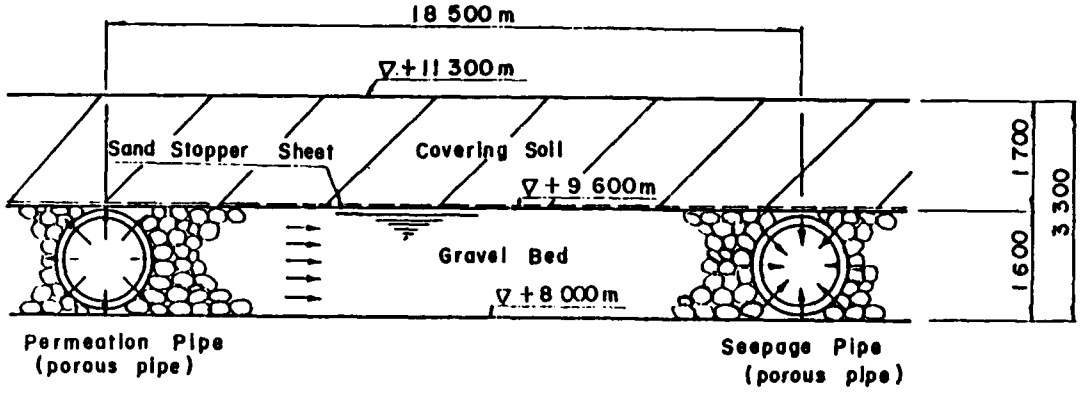


Figure 8. Cross-sectional view of a gravel bed

led to the gravel bed, it will shorten the service life of the system. In such a case, the dam is flattened automatically so that the turbid water is kept away from the facility.

Gravel Beds

The pollutants contained in the flow are removed by contact settling/adsorption in the beds, while the organisms in the biofilm covering the gravel surface contribute to purification.

Outlet Work with Sill

The DO in the inflow is consumed in the purification process by the gravel bed organisms. At the outlet, the discharging effluent is re-aerated to recover the lost DO by falling over the weir.

ROLE AND EFFECT OF NHO RIVER PURIFICATION FACILITY

The Nho River, beginning in the Kokubunji City area, flows through the densely populated Koganei City, joining with the Sen River before it flows into the mainstream of the Tama River at a location off-Kamata, Setagaya District of Tokyo Metropolis. The Nho River, as one of the most polluted tributaries of the Tama River, is a source of pollution raising BOD concentration of the Tama River in the middle basin areas.

The Nho River Purification Facility was planned to treat 1.0 m^3/sec, of the 1.15-m^3/sec flow in Nho River, and remove 75% of the BOD and 85% of the SS. For instance, when the Nho River water contains 13 mg/ℓ BOD and 16 mg/ℓ SS and the facility is working according to plan, it will improve the water to 3.25 mg/ℓ BOD and 2.4 mg/ℓ SS. The result is that the mainstream pollution of the Tama River will be decreased by about 10% by dilution of treated effluent (Figure 10).

TRACK-DOWN INVESTIGATION OF NHO RIVER PURIFICATION FACILITY

The outline of the track-down investigation for 13 months, from start-up to July 1984, will be described here.

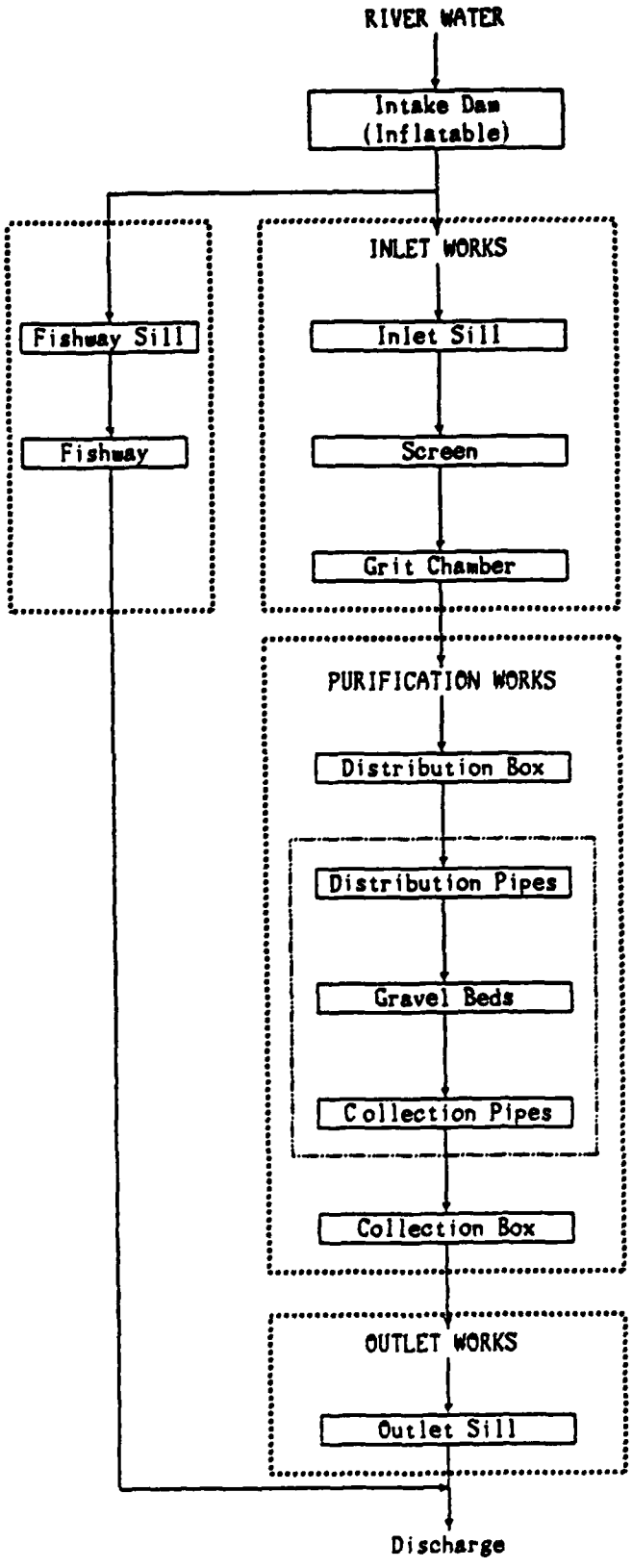


Figure 9. Flowchart of GCP plant

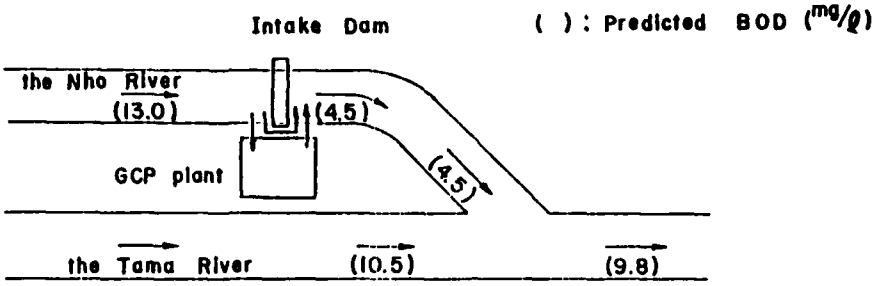


Figure 10. Predicted BOD after construction of the GCP plant

Operation of Facility

During 13 months of operation after start-up, the dam's automatic deflation took place 24 times due to substantial rainfall. All cases registered a precipitation exceeding 10 to 20 mm/day, according to Setagaya District's record. In addition, 24 intentional deflations of the dam, unrelated to rainfall, were conducted for construction, experiment/inspection, and cleaning of the intake equipment. Excluding the period of construction, the facility was operational for 272 days, showing a 77% efficiency.

Nho River Flow

The flow fluctuated between 0.16 and 1.50 m^3/sec, making the average 0.60 m^3/sec, which was about 50% of the design value. As for the overall changes, the average flow of 0.79 m^3/sec in July-November 1983 fell to 0.47 m^3/sec, 60% of the mentioned figure, in December 1983-July 1984 period. The noticeable decrease after December in the previous year was obviously due to low precipitation (Figures 11 and 12).

Change of Treated Flow

The inflow to the beds, or the treated flow, during the period was 0.14 to 0.83 m^3/sec, an average of 0.41 m^3/sec equal to 40% of the design value. The fishway flow was 0.01 to 0.44 m^3/sec with an average of 0.17 m^3/sec. Decreased flow in the Nho River due to low rainfall affected the treatment facility, along with the unexpectedly large quantity of floating matter such as garbage and weeds which clogged the inlet screen to the gravel beds.

Water Quality of Nho River

The water quality sample was taken at a spot near Nogawa-Suido Bridge, 700 m upstream of the intake where the backwater's influence was minimal. Figure 13 shows the monthly average from July 1983 to July 1984. The BOD for July to November 1983 closely followed the planned BOD value of 13 mg/ℓ, but from December 1983 to May 1984 it jumped above the planned value.

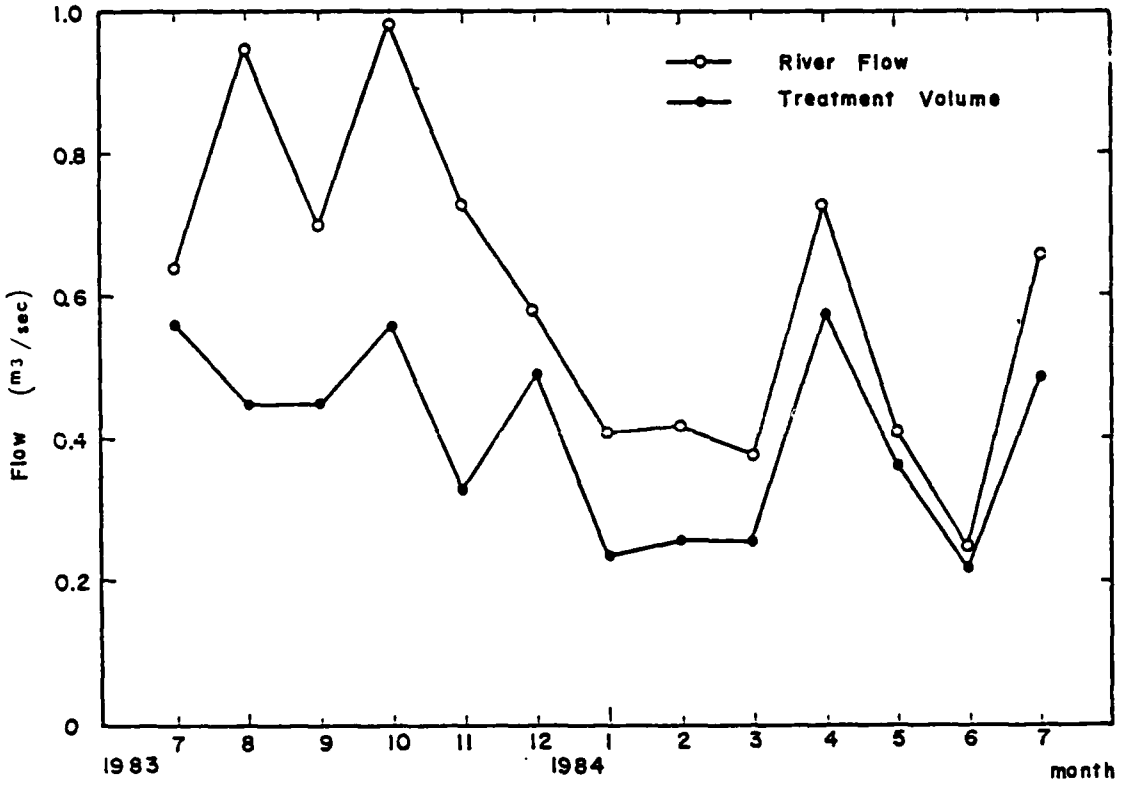


Figure 11. Monthly change of river flow of the Nho River and treatment volume by the GCP plant

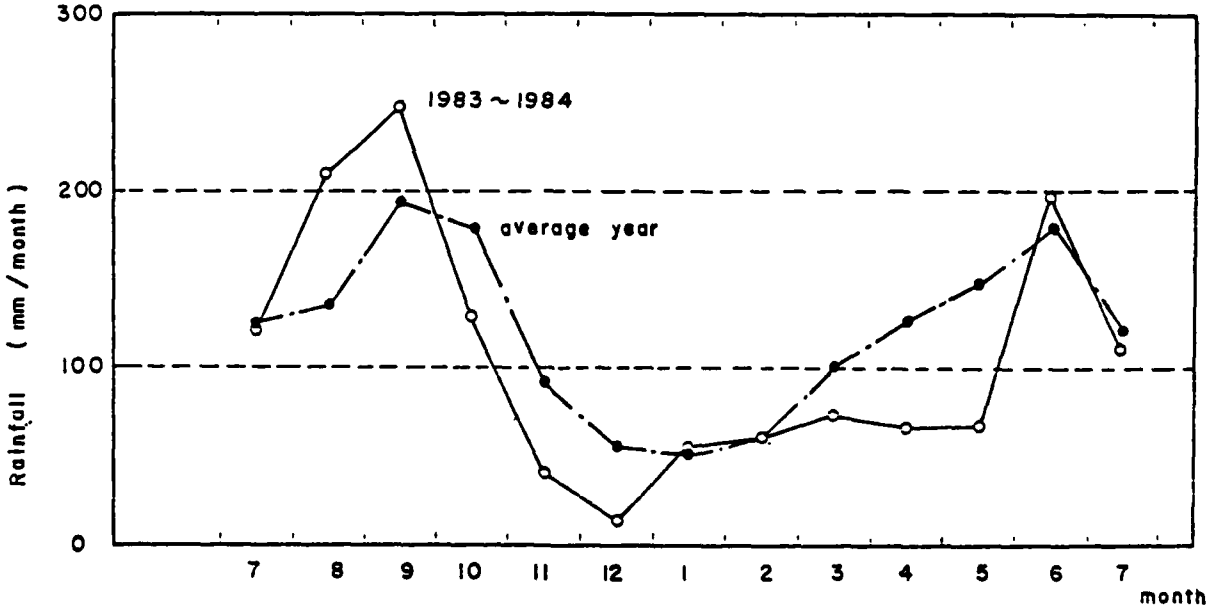


Figure 12. Change of monthly rainfall (at Setagaya Meteorological Station)

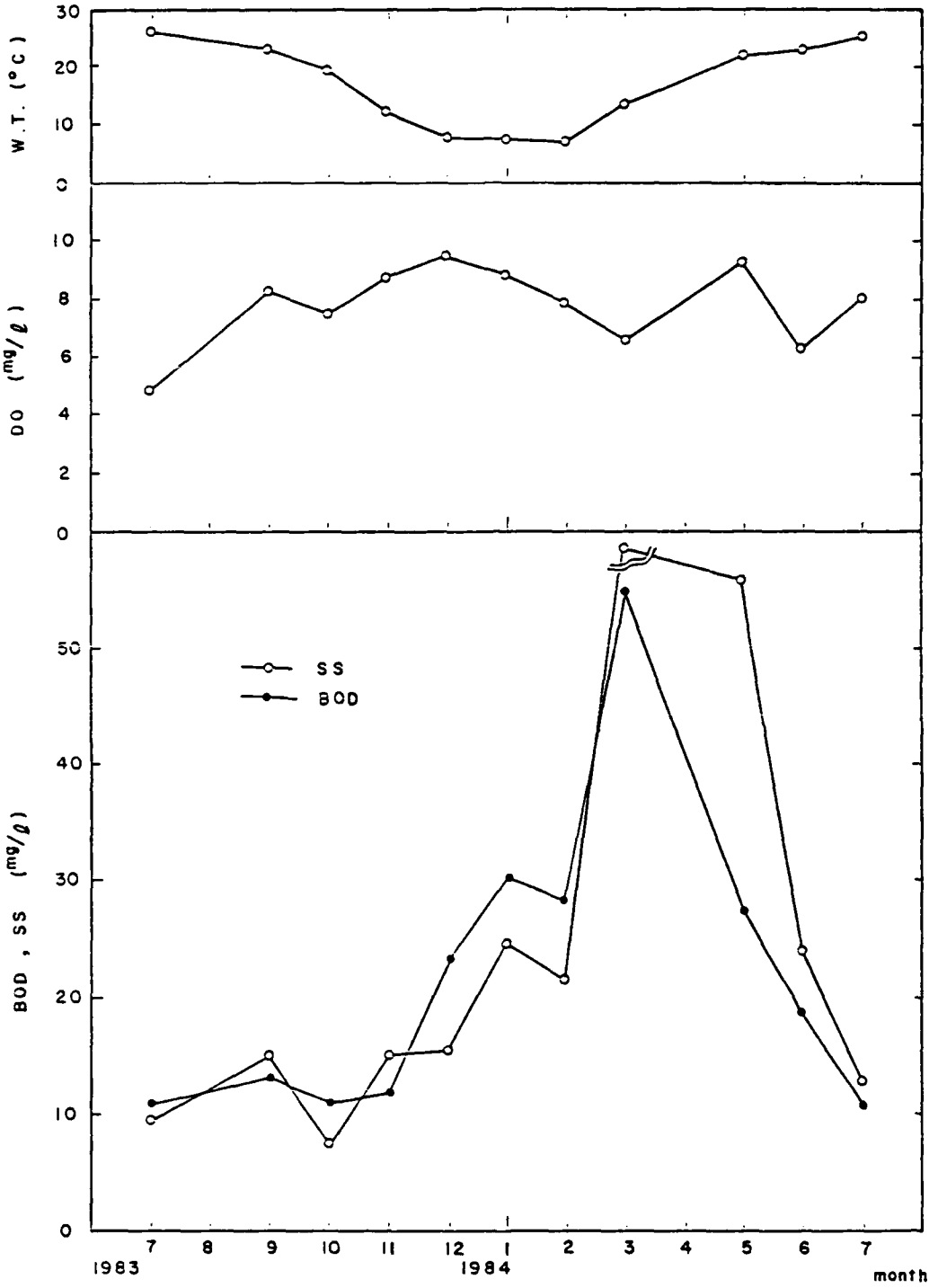


Figure 13. Monthly change of water quality of the Nho River (at the Nogawa-Suido bridge)

The SS values from July to December 1983 were below the planned value of 16 mg/ℓ and rose higher than the 20-mg/ℓ level from January to June 1984. An extraordinary high value recorded in March is thought to have been caused by construction work at the upstream river bed.

Water quality generally deteriorates in winter as the low water temperature makes organisms less active in the river bed and in widely used domestic wastewater purification tanks, resulting in reduced natural purification potential. The main reason for the continued high values in May/June is the low rainfall, about half that of an ordinary year.

Effectiveness of Nho River Purification Facility

The expected effect of purification in the facility is due first to settling in the ponded water and second to settling/adsorption and biochemical oxidation in the gravel beds. Table 3 shows the results of water quality analysis of the samples taken at the river and at the inlet and outlet of the beds, represented by the monthly average values.

Comparing the river water with the inlet water, one can see the effect of impoundment, and the comparison of the inlet water and the outlet water shows the effectiveness of purification by flowing through the gravel beds.

Purification through Impoundment by the Dam

The BOD in the Nho River of 9.3 to 31.5 mg/ℓ with an average of 19.0 mg/ℓ changes to 6.9 to 17.4 mg/ℓ with a 12.0-mg/ℓ average in the inlet. The SS values of 6.0 to 67.1 mg/ℓ with a 20.8-mg/ℓ average in the river is reduced to 4.3 to 25.7 mg/ℓ with 9.7 mg/ℓ average in the inlet.

The water quality of the Nho River itself fluctuates substantially. The recorded averages of BOD and SS in the inlet, 12.0 and 9.7 mg/ℓ, respectively, are rather lower compared to the design values of 13 mg/ℓ for BOD and 16 mg/ℓ for SS, possibly resulting from the stabilizing effect by impoundment.

Purification in Gravel Beds

Water quality at inlet and outlet. Figure 14 shows the monthly change of water temperature, DO, SS, and BOD at the inlet and outlet of the gravel beds.

For water temperature, the highest value of 28.3°C was recorded in August 1983 and the lowest value of 5.9°C in January 1984 during the survey period. There was no noticeable difference between the inlet's and outlet's temperature.

Regarding the monthly average DO, the highest value was 10.8 mg/ℓ, the lowest value was 1.5 mg/ℓ, and the average was 6.1 mg/ℓ. Low DO was experienced in May to July 1984 and, in the worst cases, DO dropped to zero. The extremely low DO results from consumption by the settled sludge at the intake dam's impoundment, under the low flow of Nho River containing high BOD and SS concentration during the scarce rainfall period of those months.

The GCP method requires dissolved oxygen in water since purification is promoted by contact oxidation of the biofilm covering the gravel surface. Low DO will certainly adversely affect maintenance of the purification capability.

TABLE 3. FINDINGS OF WATER QUALITY SURVEY FOR GCP PLANT AT NHO RIVER

Items	Point	July	Aug.	Sept.	Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June	July	Aver.
Water Temp. (°C)	River	26.2	--	23.1	19.5	12.2	7.9	7.9	7.6	13.7	--	21.8	22.9	25.3	17.1
	Inlet	22.2	28.3	23.2	19.6	11.9	6.5	5.9	6.5	9.2	15.5	20.1	22.9	24.1	16.6
	Outlet	21.5	26.3	23.4	18.0	12.6	6.7	6.1	7.1	9.7	14.2	20.6	21.9	23.6	16.3
DO (mg/ℓ)	River	4.9	--	8.3	7.5	8.8	9.5	8.9	7.9	6.7	--	9.3	6.3	8.0	8.1
	Inlet	5.3	10.8	6.7	6.9	7.3	8.8	6.8	5.9	6.4	6.1	4.3	1.5	2.8	6.1
	Outlet	4.8	4.0	5.0	5.2	5.6	6.3	5.3	5.2	4.6	4.3	3.7	2.6	4.5	4.8
SS (mg/ℓ)	River	9.5	--	15.2	7.4	15.4	15.6	24.9	21.4	352.0	--	55.9	24.0	12.8	20.8
	Inlet	8.0	17.7	10.5	9.8	7.8	13.8	17.7	8.6	5.2	9.0	8.4	11.3	5.8	9.7
	Outlet	6.1	5.0	1.2	0.7	2.6	6.1	2.9	2.3	3.7	4.5	3.2	2.2	2.3	3.9
BOD (mg/ℓ)	River	11.1	--	13.3	11.0	12.0	23.5	30.5	28.3	55.0	--	27.3	18.8	10.9	19.0
	Inlet	10.4	11.2	11.2	7.2	8.2	14.9	16.1	14.1	16.6	16.7	12.8	14.0	8.5	12.0
	Outlet	5.3	3.3	3.3	1.3	2.8	4.6	4.9	5.4	8.6	8.3	9.1	7.3	4.2	4.9
D-BOD* (mg/ℓ)	River	5.2	--	6.4	4.5	4.2	9.8	14.2	7.6	11.1	--	6.7	8.9	5.6	7.5
	Inlet	4.8	4.1	3.2	3.5	3.3	8.3	12.0	8.1	9.6	8.6	6.9	6.3	4.7	6.1
	Outlet	2.8	2.0	2.2	1.0	1.'	2.1	3.9	3.5	4.7	5.3	7.0	4.4	3.5	3.3

*D-BOD = dissolved BOD.

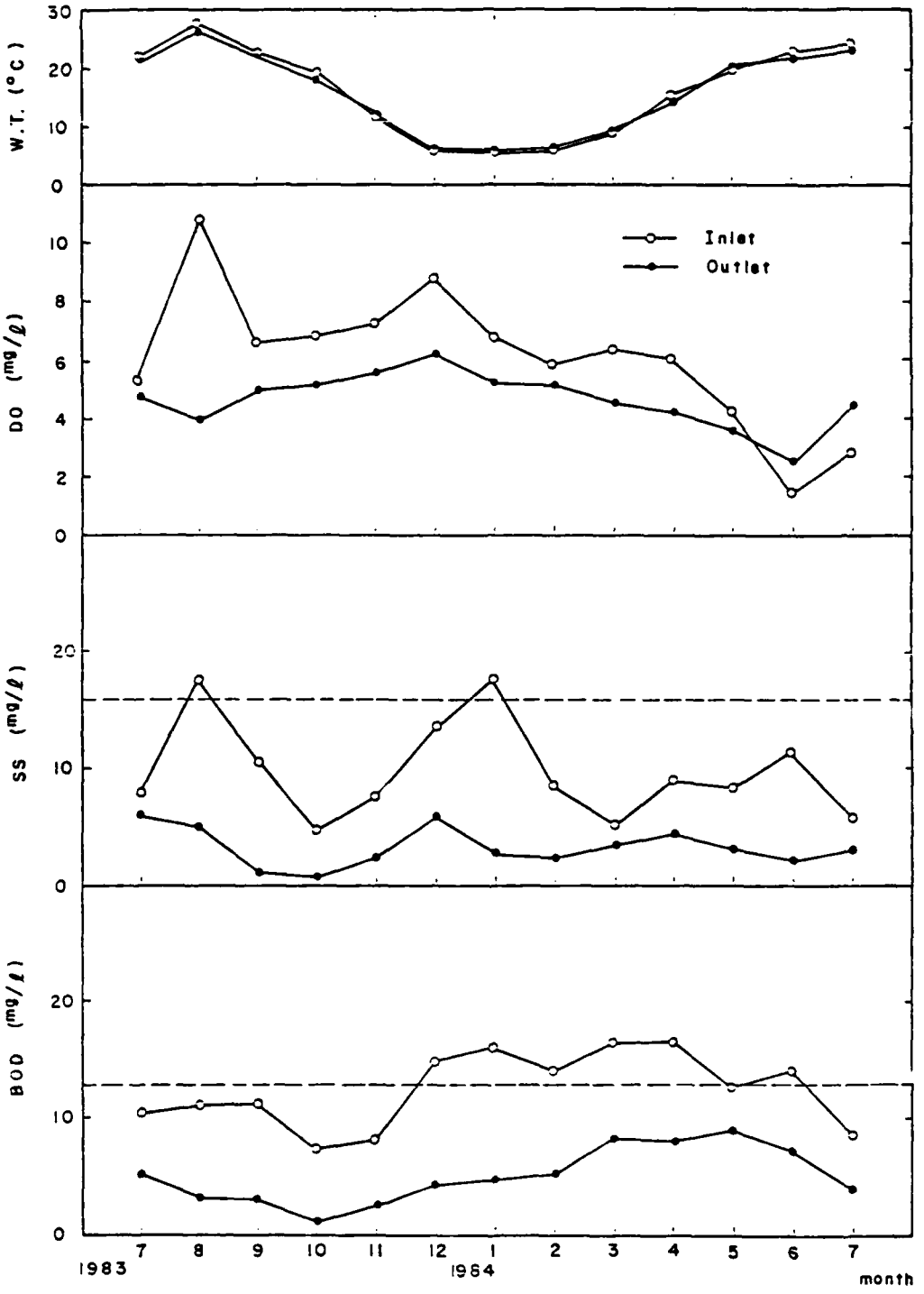


Figure 14. Average monthly change of water quality at inlet and outlet of the gravel beds

The DO in the gravel beds was kept more or less above 1 mg/ℓ in July to December 1983, but dropped to 0 to 0.5 mg/ℓ when BOD was maintained at a high level.

As mentioned before, an outlet sill was installed at the effluent discharge point and, because of it, DO of the discharged effluent was restored to an annual average of 4.8 mg/ℓ.

The inflowing BOD was close to the design level of 13 mg/ℓ, as the highest was 16.7 mg/ℓ, the lowest 7.2, and the annual average 12.0 mg/ℓ. When considering the entire survey period, a tendency was found in that BOD was high in winter and low in summer, as generally seen with rivers in urban areas of the country.

The high BOD recorded from April to June 1984 is thought to be unusual. BOD in treated water averaged 4.9 mg/ℓ for the 13 months. Although higher than the design value of 3.25 mg/ℓ, it was still acceptable. However, from March to June 1984, the value was approximately 7 mg/ℓ. During this period, the effluent was characterized by 55 to 75% of the discharging BOD being dissolved BOD (D-BOD) which needs an ample supply of dissolved oxygen. D-BOD can be removed only through decomposition by aerobic organisms, while BOD contained in suspended matter is removed by settling. Incidentally, DO was consumed completely in the gravel beds. In fact, anaerobic conditions in the beds were so extreme that the water generated the odor of hydrogen sulfide. As the inflowing DO was restored, the odor disappeared.

The inflowing SS showed an annual average of 9.7 mg/ℓ, a value lower than the design value of 16 mg/ℓ. The effluent was approximately 3 mg/ℓ, except in the start-up period.

Effectiveness of purification. The effectiveness of purification was studied for four successive periods after 13 months of operation (Table 4): (a) the initial period, (b) a normal operation period, (c) a low water temperature period, and (d) an abnormal water quality period.

For the initial period, 2 weeks after start-up, the clay and silt attached to the gravel were being washed and the biofilm of organisms was in the development stage. This condition caused low removal effects.

When the start-up period was over, the biofilm was formed and became effective in adsorption and settling (normal operation period, July 1983-November 1983). The inflowing BOD and SS were, on the average in the period, 9.3 and 9.1 mg/ℓ, respectively, lower than the design values, and the effluent BOD and SS were 2.9 and 2.7 mg/ℓ. From these figures, the removal rates for BOD and SS are 68.8 and 70.3%, respectively.

Starting in December 1983 (low water temperature period, December 1983-March 1984), the water temperature dropped below 10°C and the inflowing water quality worsened. The average D-BOD was especially high, 9.5 mg/ℓ. BOD and SS removal rates were 63.8 and 69.7%, respectively.

TABLE 4. REMOVAL RATES OF BOD, D-BOD, AND SS BY GCP PLANT AT NHO RIVER
(JULY 1983-JULY 1984)

	BOD			D-BOD			SS		
	Inlet (mg/ℓ)	Outlet (mg/ℓ)	Removal (%)	Inlet (mg/ℓ)	Outlet (mg/ℓ)	Removal (%)	Inlet (mg/ℓ)	Outlet (mg/ℓ)	Removal (%)
Initial phase (2 weeks)	11.2	6.9	38.4	4.9	2.8	42.9	8.8	8.3	5.6
Jul - Nov	9.3	2.9	68.8	3.8	1.9	50.0	9.1	2.7	70.3
Dec - Mar	15.2	5.5	63.8	9.5	3.4	64.2	12.2	3.7	69.7
Apr - Jul	13.4	7.3	45.5	6.8	5.1	25.0	8.6	3.4	60.5
AVERAGE	12.0	4.9	59.2	6.2	3.3	46.8	9.7	3.2	67.0

April to July 1984 was categorized as the abnormal water quality period. In most years, however, the river water quality improves as the days proceed in this season. However, the Nho River in this season was, as far as water quality was concerned, very similar to the winter season's. Moreover, as the water temperature rose, the consumption of DO increased causing a reduction in inflowing DO. The average BOD and SS values of the inflow were 13.4 and 8.6 mg/ℓ, respectively, and the removal rates were 45.5% for BOD and 60.5% for SS.

BOD in the effluent was remarkably high (7.3 mg/ℓ), and 5.1 mg/ℓ of the BOD was D-BOD in nature. This showed that, because of the scarcity of the inflowing DO, even after all DO had been consumed in the gravel beds, D-BOD was not removed from the water.

In the last days of this period, the rainy season started and the river water quality improved. Operation was restored to normal as the consumption of DO in the impounded area decreased.

Effect of Recovering DO by Outlet Sill

The aerobic organisms oxidize and decompose the pollutants such as organic matter in the gravel beds, consuming existing DO. For the normal life of a fish, greater than 30% saturation ratio of DO (about 3 mg/ℓ) is said to be required. This necessitates the recovery of DO for the treated effluent.

In the facility, reaeration of the effluent by the 1-m-high outlet sill is helping the recovery of DO, as summarized in Table 5 and Figure 15.

TABLE 5. REAERATION EFFECT OF OUTLET SILL

	DO, mg/ℓ			
	Inlet	Inner Beds (8.5 m)	Upper Sill	Down Sill
Average	5.7	1.3	1.4	4.8
Maximum	9.2	4.5	4.7	6.5
Minimum	0.5	0.0	0.0	2.1

The average of 5.7 mg/ℓ DO at the inlet dropped to 0 to 4.5 mg/ℓ with the average of 1.3 mg/ℓ as it was consumed in the beds. The effluent recovered DO to 2.1 to 6.5 mg/ℓ with the average 4.8 mg/ℓ by the outlet sill, and the process proved its worth.

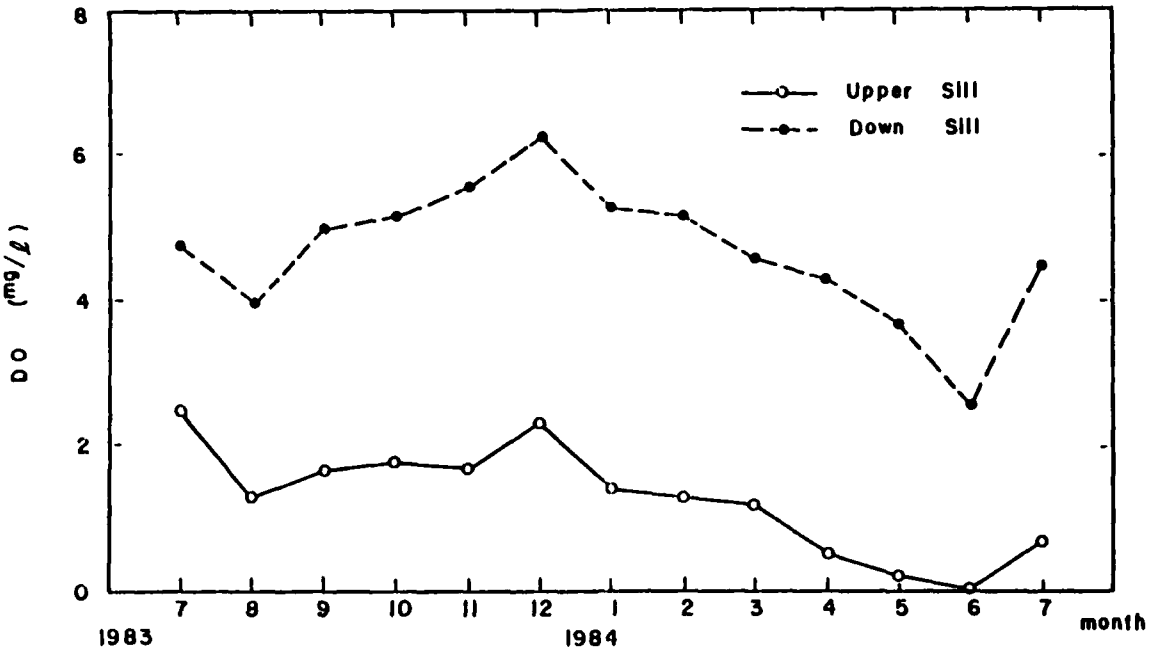


Figure 15. Effect of reaeration by outlet sill

EFFECT OF IMPROVEMENT ON TAMA RIVER MAIN

To determine the facility's effect on the Tama River's water quality, the water quality at both upstream and downstream points of the confluence, and both before and after the operation, were compared. The selected points upstream and downstream were Tama-Suido Bridge and Futako Bridge, respectively.

The difference in BOD at the two points before the operation was 2.6 mg/ℓ as a 5-year average. One year after the operation, the difference decreased to 2.0 mg/ℓ as a year's average. The decrease of 0.6 mg/ℓ equals approximately 9% of the value at Futako Bridge, the downstream point of the confluence. In the original plan 10% was the target; therefore, the facility was as effective as planned.

OUTLINE OF MAINTENANCE WORKS

Inspection on Patrol

The patrolling operators inspected the intake dam, water level meter, and recorder periodically. As the facility uses few mechanical parts, the inspection work was simple and quickly completed. A telemeter system is used to detect the water level at the site and signal the readings to the supervisor at the Keihin Work Office.

Intake Dam

The dam is automatically deflated by sensing the rising water level. After the water level lowers to a certain level the dam is inflated by onsite activation of the operator. Though the operation can be completed in 30 min, 1 to 2 hr is needed before the impounded water starts flowing into the gravel beds and an additional 2 to 3 hr is needed before the effluent resumes discharging.

Screen cleaning

The most troublesome maintenance work of the facility is cleaning the inlet screen. In the worst case, the cleaning work was carried out weekly.

The seasonally changing garbage removed by screening is composed of:

a. PVC bags, waste papers, foam styrene, and empty bottles and cans throughout the year.

b. Fallen twigs and leaves, mostly from October to December.

c. Cut weeds, mostly in July and August.

d. Aquatic plants, mostly from September to March.

As some garbage is thrown into the river with obvious intention, education and public relations programs to obtain the public's cooperation are necessary, not only for maintaining such facilities, but also preserving the river's cleanliness.

To prevent inflowing floating debris, a few tests were conducted. One was installing a skirtlike board which intended to cover 50 cm of depth below the water level. It did not collect much. Another was placing a cagelike screen or a net behind the bar screen. It collected too much, clogged, and hindered the flow. At present, these are used in combination to control inflowing garbage and remove collected debris easily. However, improvement is needed in garbage removal.

After 13 months' experience, the maintenance and prevention needs are:

a. Removing screened garbage.

b. Coping with the accumulated garbage at the dam pond to preserve aesthetic scenery.

c. Preventing foaming at the outlet.

d. Preventing scum and odor generation at the dam pond.

The maintenance work is concerned more with the fringe matters than with the facility itself, which is peculiar to the general concept of maintenance.

LOW INFLOWING DO AND ITS ADVERSE EFFECT

From the end of May to the middle of June 1984, the odor of hydrogen sulfide was generated in the effluent of gravel beds and sulfur bacteria growth was so remarkable that a stretch of river bed 100 m long downstream of the facility was colored white.

A quick investigation revealed that:

a. Due to the slight rainfall, the rubber dam was deflated only infrequently and the deteriorating water quality of the Nho River caused an accumulation of sludge on the bottom of the pond.

b. A sudden water temperature rise at the end of May stimulated DO consumption of the deteriorating river water and sludge in the pond. As a result, the inflowing DO to the beds was nearly zero.

c. The lack of inflowing DO made the gravel beds extremely anaerobic and generated hydrogen sulfide.

d. The effluent containing hydrogen sulfide was discharged to the river and encouraged the growth of sulfur bacteria, changing the color of the river bed. At first the river bed changed to white by the growth of sulfur bacteria under anaerobic condition, and after a few days, regain of outflow DO changed the color of the river bed to black.

e. The shortage of DO obviously caused the anaerobic condition of the gravel beds. If DO is preserved in the inflow, the generation of odor by hydrogen sulfide can be prevented.

f. The BOD and SS percent removal was lowered somewhat, but the purification function was only slightly affected.

These conditions resulted in a study to find a solution and to determine countermeasures including installation of an emergency aerator. In practice, however, the problem was solved naturally. The rainy season started at the end of June and improved the pond's condition and inflowing DO and, as a result, the unpleasant odor and white color of the river bed disappeared.

GENERATION OF SMELL AT THE POND

At the end of May 1984, a large amount of scum appeared on the pond and was gathered by the wind, raising a rotten smell. The direct cause was sudden decomposition of the inflowing organic matters and the sediments in the pond. The smell has stopped now because of the improved flow in Nho River.

The operation practice of the intake dam and means to prevent accumulation of the sediments are being studied at present.

CONCLUSIONS

Since this was the first practical installation of the GCP method in the country, the expectation of finding new facts and learning new experiences has co-existed with an uneasiness of unpredictable happenings during the 13-month

study. However, as anticipated, the facility has proved its expected usefulness without much maintenance. On the other hand, the burden related to cleaning the inlet has been greater than foreseen initially. Future improvement is needed for this aspect.

Also unexpected were the appearance of large amounts of scum in the pond and the generation of hydrogen sulfide odor from the gravel beds. Even though the facility is operating normally at present, unexpected happenings may still occur, partly due to the lack of experience. Hopefully, in the future, improvements and better maintenance will make the facility even more effective.

For the promotion of the GCP method and other methods of river water quality improvement, the reader's comments and advice are welcomed.

Photos 1-4 show the Nho River Facility.

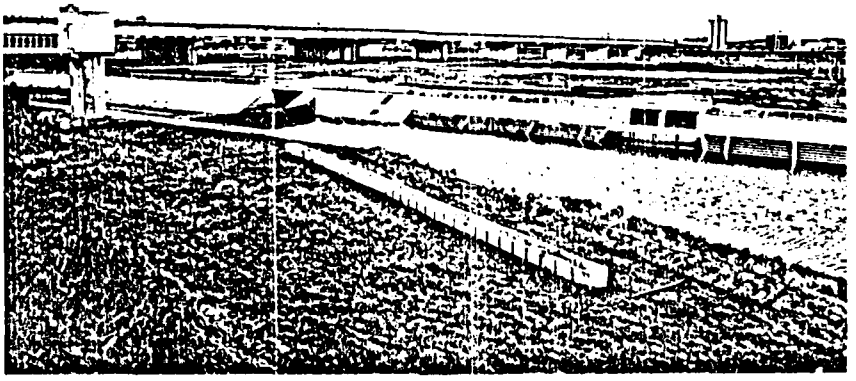


Photo 1. The purification facility at Nho River

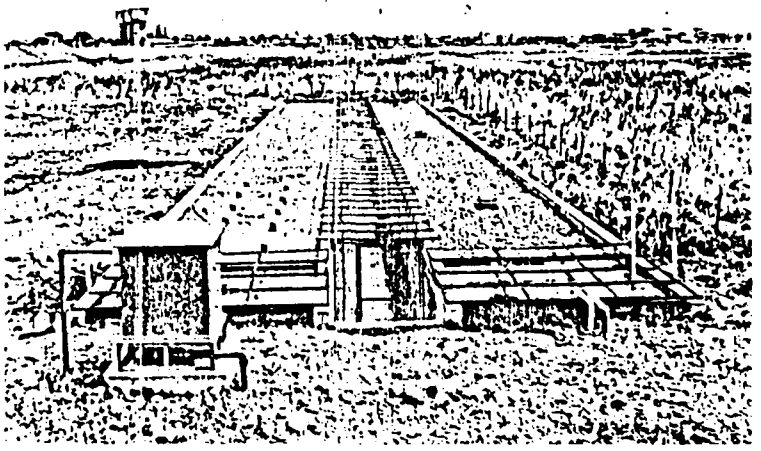


Photo 2. Experimental installation of GCP method

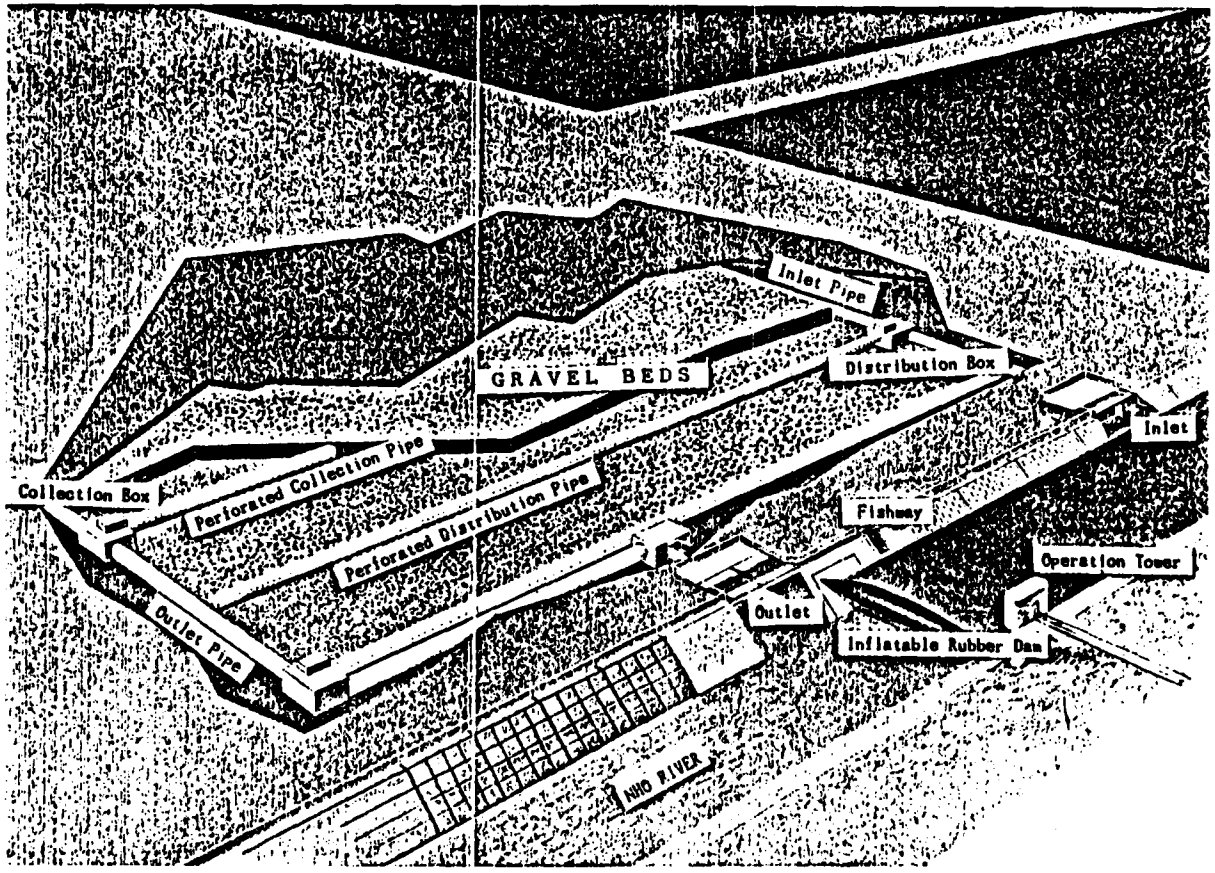


Photo 3. A bird's-eye view of GCP plant at Nho River

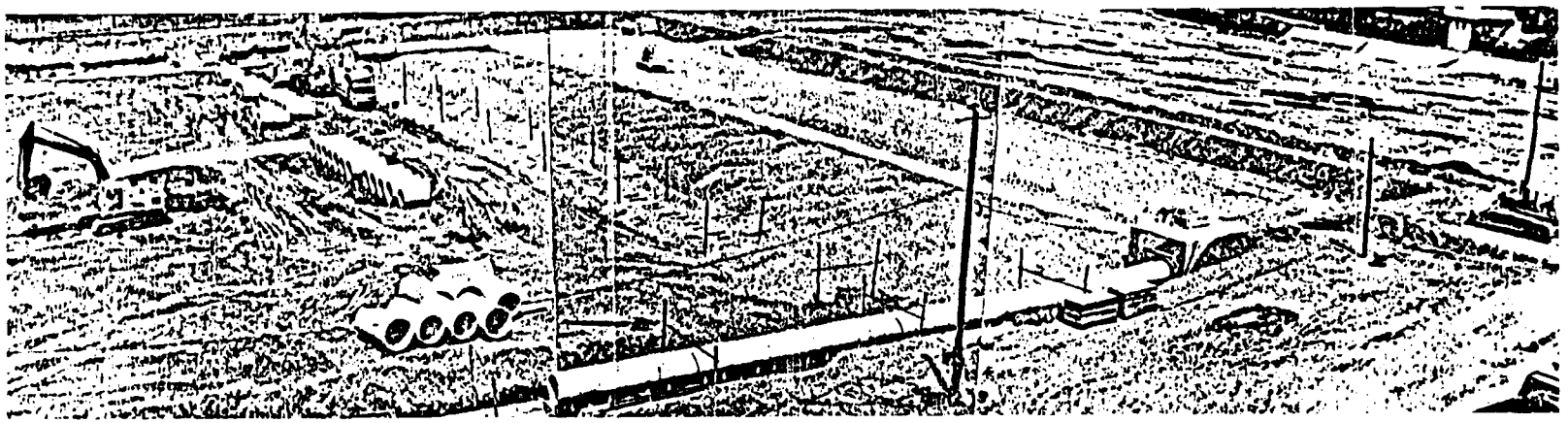


Photo 4. A view of construction of GCP plant at Nho River

PRINCIPLES AND PRACTICES FOR THE OCEAN DISPOSAL OF DREDGED MATERIAL

THE LONDON DUMPING CONVENTION RE-EXAMINED

H. R. Haar, Jr.

Board of Commissioners of the Port of New Orleans
P.O. Box 60046, 2 Canal Street
New Orleans, Louisiana 70160

ABSTRACT

Members of the International Association of Ports and Harbors (IAPH) and the American Association of Port Authorities (AAPA) have, since 1979, been actively engaged with various activities involving the London Dumping Convention. Their continued success in developing international principles and practices for the ocean disposal of contaminated and non-contaminated dredged material will ensure that national and international port views are taken into consideration, and that such views influence policy decisions being made. Many problems have arisen over the years in regard to the London Dumping Convention Treaty requirements and United States implementing laws of this treaty for ports, not only in the United States, but in other areas of the world as well. In response to these problems, it has been necessary for IAPH and AAPA to engage the services of scientific and legal professionals to assist in examining and proposing viable solutions capable of bridging differences between ports' interests worldwide and those of the U.N. treaty organization. The results of this mission--addressing past activities and current issues--are presented in the following sections of this paper which expands on my November 1982 presentation at the Eight U.S./Japan Experts Meeting entitled "Rescuing the Ports - An Update."

INTRODUCTION

Ports throughout the world play a major role in a nation's economy. To remain open to waterborne traffic and commerce, most ports must conduct periodic construction and maintenance dredging activities. Such work requires suitable means of dredged material disposal. The ports of many nations, and, particularly, ports in the United States have experienced difficulty, delays, and increased costs in conducting necessary dredging. Underlying these difficulties are the need to dispose of dredged material into ocean waters and

worldwide concern over the impact of ocean dumping on the environment. Disposal practices of many world ports are determined from the requirements established in the London Dumping Convention (LDC).

In the United States, the environmental movement of the early and mid-seventies began exacting its toll on the ports of the country in the late seventies. This toll is in the form of time delays in obtaining dredging and dredged material disposal permits, denial of permits, delayed capital investment improvements, increased investment and operation and maintenance costs, and low revenues. To counter these impacts and to seek state-of-the-art practices in dredging and dredged material disposal activities, both the American Association of Port Authorities and the International Association of Ports and Harbors established ad hoc dredging committees. Since late 1979 these two organizations separately and jointly have pursued similar goals to obtain political recognition and acquire influence to alter United States legislation and international convention as established in the LDC.

Decisions governing ports and port operations engaged in international trade must be made in the overall public interest and welfare. These decisions must not be excessively hampered by environmental considerations alone. Achieving organizational goals will require continued effort, organizational funding, and exploitation of opportunities to tell the story.

LDC BACKGROUND HISTORY

The London Dumping Convention was enacted in December of 1972 as a result of the growing realization by the nations of the world that the ocean did not have an endless capacity to assimilate man's waste and still regenerate natural resources. It closely paralleled legislation adopted several months earlier (October 1972) in the United States to establish a program for the control of ocean dumping in domestic waters and territorial seas. The LDC was opened for signature on December 29, 1972, with 27 states signing that day. The LDC entered into force on August 30, 1975, when it was ratified by the required number of 15 states. To date, 53 countries have ratified or acceded to the Convention.

The LDC relies heavily upon implementation by member states according to their national authorities. It is implemented in the United States through the Marine Protection, Research, and Sanctuaries Act 1972 (MPRSA) which required only minor amendments to ensure consistency with the LDC. The key provisions of the LDC relate to the prohibitions and permit requirements set forth in Article IV, and its accompaning Annexes, I, II, and III. Article IV (1)(a) prohibits the disposal of certain "blacklisted" substances set forth in Annex I (e.g., mercury, cadmium, organohalogens, petroleum products, and high level radioactive wastes) unless, in most cases, the substances are present as only "trace contaminants" (Annex I, para. 9) or are "rapidly rendered harmless" upon disposal (Annex II, para. 8). Article IV (1)(b) provides that for the "grey list" of substances described in Annex II, "special care" in disposal is required. Annex II substances can only be disposed under a "special permit." Under Article IV (1)(c), all other substances are to be disposed under a general permit. The factors specified in Annex III are to be considered in the issuance of both general and special permits.

The LDC does not specify particular standards and criteria that must be applied in determining the "trace contaminants" and "rapidly rendered harmless" questions under Annex I. That is left, by and large, to the decisions of national authorities. "Interim Guidelines" have been adopted to serve as "guidance" for the interpretation and implementation of paragraphs 8 and 9 of Annex I, but they establish no fixed standards. In the early years of the Convention, the Contracting Parties appear to have construed the "trace contaminant" provision of paragraph 9 as providing a categorical exclusion of dredged material from Annex I. The original draft guidelines for the implementation of paragraphs 8 and 9 adopted at the Second Consultative Meeting in 1977 contained an exclusion for sewage sludge and dredged spoil. They were endorsed in principle by Contracting Parties for further consideration as a priority item. At the Third Consultative Meeting in 1978, Contracting Parties adopted final Interim Guidelines. However, a substantial change was made. Test procedures for determining "trace contaminants" were extended to dredged material. This departed from the original understanding of the dredged material exclusion endorsed in principle by Contracting Parties the previous year.

This action at the Third Consultative Meeting created a serious dilemma for many ports. As a result of growing industrialization and increases in water-borne trade and commerce, many ports find that their harbor sediments may contain substances listed in Annex I in varying degrees. This may present special problems for those ports which rely upon ocean disposal of dredged material for continued operation. The critical determinations are those relating to the "trace contaminants" and "rapidly rendered harmless" questions. If an especially stringent standard is applied, it may trigger the Annex I prohibitions and prevent ocean disposal of the dredged material--even at the expense of continued port operations.

Concerns of this very nature began to arise shortly after adoption of the Interim Guidelines at the Third Consultative Meeting. In 1979, environmental groups in the United States insisted that Annex I barred the ocean disposal of dredged material at the Port of New York-New Jersey and at the Port of Lake Charles, Louisiana, because the dredged material allegedly contained Annex I substances that exceeded "trace contaminant" levels. It was against this background that IAPH and AAPA formed their respective Ad Hoc Dredging Task Forces whose efforts over these past five years have been directed toward achieving a more realistic and practical treatment of dredged material under the London Dumping Convention.

ORGANIZING FOR SURVIVAL

The American Association of Port Authorities

In response to such ever-increasing problems of delays and escalating costs and to continue efforts of those proposing more stringent, if not always applicable, testing procedures, the American Association of Port Authorities (AAPA) established an Ad Hoc Committee on Dredging in June 1979. (The Committee is now known as the Special Dredging Committee.) Its establishment was recognition that the then existing AAPA Committee structures and ensuing resolutions were ineffective in moderating the trend toward increasing environmental restrictions of dredging activities. Early-on, goals were established. These goals included the identification and documentation of those laws,

rules, regulations, agencies, procedures, and agreements which are creating dredging problems. Targeted for study were concerns over mitigation, compensation, endangered species, bioassay test criteria, local costs, permit delays, and interagency agreement. Additionally, the Committee was charged with developing recommended revisions to existing regulations and procedures that would provide needed relief as well as the necessary documentation to support those revisions. Finally, the new Committee was instructed to develop a strategy to be used to achieve adoption and implementation of these revisions and to compile data on key legislators, committees, boards, and administrators to whom these revisions must be officially transmitted.

The International Association of Ports and Harbors

In early 1980, a similar committee to coordinate on the international scene was established by the International Association of Ports and Harbors (IAPH).

At the IAPH Executive Board Meeting in Brisbane, Australia, in April 1980, there was considerable discussion of the problems that the United States' ports had been encountering in their attempts in recent years to dredge their facilities. The Board recognized that those difficulties stemmed in a large measure from the United States being party to the LDC. Further, the Board agreed that it would be to the benefit of the IAPH membership to develop a better understanding of port dredging practices and the relationship of those practices to the terms of the London Dumping Convention.

The missions of the International Ad Hoc Dredging Committee are:

1. To review, report, advise, and submit recommendation on major matters relating to seaport and inland port dredging and dredging equipment.

2. To meet with and coordinate with the London Dumping Convention and the Inter-Governmental Maritime Consultative Organization (IMO), the latter being the organization designated to serve as the Secretariat to the LDC.

3. To develop a program on disposal of dredged material problem areas for inland ports.

4. To publish an inventory of dredging equipment owned by dredging companies worldwide, including a special section on new, innovative equipment.

5. To collect and publish information on state-of-the-art techniques.

6. To publish an information brochure on sources of information and assistance on dredging techniques and types of equipment best suited for given situations.

MOVING TOWARDS THE INTENDED GOALS

These two committees, the Ad Hoc Dredging Committee of the IAPH and the Special Dredging Committee of the AAPA, have pushed forward in their effort to resolve regulatory problems confronting the industry and to seek, as well,

The LDC does not specify particular standards and criteria that must be applied in determining the "trace contaminants" and "rapidly rendered harmless" questions under Annex I. That is left, by and large, to the decisions of national authorities. "Interim Guidelines" have been adopted to serve as "guidance" for the interpretation and implementation of paragraphs 8 and 9 of Annex I, but they establish no fixed standards. In the early years of the Convention, the Contracting Parties appear to have construed the "trace contaminant" provision of paragraph 9 as providing a categorical exclusion of dredged material from Annex I. The original draft guidelines for the implementation of paragraphs 8 and 9 adopted at the Second Consultative Meeting in 1977 contained an exclusion for sewage sludge and dredged spoil. They were endorsed in principle by Contracting Parties for further consideration as a priority item. At the Third Consultative Meeting in 1978, Contracting Parties adopted final Interim Guidelines. However, a substantial change was made. Test procedures for determining "trace contaminants" were extended to dredged material. This departed from the original understanding of the dredged material exclusion endorsed in principle by Contracting Parties the previous year.

This action at the Third Consultative Meeting created a serious dilemma for many ports. As a result of growing industrialization and increases in water-borne trade and commerce, many ports find that their harbor sediments may contain substances listed in Annex I in varying degrees. This may present special problems for those ports which rely upon ocean disposal of dredged material for continued operation. The critical determinations are those relating to the "trace contaminants" and "rapidly rendered harmless" questions. If an especially stringent standard is applied, it may trigger the Annex I prohibitions and prevent ocean disposal of the dredged material--even at the expense of continued port operations.

Concerns of this very nature began to arise shortly after adoption of the Interim Guidelines at the Third Consultative Meeting. In 1979, environmental groups in the United States insisted that Annex I barred the ocean disposal of dredged material at the Port of New York-New Jersey and at the Port of Lake Charles, Louisiana, because the dredged material allegedly contained Annex I substances that exceeded "trace contaminant" levels. It was against this background that IAPH and AAPA formed their respective Ad Hoc Dredging Task Forces whose efforts over these past five years have been directed toward achieving a more realistic and practical treatment of dredged material under the London Dumping Convention.

ORGANIZING FOR SURVIVAL

The American Association of Port Authorities

In response to such ever-increasing problems of delays and escalating costs and to continue efforts of those proposing more stringent, if not always applicable, testing procedures, the American Association of Port Authorities (AAPA) established an Ad Hoc Committee on Dredging in June 1979. (The Committee is now known as the Special Dredging Committee.) Its establishment was recognition that the then existing AAPA Committee structures and ensuing resolutions were ineffective in moderating the trend toward increasing environmental restrictions of dredging activities. Early-on, goals were established. These goals included the identification and documentation of those laws,

rules, regulations, agencies, procedures, and agreements which are creating dredging problems. Targeted for study were concerns over mitigation, compensation, endangered species, bioassay test criteria, local costs, permit delays, and interagency agreement. Additionally, the Committee was charged with developing recommended revisions to existing regulations and procedures that would provide needed relief as well as the necessary documentation to support those revisions. Finally, the new Committee was instructed to develop a strategy to be used to achieve adoption and implementation of these revisions and to compile data on key legislators, committees, boards, and administrators to whom these revisions must be officially transmitted.

The International Association of Ports and Harbors

In early 1980, a similar committee to coordinate on the international scene was established by the International Association of Ports and Harbors (IAPH).

At the IAPH Executive Board Meeting in Brisbane, Australia, in April 1980, there was considerable discussion of the problems that the United States' ports had been encountering in their attempts in recent years to dredge their facilities. The Board recognized that those difficulties stemmed in a large measure from the United States being party to the LDC. Further, the Board agreed that it would be to the benefit of the IAPH membership to develop a better understanding of port dredging practices and the relationship of those practices to the terms of the London Dumping Convention.

The missions of the International Ad Hoc Dredging Committee are:

1. To review, report, advise, and submit recommendation on major matters relating to seaport and inland port dredging and dredging equipment.

2. To meet with and coordinate with the London Dumping Convention and the Inter-Governmental Maritime Consultative Organization (IMO), the latter being the organization designated to serve as the Secretariat to the LDC.

3. To develop a program on disposal of dredged material problem areas for inland ports.

4. To publish an inventory of dredging equipment owned by dredging companies worldwide, including a special section on new, innovative equipment.

5. To collect and publish information on state-of-the-art techniques.

6. To publish an information brochure on sources of information and assistance on dredging techniques and types of equipment best suited for given situations.

MOVING TOWARDS THE INTENDED GOALS

These two committees, the Ad Hoc Dredging Committee of the IAPH and the Special Dredging Committee of the AAPA, have pushed forward in their effort to resolve regulatory problems confronting the industry and to seek, as well,

solutions that are environmentally and economically sound. The following is a brief chronology of IAPH/AAPA efforts in seeking to achieve a more realistic and meaningful treatment of dredged material under the LDC.

1. In the fall of 1979--shortly after the Annex I prohibitions were asserted for the first time to halt essential port operations in the United States--AAPA sought and obtained representation on the EPA-chaired U.S. Ocean Dumping Advisory Committee, which is responsible for formulating U.S. positions under the LDC. It was the hope of AAPA to be able to influence U.S. positions to ensure that they would have proper regard for port concerns. AAPA also requested a place on the U.S. delegation to the Fourth Consultative Meeting of the LDC later that year, but was refused--even though a representative of a major environmental group adverse to ocean dumping of dredged material was included on the delegation.

2. In 1980 IAPH made use, for the first time, of its "observer" status under the LDC to attend the Fifth Consultative Meeting of Contracting Parties. In its initial attendance at the meeting, IAPH emphasized the drastic effect upon port operations that could occur if Annex I were applied to halt needed dredged material disposal. IAPH also suggested the possibility of using certain "special care" techniques (such as clean material capping) for the ocean disposal of highly polluted dredged material. The IAPH submission was well received, and IAPH was asked to make a more detailed presentation at the next meeting of the Scientific Group.

3. In May of 1981, IAPH attended the Scientific Group meeting in Hailfax, Nova Scotia, and presented a detailed paper on the use of "special care" measures. This focused scientific attention upon these techniques and has resulted in their growing study and use since that time.

4. In October 1981, IAPH attended the Sixth Consultative Meeting in London. IAPH reported upon additional experience with "special care" techniques and also suggested that the "emergency" provisions of the LDC should apply when a port had no alternative means of disposing of polluted dredged material other than dumping at sea. Although this construction of the "emergency" provisions was not approved, Contracting Parties did express the view that such situations might appropriately be handled through the use of "special care" measures.

5. In September 1982, IAPH attended the meeting of the Scientific Group in Paris, France. The Paris meeting was one of extreme significance. In addition to the IAPH submission on "special care," great interest was expressed in developing additional "criteria" for classifying substances to Annexes I & II, with emphasis upon the use of numerical standards. This renewed consideration of the basis for "classification" presented a major opportunity for IAPH to make a new case for a separate treatment of dredged material. IAPH seized this opportunity and offered to prepare a scientific paper addressing the special features of dredged material in the context of classification criteria.

6. In February of 1983, IAPH attended the Seventh Consultative Meeting in London. In addition to presenting port positions on the use of "special care" techniques and on the preparation of the proposed paper for the development of "classification criteria," deep concern was expressed over a proposal

by two small Pacific islands, Kiribati and Naura, to ban all disposal of radioactive material in the oceans in terms so sweeping that it could be construed to apply to naturally occurring radioactive isotopes which are present in all harbor sediments. Other nations concurred in the IAPH concerns, and no binding action on the radioactivity issue was taken at the Seventh Consultative meeting.

7. At the Seventh Meeting of the Scientific Group in October 1983, in London, IAPH submitted two technical papers relating to dredged material. The first was entitled "A Special Report on Application of Classification Criteria to Dredged Material with Emphasis Upon Petroleum Hydrocarbons and with Additional Consideration of Lead in Dredged Material." In it, IAPH described the unique characteristics of marine sediments that serve to tightly "bind" and "hold" Annex I substances so that they are essentially "unavailable" to the marine biota when disposed at sea. IAPH demonstrated that these recognized mitigative features allowed the disposal of dredged material containing Annex I substances without significant risk to the marine environment. In the second paper--entitled "An Updating of Special Care Measures for Disposal of Polluted Dredge Material in the Marine Environment"--IAPH focused upon the use of a particular "special care" measure, "clean material capping." IAPH reported upon its effectiveness in reducing the disposal of contaminated dredged material to a low risk status. The Scientific Group agreed that an interim evaluation had shown that "capping" is technically and scientifically feasible and is a useful mitigative measure that shows promise as a long-term management strategy for the ocean disposal of contaminated dredged material.

8. In December of 1983, IAPH attended a meeting of the Ad Hoc Group of Legal Experts that was convened to consider legal issues relating to proposals for the sub-seabed disposal of high level radioactive wastes. At this highly emotional meeting, the Nordic countries introduced a resolution to ban all seabed disposal of high level radioactive wastes and all other wastes listed in Annex I as well. Because of analogies that had been drawn between "capping" and "seabed disposal," this presented a direct threat to the continued use of capping as a means of disposing contaminated dredged material at sea. IAPH expressed its strong opposition to the Nordic resolution and the need to exclude dredged material from its terms.

9. At the Eighth Consultative Meeting in February 1984, IAPH presented a major new recommendation. Based upon the conclusions reached in its classification criteria paper, IAPH expressed the view that dredged material containing Annex I substances should not be subject to the strict prohibitions of Annex I but should be regulated under the "special permit" provisions of Annex II. IAPH examined the dramatic advances in scientific knowledge of dredged material since the original drafting of the Convention in 1972 and emphasized the unique properties of marine sediments that mitigate the effects of Annex I substances. The meeting took note of the IAPH presentation and agreed that the IAPH recommendations should be considered by an intersessional working group that was established to continue the development of criteria for the classification, addition, and deletion of substances to the Annexes.

The Eighth Meeting also took note of the findings of the Scientific Group that an interim evaluation of "capping" had demonstrated that it is a feasible and effective technique and members endorsed the continuing use of "capping" on a research basis. IAPH also expressed its support for the view expressed

by the French delegation that many of the Annex III guidelines (which include factors to be considered in granting general and special permits) cannot be applied to dredged material, which properly requires separate guidelines. IAPH expressed its support for this proposal and offered, subject to receiving authorization and funding from its sources, to assist in such work. Finally, IAPH renewed its objection to the Nordic resolution (which was re-introduced) that proposed an absolute ban on seabed disposal of high level radioactive wastes and all "other wastes" listed in Annex I. IAPH achieved a major success in having the language relating to "other wastes" deleted from the resolution. Dredged material was thereby excluded from the intense debate regarding the sub-seabed disposal of high level radioactive wastes.

A CALL FOR EVALUATION--THE ISSUES UNDER SCRUTINY

We are determined and are currently continuing our productive and timely efforts to achieve our goals. In the dozen or so years that have passed since the drafting of the Convention, Contracting Parties have continued to express an interest in reviewing and evaluating the bases for the several criteria originally devised for allocating substances to Annexes I and II. Thus, with new member nations involved in plenary deliberations, Contracting Parties wish now to be assured that the extant criteria are technically sound and compatible with current knowledge. Hence, they have called for a penetrating review of the existing criteria. They wish to receive also from the Scientific Group recommendations as to appropriate additions to the criteria if and when new knowledge or concepts justify such additions. Responsibility for conducting the formal study of these issues has been assigned to a select working group that will report its findings to the Scientific Group for definitive discussion at its meeting in February 1985.

As requested at the Eighth Consultative Meeting and in preparation for the July 1985 Scientific Group meeting, the Ad Hoc Expert Group on the concepts and purpose underlying the annexes to the LDC met in London in July 1984. Thirteen international scientists were in attendance, including Dr. Willis E. Pequegnat, world renown oceanographer assisting IMO as a technical consultant. The purpose of that meeting was to review current papers addressing the issues of classification, addition, and deletion of substances to the Convention's Annexes and provide the basis by which to establish the direction and objectives of the proposed paper requested by the Contracting Parties.

This new document, "Some Suggested New Annex Allocation Criteria of LDC Related to the Toxicant Binding Properties of Dredged Material," has been prepared by Dr. Pequegnat, and is currently undergoing review prior to its official release at that March 1985 IAPH Scientific Group Meeting. The paper attempts to respond substantively to both requests of the Contracting Parties, i.e., to evaluate present criteria and delineate additional criteria or, at the very least, suggest how those already under consideration should be employed. Four very different but interrelated issues are dealt with, to wit:

1. Evaluation of the present list of substances in the Annexes (does not deal with radioactive materials or with persistent plastics).

2. Evaluation of the present three allocation criteria.

3. Justification for adding particular new allocation criteria to the present list.

4. Establishing the relationship of the provisions of the Annex III guidelines to the above three considerations.

Although the four issues stated above constitute implicit objectives of this study, it is appropriate to be explicit as to the topics which received most emphasis. There is a strong conviction that it is not scientifically sound to expect that criteria for allocation of substances to the Annexes can be made to apply in an equal manner to a given toxicant when it is disposed into the ocean in different forms of carriers. This point is recognized in Annex III and should be given careful consideration when one is required to hand down a regulatory decision from the provisions of Annex I. When chemical form is linked with the concept of bioavailability, we achieve a rational solution to the nagging problem of how to make full use of the ocean's capabilities and still protect its living resources and prevent hazards to human health. To achieve this goal, Dr. Pequegnat examined:

1. The concept of toxicant bioavailability and demonstrated how its magnitude is dependent in part upon the geochemical properties of the carrier or milieu in which it is measured. Here the emphasis is upon certain types of marine sediments.

2. In considerable detail some of the geochemical properties of the above marine sediments that account for their ability under certain conditions to provide the first line of defense against the harmful effects that Annex I toxicants would otherwise have upon marine life.

3. Explained in reasonably understandable terms the functional (physiological) mechanisms that permit marine life to erect a second line of defense by detoxifying or otherwise reducing the hazards of absorbed organic and inorganic toxicants.

4. Why the concept of biopersistence may be more realistic than environmental persistence as it relates to toxicity and the food chain.

5. How these geochemical properties of the carrier can modify the regulatory application of the terms toxicity and bioaccumulation, recognizing that pharmacological toxicity may be absolute but ecological toxicity is relative.

SUMMARY OF STUDY'S FINDINGS

"Some Suggested New Annex Allocation Criteria of LDC Related to the Toxicant Binding Properties of Dredged Materials" is an ambitious study with landmark implications. The major conclusion drawn from Dr. Pequegnat's study indicates that in many instances the disposal of toxic dredged waste into the ocean should be the preferred alternative over any reasonable type of upland disposal. This conclusion is in conflict with related international policy proposals currently being considered and discussed by the LDC. Specific findings of that study which support this conclusion are as follows:

1. It is not scientifically or pragmatically sound to expect that a criterion for allocation of substances to the Annexes should routinely apply in an equal manner to a given toxicant when it is disposed into the ocean as a component of different wastes. Unfortunately, the point is valid that the Convention fails to make a definitive separation between substances and wastes, either in Article IV or the Annexes. Yet this is a critical issue to some governmental and industrial groups.

2. A waste per the Convention must be defined as the toxicant or toxicants of concern (e.g. an organochlorine) and the "carrier" (sewage sludge, dredged waste, etc.) in which it is disposed. Thus, carrier plus substances listed in the Annexes comprise "the waste." As we shall see, then, it is the waste, the ecological entity, that must be tested for potential impacts.

3. Thus, the nature of the carrier plus the chemical species involved generally determine whether or not a toxicant will be available to the biota. Therefore, bioavailability and physicochemical form of the carrier are perhaps the two most important considerations in dealing with disposal criteria. In general the carrier determines availability. So if a toxicant is not bioavailable, the other criteria should not apply.

4. Marine sediments, especially of estuarine origin, characteristically contain clay minerals, such as montmorillonite and vermiculite, humic acids, and sulfides, all of which are effective at reducing the availability of toxic metals or synthetic organic compounds to the biota.

5. When because of the composition of the carrier a substance does not have bioavailability, there can be no absorption into the organisms' tissues. This physicochemical suite of carrier properties comprises the "first line of defense" against environmental impacts.

6. Given that a toxicant is not bioavailable to marine plants or metazoan animals, it follows that attempts to apply the extent three criteria (toxicity, bioaccumulation, and persistence) are misguided and confusing. Realistically the waste has satisfied the intent of paragraphs 8 and 9 of Annex I.

7. Even when trace amounts (or more) of a toxicant in a waste are bioavailable and have actually been absorbed they will evoke the second line of defense, which is physiological. This defense mechanism protects the metabolic enzyme centers (ENZ). These enzymes control all of the cellular functions.

8. When toxic metals are absorbed, they are tied up by sulfur-bearing, low molecular weight proteins called metallothioneins, which are synthesized by liver and kidney cells. So long as the Metallothionein Pool (MT) is not overloaded, the ENZ pool is protected. Evidence thus far researched indicates that only a small percentage of the metallothionein proteins are tied up in natural populations of marine animals, even in polluted regions.

9. In similar manner, organic toxicants are bound in one or both of two places. If they are soluble in lipids (e.g. PCBs), some are stored in fat tissues, production of which appears to be stimulated by the uptake of these organics as well as by some metals like cadmium. Synthetics are also

sequestered by the glutathione pool. Both of these mechanisms are also part of the second line of defense. In addition, it must be pointed out that organisms can break down (metabolize) some toxicants. Frequently, however, in the case of synthetic organics, the metabolities may be more toxic than the parent compound (e.g. DDE from DDT).

10. It is for the above reasons that investigators have rarely observed serious impacts upon the marine biota, especially the benthic component, even in polluted sediments containing Annex I toxicants. The few exceptions are acute cases where the loading of toxicant into the environment was not only massive but also occurred in a short interval of time.

11. Clay minerals micelles like those of montmorillonite and vermiculite have huge external and internal surface areas (800 or more square meters per one gram dry weight of material) available for binding toxication of metals. Other binding agents are sulfide and hydrous oxides, and particularly humus.

12. Humus micelles are also efficient binders of various toxicants, particularly the synthetic organic compounds.

13. Dredged materials that carry significant loads of toxicants will more frequently than not possess substantial percentages of clay minerals, humic acids, and other sequestering agents. Moreover, this binding capability of the material is measurable.

14. It is proposed that when a dredged material, as carrier, contains toxic substances, that is, when the "waste" has in it Annex I toxicants, effectively bound by the above agents as evidenced by predetermined percentages of clay minerals, humics, etc., it should be considered a "naturally acceptable waste" that is eligible for ocean disposal under the permit and care procedure of Annex II.

15. It is even possible that wastes containing Annex I substances and deemed unacceptable as they are for safe ocean disposal can be upgraded for such disposal by the addition of calculated amounts of effective sequestering agents, depending upon the particular toxicants involved.

IN CONCLUSION...

These IAPH/AAPA efforts have been undertaken against a history of port "uninvolvement" in ocean dumping matters. They have come at a time when pressure for stricter environmental controls has not abated, but remains on the increase. These problems are still not yet finally settled. New issues are constantly emerging. One need only look at the Agenda of next year's Scientific Group meeting in London, and at the Agenda of the 9th Consultative Meeting next year to appreciate how much consideration is being given to matters that affect port interests. These include continued evaluation of "special care" measures mentioned (along with the attendant legal questions that have been raised as to their routine use), further inquiry into "trace contaminant" levels, further development of new classification criteria, consideration of the transfer of new substances (such as lead) to Annex I, and the question of radioactive waste disposal. Decisions will hopefully be made upon these issues within the next year. On the domestic front, efforts are continuing to include greater environmental restrictions upon dredged material in the MPRSA;

and the EPA is still planning to publish a proposed revision to its ocean dumping criteria--the first revision since the environmental changes adopted in 1977.

In the midst of this legal and regulatory activity, what is significant is that this time ports are not absent. They are not uninvolved. Port views are being expressed. The port voice is being heard. The challenge is here. It cannot be avoided.

We have been most appreciative of Dr. Sato's support from IAPH Headquarters in Tokyo and know that we can also count on the Japanese government's and ports' support on these vital issues when they are debated at IMO in London.

PROGRESSIVE TRENCHING USING A SCRAPER DIGGER

T. Inoue

Japan Sediments Management
Association

ABSTRACT

The progressive trenching method developed in the United States was introduced by Mr. Calhoun in this meeting. After his presentation the interest in it rose in our country. In the management of a disposal area in Tsu-Matsuzaka Harbor the progressive trenching method was first applied in our country to promote the dewatering of dredged material. In this case the digging of the trench was performed by a floating scraper. This paper reports how progressive trenching using a scraper digger was carried out in Japan.

BACKGROUND OF MANAGEMENT

In order to restore the estuaries of Shitomo River the sediment was removed in 1980-1982 (Figure 1). The removed quantity of sediment was

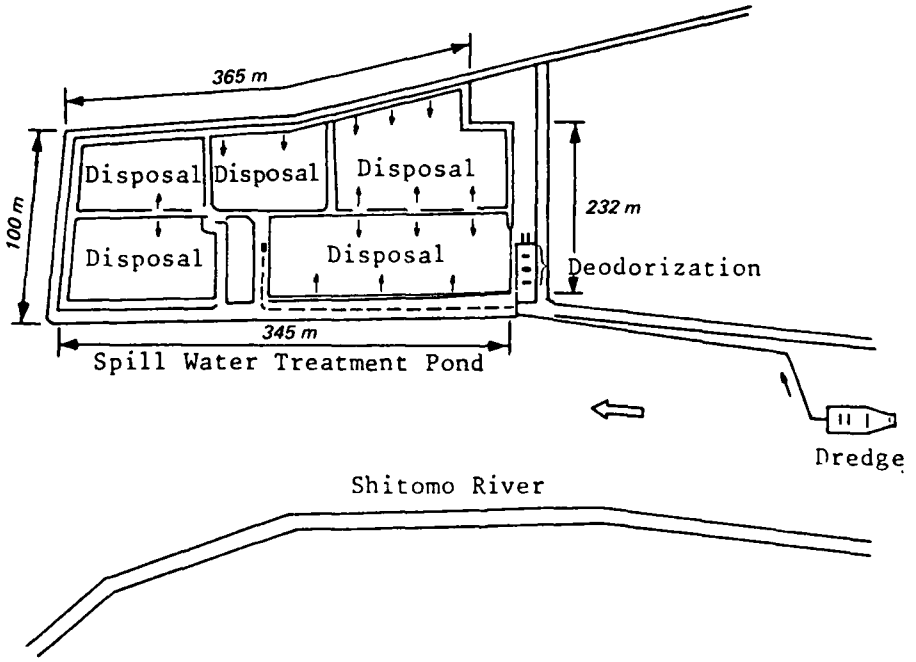


Figure 1. Dredging at Shitomo River estuaries

322,200 m^3 and its removal depth averaged 4 m. The dredged material was transported hydraulically to a spill water treatment pond through a deodorization plant. At the disposal area the progressive trenching method was applied using a scraper digger to promote dewatering of the dredged material.

SCRAPER DIGGER

In the application of the trenching method, the most important factor is how to dig the trenches. In a large disposal area a regular trenching machine may be used (Figure 2). However, in our case, because of the small-scale disposals, it was necessary to contrive a cheaper trenching method. From this requirement, a floating scraper digger was devised. The scraper is shaped like a boat with its head bow-shaped to reduce soil resistance while advancing. The weight is controlled by ballast according to the required trench depth, considering the buoyancy acting on the float and contact pressure of the surface. Figure 3 shows the appearance of the scrapper.

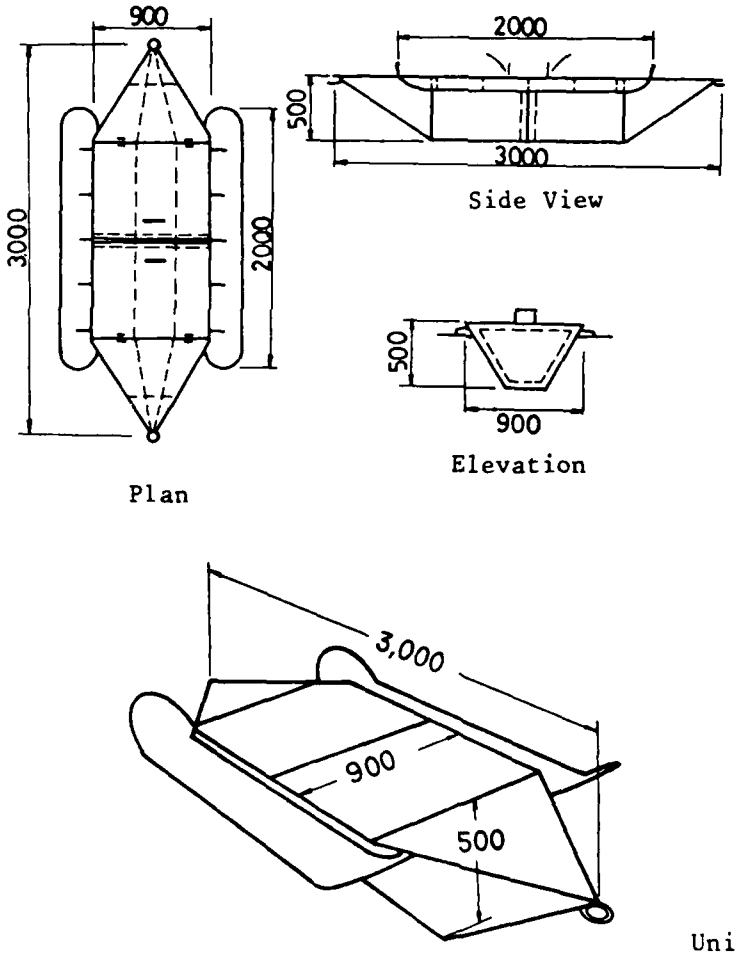


Figure 2. Scraper digger

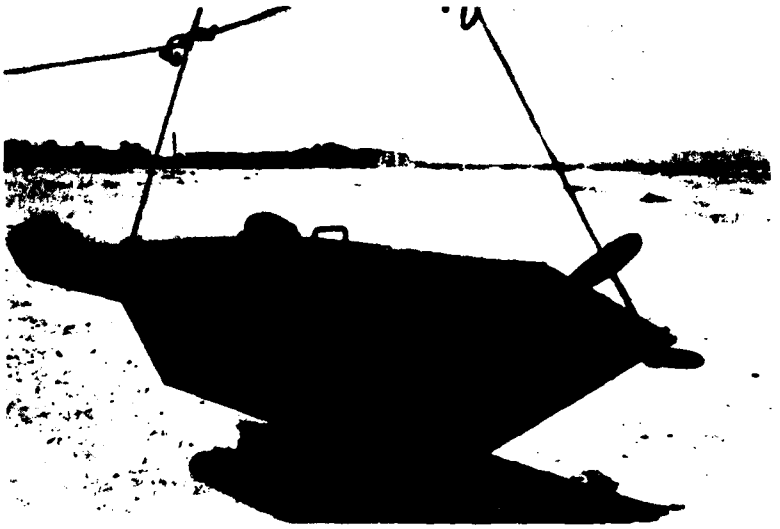


Figure 3. Appearance of scraper

The bottom is made of a 30-cm plate to obtain a good finished cut. The scraper is equipped with wings at both sides for stability. Trench excavation is conducted such that two crawler cranes standing on facing banks draw the scraper digger in succession (Figure 4).

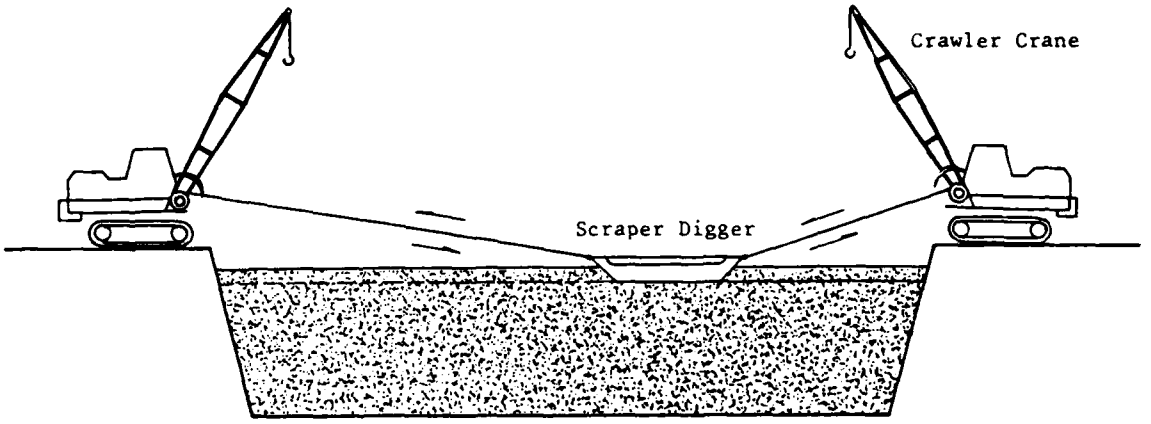


Figure 4. Excavation of trenches

TRENCHING WORK

The basic concept of progressive trenching is described by Mr. Calhoun in his paper. Therefore, it need not be repeated here, but it is depicted diagrammatically in Figure 5.

The inside walls of the pond were covered by vinyl sheets (4 mm) to prevent water permeation from dredged material to the surrounding area. Therefore, water contained in the dredged material had to be dewatered or solidified by solidificants. Moreover, since the project was conducted during the

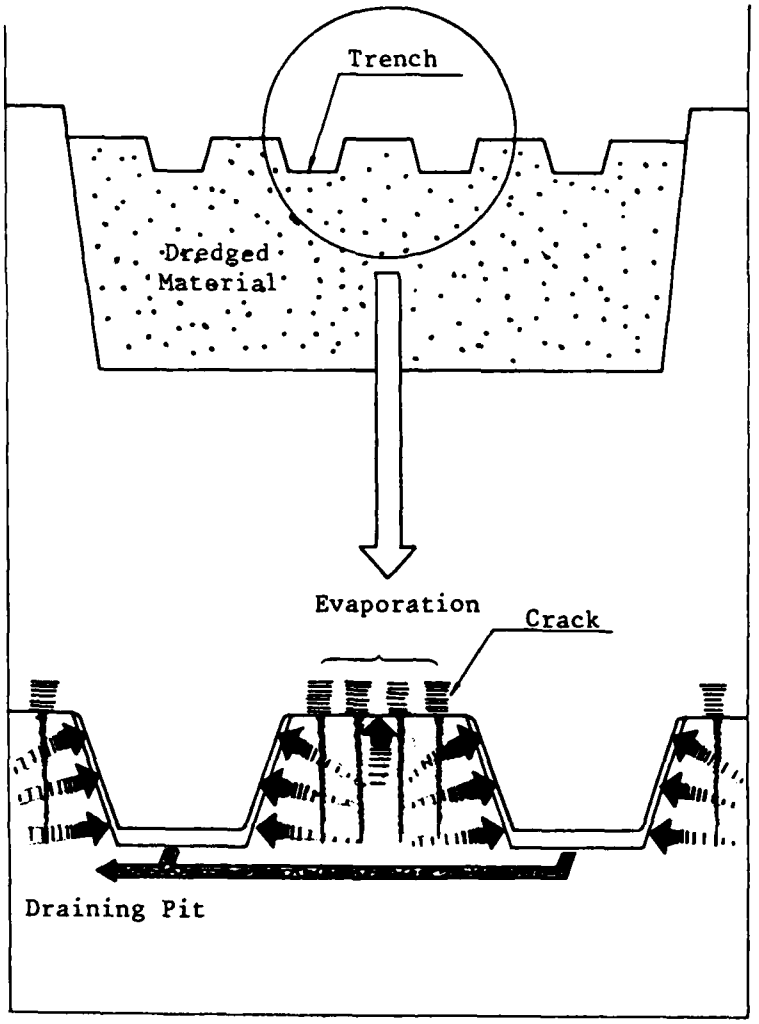


Figure 5. Principle of dewatering by trenches

winter, with its short days and low temperatures, desiccation by sunshine was difficult. Because of these circumstances, the progressive trenching method was adopted to accelerate the dewatering of dredged material.

The trenches were divided into mains and branches (Figure 6). Main tunnels were excavated by a clamshell, and branches by a scraper digger. Total length of the main trenches was 1,027 m and the branches was 4,444 m. The pitch of the branch trenches was 50 m. The excavation of trenches was conducted five times progressively over an interval of about 25 days.

The area of the filled land was 76,000 m^2 and the quantity of the filled material was 220,000 m^3. Dividing the filled area into five divisions the trenching work was conducted in order from V_1 to V_5 (Figure 7). The trench sections excavated are shown in Figure 8. Figure 9 shows a photograph of an area being worked by a scraper digger.

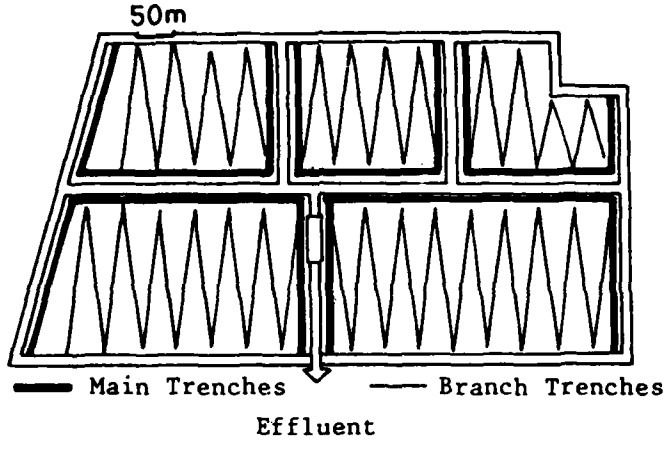


Figure 6. Main trenches and branches

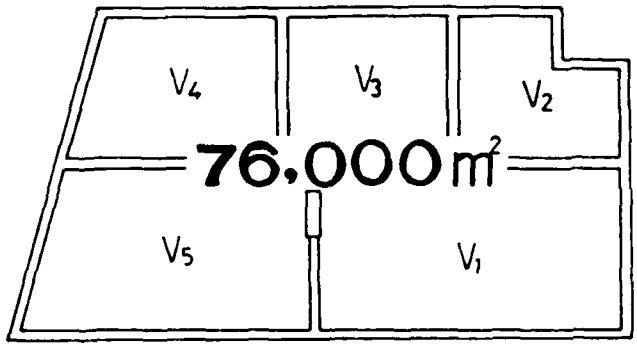


Figure 7. Disposal area divided into five divisions

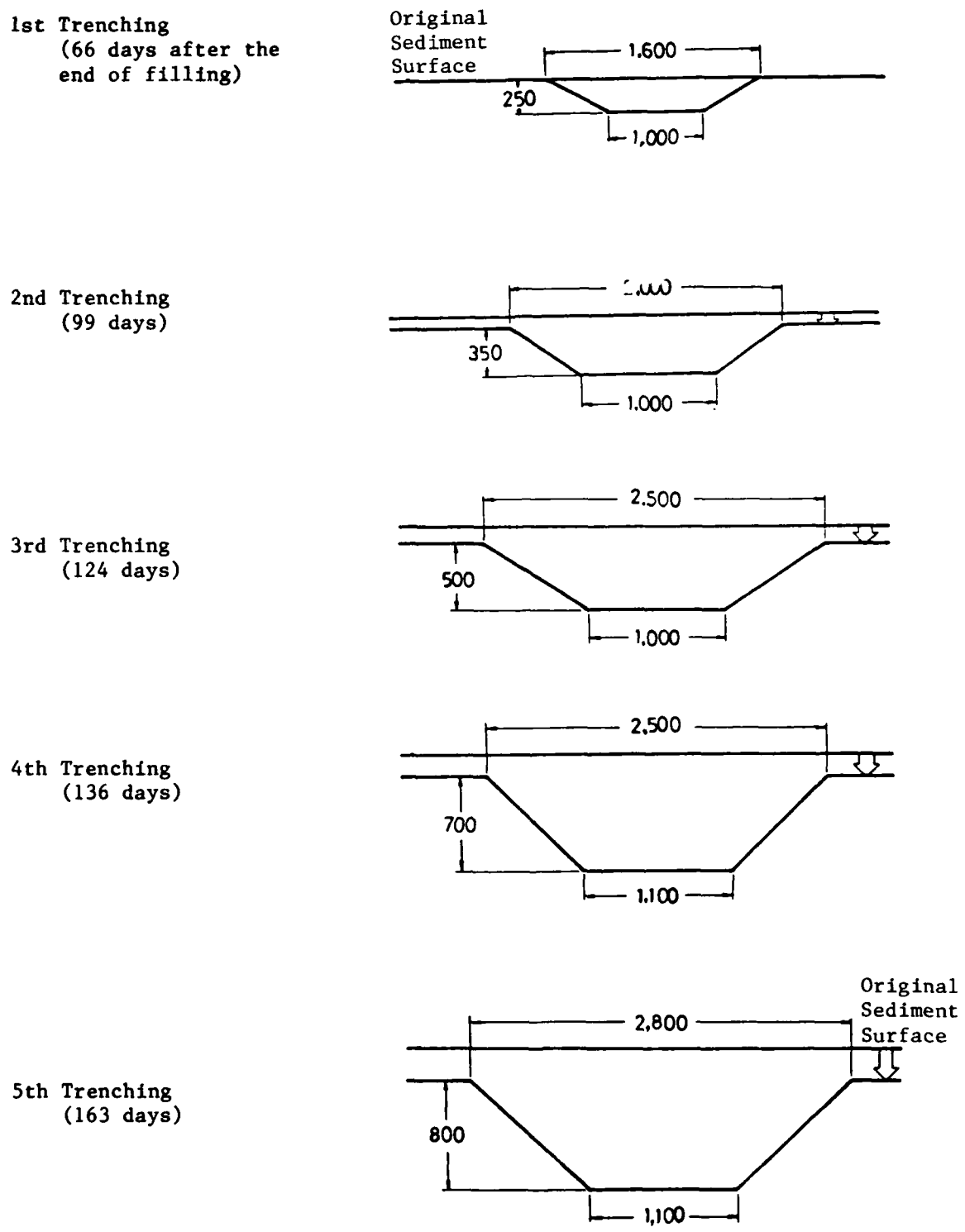


Figure 8. Trench sections (in millimeters)



Figure 9. An area being worked by a scraper digger

The depth of each trench was 250 mm after the first trenching and 800 mm after the last one. The area of excavated sections changed from 3,250 cm^2 to 15,600 cm^2.

RESULTS

Water content 30 cm below the surface was 400% at the end of filling and 261% just prior to the first trenching (Figure 10). Water content decreased gradually as trenching advanced:

1st Trenching	(66 days)	261%
2nd Trenching	(99 days)	211%
3rd Trenching	(124 days)	188%
4th Trenching	(136 days)	185%
5th Trenching	(163 days)	180%

The water content after 185 days was 149%. Consequently, as the result of trenching, a reduction in water content of 43% $\left(\frac{261 - 149}{261}\right)$ was attained. Water content at a point 30 m away from a trench was approximately equal to the water content at a point near the trench in shallow trenching. However, as the depth of the trench progressed, the water content became lower at the point near the trench and increased proportionally with distance.

Figure 11 compares water content with distances from the 4th and 5th trench. This distribution indicates that the water content in the disposal area begins to drop at points adjacent to the trench and gradually extends to distant areas. Figure 12 shows the crack generation on the surface. The settlement measured is shown in Figure 13. Settlement after 185 days in 18 cm. The mean settling rate is about 0.15 cm/day.

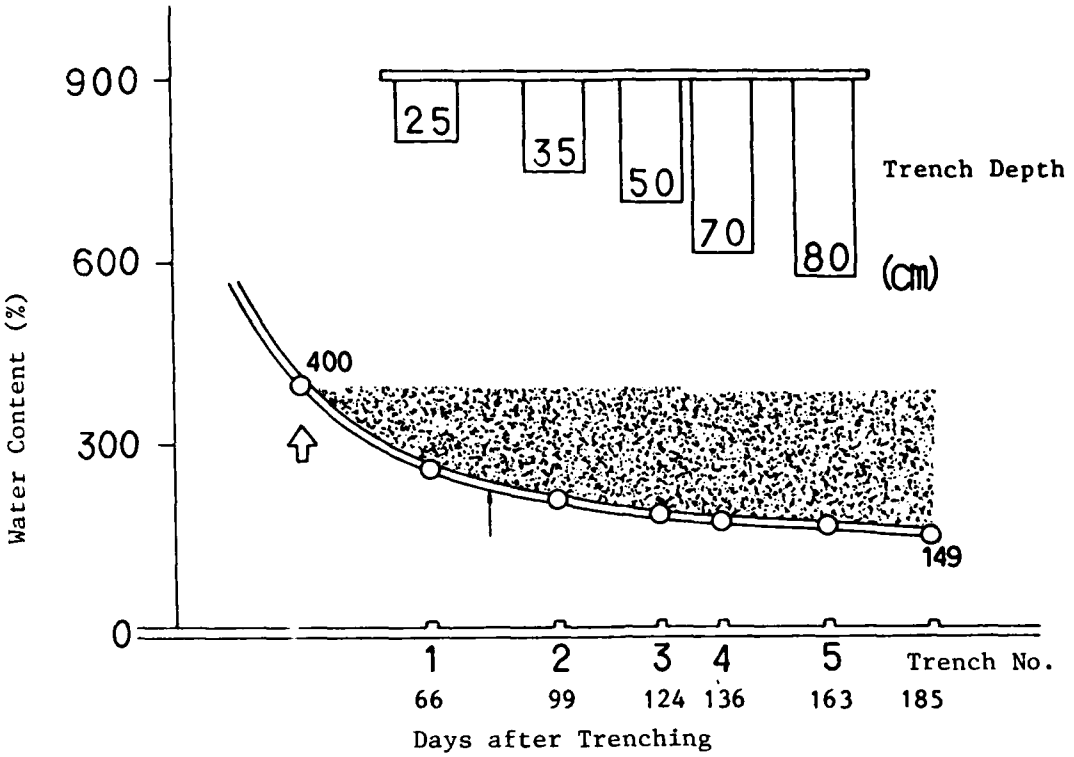


Figure 10. Change in water content

Figure 14 shows the results of cone penetration tests. The cone penetration stress was 0.25 kg/cm^2 at the beginning and increased to 0.66 kg/cm^2 by the 5th trenching and finally to 0.9 kg/cm^2. This means that in this case the filled area has stabilized well.

COVERING WORK

On the surface of dewatered dredged material permeable sheets were stretched and then foreign soil 1 m thick was placed on top of the sheets. The sheets used had a good permeability and a tensile strength of over 100 kg/5 cm^2 as determined by the Japanese Standards Association test methods (Figures 15 and 16). (See Appendix A for test methods.)

When the covering work has to be finished quickly, it should be remembered that the thicker covers take longer to apply. In such cases an unequal settlement sometimes occurs due to the ubiquitous load of the soil. To avoid such an unexpected risk, a method of sheet stretching was adopted. The thickness of the soil cover was 30 cm for the first layer and 20 cm for the second, and 50 cm for the final layer, a total of 1 m. For transporting and spreading the soil a small dump car with eight wheels and a 1-m^3 capacity was used. For covering the second layer a small bulldozer of 3-ton capacity was used. For the final layer an 11-ton bulldozer was used for soil spredding.

To prevent whirling up of sandy dust, grass was planted on the ground.

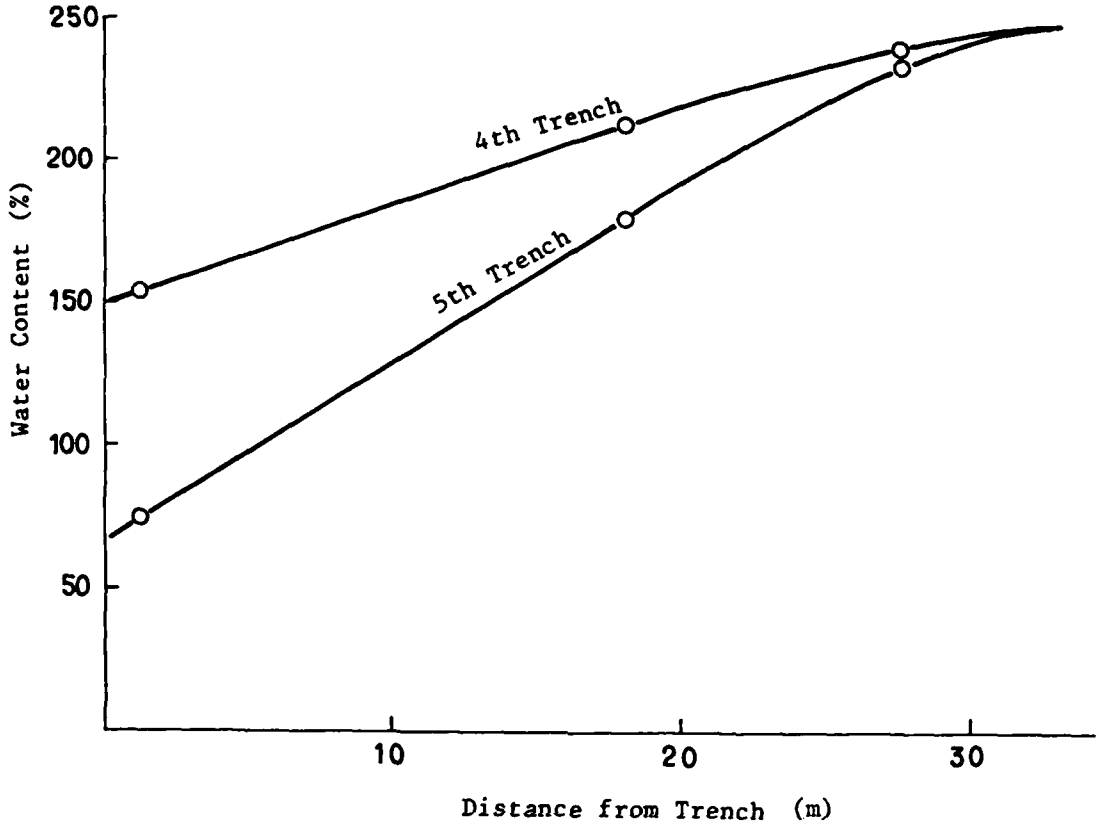


Figure 11. Water content near the trench



Figure 12. Cracks on the surface

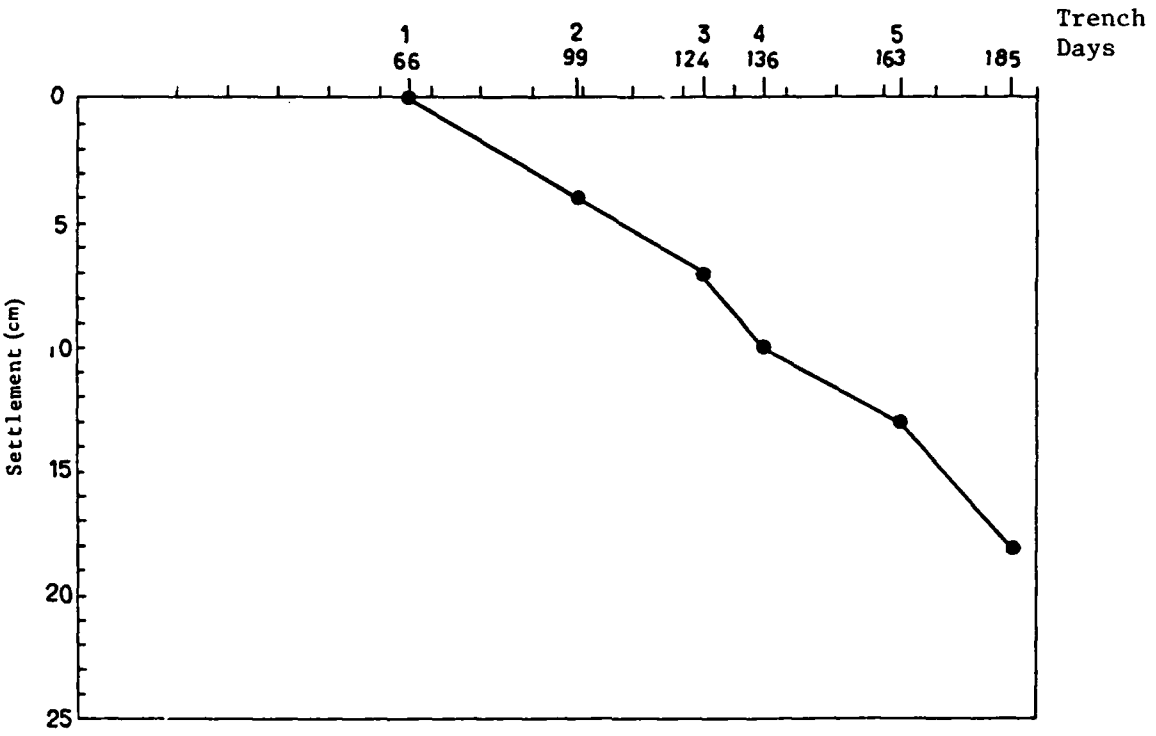


Figure 13. Settlement by trenching

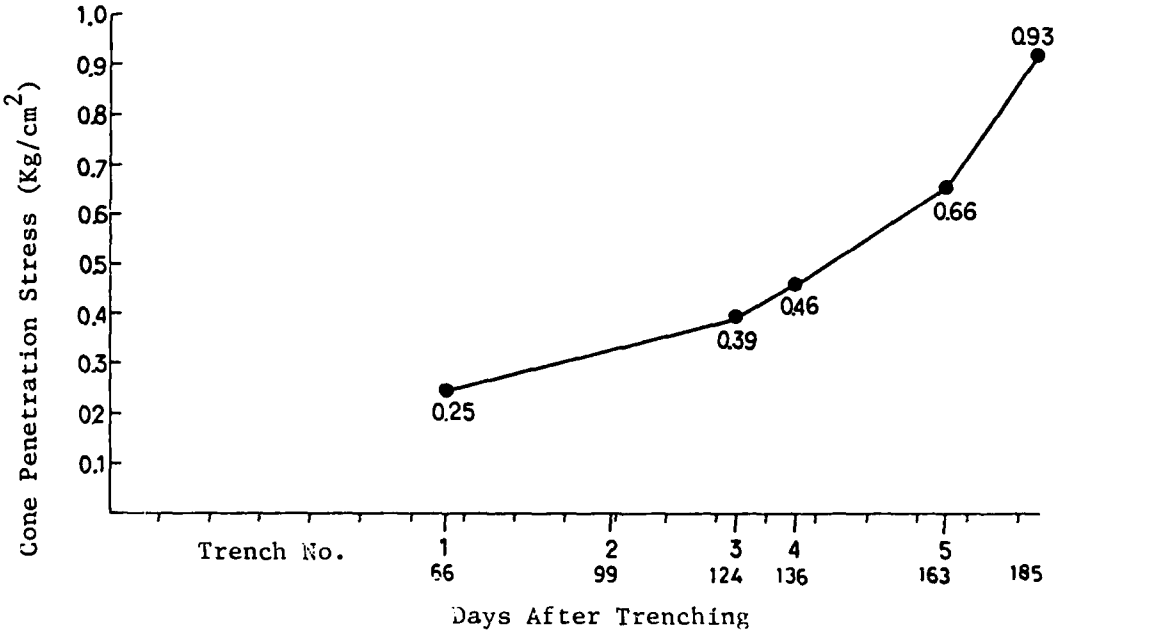
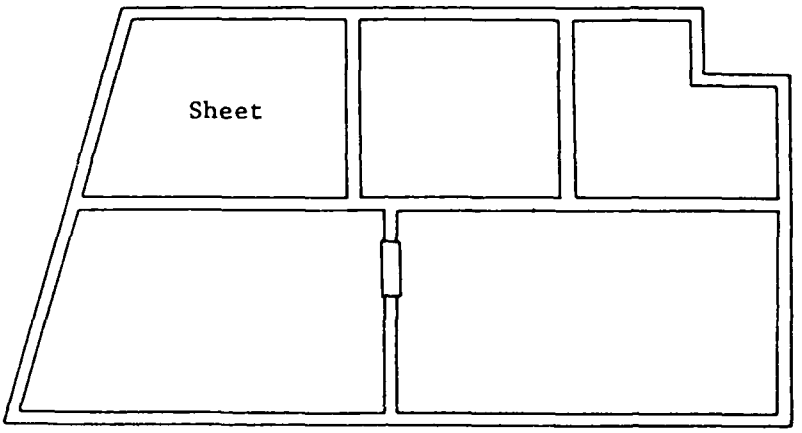


Figure 14. Cone penetration test results



Covered area 69,210 m^3

Sheet strength 100 kg/5 cm^2

Figure 15. Diagram of sheet stretching



Figure 16. Photo of sheet stretching

CONCLUSIONS

This project was the first instance of the progressive trenching method by a scraper digger attempted in our country. The main attribute of this method is that no expensive trenching machine is required. Therefore, it can be used economically for any small-scale landfill.

Although some room for improvement remains, the acceptability of the new method for practical use was verified by this study.

APPENDIX A: TEST METHODS

The test methods for tensile strength of fabric are standardized as follows by the Japanese Standards Association in our country. For heavy fabric, if the fabric is subject to raveling, the test specimen is prepared by retaining four or more yarns at both edges outside the test width (3 cm) and then cutting the yarns 5 cm from the ends of clamping jaws, as shown in Figure A1.

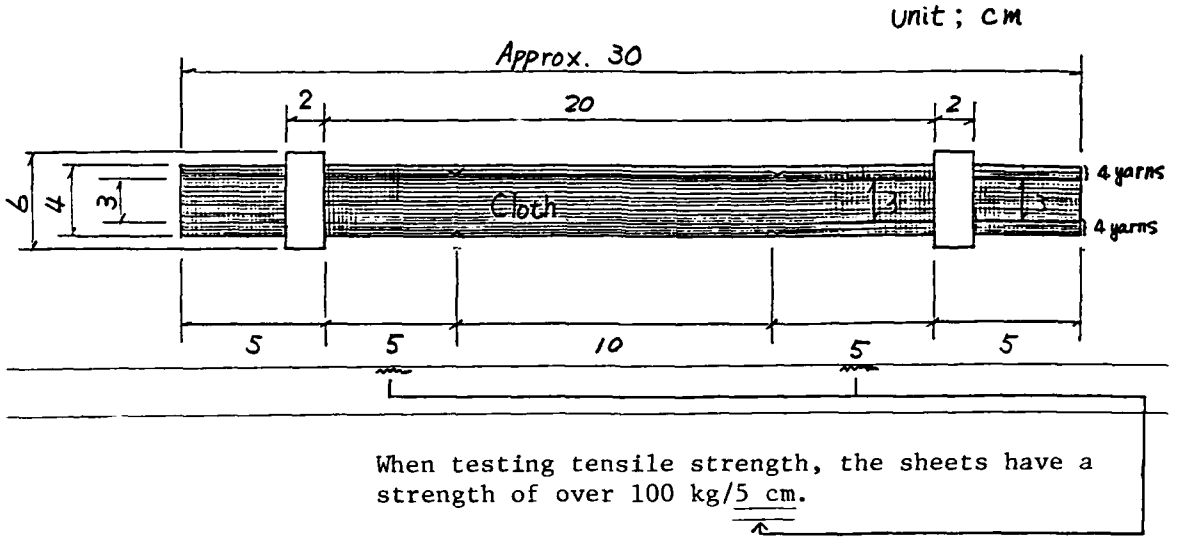


Figure A1.

When testing tensile strength, the sheets have a strength of over 100 kg/5 cm.

ENVIRONMENTAL CONSERVATION IN THE PORT OF MAIZURU

T. Matsuoka

Civil Engineering and Construction Department
Kyoto Prefectural Government

INTRODUCTION

The Port of Maizuru, located in the central part of the Japan Sea coast of the Main Island, is a natural port blessed with good geographical conditions. Maizuru Bay is narrow at the mouth and runs deep and wide into the inland in two directions forming two separate bays called Nishiko (the west port) and Higashiko (the east port). The name of the region, "Maizuru," is said to have originated from the geographical shape, which resembles a crane with its wings spread.

Maizuru City is the key city in the northern part of Kyoto Prefecture with a population of 98,000. With the current economic growth and increased contact with neighboring countries across the Japan Sea and other foreign countries, Maizuru City is becoming a developed area. In this area a high priority is attached to the improvement of the port as well as to the development of a brisk, affluent, and peaceful industrial port city where life is comfortable.

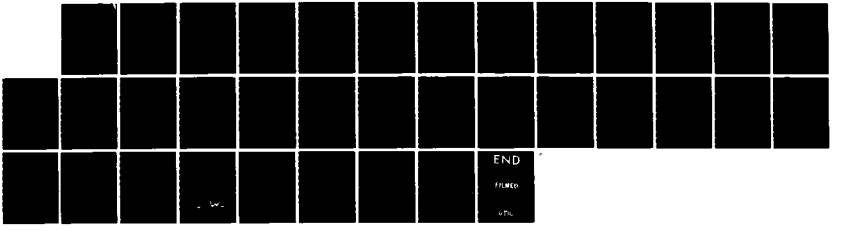
Higashiko Bay was first developed as a naval port with the construction of a naval station there in 1901, while Nishiko had developed as an important trading port on the Japan Sea coast, primarily for trade with neighboring countries, since a pier for large-sized ships was constructed in 1913. By the end of the Taisho Era (1912 - 1926), regular services between Maizuru and the Ports of Vladivostok, Chongjin, and Talien were launched successively, and Nishiko thrived on trades with neighboring countries (Figure 1).

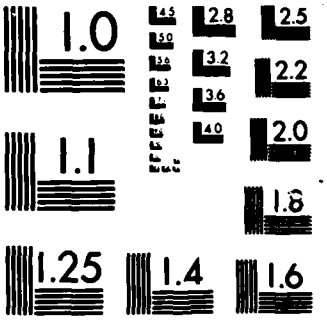
At the end of World War II, with the closure of the naval port, both Nishiko and Higashiko were redesigned to be commercial ports, and the Port of Maizuru was designated to be one of Japan's major ports in 1951.

In the 1960's, following the inauguration of the Hokkaido Regular Service Line and the designation as a port-of-call for the Japan-U.S.S.R. Regular Service Liners, domestic and foreign service lines have been launched. The completion of Pier No. 3 and Pier No. 4 was followed by a marked increase in cargo volume handled in this port (Figure 2).

The Port of Maizuru is particularly important as one of the active ports handling timber in this country. However, due to the current stagnant activities in the wood business, the volume of timber handled this is on the decline.

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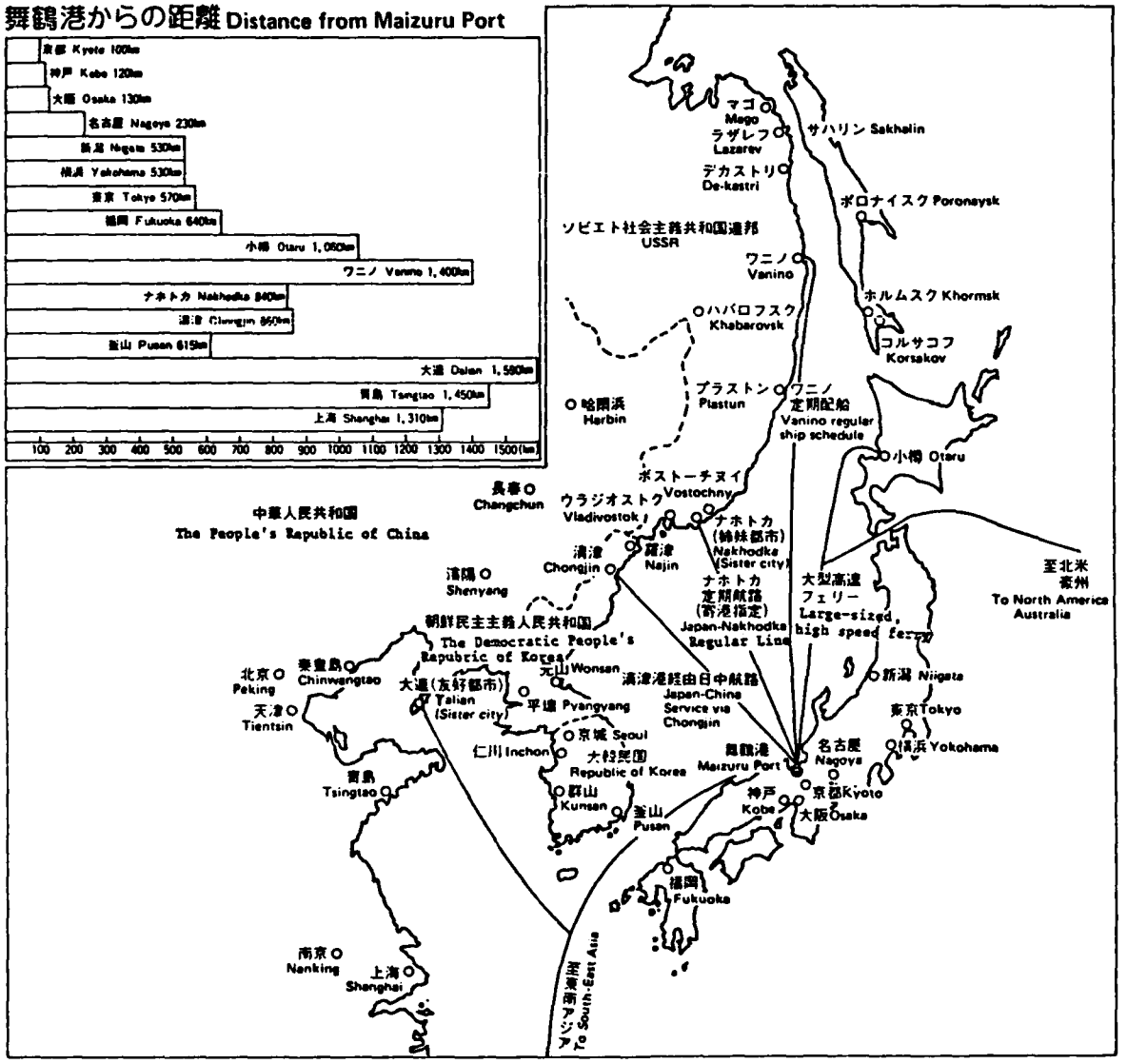


Figure 1. Geographical relation of the Port of Maizuru with neighboring ports

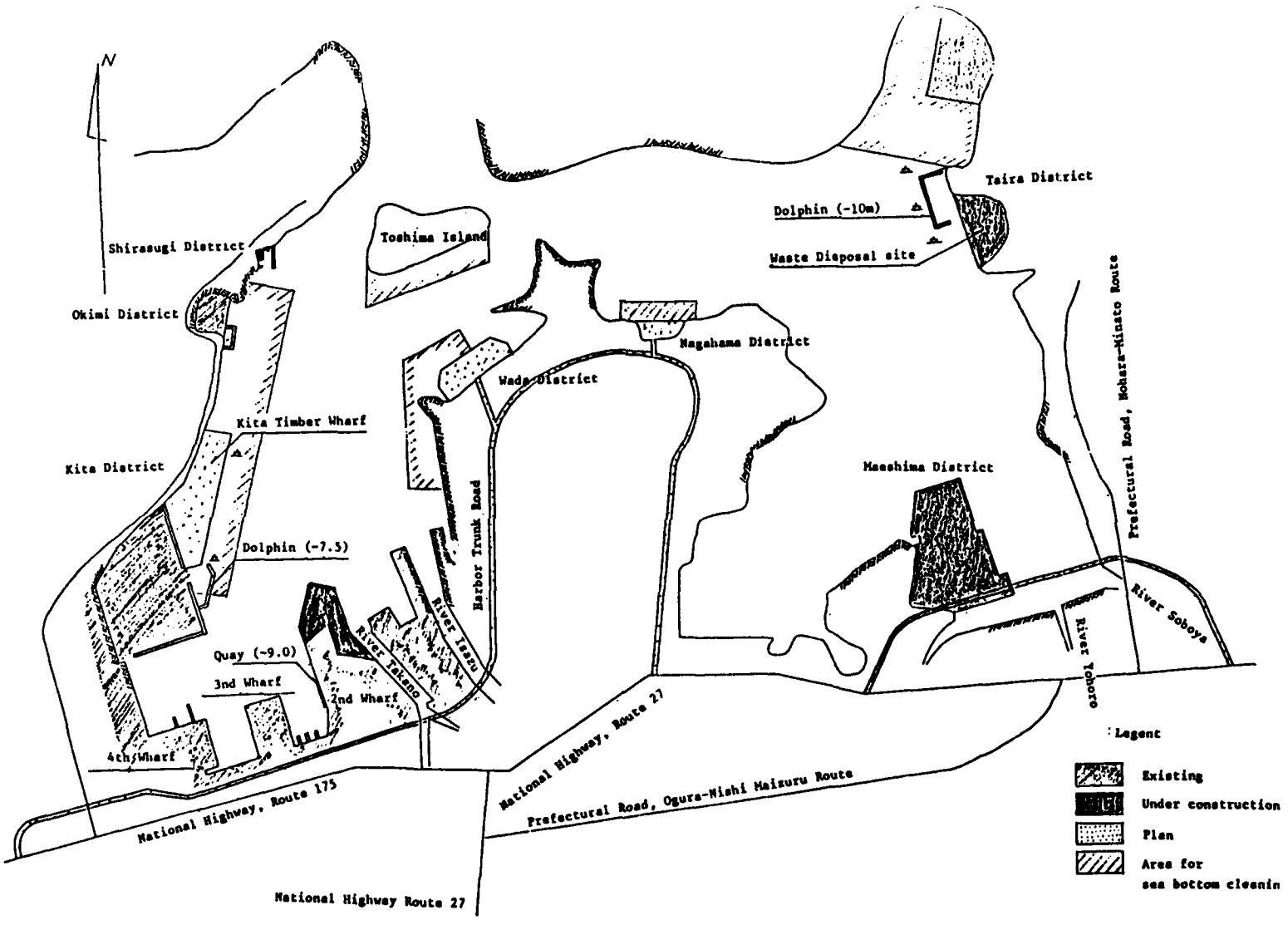


Figure 2. The Port of Maizuru improvement plan

IMPROVEMENT OF THE PORT OF MAIZURU

The port of Maizuru has diversified functions including fishery, Maritime Self-Defense Force, and marine recreation. It is, therefore, necessary to have a port improvement plan based on a long-term view. In 1976, the port improvement plan underwent an overall change where Nishiko was designed to be a commercial port for external trade with neighboring countries, while Higashiko was designed to be a commercial port for the domestic trade as well as a port for ferry boats between Maizuru and Otaru currently using Pier No. 2 in Nishiko.

According to the Sixth Five Year Plan for Port Improvement, the extension and redevelopment of Pier No. 2 and the construction of a pier in the Maeshima District are now under way along with the construction of the Maizuru Route of the Kinki National Highway. In this way, the port improvement plan is being steadily developed with the aim of making this port the key port for distribution of goods in the northern Kinki District as well as a port for international trade (Figure 3).

RATIO OF WOOD TO THE TOTAL CARGO VOLUME HANDLED IN THE PORT OF MAIZURU

In 1958 the Port of Maizuru began handling mainly northern wood from U.S.S.R. The volume of timber handled steadily increased each year from the initial volume of 14,068 tons to the peak volume reached in 1970 of 1,233,262 tons. After 1970 business activity in the domestic wood market slackened, and the volume of wood handling kept decreasing at a moderate pace. It has slightly picked up since 1982 (Table 1).

The data in the past 10 years show that wood accounted for 28.2% (79.0% of the total import volume) of the total cargo volume handled there in 1974, and 19.5% (92%) in 1983, a drop of approximately 9%. Meanwhile, the ratio of wood to the total import volume increased by 13% over these 10 years, indicating that the drop in the total import volume surpassed that of the total volume of wood handled (Table 2).

LOADING AND UNLOADINC OF WOOD ON THE SEA SURFACE AND ENVIRONMENTAL CONSERVATION

As mentioned, the Port of Maizuru has been handling mainly northern wood from U.S.S.R. since 1958. However, because of a lack of exclusive berths and dolphins for wood, timber was unloaded onto the sea surface (wood was discharged to the sea surface and carried in floats to timber ponds). This led to a massive deposition of sinking wood, bark, and wire on the sea bottom, which degraded the seawater environment in the bay and caused serious adverse effects upon the fishery.

To cope with this situation, the Kyoto Prefectural Government laid down "The Outline for Wood Handling" in 1969, followed by the organization in 1972 of "The Port of Maizuru Wood Pollution Prevention Committee" consisting of longshoremen and wood processors. A clean-up of sinking wood and bark on the sea bottom has been conducted since 1974, primarily by the polluter assisted by Kyoto Prefecture and Maizuru City. Table 3 shows the volume of wood handled on the sea surface and the cleaning operations of the sea bottom.

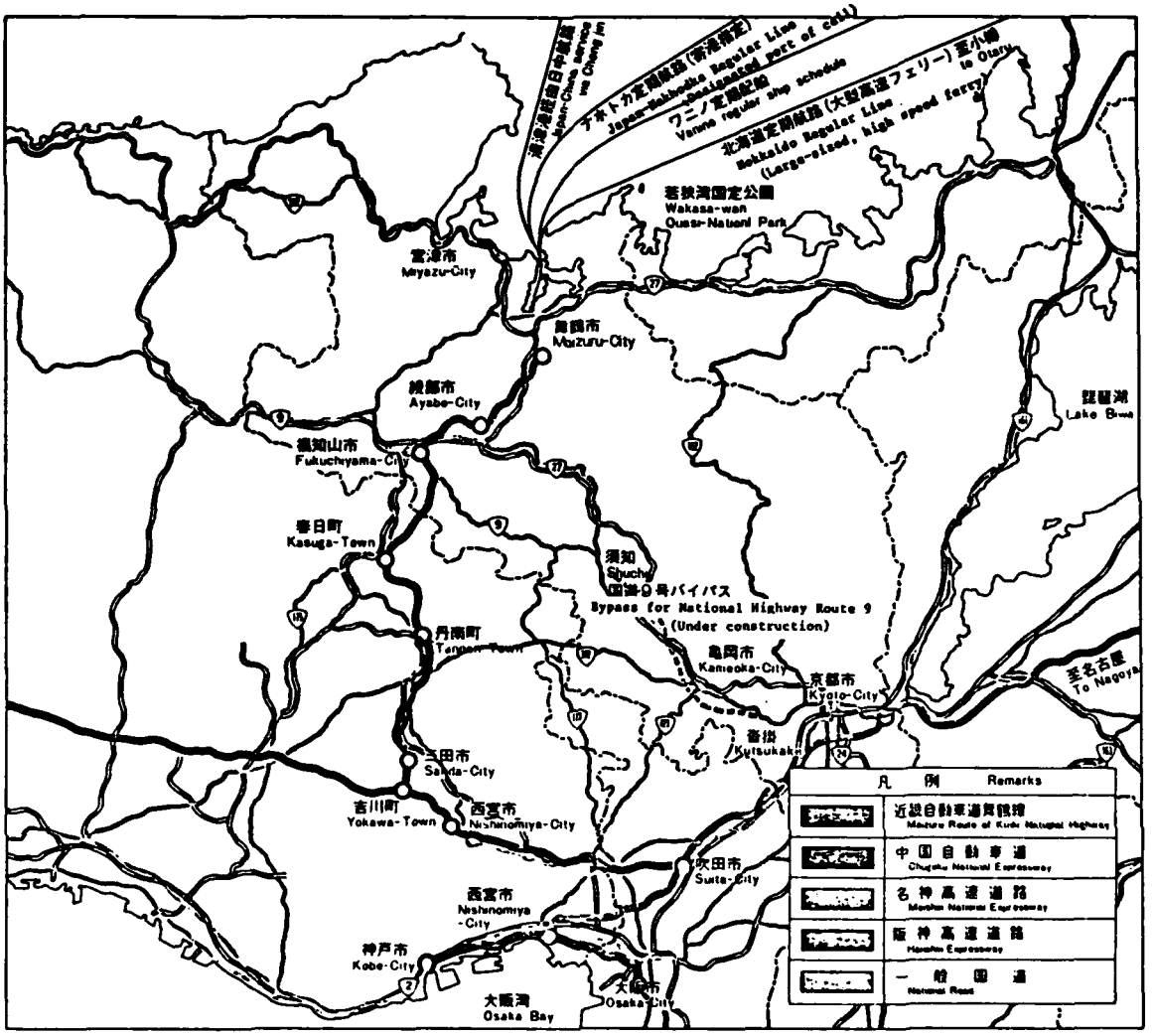


Figure 3. Progress of improvement in the highway network leading to the Port of Maizuru

The Port of Maizuru Wood Pollution Prevention Committee promotes the cleaning operations by consigning the work to the Maizuru Fishery Cooperative twice a year, in summer and in winter. Special nets for cleaning such as trawl nets are employed to sweep the sinking wood, bark, and wire from the sea bottom.

In addition to these operations, the Kyoto Prefectural Government (the port management body) constructed a cleaning boat, "Sokaku Maru" (with a cleaning capacity of 12 m^3/day, 8-ton gross) and an incinerator (Table 4) with the daily capacity of 7.5 tons/day. Management of both was assigned to Maizuru City. Maizuru City then consigned the actual operation of this

TABLE 1. CHANGE IN THE VOLUME OF IMPORTED WOOD HANDLED AT THE PORT OF MAIZURU

Year	Wood of Northern Origin		American Wood		Wood of Southern Origin		Total	
	No. of Wood Carriers	Volume of Wood tons	No. of Wood Carrier	Volume of Wood tons	No. of Wood Carriers	Volume of Wood tons	No. of Wood Carriers	Volume of Wood tons
1958	4	14,068	--	--	--	--	4	14,068
1959	15	46,001	--	--	--	--	15	46,001
1960	19	53,044	--	--	8	9,965	27	63,009
1961	21	64,336	8	32,959	12	17,804	41	11,099
1962	47	167,283	14	58,740	12	20,538	73	246,561
1963	59	222,964	8	25,298	12	19,137	79	267,399
1964	78	309,245	15	37,484	15	22,557	108	369,286
1965	76	318,567	15	34,596	24	38,533	115	391,696
1966	87	378,517	18	60,239	18	37,414	123	476,170
1967	116	524,022	31	115,809	28	80,149	175	719,980
1968	128	534,310	38	153,615	69	193,447	235	883,372
1969	147	568,708	29	109,509	84	232,110	260	910,327
1970	201	804,886	46	185,277	78	243,099	325	1,233,262
1971	203	798,433	29	147,646	61	196,666	293	1,142,745
1972	217	796,959	34	186,088	70	222,905	321	1,206,952
1973	240	844,716	38	160,266	66	224,732	344	1,229,414
1974	178	615,749	31	157,732	59	232,396	268	1,005,877
1975	165	617,805	29	142,367	40	154,920	234	915,092
1976	157	616,401	28	186,355	50	173,792	235	976,548
1977	166	710,199	36	218,382	62	210,660	264	1,139,241
1978	158	632,151	25	144,128	63	210,350	246	986,629
1979	157	600,314	43	179,656	82	238,397	282	1,018,867
1980	134	501,422	39	164,489	66	221,512	239	887,423
1981	121	442,633	22	88,384	47	138,947	190	669,964
1982	137	506,762	21	96,310	38	158,014	196	761,086
1983	161	582,394	19	72,603	30	127,208	210	782,205

Note: The Port of Maizuru began importing foreign wood in 1958.

TABLE 2. CHANGE IN CARGO VOLUME (TONS) HANDLED IN THE PORT OF MAIZURU

Year	Volume of Cargo Handling													
	Total	Import	Wood		Volume of Ferry Cargo		Agricultural and Fishery Products		Mineral Products		Metal and Machinery		Manufactured Goods	
	tons	tons	tons	%	tons	%	tons	%	tons	%	tons	%	tons	%
1974	3,570,134	1,274,041	1,005,877	28.2	1,500,273	42.0	51,740	1.5	361,575	10.1	64,619	1.8	586,050	16.4
1975	3,703,386	1,217,213	915,596	24.7	1,647,082	44.5	53,988	1.5	398,403	10.7	51,836	1.4	636,481	17.2
1976	4,110,779	1,219,942	977,694	28.3	1,997,760	48.6	48,213	1.2	353,646	8.6	42,102	1.0	691,364	16.8
1977	4,493,669	1,428,709	1,141,231	25.4	2,118,552	47.1	48,078	1.1	454,529	10.1	39,036	0.9	692,243	15.4
1978	4,131,152	1,081,354	987,362	23.9	2,121,112	51.3	39,623	1.0	252,376	6.1	15,422	0.4	715,257	17.3
1979	4,403,536	1,119,562	1,021,067	23.2	2,169,623	49.3	57,642	1.3	359,840	8.2	26,594	0.6	768,770	17.4
1980	4,124,086	786,481	888,364	21.5	2,071,758	50.2	51,730	1.3	289,192	7.0	45,390	1.1	777,652	18.9
1981	3,866,251	753,829	678,030	17.5	2,018,004	52.2	64,170	1.7	326,051	8.4	54,735	1.4	725,261	18.8
1982	3,870,614	817,300	761,571	19.7	2,012,028	52.0	55,258	1.4	274,345	7.1	47,952	1.2	719,460	18.6
1983	4,015,728	851,892	782,673	19.5	2,035,508	50.7	81,728	2.0	325,976	8.1	39,646	1.0	750,197	18.7

TABLE 3. CLEANING OPERATIONS OF SINKING WOOD AND BARK IN THE PORT OF MAIZURU

Year	Volume of wood handled on the sea surface (m^3)	Expenditure for the sea bottom cleaning operations (One thousand yen)	(Recovered Volume) Work Items		Remarks
			Sinking wood (m^3)	Bark (m^3)	
1974	857,657	9,000	333		Wood 1.133 m^3/ton Sinking wood, the number of pieces of round timber (calculated by using the mean of 0.25 m^3/piece for northern wood and 1.4 m^3/piece for American wood) 404
1975	805,426	9,000	459	1,449	556
1976	860,000	8,500	271	200	328
1977	936,564	11,000	469	220	568
1978	770,064	16,500	431	226	523
1979	806,577	36,600	848	123	1,028
1980	570,349	36,482	629	126	763
1981	312,723	36,203	522	232	633
1982	368,[illegible]41	36,600	360	159	436
1983	320,429	36,600	620	116	752
Total	6,608,230	236,485	4,942	2,851	5,991
Mean	600,823	23,649	494	285	599

TABLE 4. OPERATION OF THE BARK INCINERATOR

Year	Recovery Site*		Volume of Bark Incinerated
	On Shore m^3	On the Sea m^3	m^3
1981	4,981	1,043	6,024
1982	5,445	1,219	6,664
1983	4,966	1,141	6,107

* On shore = bark recovered at wharves.
On the sea = floating bark recovered by the cleaning boat "Sokaku Maru."

equipment to the Port of Maizuru Wood Pollution Prevention Committee to carry out cleaning and incinerating of sinking wood and bark produced during daily loading and unloading work on the sea surface.

Two thirds of the expenditure necessary for these operations is shouldered by the polluter, the Port of Maizuru Wood Pollution Prevention Committee.

NOXIOUS MATERIALS RELEASED FROM SINKING WOOD AND BARK

To assess the effectiveness of the current cleaning operations in the Port of Maizuru, the potential release load of organic matter and noxious substances from the wood and bark into the seawater should be estimated. However, since no such investigations have been conducted, the following estimations were made based on the investigation data obtained at other ports.

The sinking wood and bark cause two adverse effects. First, deposition on the sea bottom has a considerable influence upon benthos and consequently upon fishery. Second, organic matter released from sinking wood and bark causes a degeneration in the quality of the seawater environment (increase in COD) in the bay.

A particularly large amount of pieces of bark resulted from the pusher and barge carrying system adopted around 1975 for unloading timber from the U.S.S.R. This system discharged wood in large quantities onto the sea surface at one time, thereby peeling off large amounts of bark pieces by friction during discharge operations. At the same time a heavy load of bark within the cargo hold was flushed into the sea. This method naturally invited opposition from fishermen. Today, unloading is conducted by the conventional method: timber is dropped onto the sea surface by a derrick crane on a wood carrier or a mobile crane on a wharf.

Bark is peeled off either inside the cargo holds during loading and transporting or at wharves or in anchorages where wood is discharged onto the surface.

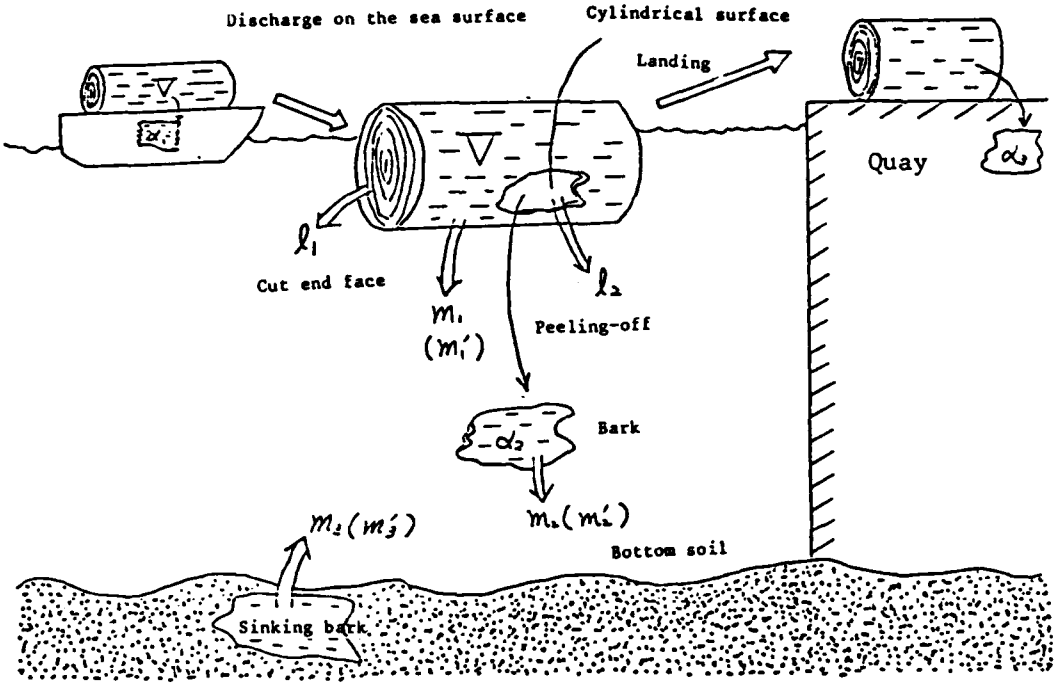
The ratio of the total volume of bark to wood is usually about 12%. The ratio of bark peeled off in a cargo hold and at a wharf to the total volume of wood unloaded is estimated to be 1% and 0.5 - 2.0%, respectively, although, in the latter case, the value varies largely depending on how the timber is handled (for example, the volume of peeled off bark increases when the timber is counted, measured, and classified because the timber is handled piece by piece). The peeling-off rate for northern wood is estimated to be 0.5 - 0.8%. The peeling-off rate of bark during unloading of timber onto the sea surface varies by the kind of timber, and, according to the investigation at the Port of Maizuru, is approximately 0.5% for northern wood from U.S.S.R., approximately 0.3% for American wood, and approximately 0.1% or less for southern wood. The small value for southern wood (mainly from Malasia) may be explained by the fact that lauan, which comprises the majority of the southern wood unloaded at the Port of Maizuru, is shipped primarily with the bark stripped off at the port of shipment. However, Joncon and Jelton, new offerings in the southern wood market as alternatives for lauan, leave a heavy load of bark behind, sometimes exceeding 5% of the total cargo volume.

These discussions include only bark; however, organic substances are also released from the timber itself. It should be noted that the release occurs from sinking wood as well as from wood in the timber ponds.

Figure 4 is a simplified illustration of this release mechanism. Admittedly, nature's self-cleaning system (for example, bacteria) contributes to the process of conversion of organic matter into inorganic matter. This process, however, will not be included in the discussion below.

When $V(m^3)$ of wood is transported by a timber freighter, $\alpha_1(\%)$ of bark will be left inside the hold. During the several months of storing in timber ponds following the discharge onto the sea surface until the time of landing, $\alpha_2(\%)$ of bark peels off the wood to sink or float in the water. During this period, ℓ_1, ℓ_2, and m_1 $(mg/cm^2/day)$ of COD are released from the cut end peeled-off sections on the cylindrical surface, and bark surface of wood, respectively, while m_2 $(mg/cm^2/day)$ of COD is released from the free bark into the water. The COD release from the sinking bark in the past is $m_3(mg/cm^2/day)$. In terms of weight, the COD release from a unit weight of bark is m_1' (mg/g/day) for the cylindrical surface, m_2' (mg/g/day) for the both surfaces of the free bark, and m_3' (mg/g/day) for the sunken bark on the sea bottom.

To obtain a unit volume for contamination such as COD released from wood, experimental volume for unit area of the cut ends, peeled-off sections on the cylindrical surface, and bark surfaces (ℓ_1, ℓ_2, m_1, m_2, m_3) must be determined. At the same time the cut end area (S_A cm^2), the cylindrical surface area (S_B cm^2), the area of peeled-off sections on the cylindrical surfaces (S'_B cm^2), the area of unpeeled sections on the cylindrical surface (outer



V: Volume of wood (m^3)

Peeling-off rate for bark

α_1: In a cargo hold (%)

α_2: In timber ponds (%)

α_3: During wood handling, released load of COD from wood (%)

ℓ_1: From the cut end face (mg/cm^2)

ℓ_2: From the peeled-off sections on the cylindrical surface of wood (mg/cm^2)

Release load of COD from bark

m_1(mg/cm^2), m_1' (mg/g): From the surface of bark

m_2(mg/cm^2), m_2' (mg/g): From both surfaces of free bark in the water

m_3(mg/cm^2), m_3' (mg/g): From surfaces of sinking bark

Figure 4. Release mechanism of organic matter due to wood handling on the sea surface

bark surface) (S_C' cm^2), both surface areas of free bark (S_D cm^2), and surface areas of sinking bark (S_E cm^2) per 1 m^3 of wood should also be calculated.

The wood contamination unit per 1 m^3, W, is obtained from the following equation:

$$W = \ell_1 \cdot S_A + \ell_2 \cdot S_B' + m_1 \cdot S_C' + m_2 \cdot S_D + m_3 \cdot S_E \quad (1)$$

Assuming the release load per unit area of the heartwood and sapwood (see Figure 5) on the cut end to be ℓ_{11} and ℓ_{12}, and the heartwood rate H, the equation will be:

$$W = \ell_{11}HS_A + \ell_{12}(1 - H)S_A + \ell_2 S_B' + m_1 S_C' + m_2 S_D + m_3 S_E \quad (2)$$

Using S_A and S_B, Equation 2 can be expressed as follows:

$$W = \ell_{11}HS_A + \ell_{12}(1 - H)S_A + \ell_2 \sum_{i=1}^{2} \alpha_i S_B + m_1\left(1 - \sum_{i=1}^{2} \alpha_i\right)S_B + m_2 S_D + m_3 S_E \quad (3)$$

Assuming that all the free bark sinks rapidly, and neglecting the release load from the accumulated sunken bark, we may write as follows:

$$W = \ell_{11}HS_A + \ell_{12}(1 - H)S_A + \ell_2 \sum_{i=1}^{2} \alpha_i S_B + m_1\left(1 - \sum_{i=1}^{2} \alpha_i\right)S_B + m_3 \sum_{i=1}^{2} \alpha_i S_B \quad (4)$$

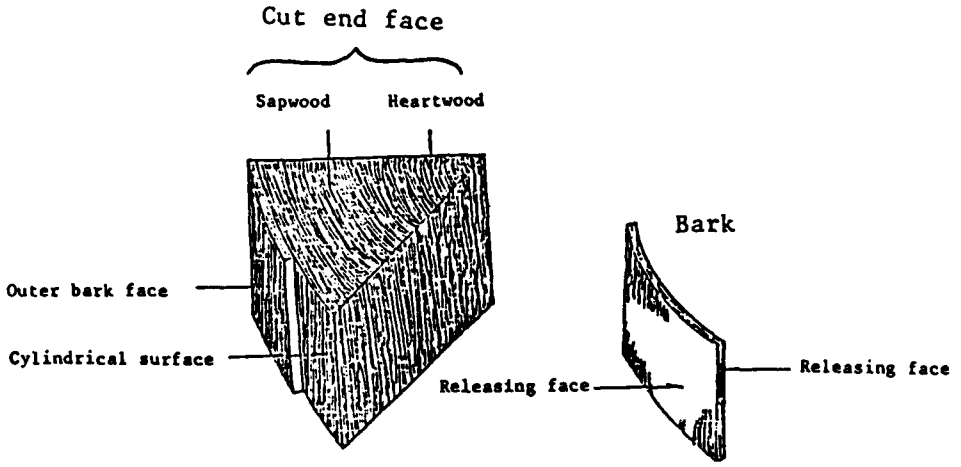


Figure 5. Names for releasing faces

The last term of Equation 4 will be expressed as $m_3 \cdot \alpha_2 S_B$ in the case where bark remains inside a hold.

In the experiments conducted by the First District Port Construction Bureau, Ministry of Transport, the values shown in Table 5 are proposed as the release values per unit area of each section of wood for each kind of wood. The total release per 1 m^3 of wood is shown in Table 6 (the influence of bacteria is eliminated by sterilization).

TABLE 5. RELEASE PER UNIT AREA OF AN INDIVIDUAL SECTION OF WOOD ($mg/cm^2/day$)

	Larch (Wood of Northern Origin)	Western Hemlock (American Wood)	Lauan (Yellow Seria)	Lauan (Red Lauan)	Cedar (Domestic Wood)
Cut end face					
ℓ_{11} heartwood	0.823	0.18	0.045	0.0345	0.18
ℓ_{12} sapwood	--	0.085	0.12	--	0.095
ℓ_2: peeled-off sections of the cylindrical surface	0.113	0.095	0.20	0.012	0.04
Bark					
m_1 outer bark	0.0358	0.065	--	--	0.03
m_3 free bark	0.039	0.08	--	--	0.06

The volume of timber handling on the sea surface in the port of Maizuru was, as seen in Table 7, approximately 320,000 m^3 (290,000 tons) in 1983, whose unloading operation was conducted at three separate places (Figure 2).

Each handling site has its own adjacent timber pond where unloaded wood is towed. In addition to these, in Ookimi District, at the exclusive dolphin for wood, timber is unloaded inside the water area separated from the anchorage by sheet piles, and stored in the Ookimi Timber Pond.

Using a conversion factor of 1.13 cubic meters per ton, Table 7 shows the volume of imported wood, classified by wood origin, and the total number of wood carriers employed over a 3-year period. The annual volume of wood handled at these sites over the 3-year period was 320,000 m^3 for the wood of northern origin, 70,000 m^3 for American wood, and 150,000 m^3 for the wood of southern origin. The annual volume of bark peeled off from this volume of wood is estimated to be 1,600 m^3 (650 m^3 excepting Ookimi) for northern timber, 220 m^3 (180 m^3) for American wood, and 150 m^3 (150 m^3) for southern timber, totaling 1,970 m^3 (980 m^3).

TABLE 6. TOTAL RELEASE PER 1 m^3 OF WOOD (EXPERIMENTAL CONDITION: IN SEAWATER, WATER TEMPERATURE OF 20°C, NO MIXING)

Type of Wood	Wood Contamination Unit*		Characteristics of Wood
North American Wood (Western hemlock)	10,400mg/day/m^3	11,580mg/day/ton	Radius γ = 20 cm, Heartwood rate H = 0.4 Length ℓ = 400 cm, Peeling-off rate α = 0.3
Wood of southern origin (Lauan, yellow seria)	8,200mg/day/m^3	9,130mg/day/ton	Radius γ = 50 cm, Heartwood rate H = 0.5 Length ℓ = 800 cm, Peeling-off rate α = 0.1 Log without bark
Domestic wood (Cedar)	8,130mg/day/m^3	9,050mg/day/ton	Radius γ = 10 cm, Heartwood rate H = 0.6 Length ℓ = 400 cm, Peeling-off rate α = 0.1
Wood of northern origin (Larch)	7,200mg/day/m^3	8,010mg/day/ton	Radius γ = 15 cm, Peeling-off rate α = 0.2 Length ℓ = 400 cm
Wood of southern origin (Lauan, red lauan)	360mg/day/m^3	400mg/day/ton	Radius γ = 50 cm, Peeling-off rate α = 1 Length ℓ = 6 cm, Log without bark
Bark of western hemlock	32,000mg/day/m^3 bark (2,400 rough timber)	35,600mg/day/ton bark (2,670 rough timber)	Thickness of bark = 2.5 cm Surface area = 400,000 cm^2
Bark of cedar	20,000mg/day/m^3 bark (1,200 rough timber)	22,000mg/day/ton bark (1,340 rough timber)	Thickness of bark = 0.3 cm Surface area = 3,000,000 cm^2
Bark of larch	26,000mg/day/m^3 bark (630 rough timber)	28,900mg/day/ton bark (700 rough timber)	Thickness of bark = 1.5 cm Surface area = 670,000 cm^2

* Wood contamination unit includes COD release load from free bark.

TABLE 7. WEIGHT OF IMPORTED WOOD HANDLED AT THE TIMBER PONDS IN THE PORT OF MAIZURU

Year	Name of Wood Handling Zone	Wood of Northern Origin		American Wood		Wood of Southern Origin		Total	
		No. of Wood Carriers	Weight of Wood tons	No. of Wood Carriers	Weight of Wood tons	No. of Wood Carriers	Weight of Wood tons	No. of Wood Carriers	Weight of Wood tons
1981	Ookimi	41	136,638	6	21,773	0	0	47	158,411
	Kazura	11	45,823	5	18,815	0	0	16	64,638
	Tahira	0	0	6	32,059	36	109,660	42	141,719
	Kita	18	59,094	3	10,563	0	0	21	69,657
	Total	70	241,555	20	83,210	36	109,660	126	434,425
1982	Ookimi	46	159,399	4	16,589	0	0	50	175,988
	Kazura	9	42,933	6	25,503	0	0	15	67,936
	Tahira	0	0	6	26,142	38	158,014	44	184,116
	Kita	21	73,139	0	0	0	0	21	73,139
	Total	76	257,471	16	68,234	38	158,014	130	501,719
1983	Ookimi	50	209,264	2	6,493	0	0	52	215,757
	Kazura	7	30,322	7	29,547	0	0	14	59,969
	Tahira	0	0	2	7,275	30	127,208	32	134,483
	Kita	26	88,463	0	0	0	0	26	88,463
	Total	83	328,049	11	43,315	30	127,208	124	498,572

Table 3 shows the recovered volume of sinking bark and Table 4 the volume of floating bark recovered by the cleaning boat, "Sokaku Maru." The total volume of the recovered bark was 1,275 m^3 in 1981, 1,378 m^3 in 1982, and 1,257 m^3 in 1983.

If no recovery operation were carried out and peeled-off bark were allowed to accumulate to 30 cm thick, the sea bottom area covered with peeled-off bark would be approximately 6,000 m^2, and the annual volume of organic matter released into the seawater from free bark would be (when converted into COD) 73.4 kg for northern timber and 64.4 kg for American timber, with a total volume of 137.8 kg (since the southern timber unit is not known, it was not included in this estimation). From this, it can be concluded that bark recovery operations may be effective in reducing this potential contaminant release load into the seawater.

The annual mean recovery volume of sinking wood over the past 3-year period was 500 m^3. When converted into volume of organic matter released by using the release rate for northern timber, the value is 1.31 kg annually, and is 1.60 kg annually when converted by using the release rate and the cargo rate for the wood of northern, southern, and American origin. These values suggest the effectiveness of the recovery work in reducing the potential organic load.

For reference, seawater quality results determined in the Port of Maizuru are shown in Figures 6 and 7. The marine environmental standards for the port of Maizuru were set in 1975. Since then, periodic investigations of seawater quality have been conducted to monitor the achievement rate of these standards.

Furthermore, in Maizuru, the public desire for pure water is particularly strong when compared with the people in other Districts. The carp which are stocked in the River Takano flowing into Maizuru Bay are one example of the wisdom of the local people to help the self-cleaning system of nature, and it goes without saying that nobody would dare catch the carp.

FUTURE ENVIRONMENTAL CONSERVATION IN THE PORT OF MAIZURU

Since the Port of Maizuru handles a large volume of wood, which accounts for a major portion of the imported goods, there are demands for environmental protection and improvement of the bay to counteract the seawater pollution caused by wood handling on the surface. This local demand is mounting as the nationwide social climate to place more importance on the environmental conservation grows. In the course of negotiations for fishery compensation relating to port improvement, the fishery cooperatives operating in the bay strongly demand marine environmental protection as well as fishery promotion policies.

To meet these demands from the local residents, the authors are committed to promote the port improvement plan by constructing an exclusive berth for timber in the Kita District and dolphins in the Kita and Taira Districts with the aim of eliminating timber handling on the sea surface.

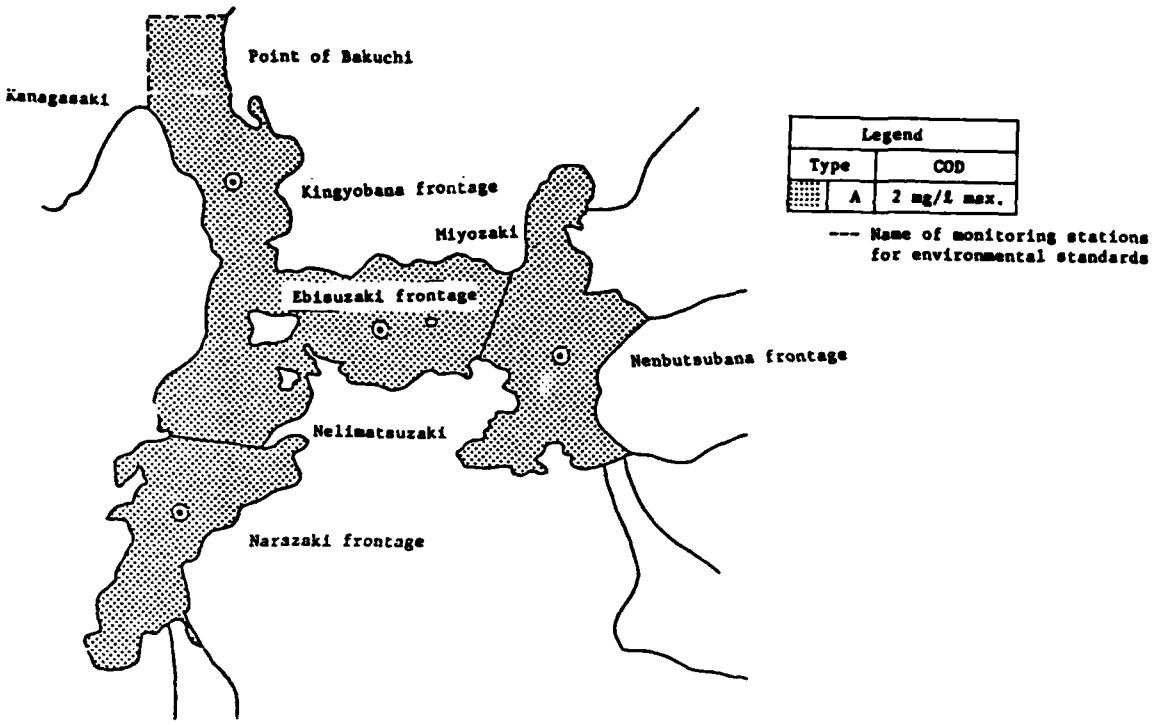


Figure 6. Monitoring stations for environmental standards (Type A according to Japanese Environmental Water Quality Standards as amended 1975.)

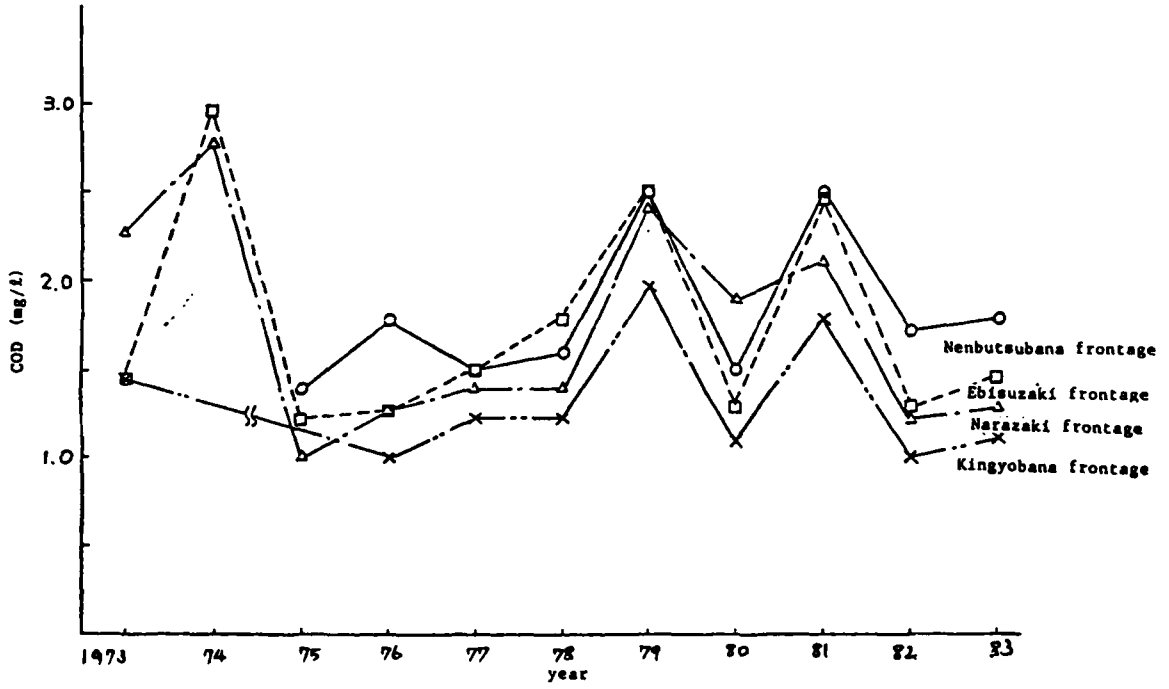


Figure 7. Change in mean COD at monitoring stations

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THE MUSSEL WATCH PROGRAM/MONITORING THE LONG-TERM EFFECTS OF DREDGED MATERIAL DISPOSAL IN NEW ENGLAND WATERS

R. C. Semonian
C. J. Lindsay

US Army Corps of Engineers, New England Division
424 Trapelo Road
Waltham, Massachusetts 02254-9419

ABSTRACT

The New England Division Corps of Engineers (NED) began studying the effects of disposal of dredged material in marine waters in 1969 and has continued this effort under the Disposal Area Monitoring System (DAMOS).

The "Mussel Watch" Program is one of the state-of-the-art techniques incorporated into DAMOS to assess bioaccumulation due to disposal of dredged material. Findings to date show the mussel to be an excellent test organism which, together with suitable field techniques, may be used to monitor effects of disposal over the long term.

INTRODUCTION

The purpose of monitoring the disposal of dredged material in ocean waters is to identify, measure, and quantify significant impacts to the marine environment surrounding the point of discharge. It is reasonable for a project manager charged with the task of designing a monitoring plan to consider obvious factors such as the character of the material to be dredged and the hydrological and biological regimes of the disposal sites. He can then base his scientific design on a program of field observations and measurements in terms of physical, chemical, and biological parameters utilizing proven methodologies and standardized practices. This was the approach used by the New England Division in early monitoring efforts (1969) and it was the deficiencies of this approach that led to the formalizing of our Disposal Area Monitoring System and the evolution of our mussel program.

Some 15 years ago the New England Division began its pursuit of the environmental consequences of dredged material disposal by contracting marine laboratories and universities to conduct field observations of specific dumping events. Simply put, our approach was to develop a predisposal baseline of the disposal site in physical, chemical, and biological parameters against which subsequent measurements taken during and after disposal might be compared. At this point in time, dredged material monitoring was in its infancy

and, since there were no established guidelines to follow, principal investigators were allowed much latitude in the experimental design of each contracted task. As studies were reported we became aware that there were few standardized methods in the marine community for acquiring data and that scientists of the same discipline used different methodologies for similar tasks. Further, to our dismay, we found that scientists of differing disciplines were proprietary in their thinking and rarely gave consideration of possible interaction with other sciences involved. Although data acquired were extensive and reported findings accurate, we realized that our efforts were too fragmented and lacked compatible continuity. Recognizing that a more structured and interdisciplinary program would contribute significantly to our monitoring efforts, the Division formalized the Disposal Area Monitoring System, or DAMOS.

DAMOS

DAMOS, formally organized in 1977, remains as the monitoring arm of the New England Division. Through it we have standardized our monitoring designs at ten open water sites along our coastline and have applied a multidisciplined approach to all investigations. Scientists participating in the DAMOS program constitute a highly experienced, multidisciplined team that meets at least annually for a peer review of findings and to collaborate on the design of upcoming activities. Team members are encouraged to suggest new concepts and innovative ideas for consideration by the team. As a result, DAMOS has been a learning experience, developmental, and often at the state of the art. The use of mussels as pollution indicators in dredged material management stands as an example of the DAMOS approach in that it represents monitoring concepts that have the potential of integrating physical, chemical, and biological processes that occur with the disposal of dredged material in the marine environment.

The role of marine bivalve molluscs in extracting trace metals from the marine environment and concentrating the metals in their tissues had been well established prior to our integrating a mussel program in DAMOS. And, in fact, bivalves had been used as sentinel organisms in monitoring environmental quality. Studies had shown that the uptake of trace metals by bivalve molluscs is reflective of metals in the environment--water temperature, physiological state, and organism size are other factors--and that the uptake reversed as the concentration of metals diminished. Evidence seemed to indicate that an appropriate bivalve mollusc might be ideally suited to monitor the transport and fate of those contaminants of concern perceived to be released into the marine environment by the dredging and open water disposal of enriched sediments. With this premise in mind, the DAMOS mussel program was initiated in 1978 under the direction of Dr. S. Y. Feng of the University of Connecticut. Initial study goals were to determine if there were significant increases in trace metals in our organisms attributable to the disposal of dredged material and to develop effective field deployment and retrieval for the mussel monitoring platforms (2,5).

Although several species of filter-feeding bivalve molluscs are known to reflect changes in exposure to heavy metals, not all are suitable for long-term monitoring. Robust organisms, active through a major portion of the year and of sufficient size and local abundance to permit frequent sampling, are needed. The horse mussel (*Modiolus modiolus*) and the blue mussel (*Mytilus*

edulis) were selected for monitoring DAMOS sites located north and south, respectively, of Cape Cod which separates the two tidal systems impacting the New England coastline.

The plan was to establish experimental stations at four northern and six southern disposal sites. Prior to deployment, single stock populations of *M. modiolus* with relatively low trace metal content were selected for the northern sites and designated as a reference population. Populations of blue mussels, *M. edulis*, from a single site in Long Island Sound were selected as the reference stock for the disposal sites in that estuary.

Groups of 50 mussels were placed in 35 polyethylene mesh bags and attached to polyvinyl chloride (PVC) platforms (1 m × 1 m × 1 m) with concrete footings. Platforms were lowered by shipboard winch and Loran C positions recorded. Subsurface pop-up floats, detachable surface buoys, and subsurface sonic pingers were incorporated in the platforms to facilitate retrieval by divers using SCUBA for subsequent sampling of the mussels. Initially we sampled mussels on a quarterly basis but have found that monthly sampling is better suited to observing subtle temporal variations (2,5,7).

During each sampling period, duplicate samples of 8 mussels were collected and, for baseline data, 10 replicates were used. Mussels were cleaned, measured, shucked, and freeze dried. The process followed for the trace metals analyses is too extensive to include in this paper, but is thoroughly discussed in DAMOS reports--numbers 20 thru 22 and the 1980 annual report, among others. However, we can emphasize that extreme care was exercised to minimize contamination (2-5).

Since mussels were collected from natural (reference) populations and experimentally transplanted to dump sites, we were ultimately studying the effects of dredged material disposal by comparing fitness of one population in two environments.

One of the major difficulties in the interpretation of results is the separation of the effects of normal physiological activities on trace metal uptake from uptake that is ascribable to anthropogenic input. Implicit in field experiments is the fact that data sets are correlational and cannot be assumed, unlike laboratory experimentation where independent variables can be carefully controlled and altered one at a time and the response of the organism to them accurately measured.

Two issues are addressed in the analysis of mussel trace metal data. First, we identified factors such as wet/dry ratio, length of mussels, and dredge volume, which singly or in combination with river runoff and temperature could be correlated to metal concentrations in mussels. Secondly, differing metal concentrations in mussels were determined.

Independent variables were categorized as intrinsic--i.e., variables associated with the physiological condition and growth of mussels expressed as the wet/dry ratio of tissues and shell length--and extrinsic factors that are variables in the environment, such as dredged material volume, river runoff, and water temperature. Since the purpose of the study was to determine

whether dredged material disposal, an extrinsic factor, is related to metal concentrations in mussels, it was extremely important that variance in metals concentrations due to intrinsic factors be accounted for or removed.

Data were subjected to intensive statistical analysis. The results indicated that peak concentrations tend to occur during periods of increased suspended load in the water column attributable both to natural meteorological events and dredged material disposal. These observations, when compared with sediment transport studies being conducted under the DAMOS program, showed that spatial and temporal effects of dredged material are several orders of magnitude smaller than those induced by natural storm events (4,7).

We have used mussels as sentinel organisms for 6 years now and have expanded their applicability to encompass site designation investigation, capping studies, and a cooperative effort with the joint Environmental Protection Agency/Corps Field Verification Program being conducted in Long Island Sound. In addition, levels of polychorinated biphenyls (PCB's) have been monitored and Dr. Feng's research has carried into mussel histology, in particular gonadal development and reproductive pathology (1,6,7).

Our current program is directed toward a single site in western Long Island Sound where three platforms have been deployed--on the mound center, at the mound periphery, and at the area reference station. An additional platform is maintained at the master reference station in eastern Long Island Sound to provide data on the stock population. Samples are collected monthly and analyzed for heavy metals and PCB's and subjected to histological examinations. It is hoped that the data will allow the perspective evaluation of disposal effects relative to the effects of other significant variables at this already stressed area.

Future plans will consider analysis of polynuclear aromatic hydrocarbons (PAH's) and metallothioneins in addition to the effort described above. Such changes will be assessed relative to program objectives and application to disposal concerns.

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ENVIRONMENTAL AND WATER QUALITY OPERATIONAL STUDIES (EWQOS) - APPLICATION TO LAKE RESTORATION

J. L. Mahloch

US Army Engineer Waterways Experiment Station
PO Box 631, Vicksburg, Miss. 39180-0631

ABSTRACT

EWQOS is the largest funded Corps research program directed at solving the environmental quality problems of reservoirs and inland waterways. Application of EWQOS technology to lake restoration falls into two categories: (a) evaluation for management decisions, and (b) specific techniques for improving environmental quality. Management decisions revolve around the ability to predict or monitor the response of a lake to proposed restoration techniques. Specific techniques to improve lake water quality such as aeration, destratification, and localized mixing were also developed under the EWQOS Program, and design procedures are available. A common manifestation of lake water quality problems is the appearance of nuisance algae blooms. Research in EWQOS has produced a review of common algal control techniques, algae succession data in response to eutrophy, and a new technique using destabilization and nutrient additions to promote the dominance of more desirable algal species. Many of the techniques applicable to lake restoration have been field tested or verified under EWQOS to facilitate their implementation.

INTRODUCTION

The Environmental and Water Quality Operational Studies (EWQOS) Program is currently one of the largest research and development efforts of the Corps of Engineers (CE). EWQOS has as an objective to develop technology for the CE to attain national environmental quality objectives while meeting authorized project purposes. While the scope of EWQOS covers all inland water resource projects of the CE, this paper will be confined to results from research on reservoir projects.

Reservoir projects of the CE face a myriad of environmental quality problems, ranging from impacts of releases on downstream aquatic habitat to problems associated with fluctuating pool elevations. These problems are compounded by operational constraints required to meet project purposes. Often, proposed solutions to water quality problems must be modified so as not to impose an unreasonable or uneconomical constraint on operational personnel.

One key problem facing CE reservoir projects involves the environmental impacts associated with increasing eutrophication. In developing solutions to this problem, differences between natural lakes and reservoirs must be kept in mind to avoid conflicts mentioned above; nevertheless, solutions to eutrophication problems developed within EWQOS have potential application to lake restoration.

Research conducted on reservoir projects within EWQOS has focused on a number of environmental problems. In addition to research tasks that address such topics as numerical modeling, solutions to specific problems, or design and operational guidance, a key area of reservoir research has been extensive and comprehensive field studies. Field studies on reservoir projects complement research tasks by providing necessary background data to evaluate problems, develop solutions, and permit verification of many techniques developed. While many reservoirs were used in these field studies, Eau Galle reservoir in Wisconsin provided a majority of the information applicable to lake restoration. Field studies, by virtue of the experience gained, also yield valuable insight into the development of monitoring guidance and procedures required to solve specific environmental problems.

APPROACH TO LAKE RESTORATION

In approaching the problem of lake restoration in response to eutrophication, a number of factors must be considered in developing a comprehensive solution: (a) a lake exhibits a unique condition or problem, often as a function of its design or operation, (b) given this condition, project modification (action) to improve environmental quality is performed, and (c) evaluation techniques are required to bridge the gap between condition and action to determine if solutions applied are feasible or successful.

Condition of CE Reservoirs

CE reservoirs often form unique ecosystems because of their design and operation. In comparison to natural lakes, they are generally larger, deeper, morphologically more complex, and have shorter hydraulic detention times (4). Many reservoirs are dominated by an advective flow that affects nutrient transport and biological response of the project. In the United States, many reservoir projects are constructed in areas where there are few natural lakes and the population is dense, often to meet project objectives such as flood control or water supply. Because of this situation, CE reservoirs have historically received large nutrient loads either by point or nonpoint sources. These characteristics have made eutrophication a major problem for many CE projects.

Eutrophication may adversely impact projects in a number of ways, although the primary symptomatic response is increased blooms of nuisance algal species. Nuisance algae species generally include the blue-greens and dinoflagellates. Increased productivity of nuisance algae may affect project activities by impacting recreation or causing taste and odor problems for water supply, and degrading biomass may adversely affect project water quality (18). Methods to control nutrient loadings are often outside the agency mission, and internal nutrient cycling can override any external controls; consequently, the only remedial action remaining is the alteration of algal production or succession to provide symptomatic relief for the problem.

Evaluation Techniques

Evaluation of the response of reservoirs to changes designed to reduce impacts associated with accelerated eutrophication is critical to any program of remedial action. In many areas the use of evaluation procedures can be very effective in determining the types of remedial action that are likely to be successful for a project. These procedures offer unique advantages to the decision-maker who is faced with several alternatives that may be costly to implement. Two principal techniques for project evaluation that have been developed from EWQOS research are nutrient loading models and simulation models.

Nutrient loading models developed under EWQOS were based on analysis of a comprehensive water quality data base (20, 21). Regression equations were developed to assess reservoir response to changes in nutrient concentration, in terms of chlorophyll a, transparency, organics, and hypolimnetic oxygen demand. An example of the network of equations developed for this purpose is presented as Figure 1 (22). While these nutrient loading models may be very

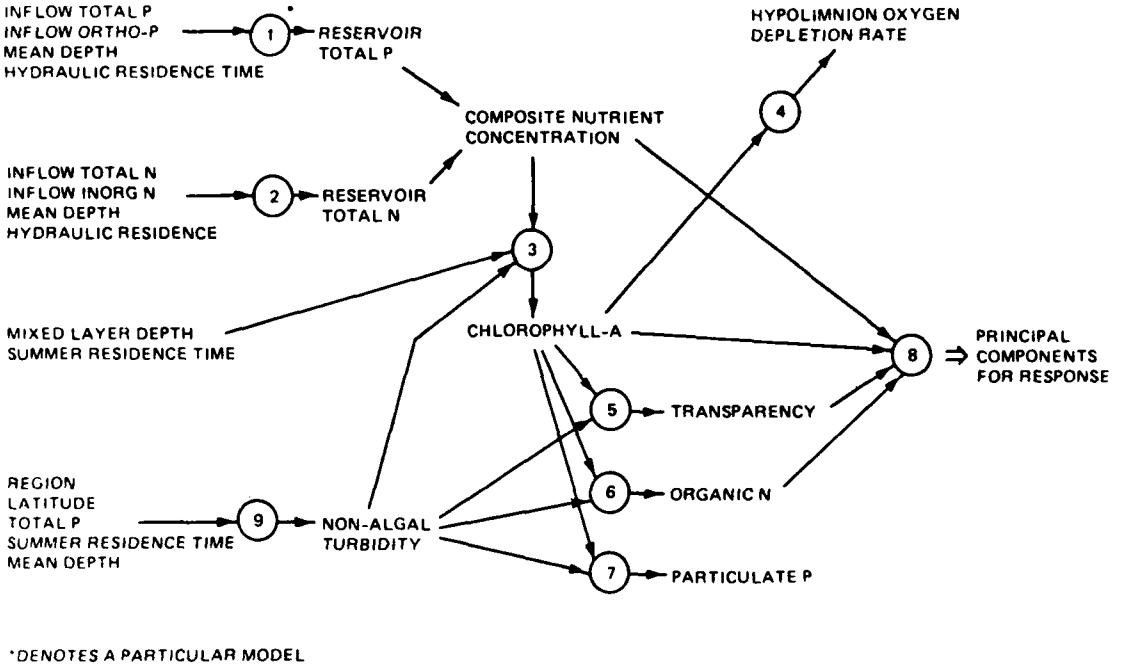


Figure 1. Network of nutrient loading models for evaluating reservoir eutrophication.

helpful in identifying and pinpointing causal relationships associated with reservoir eutrophication, they are based on certain assumptions; therefore, their use is limited. Primary limitations result from the range of the data base employed, the fact that conditions are reservoir averaged, and errors are associated with model use. Nevertheless, nutrient loading models represent excellent procedures for screening alternatives and conducting an initial evaluation of project response.

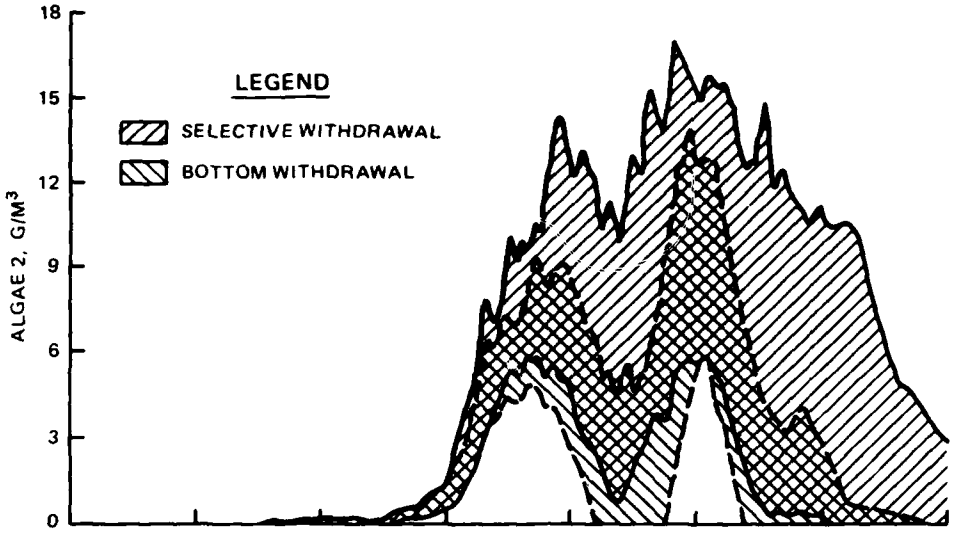
Simulation models are numerical techniques designed to evaluate the water quality of a reservoir in response to changing conditions. By their very design, simulation models are more general and useful than regression models; however, they are also more costly and time consuming. The water quality models developed under EWQOS are capable of dynamically simulating reservoirs in a one- or two-dimensional sense. In addition to complete prediction of common water quality parameters, higher trophic levels, such as algae, are also simulated by these models (7, 9, 10). In this case, the magnitude, composition, and timing of algal blooms and factors controlling algal growth can be predicted. An example of these types of predictions for structural and operational changes to proposed CE reservoirs is shown in Figure 2 (18). These models can give detailed information on the response of a project to proposed changes for relieving the symptoms of eutrophication. In many cases the simulation models developed under EWQOS have had extensive verification, and rate coefficients required for application are based on results of extensive field studies mentioned earlier. Although this imparts a significant advantage to model use, the user must always remember that certain aggregations are inherent in model formulation; therefore, interpretation of output can be tedious and complicated.

Remedial Actions for Reservoirs

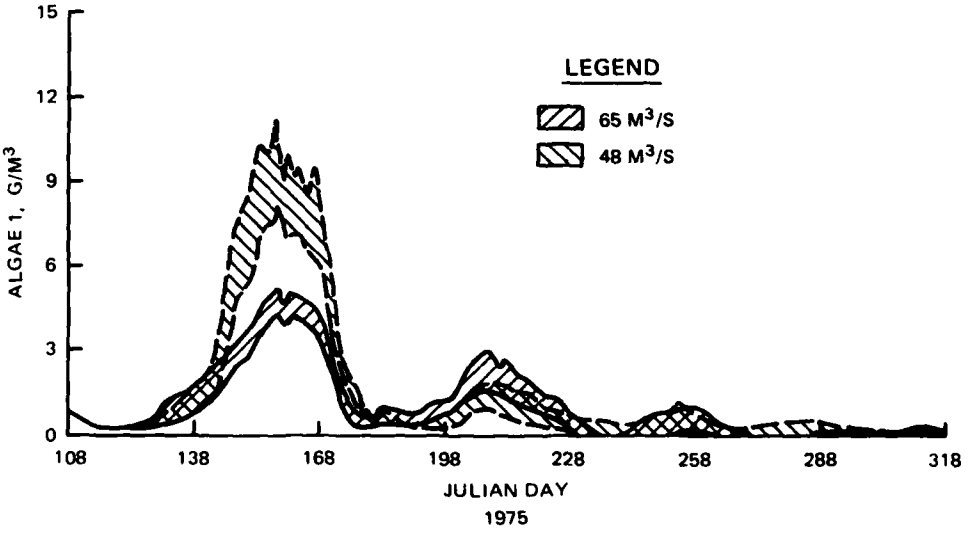
Faced with a lake or reservoir exhibiting accelerated eutrophication, an agency may have several options for remedial action. The remainder of this paper discusses options designed to retard the development of nuisance algal blooms associated with eutrophication.

In many instances, CE reservoirs follow a rather typical succession of algae common to many dimictic, North temperate lakes. These reservoirs exhibit an early spring bloom of diatoms, followed by green, blue-green, and dinoflagellate algae. After fall turnover, a diatom bloom may reoccur. The blue-greens and dinoflagellates are the most troublesome and would be the primary target for remedial action. Remedial actions might include elimination of these species in the successional cycle and/or reduction in the total amount of algal biomass produced. Major reductions in productivity may be undesirable for most CE projects because of associated impacts (e.g., reduction of reservoir fisheries); consequently, it is more desirable to alter algal community composition than to reduce overall productivity.

Early in EWQOS, a summary of common algal control and management techniques was compiled (6). Techniques surveyed included biological, chemical, and physical methods and a combination of these for algal control. This summary offers an excellent initial assessment of commonly used control techniques. Because of differences between natural lakes and reservoirs (primarily operational), many of the techniques presented in this summary may not be directly applicable to CE reservoirs. Therefore, a workshop on algal management and control was conducted (19) to evaluate major algal control techniques. The workshop concentrated on techniques for nutrient control, aeration/destratification, biological control (i.e., grazing, cyanophages), and chemical usage. Although the success of many techniques was documented for specific cases, no technique appeared universally successful or appropriate. A conclusion of the workshop was that emphasis needed to be placed on



a. STRUCTURAL MODIFICATION



b. OPERATIONAL MODIFICATION

Figure 2. Algae response to project changes.

algae control techniques that minimize adverse environmental impacts, specifically those techniques that take advantage of factors influencing algae composition and productivity.

In many cases, accelerated eutrophication in reservoirs is reflected in poor water quality. Often most significant is a depression of hypolimnetic dissolved oxygen. Remedial techniques may be applied to improve water quality, but related (specifically, biological) impacts have been only superficially considered. Technology related to reservoir mixing, destratification, and aeration has been developed within EWQOS, and a variety of documentation exists as guidance in the application of these techniques (3, 5). The environmental aspects of applying these techniques have also been evaluated (11).

Another example of a physical solution to nuisance algae control is the use of pool elevation control (6). While this may be successful in altering successional patterns of algae, care must be exercised in timing and the degree of change. A recent EWQOS publication has documented the effect of pool elevation changes on reservoir fisheries (12) and can be used for an initial evaluation if this technique is applied.

STUDIES AT EAU GALLE RESERVOIR

On the basis of available information, a comprehensive study of algal control was undertaken at Eau Galle reservoir, one of the major reservoir field study sites in EWQOS. Two primary factors influenced the direction of the work performed at Eau Galle reservoir. First, there was a general lack of a comprehensive study of algae control within CE impoundments. Eau Galle reservoir may be classified as eutrophic, exhibiting intense algal blooms, and is a rather small reservoir; consequently, intensive studies could be undertaken at a rather low investment in resources. Second, because of a coordinated field study, factors influencing algae succession and production could be identified, and experimental control could be attempted on the basis of this information.

Eau Galle Reservoir

Eau Galle Reservoir is located in west-central Wisconsin and was created by the impoundment of the Eau Galle River in September 1968 (8). The primary purpose of the project is flood control. Normal releases are made through an uncontrolled morning-glory structure and a low-level gated outlet conduit. The lake is dimictic, and ice cover usually persists from December until late March. A summary of the physical and water quality characteristics of the lake is provided in Table 1.

Algae Composition and Succession

A major prerequisite for establishing a control program for algae is a thorough understanding of phytoplankton ecology, including factors controlling growth and succession. A comprehensive study of phytoplankton composition and succession was undertaken at Eau Galle reservoir in association with other limnological studies. Methods included algal enumeration, taxonomic identification, biomass, chlorophyll *a*, and productivity, which are documented in

TABLE 1. CHARACTERISTICS OF EAU GALLE RESERVOIR

Parameter	Value
Physical	
Surface area	0.6 km^2
Volume	1.5×10^6 m^3
Depth (mean, max.)	3.2, 9.0 m
Shoreline development ratio	1.50
Drainage area	166 km^2
Residence time	0.07 yr
Water Quality*	
Total alkalinity	152.3 mg/ℓ
Total carbon	47.4 mg/ℓ
Total phosphorus	0.091 mg/ℓ
Total nitrogen	1.65 mg/ℓ
Chlorophyll *a*	31.9 µg/ℓ

* Mean surface value (0-2 m) from six stations for years 1978-1981.

detail elsewhere (1). Sampling stations are shown in Figure 3. Studies were initiated in 1981 and are continuing in conjunction with the experimental work on algae control.

Results from Eau Galle (1) showed a seasonal succession of algae typical of eutrophic, temperate lakes, with spring diatom followed by summer phytoplankton followed by fall diatoms (Figure 4). In 1981, the cyanophyte *Aphamizomenon* and the pyrrhophyte *Ceratium* predominated the summer phytoplankton. Algal biomass during periods of peak production is comparable to other temperate, eutrophic lakes. Patterns of succession were generally the same during 1982 and 1983, although they were not as pronounced (data not presented).

Development of Algal Control Experiments

The success of earlier attempts to control nuisance algae have generally been impaired due to an incomplete understanding of the specific environmental factors controlling growth and succession. In a number of situations, experiments to improve lake water quality or to control algae have been inconclusive because associated limnological data on which to adequately judge the success of techniques were not available (17).

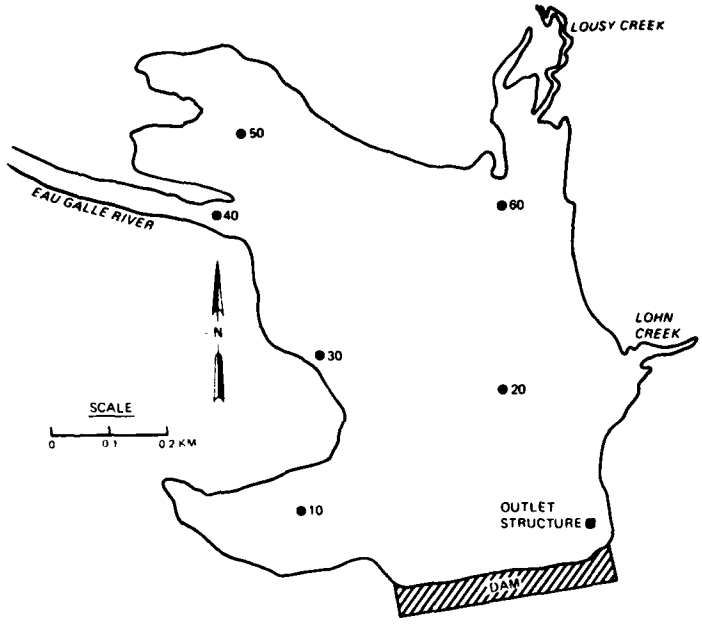


Figure 3. Sampling stations at Eau Galle reservoir.

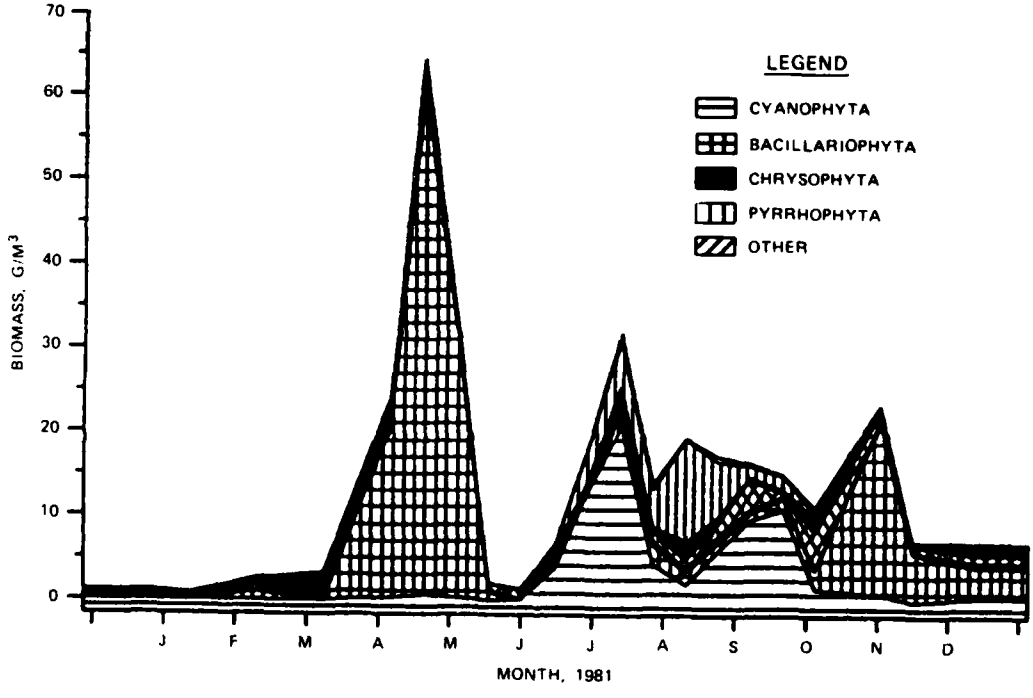


Figure 4. Seasonal succession of algae at Eau Galle reservoir.

Results of algae community studies at Eau Galle suggest that mixing (or the lack of mixing) and nutrient availability control algae succession. Diatoms tend to predominate during mixed conditions because of high nutrients (particularly silica) and suspension in the euphotic zone due to turbulence. Depth of mixed layer appears to be critical to avoiding the loss of diatoms by settling (13). As stratification occurs, species that can selectively migrate in the water column (blue-greens, dinoflagellates) are favored. Reduced nutrient availability, particularly nitrogen for the blue-greens, may contribute to their competitive advantage. Similar results have been reported elsewhere (13, 14, 15, 16), which indicates the importance of manipulating stability and nutrient availability as means for controlling algal succession and biomass. In some cases, mixing may be counterproductive. While it encourages a more desirable species (i.e. diatoms), increased nutrient availability from the hypolimnion and resuspension of resting forms (i.e. blue-greens) may offset any gain realized.

Control Experiments at Eau Galle

Based on the above, a series of experiments were implemented at Eau Galle during the summer of 1983. Experiments were conducted in polyethylene enclosures, 10 m in diameter, to minimize artificiality. A similar approach has been used by others with success (14). Treatments used were mixing alone, addition of silica, and a combination. Mixing was provided by a Garton pump with a deflector plate to minimize sediment resuspension, and silica (sodium metasilicate) was dissolved gradually in the enclosures during the experiment. A separate enclosure was used as an experimental control, and the open water served as an overall control (2).

Results of these experiments are summarized in Figure 5 (2). Silica treatment was ineffective for maintaining diatoms or suppressing the

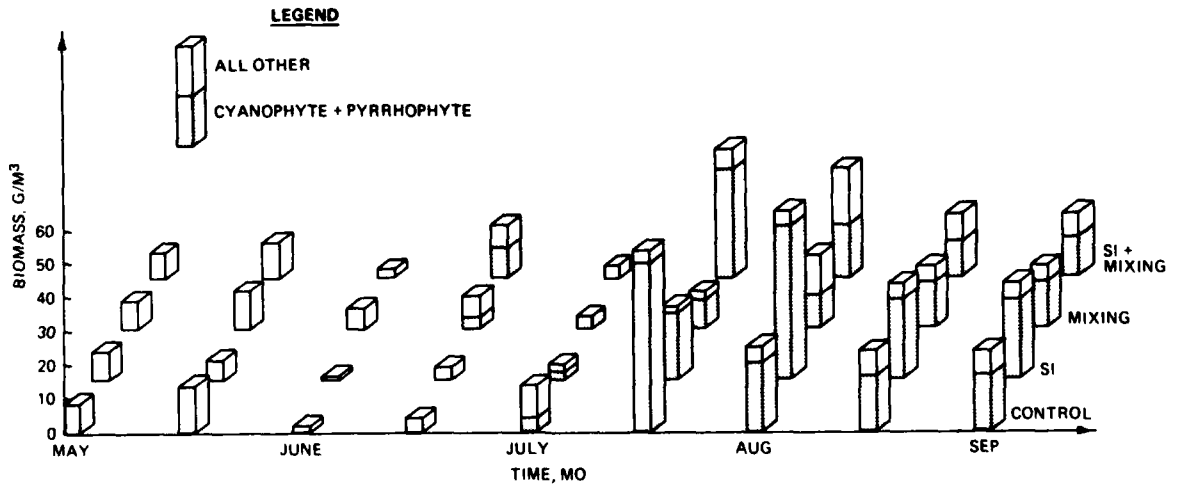


Figure 5. Experimental control results at Eau Galle reservoir, 1983.

blue-greens. Silica addition and mixing in combination did not deter the dominance of nuisance forms of algae (i.e., blue-greens and dinoflagellates), a result similar to the findings of other investigators (14, 15). Mixing alone resulted in a sustained diatom presence, but blue-green biomass was

large, although the dinoflagellates were suppressed. Mixing may have provided a source of blue-green algae as well as contributing to increased nutrient supplies. This is particularly true in the case of Eau Galle reservoir, where the water column is particularly shallow. In conclusion, results of mixing were favorable, although clearly there were other factors controlling algae succession and production. At the end of these experiments, alum was added to the silica plus mixing treatment in an attempt to control nutrient (phosphorus) availability. Although there was an immediate response (i.e., reversion to green algae), it was too late in the experiment to confirm whether this treatment would be successful over an entire season. This finding did suggest that control of phosphorus availability in conjunction with mixing may lead to more effective algae control (2).

Based on the results of experiments conducted during 1983, additional experiments were designed for the 1984 season. Treatments currently being used include silica sand addition (bottom sealing), silica sand addition plus mixing, alum addition, alum addition plus mixing, and a control. Experiments are being conducted in a fashion similar to those of the 1983 season. Alum addition is being used to control the availability of phosphorus. Silica sand addition to the bottom is an attempt to control nutrient migration across the sediment-water interface, to prevent reinnoculation of the water column, and to provide a source of silica for diatoms. The mixing treatment was continued because it appears to be a controlling factor in the maintenance of diatoms.

Preliminary results (after 6 weeks of investigation) suggest that all treatments are successful. The succession seems more favorable, and the amount of algae produced appears to be lower in all treatments. Confirmation of these preliminary results will have to await a thorough analysis of the 1984 data. A demonstration project on the whole lake may be attempted in 1985.

SUMMARY

The EWQOS Program has produced many techniques that are applicable to lake restoration. A variety of techniques plus guidance on sampling and data analysis are available to assess problems, develop potential solutions, and judge their effectiveness. Reviews of commonly employed techniques for algae control and design guidance for physical alteration of lake environments are also available. Efforts directed at controlling the primary problem associated with eutrophication, the presence and abundance of nuisance algae, are presently under way. The advantages of the experimental techniques currently under study are that they employ natural control mechanisms to affect algae succession and production, and they can be applied to lakes with minimum adverse environmental impacts.

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