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31 January - 1 February 1984
Portland, OR

SP-15

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A seminar on Applications in Water Quality Control was held on 31 January - 1 February 1984 in Portland, Oregon. The purpose of the seminar was to provide a forum for Corps of Engineers personnel who are routinely involved in water quality and water control work. Topics included cooperative efforts of local, state, and federal agencies, water quality management, data monitoring, laboratory quality control, database management, reservoir water quality control, dredging, estuarine concerns, and specific applications of reservoir water quality problems.

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31 January - 1 February 1984

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Triangle J. Council of Governments

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FOREWORD

A two-day seminar on "Applications in Water Quality Control" was held in Portland, Oregon, on 31 January - 1 February 1984. The purpose of the seminar was to provide a forum for Corps of Engineers personnel who are routinely involved in water quality and water control work.

Topics addressed during the seminar included Cooperative Efforts of Local, State and Federal Agencies to Improve Reservoir Water Quality, Water Quality Management, Water Quality Data Monitoring, Laboratory Quality Control, Data Base Management and Interpretation, Reservoir Water Quality Control, Dredging Concerns, Estuarine Concerns and Specific Applications with Reservoir Water Quality Problems. Twenty-five of the papers presented during the seminar are contained herein.

A highlight of the conference was the viewing of a video tape taken from a two-man research submersible at depths between 700 - 1200 feet on the Hawaiian coast. The footage included previously unseen fish and other biotic species from bottom environments at sites previously used for disposal of dredged materials.

Two optional field trips were arranged by Dr. Tanovan for 2 February. The trips included either a visit to Bonneville Dam or a field inspection of the dredging operations along the Cowlitz River near Mount St. Helens. The appendix includes some of the materials distributed during the trips.

The seminar was co-sponsored by the Hydrologic Engineering Center and the Committee on Water Quality. This seminar proceedings, in addition to the general seminar coordination, was organized by Mr. R. G. Willey of the Hydrologic Engineering Center. Valuable assistance was graciously provided for coordination of the separate sessions by Messrs. Richard Jackson, Wilmington District; Dave Cowgill, NCD; Mark Anthony, ORD; Robert Engler, WES; and Tom Dillion, WES. The conference room, individual rooms and all local arrangements were organized by Dr. Bolyvong Tanovan from the North Pacific Division Office.

The views and conclusions expressed in these proceedings are those of the authors and are not intended to modify or replace official guidance or directives such as engineering regulations, manuals, circulars, or technical letters issued by the Office of the Chief of Engineers.

R. G. Willey
Editor

SEMINAR

ON

APPLICATIONS IN WATER QUALITY CONTROL

31 January - 1 February 1984

Portland, Oregon

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CONTENTS

COOPERATIVE EFFORTS OF LOCAL, STATE AND FEDERAL AGENCIES TO IMPROVE RESERVOIR WATER QUALITY: FALLS OF THE NEUSE RESERVOIR, NORTH CAROLINA

By

Edward A. Holland[1]

INTRODUCTION

Corps of Engineers multi-purpose reservoirs require suitable levels of water quality to fulfill their authorized uses. The Falls of the Neuse Reservoir in North Carolina's Research Triangle Area (Raleigh, Durham, Chapel Hill) was authorized in 1965 and impounded in 1983 for public water supply, flood control, downstream flow augmentation, recreation, and fish and wildlife enhancement. Main features of the reservoir and watershed are listed in Table 1.

Falls Lake typifies the water quality dilemma faced by many Corps of Engineers reservoir projects: the Corps has little or no regulatory authority for assuring the water quality on which those projects depend. When the Falls Project was authorized, years before landmark Federal and State water quality legislation, the Nation's principal water quality concerns were the "conventional pollutants": BOD, suspended solids, and bacteria. Since then, our technical knowledge of the issues and the public's demand for clean water have increased dramatically.

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[1] Director, Resource Conservation, Triangle J Council of Governments, Research Triangle Park, North Carolina

PHYSICAL SETTING

The 770-square mile Falls watershed is located in the Piedmont physiographic province and headwaters of North Carolina's Neuse River Basin. The watershed area exceeds the reservoir surface by a factor of 40. It includes portions of six different counties and four municipal jurisdictions and contains a variety of pollution sources. Improving and protecting long-term water quality will require the cooperation of several State, Local and Federal agencies that share the fragmented responsibility for watershed protection.

Table 1: Features of the Falls of the Neuse Reservoir and Watershed

Surface Area	12,500 acres
Volume	153,800 acre-feet
Depth	12.3 feet
Average Streamflow	794 cfs
Retention Time	100 days
Watershed Area	770 square miles
Population (1980)	136,000
Land Use (1982)	
Urban, Residential	10%
Agriculture	21%
Forest	62%
Other	7%
Total	100%
Sewage Treatment Plants	6
Total Flow	10 mgd

THE POLITICAL SETTING

Although Falls Reservoir suffered no use impairment during 1983 - its first year of impoundment - public concern about water quality has remained high. The Raleigh-Durham-Chapel Hill area is one of the fastest growing regions in the Southeast. Population in the watershed is expected to increase by 50 percent during the next 20 years. Falls Reservoir will become the City of Raleigh's sole source of drinking water in 1985 and has been the subject of vigorous public and editorial concern about the effects of population growth on water quality. Raleigh is the State's capital city, and the Research Triangle Area is regarded as the "crown jewel" of North Carolina's industrial recruitment efforts. Governor James B. Hunt, Jr., and chief environmental policy makers have repeatedly challenged local governments and State agencies to protect the Falls Reservoir and the Research Triangle from uncontrolled urban growth.

The Triangle J Council of Governments, a regional planning agency for much of the Falls watershed area, was the nation's first recipient of an EPA Section 208 Areawide Water Quality Planning grant in the mid-1970's. As a voluntary association of local governments, Triangle J has no regulatory authority over water quality issues, but has developed a strong role of regional leadership and technical expertise in North Carolina.

Principal federal agencies in the Falls water quality efforts have been the Corps of Engineers and USDA Soil Conservation Service. The Corps maintains strict control over the use and access to an extensive greenbelt around the entire reservoir and supports a detailed water quality monitoring program that is closely coordinated with the State's efforts. The Soil Conservation Service recently completed a PL-566 erosion study of all agricultural land in the Falls watershed and provides technical guidance to individual farmers and county Soil and Water Conservation Districts.

THE CALL FOR ACTION

Local awareness and concern about water quality increased in early 1983 when Falls Reservoir was impounded. Elected officials and the local media focused attention on the area's intense development pressure and objected to the State's proposal to issue discharge permits for package wastewater treatment plants adjacent to the lake. Officials were concerned about the complex and fragmented array of technical and institutional issues. Under the leadership of Triangle J, several local jurisdictions petitioned the Secretary of North Carolina's Department of Natural Resources and Community Development for cabinet level State leadership in resolving the tangled array of issues. The Secretary responded by convening a special Steering Committee of county board chairmen and mayors from the principal jurisdictions in the watershed and establishing an agricultural technical committee of supervisors from the watershed's five Soil and Water Conservation Districts. The North Carolina Environmental Management Commission, the state's top environmental policy making body, initiated a reclassification procedure for the newly impounded reservoir. Issues included reclassification as a public water supply source and possible designation as "Nutrient Sensitive Waters," an action that would allow State regulation of nutrient discharges throughout the 770-square mile watershed.

WATER QUALITY ISSUES

The disparate issues of urban and rural runoff, package treatment plants, upstream discharges, construction site runoff, toxic spills and leaks are discussed under the general categories of sediment, nutrients, and toxics.

Sediment

The recently completed SCS Erosion Study estimated the sediment delivery to Falls Reservoir to be 257,000 tons per year, of which agricultural activity contributed more than 80 percent. Only 3 percent of the sediment came from forested portions of the watershed. The remaining 15 percent was attributed to urban areas, construction sites, roadsides, and streambank erosion. With a goal of reducing gross cropland erosion to five tons per acre per year, SCS estimated that total sediment loads to the reservoir could be reduced by 25 to 35 percent through the use of Agricultural Best Management Practices (BMPs) at an estimated cost of 12 million dollars spent over 10 years.

In addition to agricultural erosion control, city and county governments were encouraged to adopt and enforce strong local erosion and sedimentation ordinances for construction activities. Two of the three metropolitan counties in the Falls watershed already had effective ordinances in place.

Nutrients

One of the long anticipated problems for the Falls Reservoir has been its eutrophic potential. The Reservoir receives a phosphorus load of 397,000 pounds per year from the following sources:

Agriculture	26%
Urban Runoff	23%
Forest	7%
Wastewater Discharge	44%
Total	100%

North Carolina's Division of Environmental Management predicted total phosphorus concentrations of 0.156 and 0.080 mg/l in the upper and lower sections of the reservoir, respectively. Chlorophyll a levels were predicted to be 110 and 42 ug/l for the upper and lower segments, which would exceed the State's 40 ug/l chlorophyll a standard. State officials estimated that the 397,000 pound annual phosphorus load could be reduced 40 to 50 percent through the following methods:

Agricultural BMPs	8%
Wastewater Treatment Plant Removal	35%
(Phosphate Detergent Ban Only)	10%
Runoff Control for New Development	5%

Because the probability of nuisance algae conditions appeared to be high, the Environmental Management Commission reclassified the entire watershed Nutrient Sensitive. All existing wastewater dischargers were immediately notified that they might be required to remove phophorus down to a level of 1 mg/l, depending on the lake's trophic response over the next 2 to 5 years, and depending on local measures taken to control nonpoint phosphorus sources. Additionally, the Environmental Management Commission imposed the phosphorus removal requirement on all new dischargers in the watershed.

Toxics

Local officials and the general public have reflected the growing state and national concern about toxic materials in drinking water. Although there has been no direct evidence for a toxics problem in Falls Reservoir, many existing and proposed activities in the watershed involve the use of toxic materials. Release to the aquatic environment seemed inevitable. Concern focused on municipal and industrial dischargers, urban and industrial runoff, sanitary landfills, agricultural pesticides, and transportation spills, especially from highway crossings over the lake. Despite the lack of data indicating a problem in the Reservoir, the fear about toxics in a public water supply remains one of the strongest political motivators for water quality protection in the Falls project.

STATE/LOCAL ACTION AGENDA

Responding to the local call for leadership, State officials offered a "carrot and stick" partnership to the local communities. As outlined above, the Environmental Management Commission reclassified Falls as Nutrient Sensitive and notified existing wastewater dischargers (including four municipalities) that they might be subject to phosphorus removal at their treatment plants. Because of the lack of regulatory control over nonpoint source pollution, the State offered to negotiate with local governments: "If you (local governments) take strong actions to reduce nonpoint pollution, then we (the State) may not have to impose expensive phosphorus removal requirements at your treatment plants."

Local officials responded positively, but requested "uniform guidelines" with which to measure their compliance with the EMC mandate and to compare the progress of one local jurisdiction to another. State officials and the Triangle J Council of Governments worked together and proposed a "State/Local Action Agenda" to the local governments represented on the Falls Reservoir Steering Committee. Highlights of that document are summarized below.

State Actions

- Water quality monitoring and research to confirm suitability as a water supply, especially with respect to toxics.

- Incentive funding for individual farmers implementing agricultural BMPs.

- Additional funding and manpower for the State's erosion control program for construction activities.

- Legislative support for a phosphorus detergent ban if requested by local governments.

- General technical assistance to local governments implementing the Action Agenda.

Local Actions

- Review sewer use and industrial pretreatment ordinances for controlling toxics and synthetic organic chemicals.

- Inventory the storage of hazardous materials by local industries and institutions.

- Review emergency response capabilities for toxic spill containment.

- Resolve the critical agricultural erosion problems within each Soil and Water Conservation District.

- Adopt and enforce sedimentation and erosion control programs for construction activities in jurisdictions without such ordinances.

- Apply stricter land use control measures to "Water Quality Critical Areas" nearest to Falls Reservoir.

- Limit the impervious surface areas of new development and maintain 50-foot vegetated buffers along all perennial streams.

- Discourage urban level development in the Water Quality Critical Areas.

Local governments agreed in principle with the proposed State/Local Action Agenda, but asked for additional guidance in delineating the proposed Water Quality Critical Areas and development standards. Further collaboration by State, local, and Triangle J staff produced the following guidelines:

- Designate a Water Quality Critical Area perimeter at least one-half mile beyond the Corps of Engineers property line.

- Modify local sewer extension plans and policies to discourage urban level development in the Critical Areas.

- Restrict impervious areas for all new development to six percent in the Critical Areas. Allow no new non-residential development except small offices and neighborhood businesses in the Critical Areas.

- Limit impervious coverage in new development outside the Critical Areas to 12 or 30 percent, depending on the availability of public sewer service.

- Apply Special Use requirements to any industries that produce, store, treat, or use hazardous materials in areas draining to reservoir segments classified for water supply.

- Control the first half-inch of stormwater runoff from impervious surfaces in all new developments, with natural infiltration as the preferred method.

PRESENT STATUS

At the time of this writing, several local jurisdictions have already modified, or are in the process of changing, zoning ordinances and subdivision regulations to comply with the proposed guidelines. Although subject to considerable local debate and some resistance, recent progress has been impressive. State officials have launched a legislative budget initiative for funding and expect local support when the General Assembly convenes this Spring. The Soil Conservation Service continues to provide much needed technical support to the county Soil and Water Conservation Districts, and the Corps of Engineers is funding most of the water quality monitoring work on Falls. More precise estimates of the reservoir's toxic and trophic status will depend largely on additional monitoring and analysis. The University of North Carolina's Water Resources Research Institute is likely to fund one or more studies proposed for the reservoir.

SUMMARY

The Falls of the Neuse project typifies many Corps of Engineers multipurpose reservoirs: it requires good water quality in order to meet the project's intended uses. The Corps has little regulatory authority with which to assure this goal. Existing authorities are fragmented among State and local agencies. Existing data are insufficient to predict accurately the lake's trophic response to the nutrient reduction strategies and to answer public concerns about toxic materials. Overall needs include political and technical consensus, leadership, commitment, funding, and coordination to carry out the complex task of watershed protection. The Corps of Engineers can play an important role in this effort - through ongoing water quality monitoring and through its open and frank cooperation with State and local officials. Recent Federal requirements for front-end local cost sharing of Corps feasibility studies and water supply projects will require a high degree of cooperation between the Corps of Engineers and outside agencies. Many of the Corps' new proposals may be judged on the basis of its past performance in projects such as the Falls Reservoir.

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WATER QUALITY MANAGEMENT

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INTRODUCTION

A necessary prerequisite to any successful program is the careful organization and utilization of one's resources. This has been particularly true lately when budget cuts and manpower shortages have been the rule rather than the exception.

This presentation will attempt to discuss some of the many considerations in the management of a water quality program. It should be realized at the onset that each individual situation will dictate, to some degree, the exact nature of program needed. Thus it will not be possible to discuss in detail the "nuts and bolts" of how all water quality programs should be managed. This will, of course, vary from district to district and will depend on some of the factors we will attempt to discuss.

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District Responsibilities

In recent years, the Corps of Engineers' concern for the environment has encouraged the construction and operation of environmentally sound projects. This is particularly true in the area of water quality where numerous directives have been issued which define the Corps responsibility and offer guidance in the design and implementation of effective water quality monitoring programs. The following publications are most applicable in this regard:

1. ER 1105-2-8, 24 Sept 1973, Responsibilities for Study Accomplishment and Participation of Other Agencies. This ER fixed the Corps with primary responsibility for "the general environmental effects of its projects, including the responsibility for accomplishing related water quality studies and activities."

2. ER 1130-2-415, 28 Oct 1976, Water Quality Data Collection, Interpretation, and Application Activities. This ER directed water quality data collection at Corps water resources projects. It said: "Water quality data will be collected at existing projects in order to: a) establish baseline conditions and monitor subsequent changes; b) identify water quality environmental problems; c) provide continuing guidance to reservoir regulation elements; d) study special problems; and e) provide an adequate data base and understanding of project conditions. Field offices should take the initiative in identifying problems and formulating subsequent studies leading to the solution or control of these problems."

3. ER 1130-2-334, 16 Dec 1977, Reporting of Water Quality Management Activities at Corps Civil Works Projects. This ER established the requirement for reporting water quality management activities to OCE.

4. ER 1110-2-1402, 15 Sept 1978, Hydrologic Investigation Requirements for Water Quality Control. Among other things, this ER directed field operating agencies to: "Develop water quality management objectives relative to the specified standards and the present and prospective needs of users of the impounded and released waters."

5. ER 15-2-10, 25 May 1979, Committee on Water Quality. This ER established a committee on water quality and directed, among other things, that the committee render service on specified problems to Corps elements.

6. NCDR 1110-2-23, 7 June 1982, Water Control Management, Water Quality. This regulation furnished guidance to NCD districts on water quality surveys, funding, annual report requirements, and other actions required.

7. ER 1110-1-261, 29 Oct 1982, Control of Field Testing Procedures. This ER required standards of capability for testing by laboratories.

8. ETL 1110-2-281, 17 June 1983, Reservoir Contaminants. This ETL furnished guidance for screening of Corps reservoir projects to determine the presence or absence of contaminants. It said: "It may be necessary in some cases to assemble more information. This additional action may take the form of obtaining more difficult to locate reports or data, or it may involve additional sample collection and analysis."

The 404 Program.

In addition, with the passage of PL 92-500 (FWPCA, 1972) the Corps must comply with criteria established by respective States in our operation and maintenance of existing projects. This includes obtaining the necessary permits required for disposal of dredge spoil. As a result, the importance of establishing meaningful water quality monitoring programs is greater than ever. Due to the complexity of the subject and the limited availability of resources, it is essential that adequate steps be taken to avoid generation of data which are inappropriate or useless for the intended purpose.

Establishing Water Quality Management Objectives.

A primary objective in water quality management is planning to avert water quality problems. This process must begin very early if data collection is required, as this may take years to accomplish. Too often "after the fact" water quality monitoring programs are established in an effort to solve problems, when far less effort would have been required to avert these problems in the first place.

In order to accomplish this goal, coordination of all parties involved is critical. This coordination should not be limited to elements within the office but should include the appropriate Local, State, and Federal agencies so as to avoid both the duplication of efforts and data gaps.

Similarly, the design of monitoring programs at existing projects should begin as soon as possible and is ideally an on-going process. This not only affords continuity in the data but also eliminates last minute decision-making which so often leads to unforeseen difficulties.

It is impossible to over emphasize the importance of establishing and clearly defining management objectives prior to "jumping in" and initiating a program. One should ask the question, "What do I want to determine?" It is not always necessary to have very specific objectives in mind. For instance, it may be quite informative simply to establish baseline water quality conditions prior to project construction. It should be realized, however, that data of this sort will have limited applications and may not be appropriate to answer such questions as:

1. Will the project adversely impact existing water quality?

2. To what extent will water quality be impacted?

3. Can project operational changes reduce these impacts?

4. Can water quality impacts be minimized by using alternative design criteria?

Such questions may only be answered by designing very specific programs or by using predictive tools such as numerical models. In any event, one must first determine what is desired, then design the program accordingly.

Development of Water Quality Strategies.

The next step is to develop management methods or strategies. This involves analyzing the available resources (financial, personnel, and facilities) and determining how your objectives can best be met. In some cases, all the necessary resources may exist in-house, and it is simply a matter of mobilizing these forces. When one or more of these resources is lacking, it may be necessary to consider alternative courses of action. Some of the options available include:

1. Contracting services to State or Federal agencies, Universities, or private consulting firms.

2. Seeking assistance from the Committee on Water Quality.

3. Seeking assistance from WES.

4. Seeking assistance from HEC.

5. Seeking assistance from other Corps Districts.

In any case, having sufficient "lead time" will allow for careful consideration of each alternative and selection of the most appropriate course of action.

Monitoring and Modifying the Program As Necessary.

Once the project has begun it is quite beneficial to periodically review the progress of the work and to modify the program if necessary. Quite often interim report preparation forces the investigator to critically analyze the results and may reveal questionable data and/or data gaps. If noted in time, erroneous data may be corrected. Should the data prove to be correct, yet unexplainable, additional studies may be necessary. This technique of periodic data review is quite common when contractors are being utilized but is also particularly effective when all work is being done in-house. Too often individuals become caught up in the day-to-day activities of data collection while losing sight of the overall objectives. By taking time to study the findings during the data collection process, one may be able to identify deficiencies in time to remedy the situation. This "feed back" process results in a dynamic program which can be adjusted to meet the needs of the particular problem. For instance, sampling stations may be added or eliminated; parameters may be added or eliminated; and sampling frequencies may be increased or decreased as necessary to optimize use of resources.

Long Term Controls and Applications.

Ultimately it will be necessary to evaluate the findings in terms of your original objectives and to recommend appropriate courses of action. This may result in additional problem identification requiring more extensive sampling.

This iterative process should, however, refine the program in terms of parameters analyzed and sampling techniques employed until adequate data exist to permit problem solution.

Occasionally, however, long term solutions require continuous monitoring. Such is true at existing projects which experience project-induced water quality problems. In these instances it is desirable to limit sampling to the "problem parameters" or suitable "indicator parameters" which can be useful in documenting water quality problems. Usually these parameters can be related to human health hazards or degradation of water quality which directly impacts aquatic organisms.

From time-to-time, water quality studies also identify problems inherent in the agricultural or industrial practices in the watershed. Project-induced problems can sometimes be alleviated by structural modification or by operational changes in our dams,etc. Basin problems require legislation and/or education of the public to produce meaningful changes.

What is a Good Water Quality Program?

As is the case in so many other areas, it is difficult to develop a water quality program which would be appropriate for all circumstances. Each district has unique problems which require individualized approaches to solve. Obviously, fresh water rivers, reservoirs, and natural lakes have different problems from estuaries. Groundwater investigations are distinct from all of these. Just as there is no single approach to water quality management there is no definitive size for a water quality organization-whatever works best.

There are a few principles which we have found to be essential to any good water quality program. First, the water quality function should be assigned to one responsible element. A centralized responsibility is preferred to a fragmented one. Lines of communication are concise and there are economies of manpower and funds in such a design. There is also a good chance that a more professional organization can be developed if all water quality responsibilities are concentrated in one organization.

Secondly, adequate funding must be programmed in advance. Considerable planning is required and flexibility is a necessity. Pre-authorization water quality studies should be funded by the General Investigation (GI) program. Post-authorization studies (including during construction) should be funded by the Construction General (CG) program. Water quality at completed projects should be funded by the Operations and Maintenance (O&M) program. Often budget cuts will dictate that water quality studies be flexible.

There is no substitute for technically-qualified personnel in a water quality organization. Water quality is a complex technical subject requiring a number of scientific disciplines plus a certain amount of managerial talent. The recruitment of adequately-trained personnel is an absolute necessity.

Finally, a water quality program must be dynamic. It must be able to respond to changing levels of funding, to changing uses of water and to unexpected problems. It must continually be reassessed to determine if objectives and needs are still being met by the current program.

In the Rock Island District water quality problems are often closely tied to suspended sediment transport. Due to manpower constraints we have a small water quality group in the District. Our Water Quality and Sedimentation Section, consisting of 4 full-time individuals and part-time student aids, handles the majority of the work load through a combination of in-house studies and various contracting efforts. In is located within our Hydraulics Branch.

In recent years, study efforts have concentrated in the areas of:

1. Reservoir drawdown for storage of flood waters.

2. Reservoir conservation pool raises for water supply.

3. Reservoir release for the purpose of low flow augmentation.

4. Low head hydropower development at several dams.

5. Reservoir tainter gate usage during peak flow periods.

6. Maintenance dredging and associated water quality certification.

7. Resumption of commercial fishing at one reservoir following banning due to elevated pesticide concentrations (an environmental success story).

As a result of our efforts numerous operational changes have been implemented and several potential projects analyzed in terms of their water quality impacts. It is hoped that continued effort in the area of water quality will enable us to prevent water quality related problems from developing while continuing to solve existing problems. Only by efficient management of resources can we continue to fulfill the Corps mission.

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Environmental and Water Quality Operational Studies (EWQOS) - An Overview

by

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Introduction

The EWQOS program was initiated on 1 October 1978 (FY 78) and will be concluded during FY 85. To obtain an understanding of EWQOS and the results, it is necessary to briefly review the foundation of the program and its history. The need and scope of the EWQOS program were established in a survey of Division and District environmental quality problems that was performed from February to September 1976. Analysis of these problems produced a set of research needs and defined the scope of EWQOS. A summary of these research needs is presented in Table 1. Along with these research needs another major finding of the survey was that the CE was undergoing a shift in emphasis from predominantly a construction agency to a water resource management agency, hence a predominance of the problems were anticipated to be of an operational nature.

Table 1

Research Needs for EWQOS*

Dissolved Oxygen Problems/Corrective Procedures
Nutrients and Eutrophication
Contaminants (Sources/Effects)
Predictive/Evaluation Techniques
Reservoir Operations
Environmental Data Analysis and Management, Sampling Design
Environmental Assessment
Water Resource Management
Riverine Environmental Impacts

Source:
* WES Technical Report E-78-1

EWQOS was formally initiated in October 1978 and was originally scheduled to be completed by October 1983. Due to a shortage of funds, the program was extended two years and is presently scheduled to be completed by October 1985. A review of the major research needs in Table 1 will indicate that some change in emphasis has taken place during the course of EWQOS. Some problems have diminished in importance while new problems have emerged or other problems have increased in importance. Many changes in the direction of EWQOS have taken place in response to field office input. This input occurred as a result of semiannual meetings of a Field Review Group formed at the onset of the program. The Field Review Group is composed of the OCE Technical Monitors

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and a representative from each CONUS Division Office. Currently, a majority of the research in EWQOS has been completed and program emphasis has shifted to analysis of results, development of recommendations, and implementation of technology transfer plans. The remainder of this paper will highlight results of EWQOS and indicate future trends and applications of technology developed from these results.

Results of EWQOS Research

The EWQOS program is divided into two major research areas, one on reservoirs and the other on waterways. While the EWQOS program is represented by a number of different projects, work units, and tasks, the majority of the findings and results may be summarized under technology areas presented in Table 2.

Table 2

Technology Areas for EWQOS

Reservoirs

- Description/Purpose/Operation
  - Sampling Procedures/Design
  - Data Analysis/Management
  - Eutrophication
- Simplified Analysis Techniques
- Algae Control
- Site Preparation
- Shoreline Revegetation
- Releases/Regulation
- Fisheries Management
- Water Quality - Outlet Works and Pool

Water Quality Models

Waterways

- Environmental Aspects
  - Monitoring, Sampling, Data Analysis
- Navigation Effects
- Dikes
- Bank Protection/Revetments
- Levees
- Channelization
- Design/Construction Considerations

Environmental Assessment

A central thrust of the EWQOS program was the development of an increased understanding of the environmental aspects of reservoir and waterway projects in response to their design and operation. This increased understanding of the function of reservoir and waterway projects has lead not only to new or improved technology for correcting environmental deficiencies but also to improved procedures for conducting monitoring programs, data analysis, and sampling design. These results were based on the numerous field studies conducted under the program that contributed to the verification of technology developed but also permitted the development of cost-effective and meaningful procedures for environmental monitoring. As a result of field studies conducted under EWQOS, case studies are available on which to base field application of results. Field studies have identified the major environmental considerations for these projects; consequently, this allows improved analysis of project alternatives during the planning stage. For problems that cannot be resolved on the basis of known information, procedures developed from field studies allow the formulation of sampling or monitoring plans based on a good understanding of project operations and information requirements to solve problems in a cost-effective manner consistent with study objectives.

Within the program area dealing with reservoirs, major technology areas include numerical modeling, nutrient response and eutrophication potential, site preparation, project regulation, environmental aspects of project releases, shoreline revegetation and water quality improvement for releases and within the pool. This technology has application in project design, but probably has more profound application to project operation to correct environmental quality problems confronting field offices. Because this technology has been based on field studies and a thorough understanding of reservoirs, there is considerable potential for rapid application to newly emerging environmental problems associated with reservoir projects. Field studies of reservoirs has lead to a new understanding of their function in response to design and operation and this information has been applied to numerical modeling and other technology developed within the EWQOS program.

The integrated nature of EWQOS has fostered the development of many technologies that fit together to permit comprehensive solutions to environmental quality problems. For example, water quality models may be applied to projects in conjunction with regression techniques to analyze and evaluate project response to various operational conditions. This analysis may in turn be coupled to design procedures to improve project water quality and alternatives may be evaluated against impacts of various regulation schemes developed to judge their effectiveness. Information gained from certain technology areas, such as algae control and site preparation, has been contributed to improved methods for modeling water quality or ecosystem response. This integrated approach also minimizes the chance that recommendations to correct a specific problem, i.e. nuisance algae, will conflict with other management objectives or project purposes.

Research on waterway projects has concentrated on the environmental effects of dikes, bank protection, levees, channelization, and navigation traffic. As with studies on reservoirs, field studies have lead to an increased understanding of the environmental consequences of project features and provided the tools to permit meaningful and cost-effective studies of waterway projects. Based on information from field studies and surveys of current field office practices, corrective procedures have been developed for environmental problems associated with waterway projects. In many cases these improvements have focused on improved design or construction techniques that are applicable to new projects or maintenance of existing projects. In some cases improved understanding of environmental aspects has lead to improved management of project resources.

Future Directions

As the EWQOS program comes to a close, emphasis has shifted from performing research to transferring results to field offices for application in solving environmental problems. Recommendations have been developed with an awareness of project purposes and the often conflicting needs of projects; consequently, program results are intended to minimize conflicts between environmental objectives and other project purposes. Technology transfer is seen as a key to the success of the EWQOS program. If information developed by EWQOS cannot be readily assimilated and used by field offices, then research performed is of little benefit. To ensure success of technology transfer activities, a plan has been developed in conjunction with the Field Review Group to meet the needs of the field offices. The key elements of this technology transfer plan are presented in Table 3.

Table 3

Technology Transfer Activities Under EWQOS

Key Tasks	Subordinate Elements
Program Documentation	Technical Reports Environmental Engineering Manuals Journal Publications, other
Informing Users	Information Exchange Bulletin Computerized Information System Field Office Briefings Interagency Coordination
Training	
User Assistance	
Technology Maintenance	

Many of the technology transfer activities presented in Table 3 are presently ongoing within the program or are being actively developed. Documentation of program results through technical reports form the background for any technology transfer activity. Three Environmental Engineering Manuals will summarize results of reservoir and waterway research and provide guidance to field offices. Publications in technical journals and presentations at technical society meetings increase the scientific credibility of program results. It is critical that potential users of EWQOS technology be informed of what is available as a first step in being able to use program results. The information exchange bulletin serves the purpose of informing a wide audience on program results and their applicability to solving environmental quality problems. A computerized information retrieval system, to be in operation during FY 85, will permit access to all published program information and permit the user to rapidly identify those documents amenable to his particular problem. A series of Division/District Office briefings, to be initiated this fiscal year, will give an overview of program products and technology available to field office personnel so they can easily identify program areas suitable to present and future problems. Interagency coordination has minimized duplication of research among water resource agencies and promoted mutually beneficial research efforts. Training, User Support, and Maintenance of EWQOS technology are activities that are ongoing and are expected to continue during a post-EWQOS support program under the auspices of OCE. During this time of diminished manpower and technical resources, it is critical to establish a "center of knowledge" that can serve as a focal point to preserve and assist in use of EWQOS technology. The support effort will serve this purpose and also provide the direct user assistance to field offices. To some extent this service is already provided under the one-stop requests within the existing program, but as EWQOS is concluded the demand for assistance is expected to rapidly increase as technology is applied to environmental problems.

Summary

National environmental quality objectives continue to be an important part of the Civil Works objective of the CE and continuing emphasis is placed on operating projects to meet these objectives. The EWQOS program has developed information and technology to meet these national environmental objectives in a manner compatible with authorized project purposes. This technology and knowledge on how reservoir and waterway projects respond to various design and operational scenarios will allow field office personnel to solve current environmental quality problems and to rapidly respond to emerging problems in the future. Technology transfer efforts planned or underway will insure this information is available and will provide the necessary support for full implementation of EWQOS research results in a cost-effective manner. Technology transfer activities and information gained from EWQOS research will permit the CE to continue water resource management in harmony with national environmental quality objectives.

DEVELOPMENTS IN WATER QUALITY MODELS FOR SURFACE WATERS

By

Mark S. Dortch[1] and Jack B. Waide[2]

Construction and operation activities at Corps of Engineers (CE) water resource projects can impact the quality of surface waters. A variety of numerical water quality models are required to assess these impacts, to evaluate various structural and operational alternatives for water quality control, and to determine cause and effect relationships of water quality problems. To help meet these needs, some of the R&D activities within the CE have been directed toward developing water quality models for reservoirs, rivers, and estuaries. This paper summarizes the status and availability of the reservoir and riverine water quality models.

Reservoir Models

Several tasks within the Environmental and Water Quality Operational Studies (EWQOS) Program were devoted to the development, application, and evaluation of generalized one- and two-dimensional (1-D and 2-D) numerical computer codes for reservoir water quality. These efforts have resulted in the codes: CE-QUAL-R1 (1-D reservoir) and CE-QUAL-R2 (2-D reservoir). Both codes are based upon integrated descriptions of hydrophysical, chemical, and biological/ecological processes which regulate reservoir water quality. The codes were developed with the intent of being generally applicable to a variety of reservoirs while allowing the input of features specific to a particular reservoir. Although both codes contain fairly comprehensive transport and water quality algorithms, the computational expense of applying the codes is relatively small, thus permitting simulation of realistic time frames, such as an annual stratification cycle. Versions of the codes have been adapted for minicomputer (e.g., VAX 11/750) as well as main frame (e.g., Cyber 176) systems. Major features and status of CE-QUAL-R1 and CE-QUAL-R2 are discussed below.

CE-QUAL-R1

CE-QUAL-R1 (Environmental Laboratory, 1982) allows the user to simulate temporal changes in up to 36 water quality variables along the vertical axis in a stratified reservoir. As a 1-D model, it is appropriate for simulating water quality conditions in the deep pool near the dam and for predicting the quality of reservoir releases. The major hydrophysical, chemical, and biological processes included in CE-QUAL-R1 are listed in Table 1. Some of the typical problems which may be addressed with CE-QUAL-R1 are listed in Table 2.

CE-QUAL-R1 is a very comprehensive water quality model requiring numerous inputs and coefficients. Much information can be gained from the model but much information must also be furnished. This requires interdisciplinary training and understanding in the limnological sciences. This is true to some extent for any model so is considered a requirement rather than a limitation.

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Table 1

Major Processes Included in CE-QUAL-R1

Hydrophysical	Chemical/Biological
Solar radiation and surface heat transfer	Phytoplankton dynamics
Density stratification	Organic matter production and decomposition
Integral energy wind and convective mixing with flow and wind dependent diffusion	Nitrification and denitrification
Placement of inflows	Nutrient cycling (N, P, C, Si*)
Selective withdrawal	Carbonate equilibria involving pH and alkalinity
Pumped-storage inflows and mixing*	Biomass transfers through higher trophic levels
Coupled reservoir/afterbay system*	Reoxidation and reduction for aerobic and anaerobic conditions
Conservative substance routing (density coupled)	
Suspended solids routing and settling (density coupled)	

* Will be included in distribution version in the next update

Table 2

Typical Problems Addressed by CE-QUAL-R1

- Onset, extent, and duration of thermal stratification
- Location of selective withdrawal intakes to meet downstream water quality objectives
- Cause and effect relationships involved in reservoir water quality conditions
- Effects of structural and operational alternatives on in-pool and release water quality
- Onset, extent, and duration of anoxic conditions
- Magnitude and timing of algal blooms
- Appearance of reduced substances
- Effects of upstream land use
- Effects of storm events
- Assistance in real time management of water quality

To assist the user in determining values for model coefficients, a report summarizing relevant literature sources (Collins and Wlosinski, 1983) is available. A Monte Carlo subroutine is also available for examining effects of uncertainties in coefficients on model results. This capability allows the user to place confidence limits on predictions. To reduce model calibration costs and complication, a submodel (CE-THERM-R1) can be used to model thermal behavior, physical processes, and conservative constituents prior to full water quality simulations. The release version of the code also contains a flexible graphics package which allows the user to display simulation results in a variety of informative formats.

During the developmental years, CE-QUAL-R1 was applied to about a half dozen reservoirs. The model is currently being applied to DeGray Lake, Ark., Eau Galle Lake, Wis., and Lake Ashtabula, N.D. Results of the DeGray and Eau Galle verification studies will be provided in two EWQOS reports.

The code and user manual were initially released in April 1982 with the first revision in December 1982; both are available upon request. About 100 requests have been received as of December 1983. The next update is planned for September 1984. This update will include algorithms for ice cover, pumped-storage, afterbays, and peaking hydropower withdrawals, and improvements to several chemical/biological compartments.

CE-QUAL-R2

In many CE reservoirs, significant gradients in water quality conditions occur along the longitudinal as well as the vertical axis of the reservoir. When the purposes of a water quality study require that explicit attention be given to such gradients, a 2-D model must be employed. Although a 2-D model is more realistic of the physical conditions, it is also more costly and difficult to use. For example, a 2-D model application can be as much as an order of magnitude more expensive than that for a 1-D model. For practicality, the 2-D model is limited to about half of the number of water quality constituents as the 1-D model, thus reducing complexity and computational burden while losing some water quality information. If longitudinal definition is not required, a 1-D reservoir model should usually suffice and is recommended at this time.

With the exception of the number of water quality constituents included, the 2-D model can address the same problems as the 1-D model. Additionally, the 2-D model could be used to address questions relating to longitudinal gradients such as those shown in Table 3.

CE-QUAL-R2 is a derivative of the Laterally Averaged Reservoir Model (LARM) developed by J. E. Edinger and Assoc. (JEEA) Inc. of Wayne, PA. Years of effort by JEEA and the Waterways Experiment Station (WES) went into the evolution of CE-QUAL-R2. The code is arranged such that hydrodynamic and water quality variables can be computed simultaneously or separately in which case the hydrodynamics would be solved initially with output files used to drive subsequent water quality simulations. The latter option permits cost savings during a study. The user can also specify the level of water quality detail, for example, temperature and conservative constituents only (level 1) or these plus twelve nonconservative constituents (level 2). A third level of definition includes reduced substances under anaerobic conditions; this level

Table 3

Problems Addressed by CE-QUAL-R2

- Same as those in Table 2
- Longitudinal variations in trophic state and other conditions such as:

    Development of upstream anoxic conditions and their advection into the main pool

    Occurrence of midpool algal blooms

- Longitudinal location of project features such as recreation sites
- Occurrence of density currents and their effect on circulation and water quality

will probably remain developmental for some time in the future. The code also allows simulation of branched and looped reservoir systems. The formulation and solution schemes employed permit economical simulations relative to other 2-D models; simulations of stratification cycles are practical.

During FY 84, the water quality algorithms will be incorporated into the final version of the underlying hydrodynamic code and tested on DeGray and Canyon Lakes. A user manual and evaluation report will be completed at the end of FY 84. The CE-QUAL-R2 code will be released during early FY 85.

Riverine Models

Two unique riverine water quality models have been recently developed to address many of the waterway related questions of the CE. Both models allow simulation of dynamic conditions. One model is 1-D (longitudinal) and is referred to as CE-QUAL- RIV1; the other is 2-D horizontal (depth integrated) and is named CE-QUAL-RIV2.

CE-QUAL-RIV1

CE-QUAL-RIV1 was originally developed by Bedford, et al. (1982) for the Ohio State EPA and later enhanced for the Waterways Experiment Station. The program is actually a two part code for hydraulic routing and water quality simulation. The hydraulic routing is accomplished with the efficient and accurate four-point implicit method. Water quality transport is also done with a highly accurate scheme to reduce numerical error. Water quality constituents include temperature, algae, nutrients, DO, CBOD, and coliform bacteria. Soluble iron and manganese are being added. The code allows for dendritic (multiple branching) systems with in-stream hydraulic control structures.

This type of model would be especially useful for addressing water quality questions below peaking hydropower projects or for similar dynamic discharges and for modeling the effects of riverine control structures such as reregulation dams and multiple run-of-the-river lock and dams.

CE-QUAL-RIV1 is being applied to the Chattahoochee River (below Buford Dam to Atlanta) for the South Atlantic Division, CE, to study the effects of the proposed reregulation dam below Buford Dam. The code and user manual will be released at the end of FY 85.

CE-QUAL-RIV2

CE-QUAL-RIV2 is a two-dimensional, depth-integrated, unsteady flow water quality simulation code. This code is presently under development and is being tested for riverine conditions; however, the code permits the evaluation of both lateral and longitudinal water quality gradients in any shallow, vertically mixed water body. Through the use of a grid technique known as boundary-fitted coordinates, this model can be fairly easily applied to problems with complex geometries.

CE-QUAL-RIV2 is also two separate codes, one for hydrodynamic computation and the other for water quality computation. The hydrodynamic code was originally developed by Johnson (1980) and is known as VAHM, Vertically Averaged

Hydrodynamic Model. The water quality compartments are similar to those of CE-QUAL-RIV1. Output from the hydrodynamic code is used to drive the water quality code.

The code and user manual will be released at the end of FY 85. It is envisioned that this model would be used to address water quality questions for wide (possibly braided) rivers and shallow lakes and estuaries with complex geometries.

Conclusions

A variety of numerical water quality models are (or will be) available to address CE water quality questions. Not all water quality questions require the use of a numerical model to resolve. However, it is very difficult to make sound decisions with respect to water quality management without the benefits provided by a model of the system. There are time and cost requirements associated with setting up a model; therefore, the benefits should exceed the expenses before implementing a model. One major benefit that is often overlooked is that after a site-specific model is developed, it can be used for years to come to address pre- or post-project conditions and to assist in making future decisions. Long-term benefits of a model should be considered when trying to justify the expense and training required to apply the model.

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Laboratory Quality Control
by
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Quality Control and Quality Assurance cover a wide range of topics even if restricted to only the problem of water analysis; it also varies with job responsibility. Quality Control from the viewpoint of the lab manager is not necessarily the same as that of a data user or a requisitioner of lab service. The Corps of Engineers is not in the business of collecting data for tabulation or independent research; the data are used to drive operating decisions; and quality control programs should keep this in mind. Most importantly, quality control costs money. What I hope to do is give you some insight into quality control problems in the laboratory so you can spend your money wisely. In doing so, I will cover such topics as the statistical nature of analytical results, the sources of laboratory errors, and the implication of these in data evaluation. It is worth pointing out that quality control is at least in part a philosophical subject and that while cost benefit ratios are often implicit in many quality control decisions, it is not usually taken directly into account in a laboratory doing routine chemical analysis. I'll end with some examples and a few brief guidelines and opinions. Most of my remarks will be limited to laboratory operations, leaving to you the appropriate extrapolations to field work.

The statistical nature of any measurement is well known to all of us. If you send duplicate samples to a lab, you will probably get different results; if you send them to two different labs, you will almost certainly get different answers. On the other hand, if you try to measure a cereal box to the nearest micron, you will always get different answers; but measured to the nearest foot all cereal boxes are the same size.

The laboratory objective is to find the appropriate procedures to solve the problem. In the office, examining lab results, it is important to consider the statistical nature of that result.

A traditional way to describe two important concepts-accuracy and precision-is with bullseye diagrams. Figure 1 shows the scatter of analytical results for the extreme case of high accuracy (the average is on target) and low precision (any individual answer is very different from any other). This is typical if random errors are dominant. Figure 2 is the case for low accuracy (the average is off center) and high precision (repeated analysis are close together). This is the case in which systematic errors dominate.

There are several points connected with both of these diagrams which are often neglected. Figure 3 indicates the problem of scale and implicity asks the question, what are you going to do with the data? High precision costs money. What precision do you need to solve your problem? A second point is the source of the scatter. Suppose this represents only the laboratory scatter. In that case, each point on the diagram should be viewed not as a point but as the circle of another bullseye representing all other causes of scatter. If the scatter from these other circles is great enough, improving the quality of lab results may gain nothing. Put in statistical terms, the total standard deviation is equal to the square root of the sums of the squares of the individual standard deviations. So if the lab standard deviation is 2 and other sources of the scatter produce a standard deviation of 4, the total standard deviation is about 4.5. Not much can be gained by better lab results.

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Figure 4 is to indicate that in the case of accuracy, the targets do not represent an absolute. They are often concensus or mandated targets. Zinc and its alloys are sold by impurity content. Several years ago, the zinc association prepared a series of standards to be analyzed by emission spectroscopy. All zinc sold was referenced to these standards by this method. Whether or not these were "true" values did not matter. The EPA does the same thing with its list of methods and control samples. In an absolute sense, the answers may in fact be wrong. It is a mechanism to get labs to agree. One result of this is that procedures developed for different purposes can give seemingly inconsistent results.(It should not come as a surprise if individual phenol type compounds analyzed by gas chromatography do not add up to total phenolics measured colorimetrically.)

These diagrams are concentration dependent. Figure 5 shows a schematic plot of how the standard deviation changes with concentration for a typical analytical method. The general shape of the curve is due to the different sources of error that predominate at different concentration levels and establish the upper and lower limits of the method. These limits and, in particular, the detection limit, are a matter of choice. The percent error that the analyst considers tolerable sets the detection limit. These curves and the detection limit will vary from method to method for the same parameter.

Basic to this diagram and all statistical considerations are the definition of the population, the assumption that population is sampled randomly, and the assumptions about population distribution. Generally, curves such as these are obtained for standards not samples. They will be different for real samples. The standard deviation of a series of measurements on a standard is different from that of a real sample, and it varies from sample to sample. The repeatability of results at the detection limit is not the same from sample to sample. Normally, statistical measurements are made on standards, typically pure compounds in distilled water. It is assumed that the statistical inferences apply to samples. This assumption may be valid or may be grossly in error. The actual standard deviation on any given sample for most labs performing routine environmental analysis is rarely known.

For samples on the flat part of the curve near the detection limit, there are some interesting statistical effects. The curve in Figure 6 represents the normal distribution for sampling a population. It indicates that for any given result there is a 16 percent probability that the answer is at least 1 standard deviation too high, 2.3 percent at least 2 standard deviations too high, etc.; the same would be true on the low side. To reiterate this curve is sample dependent. In addition, it assumes a normal distribution of errors; this is not always the case.

Consider the case where the mean concentration has a value of 5 with a standard deviation of 1 and, as often happens, the instrument measures to the nearest .1 and the results are then rounded to the nearest unit. Then, 30 percent of the results will be reported as 4 or less, and 7 percent as 3 or less; 30 percent reported as 6 or more, and 7 percent as 7 or more. Suppose the concentrations were lower: if the concentration is 1, then 30 percent will be reported as 0--false negatives; if the concentration is zero, then 30 percent will be reported as 1 or more--false positives.

Now, however, we program our analytical instrument or computer to set 1 as the detection limit. Then at a concentration 1, 50 percent of the values will be reported as less than 1--false negatives, but only 30 percent of the zero will be reported as 1 or greater--false positives--an obvious bias in the data.

If the standard deviation is improved, but the other conditions held constant, the bias is even greater. There is a different set of assumptions in which the degree of bias is smaller. Unfortunately instrument manufacturers usually do not tell you how they program their machines, and it is normally not controllable. But in any case, there will usually be a bias in data at the detection limit.

Sometimes attempts are made to improve the precisions of the result by doing multiple analysis and averaging. It would take quadruple analysis to reduce by one-half the percentage error; stated alternatively, the standard deviation is proportional to the square root of the number of replicates. This is usually not a cost-effective way to improve precision.

The sources of errors in laboratory analysis are typically divided into random and systematic to which will be added outliers and sample problems. In a certain sense, all errors are random if the population is large enough for a significant number of errors of a given type to accumulate. The distinctions here are based on the fact that in typical laboratories certain classes of errors are frequent enough to be amenable to statistical measurement and others are not.

The three main sources of random error--operator skill, robust method and instrument specification-are all interrelated. The instrument specifications represent a lower limit. A few highly skilled operators may be able to do marginally better, but for well designed instruments and methods even a poorly skilled operator will get acceptable results. There are some lab operations, pipetting small amounts of liquid, cleaning glassware, etc., which are highly dependent on the technician. Robust methods refer to slight variations in procedures which have not, or cannot, be documented and which influence results. An extreme case is one in which an analysis can only be performed by few people and it cannot be duplicated by others from the written procedure. In some cases, zinc analysis for example, the limiting factor is environmental variation; zinc, in easily detectable amounts, is present in dust in the air. Random errors cannot be eliminated, only reduced, and depend on operator skill for that particular analysis.

Systematic errors are ones which can be eliminated if detected, but often go undetected for long periods. They can be caused by things such as poor choices of methods or standards, grating misalignment in an instrument, or poor housekeeping procedures which allow contamination of nutrients with nitric acid from metals analysis. For most environmental analysis, the problems of methods and standards have been defined out of consideration by using the prescribed choices. Systematic errors usually arise from management inattention or ignorance.

I've broken outliers into three classes: statistical, environmental, and gross error. In any analysis there are a large number of sources of random errors which are usually small and which tend to cancel each other; there is a small, though finite probability, that in some particular sample they will be large and additive. There are a variety of environmental flukes in any lab--a few specs of dust from an unknown and transient source contaminating a few samples in a batch, but not the quality control checks. Gross errors are those outside normal operator skill considerations--a sample is accidentally poured in the sink instead of a test tube and the error was not caught. In a study comparing Atomic Absorption and Emission Spectroscopy several years ago, 5,000 samples were split in two parts for analyses. In six of these samples, the results indicated that one of the two contained only distilled water when it arrived at the instrument.

The last category of laboratory errors are sample problems: those things which the method is supposed to account for but can't. Inhomogenous samples can never be subsampled in the lab without introducing some error--and most environmental samples are inhomogenous. Environmental samples change with time; the recommended preservatives don't always work. Matrix variation refers to those problems which are in excess of that anticipated by the method. This can range from things like unusually high manganese suppressing phosphate to too much salt or organics affecting the viscosity and hence the volume of solution asperated into an atomic absorption spectrometer.

The major elements in quality control programs fall into two categories. One might be termed general laboratory operations--personnel, procedures, methods, etc.; next are the measurements which indicate the effectiveness of the program. The first group has been detailed in a variety of documents from ETL's, to EPA inspection manuals, to various sections of the Federal Register and will not be repeated. Several things are often omitted in these documents. Five will be discussed here: personnel, results of QC measurements, standards, methods, and evaluations of laboratory evaluations.

For routine analysis, a dedicated, experienced high school graduate will often get better results from even complex instruments than PhD's who think they are doing research. The best GC/MS or emission spectrometer operators I have known never went to college, but they learned enough chemistry and enough about their instrument to produce superior results, including data interpretation.

Why does a laboratory do quality control? What do they look for? What happens if they find it? Under what circumstances do they shut the lab down to correct a problem? Under what circumstances do they rerun samples? Have they ever stopped running samples? Have they ever repeated batches of samples? Remember sooner or later everyone goofs.

What is used for standards? This is a complex problem and depends on the analysis being performed. Good practice requires that the standard match the samples as closely as possible. For some environmental samples, the distilled water standards usually used are a good approximation, provided no separation concentration steps are involved. In most cases, however, it is just the best of a bunch of poor choices. In many cases, good standards, or any standards, particularly in the case of organics, simply do not exist. There are some laboratories which try to impress you by saying that they use EPA's Q.C. sample as standard. This represents a gross misuse of these samples and indicates a lack of understanding of quality control.

The EPA methods typically used in analytical laboratories tend to be compromises rather than optimized for sensitivity or freedom from interferences. For environmental samples, the only barriers to finding any naturally occurring elements, and a variety of organics such as PCB's, chlorinated pesticides, phenols, etc., are the skill of the analyst, the skill of the sample collector, the size of the sample, and the cost.

What does the bureaucratic nitpicking that takes place at laboratory evaluations gain you? It is an insurance policy, but insurance companies have gone bankrupt and some insurance companies are out to milk the public. Professionals doing an honest job guided by experts who can properly define problems and know how to approach their solution are worth reams of QC data. With regard to experts, research labs are rarely suited for routine analysis and routine labs can't do research. Some labs are often referred to (in a derogatory sense) as "number generators." Number generators produce some of the most reliable results available--just don't expect good results from analyses they don't normally perform.

Error measurements generally involve duplicates, split samples, synthetic samples and spiked samples.

Duplicates are when you divide the sample in two parts in the laboratory, run both and perform some sort of comparison of the results. This is probably the most common form of Q.C. measurement made. If you are in the business of buying analysis you should be able to get this data ahead of time. Duplicates mainly detect random lab error, particularly poor operators. If properly performed (it often isn't) it can also detect cross contamination. The problem with duplicates is that they are usually run on real samples and there is no way to distinguish between sample inhomogenity and random error. In addition, large numbers must be run at assorted concentration to get statistically significant information.

Splitting samples with another lab is an excellent but seldom used procedure. The difficulty is insuring that the samples are indeed the same; this requires homogenous samples which do not change with time. It is an extremely powerful method of detecting laboratory bias.

Synthetic samples are probably the most cost-effective way of detecting laboratory bias. The three basic types in order of increasing effectiveness are: EPA type in which water is spiked with a known amount of analyte; analyzing a standard made and used in another lab; and NBS type which is an actual sample with known concentration. There are relatively few of the NBS type for water analysis although non-water types can often be used.

Spiked samples cover a wide range of techniques. The most common is to add a known amount of analyte to a separate portion of the sample and evaluate the difference between the spiked and unspiked results. There are many problems with this. For optimum results, the samples should not be near the detection limit and the added amount should be comparable to the amount present. If these conditions are not met, normal variations will mask any problems. The most useful aspect of this technique is to detect cases where other substances in the sample either suppress or enhance the signal from the analyte. These kinds of effects are both small and rare. If they are suspected, a careful study should be made of all related samples rather than a random spiking of samples. In organic analysis, a common technique is to spike all samples with a surrogate compound, which cannot be in the sample, and evaluate the results of that sample by the results found for the spike. This approach can detect both systematic and random error. There are several other spiking techniques which are not usually used in environmental analysis.

The three following data sets illustrate problems in converting laboratory data into meaningful information. Figure 7 gives the standard deviation expressed as a percentage of the concentration for a variety of analyses performed by the Food and Drug Administration. The quality control procedures used in their labs are probably better than most; they take great pains to insure that all of their labs will get the same result on the same sample. This probably represents a reasonable limit achievable at other labs. Note that for most substances of environmental concern variations are in the range of 30-60 percent or worse. These results are quite good when compared with the initial analysis of the moon rocks. These rocks were analyzed by the best labs in the world, but without interlaboratory quality control. The results are too embarrassing to report.

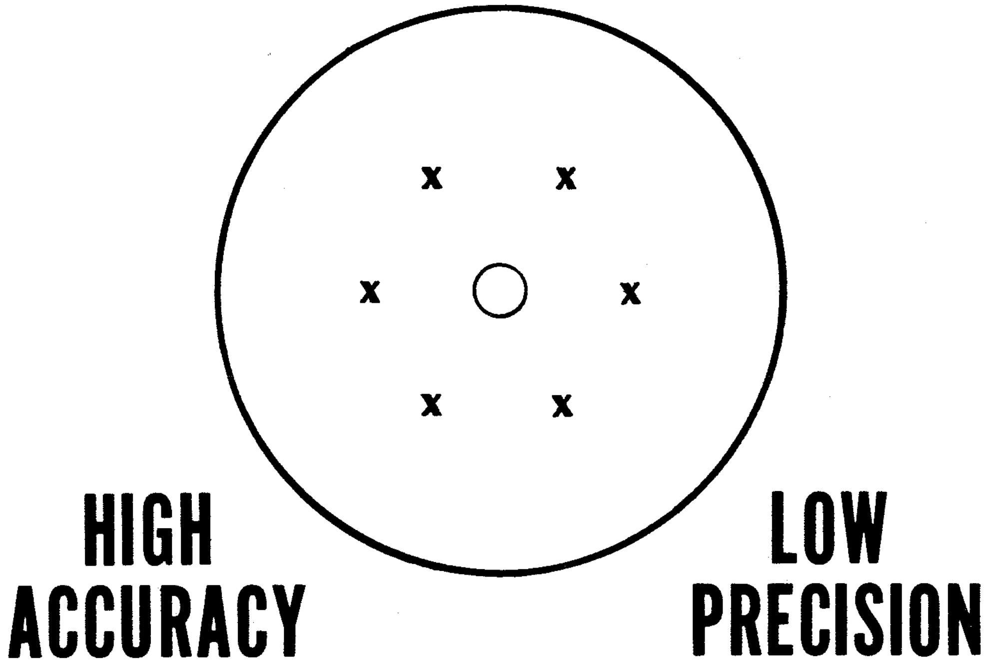
The Control Charts in Figure 8 are from an EPA Love Canal study of occurrence of priority pollutants. Each sample was spiked with a compound of fluorobenzene which would not occur in any sample. Lab A shows a much higher scatter than Lab B; B drifted out of control, then was brought back in. On the surface, it seems that Lab B was doing a better job. However, the following point needs to be clarified before the conclusion is justified. Were the samples paired, or were they at least taken during the same time frames with random chances for the sample going to either lab? If not, there are a great many other variables (e.g., groundwater flows) which could contribute to the difference.

Figure 9 seems to compare two different sampling techniques, but it really shows that the laboratory cannot overcome poor sampling choices. If the purpose was to determine the spacial distribution, a table of random numbers would give perfectly adequate results for the discrete samples. In the case of the integrated sample, it is well to keep in mind that pH 0 and pH 14 are both lethal but the average is harmless; there is also the nagging problem of how do you know there was no cross contamination after the very high samples were collected.

Finally, experimental design and data interpretation are an integral part of quality control. Unless you know the kind of information you want (not data, but how do I operate this reservoir to minimize algae growth, or does this dredge material meet the criteria for open water disposal), you cannot make the most cost-effective use of your quality control dollar. Environmental and engineering studies could benefit by medical experience which requires detailed protocols, laid out in advance, covering all aspects of the project, particularly the exact procedures for evaluating the data and drawing conclusions.

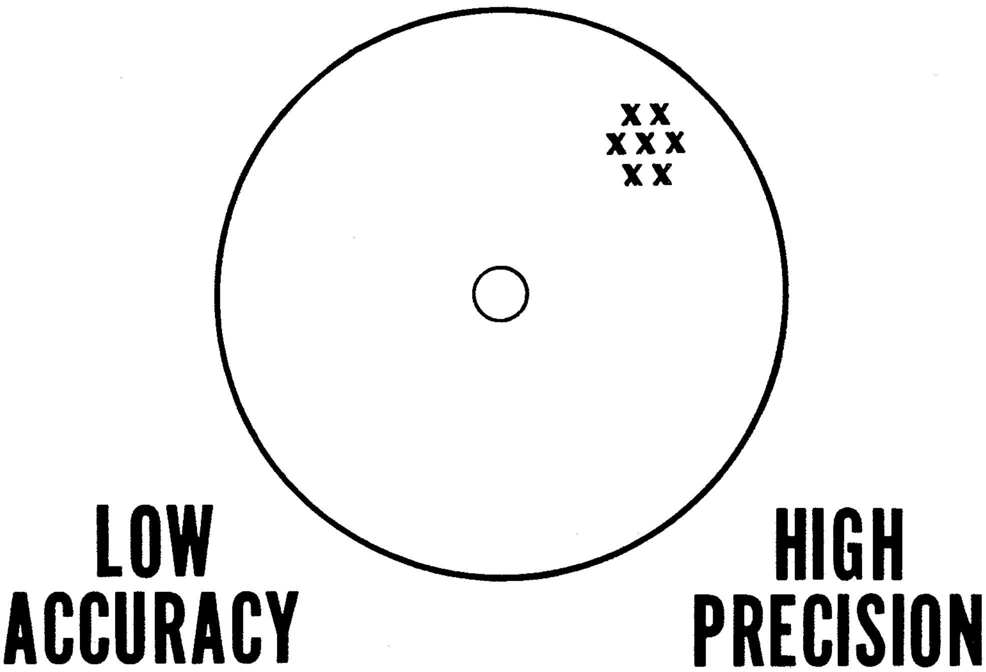
Where samples are few and costs in the millions (nuclear test), it makes sense to get the most information possible out of each sample. Where sample material is plentiful and the incremented cost of additional samples is small, a different approach is desirable.

My personal opinion is that, usually in the case of environmental samples, too much effort is placed on getting the best answer for a particular sample; not enough effort is given to the space/time variation of parameters of concern; and no effort is spent on how laboratory variation relates to space/time variation and how together they impact on decision directing information.



RANDOM ERROR

Figure 1



SYSTEMATIC ERROR

Figure 2

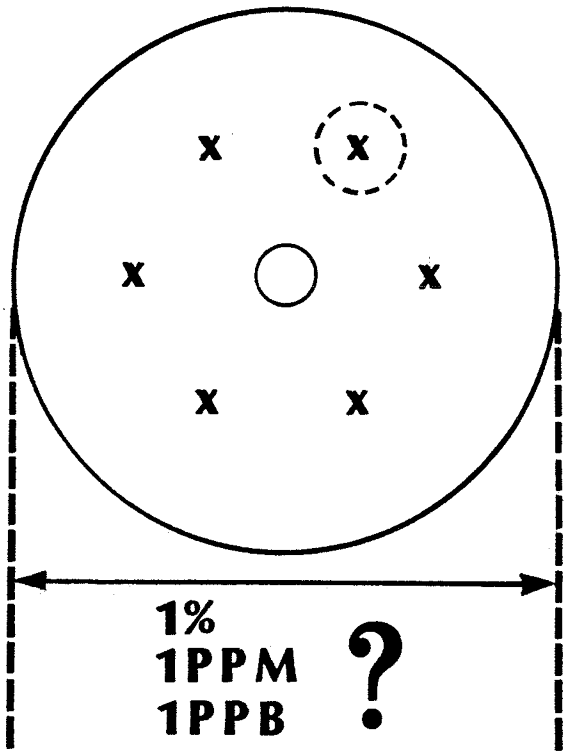


Figure 3

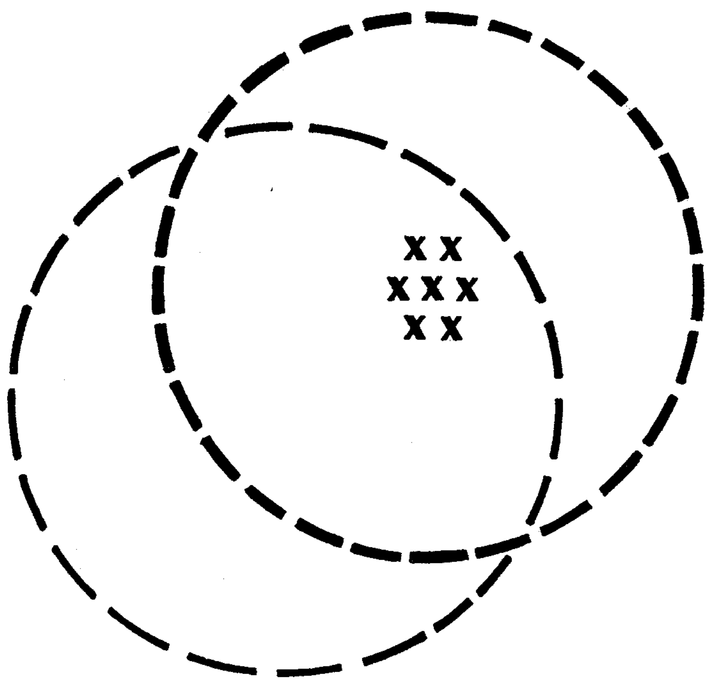


Figure 4

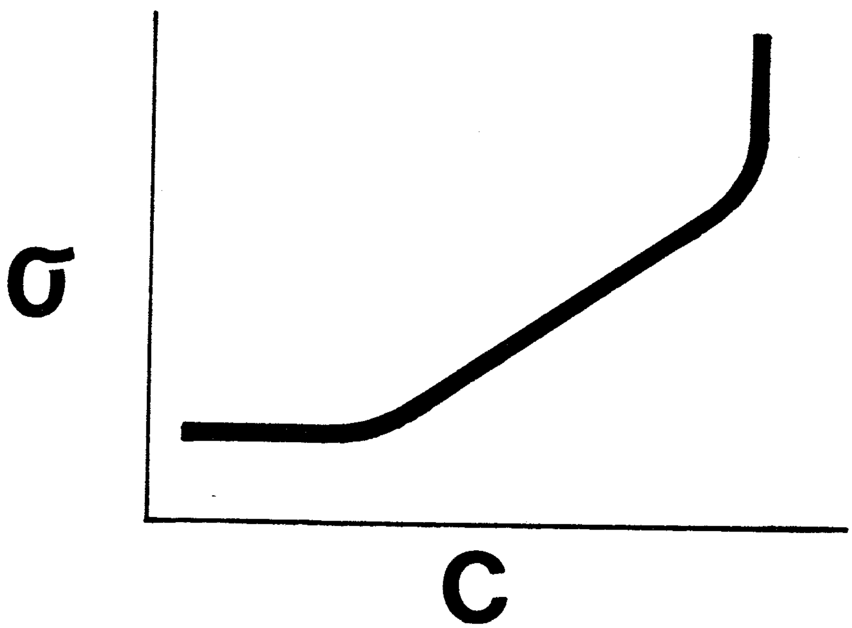


Figure 5

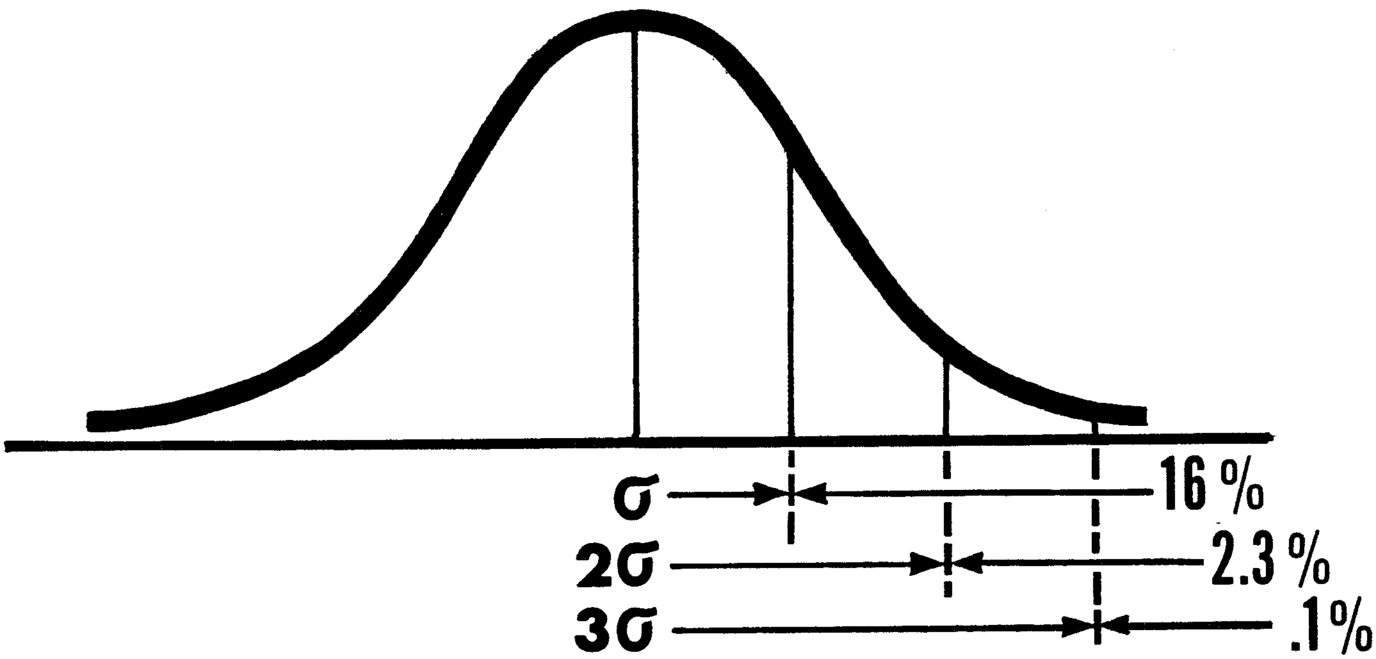
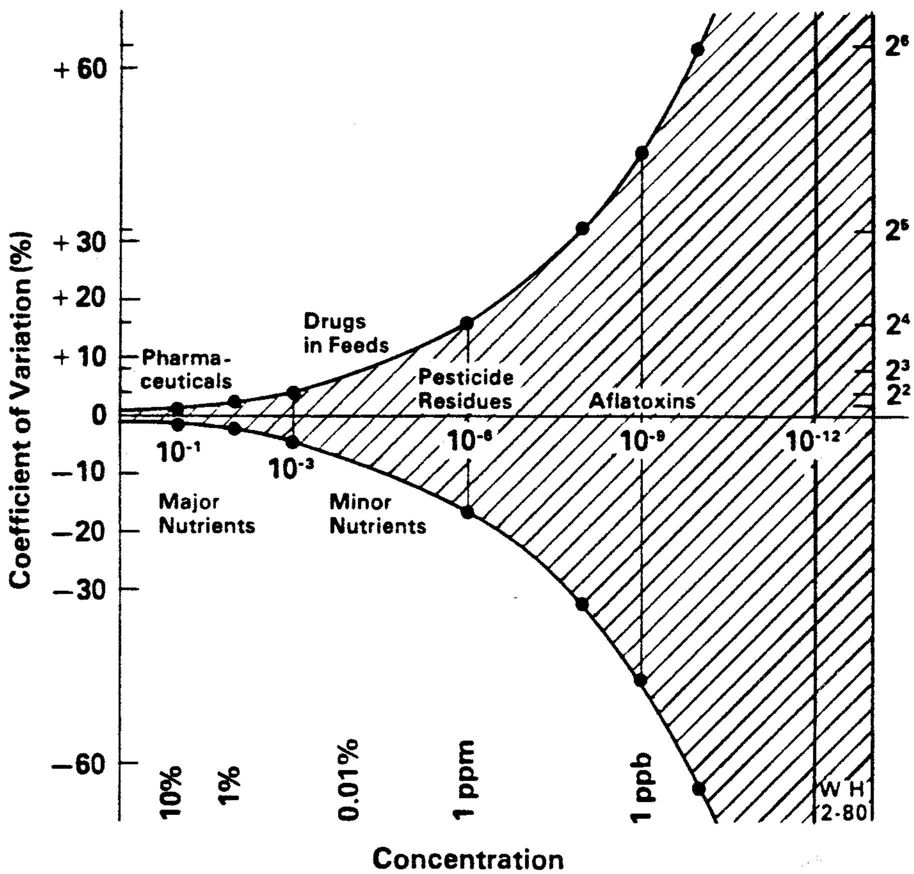


Figure 6



Interlaboratory coefficient of variation as a function of concentration.

Figure 7

Statistical control charts show recovery of surrogate fluorobenzene by laboratories A and B

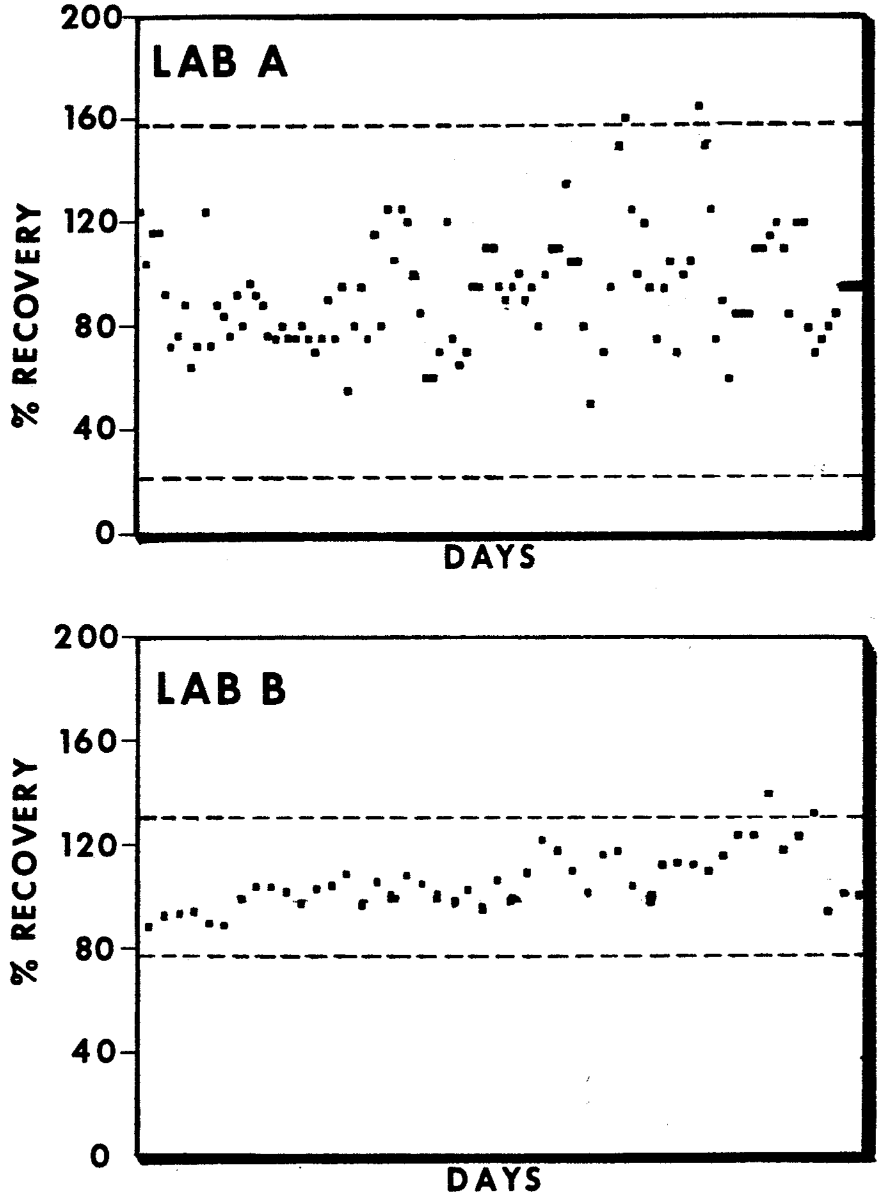


Figure 8

COMPARISON OF DISCRETE SAMPLING DEPTH MEAN WITH VALUE FROM INTEGRATED SAMPLES FOR TOTAL COLIFORMS (AFTER THORNTON ET AL. 1980).

DATA	MEAN FOR DEPTHS 0,3,5 METERS TOTAL COLIFORMS CELLS/100 ML	INTEGRATED SAMPLE OF UPPER 5 METERS TOTAL COLIFORMS CELLS/100 ML
5–27	237	5000
5–31	467	700
6–2	167	58000
6–6	333	10700
6–9	633	26000
6–12	600	–
6–15	367	52000
6–16	4233	2500
6–18	933	3000
6–21	1000	36000
6–23	5600	19800

Figure 9

MONITORING: DEFINING OBJECTIVES AND SAMPLE DESIGN

by

Robert H. Kennedy, Ph.D.[1]

The U.S.Army Corps of Engineers and other federal and state agencies are currently active in the acquisition of water quality data for the purpose of defining and documenting water quality conditions in this nation's lakes, reservoirs, and rivers, and as a means for assessing various potential ameliorative activities. While highly variable between and within agencies, regulations requiring the collection of water quality data seldom prescribe the manner in which data are to be collected. In most cases, those delegated this responsibility must apply expertise gained from a variety of technical and academic backgrounds. The result is often a lack of consistency. Problems of adequacy of the sampling program are also frequent. While accentuated by the fiscal realities of tight budgets, these problems frequently stem from poor sample design.

Sampling or monitoring programs attempt to determine characteristics of a lake or river based on an evaluation of samples. In statistical terms, qualities (e.g. mean, variance, etc.) of the target population (i.e. all possible observations) are inferred from a sample population (i.e. sampled observations). In most instances, the sample population represents an extremely small subset of the target population. If, for instance, ten 1-liter samples are withdrawn from a reservoir containing ten million cubic meters of water, the characteristics of that reservoir must be inferred from the characteristics of a volume of water representing only 0.0000001 percent of the entire reservoir! Clearly, the manner in which the samples were collected and their representativeness will have great bearing on the final outcome of any interpretation of the data. Sample design must, therefore, receive careful consideration prior to the initiation of any sample collection.

The development and conduct of sampling or monitoring programs occur in five distinct phases: problem identification, objectives definition, sample design, implementation, and data management and interpretation. The problem identification stage serves to delimit the area of direct interest (i.e. defines the target population). For instance, if algal blooms are perceived by the public as an impairment to recreational enjoyment and the

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responsible agengy desires to conduct studies of this reported problem, an understanding of the scope of the problem will be required. Should studies be conducted in all lakes and reservoirs under the agency's jurisdiction or only that lake for which complaints have been received? Are all portions of the lake(s) to be considered in these studies?

Once the problem has been clearly identified, study objectives must be unambiguously defined. Most monitoring objectives fall into three generic categories: determination of "average" conditions, identification of "extremes," and trend detection. The determination of average conditions, as might be attempted in an evaluation of trophic state, generally involves the pooling of data collected at several stations. If station locations have been assigned based on a knowledge of variability, the importance (i.e. weight) of individual observations can be scaled prior to averaging. This generally involves the use of scaling factors based on representative areas or volumes. Identification of extreme conditions in time or space, an important consideration in cases in which standards or criteria are to be enforced, assumes an understanding of spatial or temporal variability. Emphasis here is placed not so much on representativeness per se as on the range of exhibited conditions. Methods for detecting trends, generally through time, are frequently employed in the evaluation of the impacts of perturbations or operational changes.

Sample design, or the plan by which sample locations, frequencies, and variables to be studied are specified, is often the most important, yet most overlooked, step in the conduct of sampling programs. Estimates of the characteristics of the environment are inherently imprecise due to the variable nature of these systems and the manner in which information is obtained. In statistical terminology, the sources of this uncertainty are error, random variation and bias. Bias, or non-representativeness, which is often attributable to poor sample design, can be reduced by careful planning. As uncertainty is reduced, the informational value of the resulting data will be increased. In general, reductions in the degree of uncertainty are realized when sample effort is increased. While increasing the number samples and the frequency with which they are collected would appear to improve any sample design, problems of cost and redundancy can occur. Thus, sample design must provide a means by which uncertainty can be reduced within realistic cost constraints. Sample design must also allow for a quantification of variability of uncertainty, since this will provide a measure of the informational value of the resulting data.

Random sample designs, which require a random distribution of sample effort, are often employed in situations in which the heterogeneities or patterns in variability are unknown or insignificant. For instance, a completely mixed reservoir would not be expected to exhibit significant spatial patterns in the degree of variability between successive samples, and sample locations (i.e. stations and depths) could be assigned randomly. If, on the other hand, the reservoir exhibits significant spatial patterns, a stratified random sample design would provide a more efficacious and statistically defendable approach. For example, most lakes and reserviors are thermally stratified and as a result exhibit pronounced vertical differences in the concentrations and variability of several chemical, biological, and physical variables. If zones or strata having similar variabilities can be identified, then sample effort can be distributed among strata with respect to stratum variability and randomly within each stratum. Such an approach would place greatest sampling effort in strata exhibiting the greatest variability and would reduce the collection of redundant information from strata exhibiting little variability. Following the same reasoning, sampling frequencies could be assigned by considering temporal patterns in variability.

Decisions concerning the type of sample design to be employed in any particular study are facilitated by the analysis of historical data or data collected during preliminary studies. Such data provide an estimate of variability and an approximation of the anticipated mean condition, both of which can be used in the estimation of sample number. This calculation involves a consideration of the probability that the mean of a sample population having n members is not significantly different than the mean of the target population. Obviously, the number of required samples will be a function of the variability of the system under study, the variability of the sample population, and the desired probability specified by the researcher. Unfortunately, the optimal number of samples often exceeds cost limitations. In such cases the researcher must proceed with less than the optimal number of samples or re-evaluate the efficacy of initiating the sampling program. In the former case, it is critical that uncertainty or error be estimated and discussed when summary statistics are presented.

With the problem statement and study objectives clearly defined and a statistically sound sample design established, data collection and interpretation efforts can proceed using accepted standard methods. Time and effort expended in program design will facilitate these efforts and insure that meaningful results can be obtained. It is important to realize that it is extremely difficult to refute sound conclusions drawn from a well designed and implemented data collection program.

The Environmental and Water Quality Operational Studies (EWQOS) Program, sponsored by the Office of the Chief, U.S.Army Engineer and administered by the U.S. Army Engineer Waterways Experiment Station, was designed to address water quality problems common to many Civil Works projects. These studies included a number of investigations of water quality conditions in several reservoirs. Realizing the complex nature of these large, river-fed lakes, considerable effort was expended on evaluating methods for designing sampling programs that were cost-effective and statistically-defendable. Preliminary results of these evaluations have been reported, and two EWQOS Technical Reports are in preparation. These reports, which will deal with both sample design and data interpretation, will be available in FY 85.

A WATER QUALITY DATABASE MANAGEMENT SYSTEM

by

Robert C. Gunkel, Jr.*

INTRODUCTION

The U. S. Army Corps of Engineers (CE) Environmental and Water Quality Operational Studies (EWQOS) research program established a number of objectives to be met by the Reservoir Field Studies (RFS) team. In order to provide the information necessary to meet these objectives, numerous and diverse limnological data were collected at four representative (CE) reservoirs. The magnitude of such a program made the management of information a critical phase in research design. The establishment and successful operation of a database management system (DBMS) was essential for achieving research objectives. Although the RFS DBMS application is not a database management system in the purest sense, it still prescribes to conceptual characteristics of database management systems and involves the management of a large water quality database. This paper will present basic concepts for database management systems, overview the Statistical Analysis System (SAS Institute Inc., Cary, NC), and describe the RFS DBMS.

CONCEPTS OF DATABASE MANAGEMENT SYSTEMS

Several definitions may be helpful before continuing. A database is a collection of interrelated stored data used for multiple applications by some particular organization (1, 3). A database system is nothing more than a computer based system for recording and maintaining information (1). A database management system is a combination of personnel, materials, and methods that provide a structured mechanism for processing raw data into useful information for subsequent decisionmaking processes (2).

One valuable asset of a database system is that it provides centralized control of the database (1). The database administrator (DBA) is responsible for the design, maintenance, and overall control of the database system. Many advantages accrue from a centrally controlled system. Data redundancy can be controlled by either eliminating redundancy entirely or partially. If it does not benefit an organization to eliminate redundancy entirely, then the possibility of inconsistency within the data exists. Centralized control and a DBA aware of such redundancies can plan for and guarantee updating of redundant data. Data in the database are available for and can be shared by multiple users for many different applications. Not only can several users access the database at one time, but it is possible for them to actually be using the same piece of information. Measures for security can be easily established and controlled, ensuring that database access is only by authorized users. Centralized control also aids in maintaining data integrity. Again it is the responsibility of the DBA to define validation procedures on update operations in order to maintain data integrity. Overall, a centrally controlled database provides for better data management.

An effective water quality DBMS is developed through three distinct phases: 1) Data Acquisition, 2) Data Maintenance, and 3) Data Utilization (4). Data acquisition involves experimental design, sample collection, and laboratory analysis. All strategies, methods, and techniques for sample

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collection and laboratory analysis are planned in the experimental design. Data maintenance includes all handling of data in the database system. This handling includes data entry and verification, file establishment, data and file manipulation, update, retrieval, and storage. Data utilization includes analysis and utilization. Data analysis uses statistical methods to reduce, summarize, and analyze data into meaningful information. These analyzed data are then utilized for some decisionmaking process.

THE STATISTICAL ANALYSIS SYSTEM (SAS®)*

The Statistical Analysis System is a computer software package that provides utilities for data management, statistical analysis, report writing, and graphics. Although SAS® is not a pure DBMS, it does provide an organization with the utilities to manage data. Using the simple free-format SAS® language and the numerous procedures the end-user is able to manage, analyze, and present data. The job would otherwise require a computer specialist and many programming hours. In addition, SAS® can be interfaced with Fortran and PL/I program languages.

Data management utilities include data entry techniques, data and file manipulation, and documentation. The capabilities for reading data are very flexible in allowing list, column, or formatted input from various devices (i.e., cards, tape, disk). In addition, multiple observations can be created from one input record, or several input records can create one observation. Similarly, one reading of an input file can produce multiple output files, or several input files can be read simultaneously to produce one output file.

Other management utilities exist to manipulate data and files for organizing, managing, and storing data values. SAS® provides both a line and full screen editor for editing and updating data. Manipulation of data values and files is achieved by using the internal language with its many statistical and mathematical functions and expressions. Tools for transforming data values, creating or deleting variables and observations, as well as file management tools for sorting, subsetting, concatenating, match merging, and interleaving, are available.

Documentation is an important part of any system. SAS® files are automatically self-documenting, since all data values are described by variable name and whether it is numeric or character. Additionally, data values are documented as to the number of bytes used for storage, location in the file, formats used to read or print, and a 40-character descriptive label. Source statements and comments used in creating the file, time, date, and number of observations are also documented.

As a tool for data analysis SAS® provides numerous statistical procedures, ranging from descriptive statistics for data reduction and summary to more complex statistical procedures of multivariate analysis, regression, and analysis of variance. Data results can be presented as a report using SAS® report writing capabilities or the user can write tailored reports using the internal language. In addition, a graphics package provides procedures

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* SAS® is the registered trademark of SAS Institute Inc., Cary, NC 27511-8000.

for producing plots, charts, maps, contour plots, and 3-dimensional plots. Utilities for producing color graphics are also available.

RESERVOIR FIELD STUDIES DATABASE MANAGEMENT SYSTEM

We in the RFS use existing computer facilities and SAS® in order to alleviate our data management problems. SAS® not only provides us with a powerful statistical package, but it also provides us with data management utilities, report writing, and graphic presentation capabilities. Organizations such as the RFS that do not require full-blown DBMS's or do not have DBMS-trained people can take advantage of the database management utilities in SAS®. The RFS DBMS was developed using concepts described earlier.

Data acquisition involves our experimental design, field sampling, and laboratory analysis. All RFS personnel are involved in experimental design, which includes the planning and strategy of sample collection and laboratory analysis. Determining the period and frequency of sample collection, the number and location of stations, variables to be measured, and techniques and methods to be used are all part of experimental design. Sample collection involves the collection of field measurements and water samples, ensuring that proper methods and techniques are followed. Laboratory analysis ensures that proper methods and techniques are performed, as well as establishing necessary quality control. In addition to collecting and analyzing samples, field and laboratory personnel have the responsibility for recording data values accurately. The RFS's code form design incorporates the needs of both those recording data and those entering data into the database system. Errors due to transcription from field or laboratory notebooks to appropriate computer forms for system entry have been eliminated by providing a direct link between data recording and data entry.

Data maintenance consists of the overall handling of recorded values from entry to storage. Code forms are reviewed for completeness and photocopied upon arrival from field or laboratory. The copied forms are used for entering data into the database system. The computer data file is scanned for errors using an editing program written in SAS® language. This program checks for the presence or absence of identifying variables and verifies that ranges of quantitative variables are acceptable. While this program can locate many errors, subtle errors may only be detected through point by point verification. Therefore, a printout of the data file is checked by RFS personnel with the original code forms. This step may seem time consuming and tedious, but it has proven very important in ensuring that the data values are as accurate as possible. After verification and corrections have been made the files are consolidated and grouped logically by reservoir, study area, study type, and period. This file structure was chosen because it provided the end-user with a functional data package for statistical analysis.

Ultimately the files are stored on a central database disk. This central disk is composed of five mini-disks, one for each reservoir and one for miscellaneous data files. All privileged users have read access to any mini-disk by simply entering a keyword for that mini-disk. Security of the database is maintained by permitting users read only access. In addition, only the DBA or people with permission have write access to the database, thereby ensuring the integrity of the data. Protection from system failure and the loss of data files are established by maintaining tape backups on all files.

The final phase in the RFS DBMS is to analyze and utilize our water quality data. We use SAS® statistical, graphical, and report writing capabilities for processing and presenting our water quality data. The statistical procedures provide a means for analyzing large packages of data into meaningful information. We use SAS® graphics for exploratory analysis as well as presentation of research results. The report writing capabilities provide a way to format data as requested by other users.

CONCLUSION

The objective of the RFS DBMS was to provide noncomputer type people the utilities for entering, maintaining, analyzing, and presenting large amounts of water quality data. Using SAS® as a base we have included database management concepts to develop a system that solves the RFS data management problem.

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WATER QUALITY DATA ANALYSIS

by

Robert F. Gaugush*

INTRODUCTION

Statistical analyses attempt to infer the characteristics of a group (the target population) by analyzing the characteristics of a small segment of the group (the sample population). Proper execution of the inferential process, data analysis and interpretation, is essential if the data collection program is to be cost-effective and provide significant information about the target population. This paper presents a brief discussion of the exploratory and confirmatory phases of data analysis and represents a distillation of an EWQOS technical report entitled "Statistics for Water Quality Investigations" that is currently being prepared at WES.

EXPLORATORY DATA ANALYSIS

The objective of exploratory data analysis is to uncover important properties of the data with the use of simple graphical displays and basic descriptive statistics. The amount of data that results from any water quality monitoring program is extremely large and exploratory data analysis represents a means by which the data can be examined in a manageable format. This phase of data analysis is essential because it not only familiarizes the investigator with the data but it also serves to direct the confirmatory phase of data analysis.

Data Displays

Before a data set is used to calculate descriptive statistics or to perform any statistical analysis, it is extremely useful to look at various displays of the raw data. Simple graphical displays can help identify the need to edit or transform the data prior to conducting the statistical analysis. Most methods in statistics use summary values (e.g., mean and standard deviation) and if the inferences made from the statistical analysis are to be valid, then the summary values must be representative of the entire data set. Simple data displays in the selection of the proper summary statistics can help to assure that the inferences drawn from a given analysis are valid.

Many statistical procedures assume that the data are distributed normally and deviations from a normal distribution may result in invalid inferences based on the statistical procedure. Frequency histograms can be used to determine if the data on a single variable approximates the normal distribution. The histogram provides a representation of the distribution of the sample, which is of considerable value in the selection of descriptive statistics. The influence of the shape of the distribution on the selection of descriptive statistics is discussed in the section on basic descriptive statistics.

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Data displays are also useful in the identification of outliers. There is no single accepted definition of the term "outlier," but the term is used to identify observations that stand apart from the remainder of the data set. Outliers can exert more than their fair share of influence on the value of a number of statistics. Outliers must be carefully examined to determine if they are legitimate observations rather than the results of sampling, analytical, or coding errors. If the outliers can be rejected from the data set (i.e., they are legitimate observations), then it may be necessary to transform the data in order to reduce their influence. Often, the logarithmic transformation is sufficient to reduce the undue influence exerted by outliers in water quality data.

The most important role of data displays lies in their use to discover patterns in the data. Determining how the measured variables behave with respect to space and time is crucial to the interpretation of water quality data. Spatial patterns include the commonly observed changes with depth, as well as the longitudinal changes that result from advective transport which dominates many reservoirs. Temporal patterns can be diel, seasonal, or long-term (over years). The presence of consistent spatial and temporal patterns may allow for a modification of sample design that can lead to a reduction of effort. Identification of homogeneity in either space or time can be developed into a stratified design where the number of samples taken in a given area or over a given interval of time can be reduced.

Many statistics (e.g., the correlation coefficient and many statistical procedures such as regression) are basically concerned with the relationship between two variables. The simplest and most efficient way to examine the relationship between two variables, a bivariate relationship, is to use a scatter plot. A scatter plot is simply a two variable plot of the data on an x-y coordinate system. Scatter plots will indicate the nature of the relationship, if any, between the two variables as well as indicate the existence of any bivariate outliers.

Basic Descriptive Statistics

Water quality monitoring programs result in large amounts of data that must be summarized in order to effectively transfer information. In summarizing data a choice is made to sacrifice some of the information contained in the entire data set for the convenience of a few well chosen descriptive statistics. It is essential that as much information as possible be summarized by the descriptive statistics because the alternative may be a misrepresentation of the original data.

Generally, a data set can be adequately summarized by a measure of the central tendency and by a measure of the dispersion about the central tendency. Candidate statistics for central tendency include the mean, median, and mode, and the candidates for dispersion are the range, standard deviation, and interquartile range. The mean and standard deviation is justified (when the data are normally distributed) but situations can and do arise where these statistics can misrepresent the data.

A measure of central tendency is probably the single most useful statistic to summarize a data set. There is no single, unambiguous definition of the center of a data set but the concept implies either the middle of a set of points or the region where points are most common. Given that samples can exhibit a variety of distributions when plotted as a frequency histogram, there is no single measure of central tendency that is adequate for every situation. In samples that approximate the normal distribution any of the candidate statistics (the mean, median, or mode) provide an efficient estimate of the central tendency. If, on the other hand, the data are skewed either right or left, the estimators of central tendency diverge. The mean is "pulled" in the direction of the skew and no longer indicates the position of the majority of the observations. In situations where the data are skewed considerably, the median and particularly the mode are more efficient estimators of the central tendency.

Measures of dispersion are used to describe the variability of the data about the center of the distribution. The standard deviation is the most commonly used measure of dispersion, and with data that approximate the normal distribution the dispersion is effectively summarized by the standard deviation. But, like the mean, the standard deviation is considerably influenced by the presence of skew. Skewed distributions result in a standard deviation that overestimates the dispersion in the shorter tail and underestimates it in the longer tail. When the data are skewed it is preferable to use an alternative measure of dispersion that is not unduly influenced by the presence of extreme values. The interquartile range, by describing the spread about the median, is the candidate statistic that should be used to estimate dispersion in skewed samples. Use of the sample range (difference between the highest and lowest values) as the only measure of dispersion is not recommended because it is not an efficient estimator of dispersion in that it considers only two values from the entire sample.

CONFIRMATORY DATA ANALYSIS

The objective of this phase of data analysis is to statistically confirm the presence or absence of certain properties in the data. Confirmatory data analysis is directed by both sample design and the results of the exploratory data analysis. Sample design determines the extent of what can be statistically inferred from the data. For example, data acquired from a sampling program using a completely randomized design to determine the average phosphorus concentration of a given reservoir may not be able to statistically confirm the presence of spatial patterns. The patterns observed in the exploratory phase direct the confirmatory phase in the sense that they suggest which statistical techniques are best suited to confirm or deny their existence. Confirmatory techniques can be conveniently divided into parametric, nonparametric, and multivariate statistics.

Parametric Statistics

Parametric statistics is a term used to describe a body of statistical procedures that test hypotheses about population parameters by examining sample statistics. These methods are the most commonly used procedures in confirmatory data analysis and include t-tests, analysis of variance, regression analysis, correlation, and others. In fact, these methods are used so often that the underlying assumptions of these methods are usually not considered before their application.

All parametric procedures assume that the samples have been drawn at random for a population with a normal distribution. Many procedures, particularly those involving multi-sample hypotheses, also assume that the variance is homogeneous or equal between samples. These assumptions may not be met in all cases and severe deviations from normality and equality of variance may result in drawing invalid inferences about the population. Examination of the data using histograms can indicate whether or not the sample seriously deviates from the normal distribution, and tests concerning the homogeneity of variance do exist. Fortunately, most parametric procedures are sufficiently robust to withstand considerable departures from their assumptions, but situations can and will arise when the use of parametric statistics is not warranted.

Nonparametric Statistics

Nonparametric tests represent a set of statistical procedures that can be used when serious violations of the normality and/or homogeneous variance assumptions are either known or assumed. These tests do not utilize estimates of the variance, mean, or any other population parameters and do not consider hypotheses about parameters -hence the term "nonparametric." Nonparametric methods generally rely on a less stringent set of assumptions than parametric procedures and as a result these methods are sometimes referred to as "distribution-free" methods. The ability to analyze data without meeting the constraints of normality and/or homogeneous variance is not without some cost, however. Nonparametric methods usually result in 1) less specificity in the precise nature of the differences between populations and 2) less power to detect differences that do in fact exist.

Most nonparametric tests employ the ranks of the measurements rather than their actual values. For example, suppose five samples for turbidity were taken from each of three regions (headwaters, mid-pool, and near-dam) in a reservoir. All of the observations are then ranked in ascending order irrespective of the region from which they were drawn. As a result, assuming there are no tied ranks, the ranks will range from 1 to 15. If no differences exist between regions, then one would expect the means of the ranks from each region to be equal. On the other hand, if differences did exist (e.g., headwaters > mid-pool > near-dam) then the means of the ranks should reflect the underlying differences. Rank differences form the basis for most nonparametric techniques.

Multivariate Statistics

Often, studies of water quality involve multiple variables, multiple samples and/or multiple bodies of water. In situations such as these, the data analysis will be both univariate and multivariate. Multivariate statistics can be used to greatly enhance the understanding of water quality relationships in and among reservoirs. Given the existence of computer software with multivariate applications, these methods can be used as easily as their univariate counterparts. As with univariate statistics the use of multivariate methods must consider the assumptions behind their application.

The assumptions that are important to multivariate statistics are essentially the same as those of the parametric statistics. The key assumptions for multivariate methods concern normality, independence of observations (i.e., random samples), homogeneous variance, and linearity. It is important to note that the assumptions do not hold for all methods nor do they necessarily hold for all applications of the same method. As with parametric statistics, mild violations of the assumptions do not seriously affect the inferences drawn from the data.

The types of applications of multivariate statistics can be grouped into relatively few categories. Multiple and canonical correlation can be used to characterize the strength of a relationship between and/or among variables. Cluster analysis is used to classify groups of observations based on their relative similarity. It is not uncommon that data on multiple variables represent one or a few fundamental characteristics. Principal component and factor analysis can be used to extract this fundamental structure from a data set. Discriminant analysis can be employed to develop predictive relationships for assignment of new observations to predefined groups.

CONCLUSIONS

Extensive data analysis is often neglected in many water quality monitoring programs due to limited expertise and/or time and funding constraints. The existence of time and funding constraints cannot be altered but the lack of expertise can be changed. This paper represents an overview of a EWQOS technical report that will provide the necessary background information for the statistical analysis of water quality data.

AN OVERVIEW OF RESERVOIR WATER QUALITY CONCERNS AND TECHNIQUES OF INVESTIGATION

BY

Richard G. Hunter[1]

INTRODUCTION

Water quality control in reservoirs is a diverse subject lending itself to a temporally-based discussion. This paper will examine typical water quality concerns associated with various phases of the reservoir construction and operation process. It will begin with pre-impoundment studies, progress through problems with water quality in new impoundments, and culminate in an examination of problems with reservoir release schedules and low-flow regulation. Throughout the discussion various examples of problems encountered by the Tulsa District will be used to illustrate major points.

PRECONSTRUCTION ENVIRONMENTAL ASSESSMENTS

Due to the influence of weather on stream water quality, pre-impoundment studies can be performed rapidly and at reduced expense if a large historical data base is available. The District relies almost exclusively on the Environmental Protection Agency's (EPA) STORET system for such data because the two main water quality monitoring agencies in the District's region, the United States Geological Survey (USGS) and the various Health Departments, use it extensively. This information can be supplemented by existing studies on the area, such as graduate research. Analysis of historical data is initiated in the reconnaissance phase of the planning process to allow gaps in the existing data base to be supplemented, and so that known problem areas can be addressed during the remainder of the planning process.

Alteration of the existing flow regime and water quality by new or modified projects has very different impacts on the upstream and the downstream project areas. The District has dealt with two major problems from downstream effects; the first is disruption of waste load allocations for existing waste sources. The second is impact on the biota caused by seasonal hypolimnetic discharges. In most States, waste load allocations (the legal right to discharge waste to a watercourse) are based on some minimum flow duration. In Oklahoma, this is the lowest average flow expected to occur over seven consecutive days during any two years, and this value is published for various stream segments (USGS 1978). Multi-purpose reservoirs, particularly those with hydropower, frequently reduce stream flow below this 7-day, 2-year low flow. This reduces the assimilative capacity of the

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downstream reach and exacerbates the effects of legal, existing waste sources, usually by failure to meet dissolved oxygen (DO) standards. Hypolimnetic discharges are also low in DO, as well as exhibiting lowered pH and elevated iron and manganese.

The District's primary tool for predicting downstream effects and developing release schedules for operation is the EPA QUAL-II stream quality model which is available at no charge through EPA's Environmental Research Lab in Athens, GA. Tuition at user training courses is also free and the model is relatively easy to use. The model can simulate up to 13 constituents in any combination desired by the user and allows for multiple waste discharges, withdrawals, tributary flows, and incremental inflow (Roesner et al 1981).

The planned addition of hydropower generation capability to several District projects provides an example where both hypolimnetic discharges and reduced minimum flows are involved. In this case, discharges from the dam are considered the headwaters and the quality varies between hypolimnetic discharges of great volume for relatively short duration and epilimnetic discharges of small volumes for longer periods. Approximately 15 km (9 mi) downstream, a municipal source discharges waste with a biochemical oxygen demand (BOD) of 45 mg/l at a rate of 2.0 cfs. This source was permitted based on a low flow under present regulated conditions of 3.5 cfs (Oklahoma Water Resources Board 1979). The problem to be investigated is the need for flow maintenance either through pass-flows during non-generation periods or through construction of a re-regulation dam. Because the flows in a QUAL-II simulation must be fixed for each run, it is necessary to make several runs with differing flows and dissolved oxygen and temperature. Figure 1 shows the results of dissolved oxygen modeling from the headwaters to a point 11 km (7 mi) below the waste discharge. Under critical summer, low-flow conditions, DO remains above the Oklahoma standard for cool-water fisheries of 6 mg/l if flow exceeds about 3 cfs from the epilimnion. Hypolimnetic flows greater than 17 cfs, the anticipated gate leakage, also met the DO standard within 1.6 km (1 mi) of the dam.

WATER QUALITY CONTROL IN NEW IMPOUNDMENTS

The primary problem areas the Tulsa District has found in new impoundments have been associated with decaying organic matter. These problems are manifested in elevated BOD and chlorine demand in the lake water and in high concentrations of hydrogen sulfide in the hypolimnion of stratified lakes. Hydrogen sulfide production from the hypolimnion of Keystone Lake, Oklahoma, was so large that hydropower generation equipment was corroded and downstream recreational usage reduced during the first year of operation. One year after normal pool was reached following a five-year staged fill,

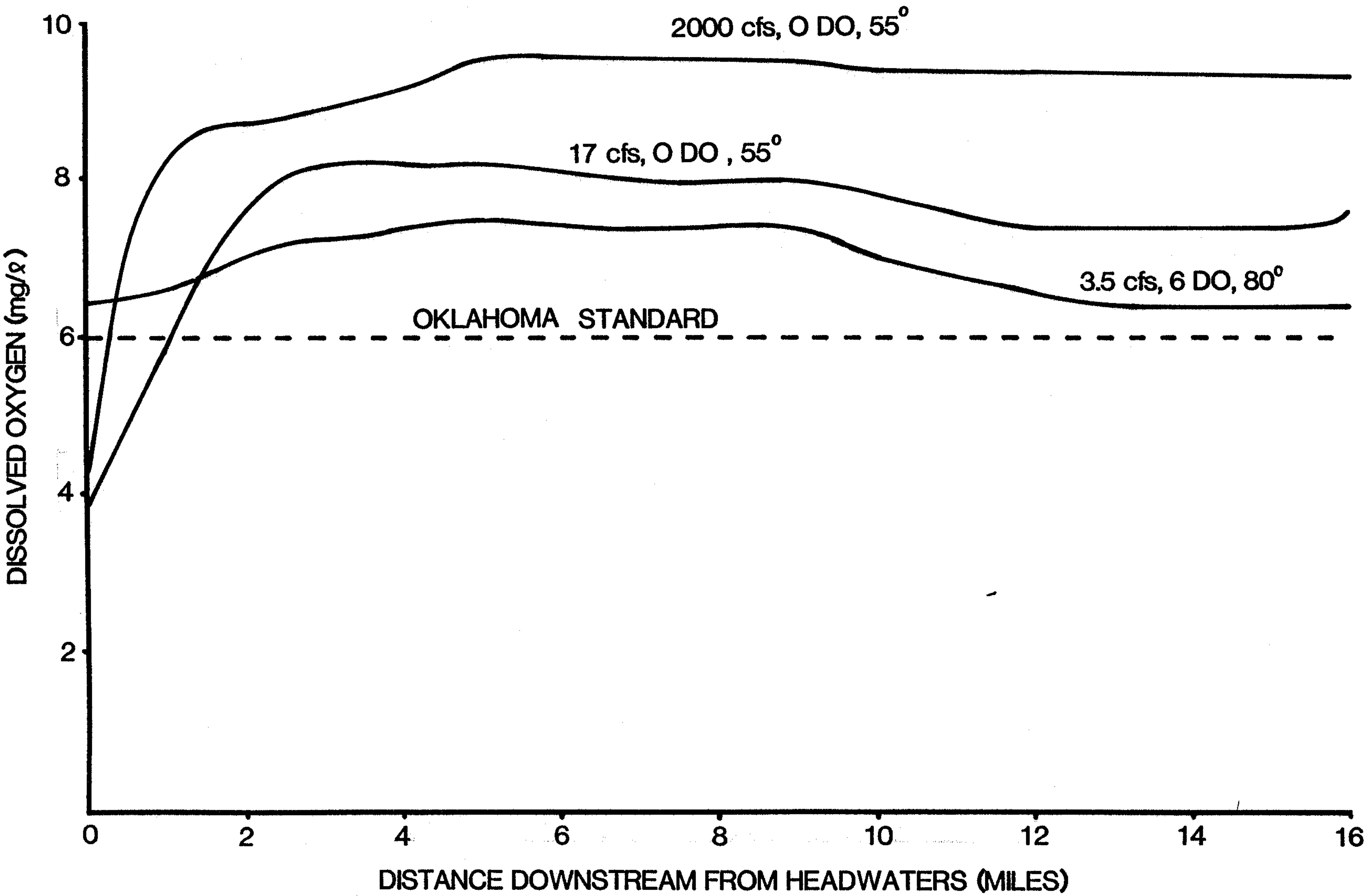

FIGURE 1. PREDICTED DISSOLVED OXYGEN BASED ON VARIED RELEASE CONDITIONS.

Waurika Lake in southern Oklahoma continued to exhibit a mean 5-day, 20°C BOD of 6.4 mg/l. This is higher than the mean of 1.9 mg/l of 14 BOD measurements from 3 nearby, older reservoirs. The increased chlorine demand can be of importance in lakes with a municipal water supply function, and there is some evidence to suggest increased taste and odor problems in municipal water are associated with this situation (Williams Brothers 1982). These not only result from decaying humic substances but also from the closer proximity of the intake to the photic zone with its higher algae populations and actinomycetes. More insidious has been the recent contention of Kansas Health authorities that inadequately cleared basins are associated with elevated trihalomethanes in municipal water. Their theory is the decaying plant material increases organic precursors which combine with chlorine in the treatment process to aggrevate trihalomethane formation. The District is fortunate that our last remaining Kansas construction project was in a relatively treeless area but this problem may arise in other Districts.

During the last 5 to 10 years it has become standard procedure in the Tulsa District to fill reservoirs in stages. This is done at the request of State agencies to prolong the dramatic growth of sportfish populations occurring in the first few years after impoundment and usually involves a rapid initial filling to a level where boat launching ramps and water intakes are operational. Normal pool level is then reached over the next three to four years. In the initial stage the volume innundated may represent 60 percent of the total normal pool, and there is little water quality benefit from this portion of the plan as compared to non-staged filling. The impact of water quality problems may be lessened by the increased water volume acting upon the narrower band of vegetation in the delayed-fill portion of the plan. The fact this usually involves temporary excursions into the vegetation at higher elevations also lessens water quality degradation.

RELEASE SCHEDULES

Many Tulsa District projects are operated under water quality release schedules which vary release volume by season. Projects constructed before the mid-1970's were justified partly to maintain downstream assimilative capacity and prevent dissolved oxygen depletion during critical, low-flow periods. The volume and schedule of these releases were determined by the EPA or its predecessor agencies. The EPA has greatly limited allowable water quality benefits in the construction of new reservoirs since the mid-1970's, when Federal efforts were aimed at eliminating point source discharges of wastes. This effort has resulted in marked improvement in water quality of many streams; yet in most cases water quality release schedules have not been modified to reflect such changes. Because improvement in point source discharges would not manifest themselves in environmental catastrophes, there is little incentive to determine modifications to the existing schedule.

Waurika Lake again offers a good example of this reasoning. Initial planning studies for the Waurika Lake project began in October, 1962, in response to a resolution by the U.S. Senate Committee on Public Works (U.S. Senate 1963). These studies identified chronic problems with low dissolved oxygen levels in Cow Creek between the cities of Duncan and Waurika and in East Cache Creek near Lawton (Fig. 2). Cow Creek is a tributary of Beaver Creek with the confluence below Waurika Dam. Claridy Creek, a tributary of Cow Creek, is an effluent dominated stream receiving effluent from the City of Duncan. East Cache Creek is a tributary of the Red River and is outside the primary basin of the Waurika Lake project. In addition to sewage treatment plant effluents from Duncan and Lawton, six smaller cities, a large oil refinery, and Ft. Sill, Oklahoma, all discharged effluents into Cow Creek, Beaver Creek, or East Cache Creek. Studies by the U.S. Public Health Service (PHS) documented the need for water quality flows in these two creeks based on a required DO standard of 4.0 mg/l. The PHS report (U.S. Senate 1963) identified a varying need for water quality flow over the life of the project. This need was dependent on the increasing demand for more important municipal water supply. The maximum requirement for water quality flows averaged 16.6 million gallons per day (mgd). Pumping for water quality augmentation was to begin in 1980 and a schedule which called for the majority of pumping to occur in the summer was supplied in the PHS report. That agency assigned an average benefit to the Waurika Lake Project for water quality control of $995,000 annually through 2000. The annual water quality benefits over the 100-year project life are $853,440 (1982 basis).

The Federal Government entered into contracts with the Waurika Project Master Conservancy District based on the study findings. This local group assumed the costs of the construction, operation, and maintenance of a pipeline to convey both municipal water and water quality flows. One branch of the pipeline runs 35.4 km (22 mi) to a discharge point on East Cache Creek near Lawton. A second branch discharges water quality flows to Claridy Creek below Duncan, a distance of 9.3 km (5.8 mi).

Between the 1963 PHS studies and 1980, some major changes influencing pollution loads in both the East Cache Creek and Cow Creek basins included:

1. Construction of an advanced wastewater treatment plant by Lawton. The plant, discharging into East Cache Creek, provides a much higher level of treatment than its predecessor. The old facility was designed

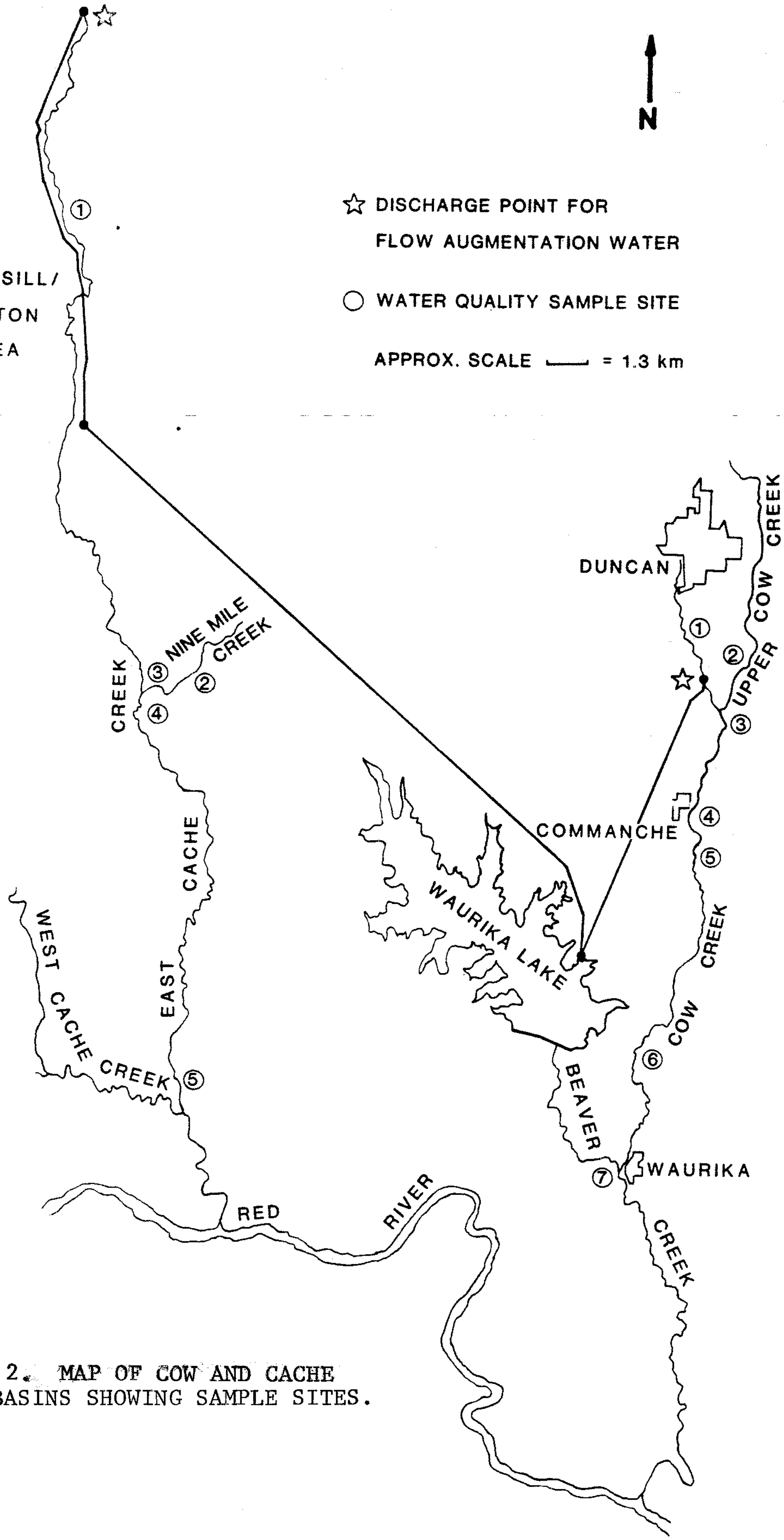

FIGURE 2. MAP OF COW AND CACHE CREEK BASINS SHOWING SAMPLE SITES.

to treat 6 mgd but had an average input of 8 mgd in 1977. The new plant is capable of treating 13 mgd.

2. An expansion and upgrading of the Ft. Sill sewage treatment plant in 1977. The plant capacity was increased by 50% and the load into East Cache Creek was measurably reduced.

3. A failure of population growth in the area to meet predictions used in the 1963 studies. The actual 1980 population level of nearby communities was only 73% of projections (Bureau of Census 1980).

4. A reduction in the Oklahoma standards for DO. Earlier standards were 5 mg/l with a 1 mg/l excursion allowed for diel fluctuations; the 4 mg/l requirement used by the PHS followed this reasoning. The 1982 Oklahoma standards allow certain classes of impacted streams (e.g. East Cache and Cow Creeks) an absolute minimum DO of 3 mg/l (OWRB 1982).

5. Dramatic increases in fuel costs leading to increased costs to pump the flow augmentation water.

The Tulsa District implemented a study in 1981 to determine if flow augmentation was desirable in light of the above changes. Detailed study methodology was reported by Hunter et al (1984). Few violations of the new Oklahoma standard for DO were found in the East Cache Creek basin and modeling efforts were directed only at the Cow Creek basin. Numerous violations of the standard occurred in the Cow Creek system creating a zone of depressed DO below the confluence with Claridy Creek. The length of this DO sag zone ranged from 0.8 to 11.6 km (0.5 to 7.2 mi).

The high BOD in the Waurika augmentation water resulted in a parabolic relationship between BOD loading, effluent flow, and pumped flow in Claridy Creek. When little or no water was pumped to the intermittent creek, the normal sewage discharge of 4 to 8 cfs dominated the stream flow and BOD loading was high for the resulting volume of water. When large amounts of the Waurika Lake water, containing an elevated BOD, was used for flow augmentation, BOD loading remained high. Optimum mixtures, resulting in the lowest BOD loading, occurred at various combinations of Claridy Creek flow and augmentation. Pumped flows of 6 to 10 cfs were most effective.

The rational method proposed by Velz (1939) was used to model DO levels in the Cow Creek system for three periods when measured DO and flow were available. These periods spanned the range of observed flows. Models for the maximum and minimum flows are shown in Figure 3. In each case, model verification was excellent with only the most downstream station showing significant deviation from observed values. This was of little consequence since the station was well downstream of the critical zone of DO depletion.

An examination of these three data sets revealed flows in Cow Creek above its confluence with Claridy Creek (upper Cow Creek) are critical to predicting DO levels downstream. The ineffectiveness of flow augmentation was shown by setting flows in upper Cow Creek to an observed critical low flow of 0.2 cfs and modeling augmentation at two flow rates: 6.6 cfs and 24.5 cfs. These are the flows for minimum BOD loading and the maximum capacity of the outlet works, respectively. In both cases DO depletion was severe and lengthy. Having demonstrated a minimum flow in upper Cow Creek at which it was impossible to prevent violation of the DO standard through flow augmentation, the model was used to determine the actual value. This flow was found to be 3.2 cfs. It was further reasoned there was a maximum flow in upper Cow Creek above which augmentation was not needed. Again the rational method was used and the critical maximum flow was found to be 7.0 cfs. Adjusting flows from a downstream USGS gage showed flows between 3.2 and 7.0 cfs in upper Cow Creek only about 4.4% of the time. The expected depletion zone length and minimum DO for the various models are shown in Table 1.

Annual costs (1982 basis) for operation and maintenance of the water quality portion of the conveyance system are $727,000 for the East Cache Creek segment and $245,000 for the Cow Creek basin. Thus, the current cost of pumping the water now exceeds the $853,440 in annual water quality benefits attributed to the project by the PHS. Unfortunately, the $972,000 spent annually on flow augmentation would be mostly wasted since the East Cache Creek augmentation was not needed and the Cow Creek portion would be ineffective the majority of time. Based on this information, four alternatives were considered:

1. Operate the flow augmentation system as per the original PHS schedule. This would have a total annual cost of $972,000 and would clearly be an undesirable use of money and energy based on the limited benefits.

2. Abandon the concept of water quality improvement by flow augmentation in both basins. This assumes any benefits in Cow Creek during the time of flow augmentation would be negated by DO depletion during

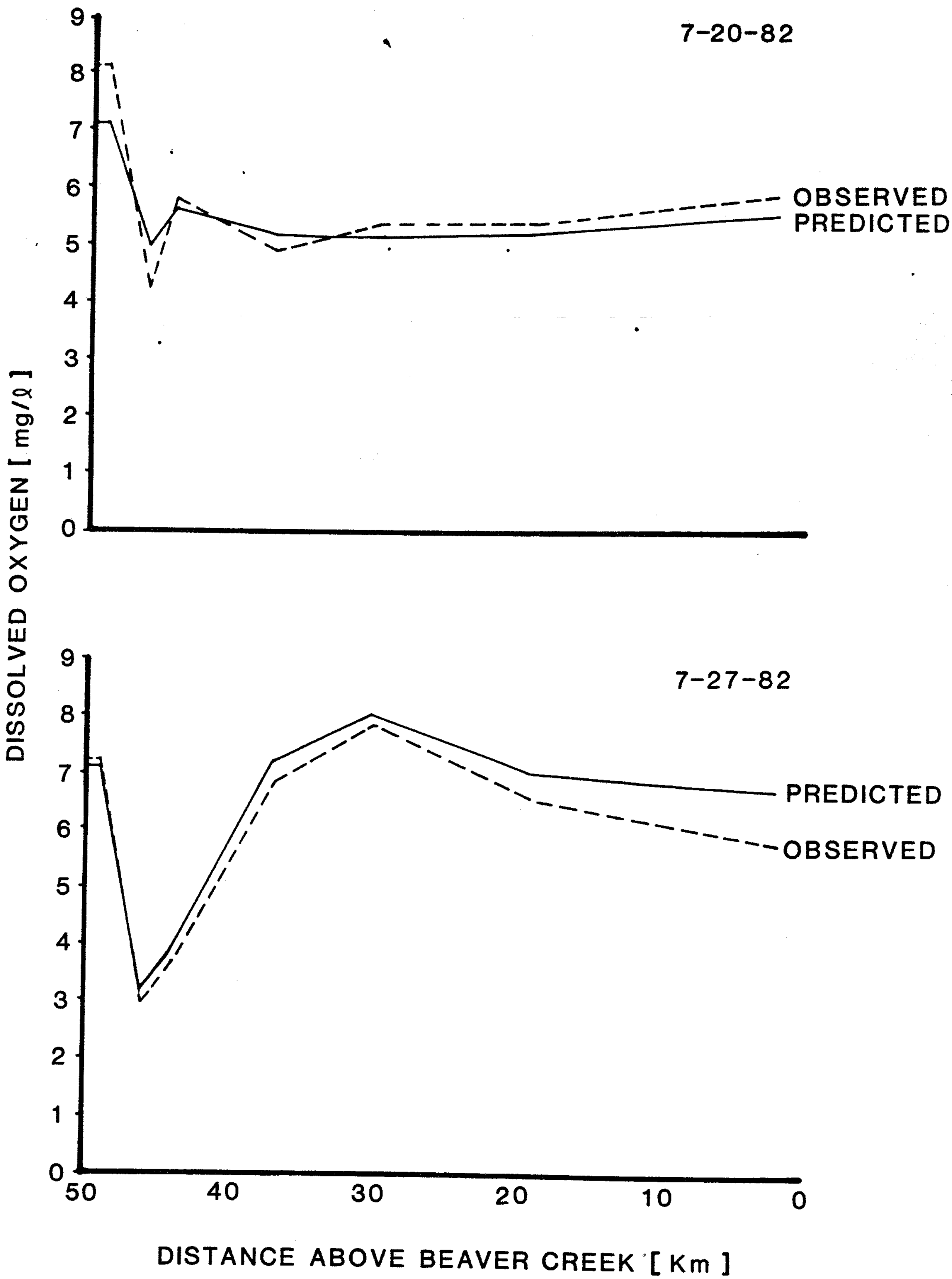

FIGURE 3. DISSOLVED OXYGEN LEVELS IN COW CREEK AT MAXIMUM (7-20-82) AND MINIMUM (7-27-82) OBSERVED FLOWS.

Table 1. Modeled DO in Cow Creek at various flows

Flow (cfs)				
pumped	Claridy Creek	upper Cow	minimum DO %	violation length (km)
0	4.4	0.2	15.1	11.6
0	7.3	2.9	39.3	0.8
0	8.0	7.0	51.5	--
0	8.0	7.1	45.0	--
0	8.0	8.8	59.7	--
6.6	5.0	0.2	9.1	11.4
10.0	8.0	3.0	30.1	2.4
24.5	5.0	0.2	4.3	8.0
24.7	8.0	3.2	43.0	0.4

critical low flows. Thus, the biota would be altered or destroyed by the low dissolved oxygen even with the flow augmentation. This alternative would have no annual operating costs, but the depreciation of the water quality portion of the project would continue.

3. Pump water quality flows only in the Cow Creek basin and only when flows in Upper Cow Creek were between 3.2 and 7.0 cfs. This would result in the minimum number of water quality violations and would cost only $13,105 annually.

4. Allow water quality flows to remain as a project purpose without formally scheduled augmentation in the two streams. This would allow emergency demands to be met and provide for any unanticipated changes that might require pumping. This alternative was approved by the OWRB.

This study illustrates the environmental planning process must continue through the construction phase and even into the early operation of some projects. Following the course of action dictated 20 years previously would have resulted in unnecessary annual costs of $972,000 for 18 years, a total of $16,524,000. It is probable other projects planned and/or implemented during the 1960's and early 1970's might be profitably re-evaluated based on later improvements in environmental quality. The District is presently using the QUAL-II model to investigate the need for water quality flows at the other 12 projects with water quality as a purpose.

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CONTROL OF NUISANCE ALGAE BLOOMS AND PRELIMINARY EVALUATION OF ALGAE CONTROL THROUGH RESERVOIR REGULATION

By

Christina C. Meshaw[1]

CONTROL OF NUISANCE ALGAE BLOOMS

Reduction of nutrients at the source is the most desirable way to prevent nuisance algae blooms. It is probably one of the most, if not the most, desirable means of controlling nuisance algae blooms over the long term after they begin occurring in a water body. There are many other ways to control nuisance algae blooms, most of which are effective only over the short term or with repeated treatment. Janik et al. (1980) listed four categories of control measures: chemical, physical, biological, and some combination of the three. The control measures included in Janik et al. (1980) are discussed in the context of controlling algae blooms in the reservoir, which is the portion of the inflow river/reservoir/downstream river system with the most frequent need for control.

Reservoirs, however, can aid control of nuisance algae blooms in the downstream river portion of the inflow river/reservoir/downstream river system. Reservoirs usually act as nutrient sinks, thus reducing the nutrient loading which is often responsible for nuisance algae blooms, to the downstream river. In a study of John H. Kerr Reservoir, Va. and N. C., annual retention of total phosphorus was 50 percent and of total nitrogen was 16 percent (Weiss, 1978). Weiss (1981) found that 60 percent of the annual total phosphorus load and 34 percent of the annual total nitrogen load to High Rock Lake, N. C., were retained. The nutrient trapping characteristic of reservoirs makes them a potentially valuable adjunct to other watershed nutrient control measures.

There is one case in North Carolina, the Neuse River downstream of Falls Lake, where control of nuisance algae blooms in the downstream river is needed. The purpose of this section of this paper is to discuss the idea that Falls Lake will act as a nutrient sink and help control nuisance algae blooms in the Neuse River downstream of Falls Lake.

The lower Neuse River has been plagued with algae blooms since the late 1970's. Algae blooms have occurred from Kinston to Oriental (see figure 1). Complaints from the public about visible algae blooms in the New Bern area in 1978 prompted an extensive investigation by the North Carolina Division of Environmental Management (NCDEM) (NCDEM 1981). The investigation led to the development of a nutrient management strategy for the Neuse River Basin. The strategey includes short-term actions, intensive planning and implementation of a water quality management plan (NCDEM 1983). One of the items addressed in the water quality management plan is to evaluate the effectiveness of Falls Lake as a nutrient trap.

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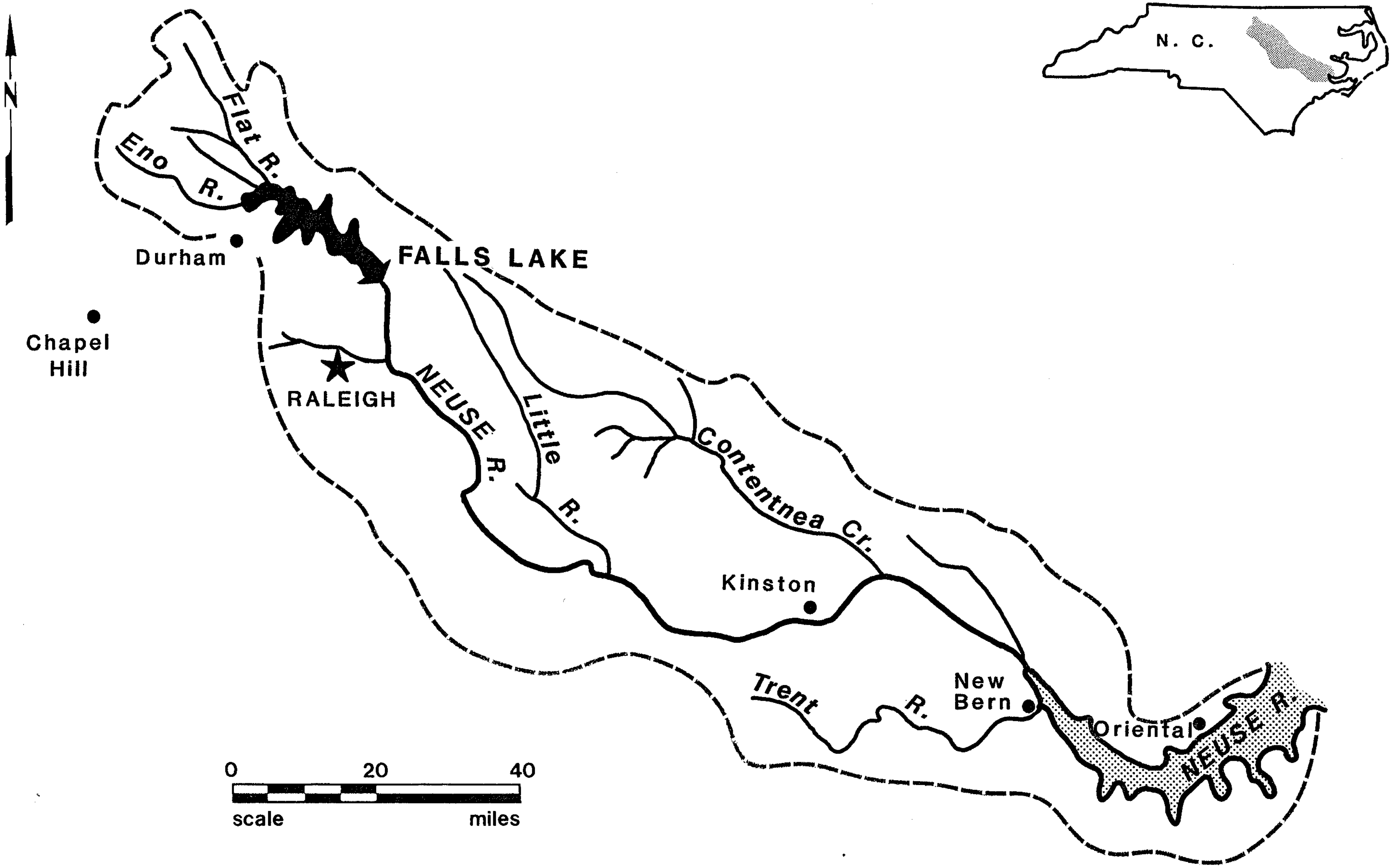


NEUSE RIVER BASIN, NORTH CAROLINA

U.S. ARMY ENGINEER DISTRICT, WILMINGTON, N.C.

Figure 1

The degree of annual nutrient trapping in Falls Lake has as an upper limit the portion of the basin annual nutrient load which enters the reservoir. NCDEM estimates that 16.95×10^4 kg/yr of total phosphorus and 8.63×10^5 kg/yr of total nitrogen enter the Neuse River Basin at Falls Lake (NCDEM 1983). The loadings are 17.6 percent and 11.7 percent of the total basin loads, respectively (NCDEM 1983).

Since Falls Lake was not impounded until 1983, there are not enough data to present actual annual nutrient retention in the reservoir, but it is reasonable to assume based on John H. Kerr Reservoir and High Rock Lake data that about 55 percent of the annual total phosphorus load and 25 percent of the annual total nitrogen load entering Falls Lake will be trapped. Thus, Falls Lake can potentially reduce the annual loading to the downstream Neuse River from 17.6 percent to 8 percent total phosphorus and from 11.7 percent to 9 percent total nitrogen.

During the summer (low flow) period, when algae blooms normally occur in the Neuse River downstream of the reservoir, it is probable that the percent retention of the total phosphorus and total nitrogen loads is even higher due to increased hydraulic retention time. Very preliminary estimates of nutrient retention by Falls Lake during the summer of 1983 are shown in table 1. The estimates are very preliminary because they are based on only three sampling dates. More than 90 percent of the total phosphorus and more than 35 percent of the total nitrogen were retained in Falls Lake on the dates for which data are available.

In conclusion, use of reservoirs as nutrient sinks where there are downstream algae blooms appears to have significant potential as a control method. There may even be some potential for nontraditional water quality benefits associated with use of reservoirs as nutrient sinks in such situations.

It is recommended that a thorough evaluation of the potential for algae blooms within the reservoir be made before proposing a new reservoir as a nutrient sink. Such an evaluation is necessary to assure that benefits of reducing downstream algae blooms via nutrient trapping are not negated by the occurrence of algae blooms in the reservoir which could impair its uses.

PRELIMINARY EVALUATION OF ALGAE CONTROL THROUGH RESERVOIR REGULATION

As discussed in the previous section of this paper, reservoirs can help control nuisance algae blooms by acting as nutrient sinks. Depending on how they are regulated, the effectiveness of reservoirs as nutrient sinks can vary. As part of its strategy to manage nutrients in the Neuse River Basin, the NCDEM is considering requiring that reservoirs be operated to maximize their efficiency as nutrient traps (NCDEM 1983). In the spring of 1983, NCDEM requested that the Wilmington District, Corps of Engineers, release water from the upper layers of Falls Lake, instead of the lower layers, in order to maximize nutrient retention by the reservoir during the critical low flow (summer) period. The basis of NCDEM's request is that nutrient

TABLE 1
Nutrient Loading, Falls Lake, NC

Date	Load Entering (kg/day)	Load Leaving (kg/day)	Retention (%)
	Total Phosphorus		
1 Aug 83	846	33	96
7 Sep 83	191	5	97
26 Sep 83	181	13	93
	Total Nitrogen		
1 Aug 83	3,120	285	91
7 Sep 83	640	56	91
26 Sep 83	597	379	37

concentrations are generally lower in the upper layers of North Carolina reservoirs than in the lower layers. This results from thermal stratification/deoxygenation and the accompanying accumulation of nutrients in the hypolimnion. The purpose of this section of this paper is to compare reservoir nutrient retention during the critical low flow (summer) period in a reservoir with surface withdrawal to one with bottom withdrawal. Falls Lake, N. C., is used as an example of a reservoir with surface withdrawal and B. Everett Jordan Lake, N. C., is used as an example of a reservoir with bottom withdrawal. It should be emphasized that this is a preliminary evaluation based on only three sampling dates during the summer of 1983.

Both reservoirs have multilevel intake structures. The Falls Lake multilevel intake structure has four water quality gates at two different elevations and two emergency gates at the bottom of the reservoir. During the summer of 1983, one or the other of the water quality gates with invert elevations at 231 ft. m.s.l. was open. Figure 2 shows that with a gate open at invert elevation 231 ft. m.s.l. withdrawal was from the upper layers of the reservoir, which had very low total phosphorus (<0.1 mg/l) and total nitrogen (<1.0 mg/l) concentrations compared to the hypolimnetic concentrations. The B. Everett Jordan Lake intake tower has eight water quality gates at eight different elevations and two emergency gates at the bottom of the reservoir. During the summer of 1983, one or the other of the emergency gates was open along with one water quality gate in the same wetwell. The Wilmington District, Corps of Engineers, Hydraulics Branch, estimates that about 90 percent of the release water comes off the bottom of the reservoir under those conditions. In contrast to Falls Lake, the B. Everett Jordan Lake nutrient profile shown in figure 3 indicates that water withdrawn from the lower layers of the reservoir had total phosphorus concentrations greater than 0.1 mg/l and total nitrogen concentrations greater than 1.0 mg/l. The concentrations were much higher than those in the surface layers and than those in the Falls Lake withdrawal zone. The outflow total phosphorus and total nitrogen concentrations for the two dates in table 2 illustrate the differences.

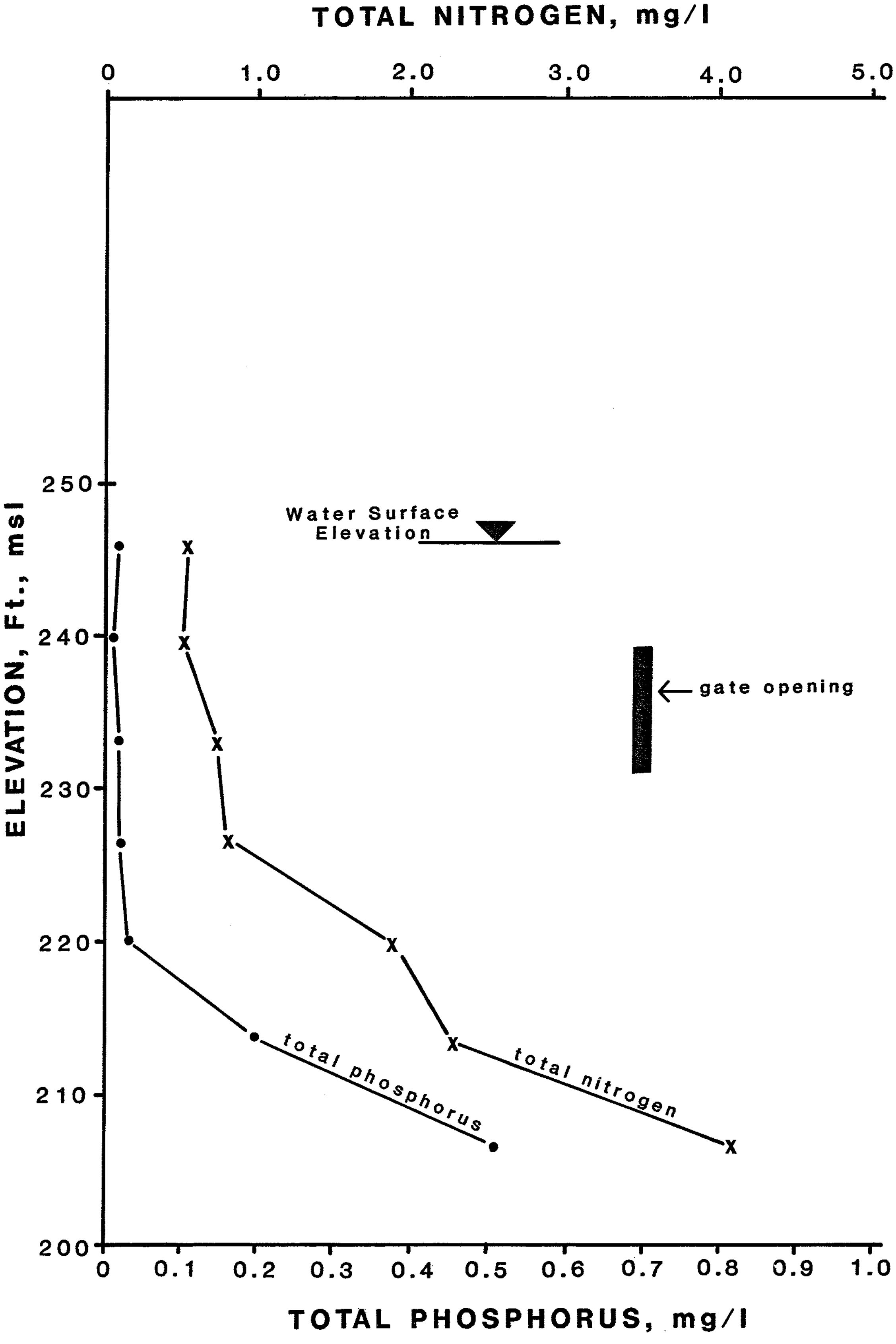
Table 3 shows that total phosphorus retention was substantially different for surface withdrawal (97 percent) than bottom withdrawal (64 percent). The range of total phosphorus retention for the three dates for which data were available was 93-97 percent for Falls Lake and 59-74 percent for B. Everett Jordan Lake.

Table 4 shows that total nitrogen retention was 91 percent for surface withdrawal and 43 percent for bottom withdrawal. The range of total nitrogen retention for the three dates for which data were available was 37-91 percent for Falls Lake and 8-43 percent for B. Everett Jordan Lake.

In conclusion, this preliminary evaluation shows that reservoirs can be regulated to maximize nutrient retention.

It is recommended that existing and projected reservoir and downstream nutrient loading be carefully considered when decisions are being made whether or not to construct multilevel intake towers at new reservoirs and at existing reservoirs where hydropower additions are being considered.

7 SEPT. 1983

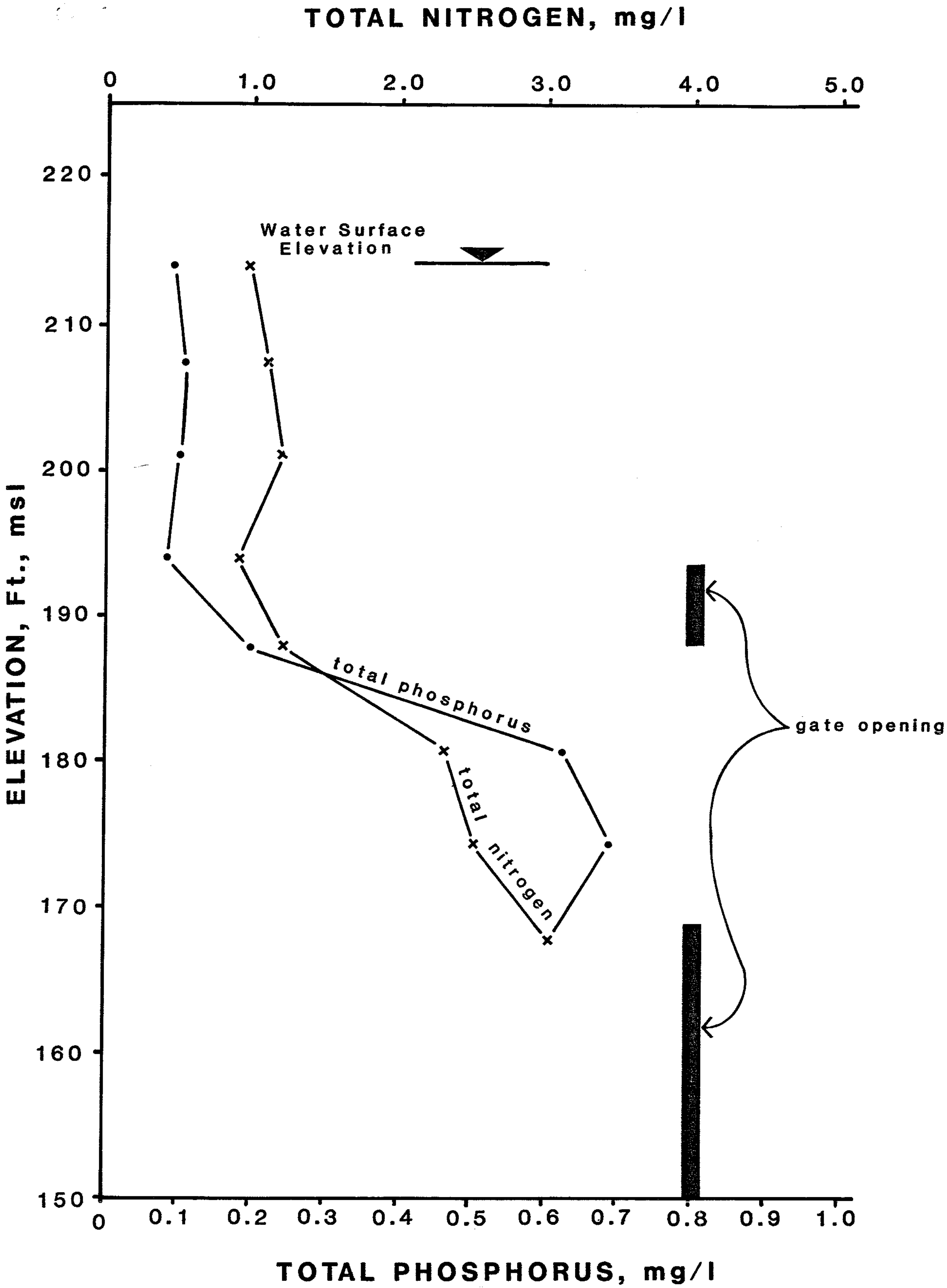


FALLS LAKE

U.S. ARMY ENGINEER DISTRICT, WILMINGTON, N.C.

Figure 2

10 AUG. 1983



JORDAN DAM and LAKE

U.S. ARMY ENGINEER DISTRICT, WILMINGTON, N.C.

Figure 3

TABLE 2

Outflow Nutrient Concentrations(mg/l)

	Total Phosphorus	Total Nitrogen
Falls (7 Sep 83)	0.03	0.35
Jordan (10 Aug 83)	0.2	1.9

TABLE 3

Total Phosphorus Retention

	Total Phosphorus (kg/day)		Retention (%)
	In	Out	
Falls (7 Sep 83)	191	5	97
Jordan (10 Aug 83)	657	239	64

TABLE 4

Total Nitrogen Retention

	Total Nitrogen (kg/day)		Retention (%)
	In	Out	
Falls (7 Sep 83)	640	56	91
Jordan (10 Aug 83)	3,944	2,267	43

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REVIEW OF METHODS OF RESERVOIR WATER QUALITY CONTROL

by

Jeffery P. Holland[1]

Problem Definition

During the late spring or early summer months many Corps of Engineers (CE) reservoirs become thermally stratified. The subsequent density stratification inhibits vertical mixing in these reservoirs, resulting in the formation of three vertical strata in the reservoir. The epilimnion, the upper region, contains warm, low-density water which is generally high in dissolved oxygen (DO) concentration due to surface exchange and wind mixing and is usually considered high-quality. The region of rapid temperature change just below the epilimnion is called the thermocline or metalimnion. The hypolimnion, the lowest region of the reservoir, consists of cooler high-density water which, due to stratification and oxygen demand, is often low or deficient of DO.

Stratification often presents a water quality problem for both in-situ and downstream releases from these reservoirs. The water released from these outlets, which often is either predominately or completely hypolimnetic, may be of generally poor quality due to its relative oxygen deficiency. During certain periods of the year, these waters may become anoxic, resulting in the potential release of high concentrations of reduced iron, manganese, and hydrogen sulfide. Further, in-reservoir aquatic biota residing in these regions are often adversely impacted by the poor water quality.

As a part of the Environmental and Water Quality Operational Studies (EWQOS) research program conducted at the Waterways Experiment Station (WES), several solutions have been considered to improve both reservoir in-situ and release water quality. Total destratification, oxygenation, structural and operational modification, and localized mixing are all feasible approaches to the solution of specific in-situ and/or release water quality problems. Each, however, has differing aspects of design and applicability which must be considered prior to their selection.

Purpose

The purpose of this paper is to review, in a synoptic fashion, the technology developed within EWQOS on the following reservoir water quality enhancement alternatives: (a) total reservoir destratification; (b) localized mixing; (c) oxygenation; and (d) structural and/or operational modification through the use of multilevel selective withdrawal. For each alternative, the pertinent EWQOS documentation will be referenced; available design concepts presented; and general applicability discussed. Specific reference to techniques for the reaeration of hydropower releases and to the reaeration potential of gated-conduit hydraulic structures will not be discussed herein. The reader is referred to Bohac et al. (1983) and to Wilhelms and Smith (1981), respectively, for details on these subjects.

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TOTAL RESERVOIR DESTRATIFICATION

Overview

For a reservoir to be totally destratified by means other than the natural overturn or local winds and meteorological conditions, sufficient energy must be added through artificial means. There are, generally, two methods of creating artificial destratification: (a) release of compressed air within the hypolimnion (pneumatic destratification) and (b) pumping water within the lake (hydraulic or mechanical destratification). The hydraulic concept involves pumping water from one region of the reservoir and jetting it into another region of different density. The buoyant water jet induces circulation and mixing. With pneumatic destratification, a buoyant air-water plume causes circulation and mixing as it rises to the surface. The results of site-specific applications of pneumatic destratification were reported by Smith and Merritt (1980). Additional research results were given by Fast and Hulquist (1982) on this methodology. Results on the initial WES efforts on hydraulic destratification were given by Dortch (1979) and by Holland, Dortch, and Smith (1981). Holland and Dortch (1984) provide updated results from additional research efforts on hydraulic destratification.

Design Information

Fast and Hulquist (1982) provide an overview of the pertinent parameters affecting pneumatic destratification. Pastorok, Lorenzen, and Ginn (1982) provide a more generalized review of the theory and experiences involved in this concept. Design information on this alternative generally consists of extrapolations from various site-specific applications.

Dortch (1979) presents, in detail, an overview of the parameters affecting hydraulic destratification performance. Holland and Dortch (1984) provide results from laboratory parametric investigations which may be used to provide a first-order approximation for the design of a hydraulic destratification system. No site-specific investigations of hydraulic destratification systems for impoundments of average CE size are known to this author.

Applicability

Pastorek, Lorenzen, and Ginn (1982) provide an excellent discussion of the utilities and limitations of total reservoir destratification based on a compilation of field experience. The primary purpose of total destratification has been to destroy the density gradients in a reservoir which inhibit mixing and oxygen replacement in order to enhance in-situ and release quality. These systems are generally employed prior to the onset of stratification as a deterrent to hypolimnetic anoxia. In this capacity they seem quite efficient, both in inhibiting stratification and in raising average dissolved oxygen concentrations. However, it is the very action of reservoir destratification which provides the limitations for this alternative. Reservoir destratification generally increases the overall heat budget of the impoundment, warming hypolimnetic waters while cooling surface waters for a given set of conditions. Pastorek, Lorenzen, and Ginn (1982) note that the effects of destratification on aquatic biota are generally unknown. However, Fast and Hulquist (1982) do report the potential for nitrogen supersaturated releases from pneumatically destratified impoundments. Dortch (1979) also notes that 100 percent destratification is impractical and suggests that these systems should be designed for the 80 percent mixed state.

LOCALIZED MIXING

Overview

Localized mixing systems are designed to destratify an impoundment only in the vicinity of the release structure. Field applications for small impoundments have shown it to be a simple, cost-effective approach to improve the quality of reservoir releases (Garton and Peralta, 1978; Dortch and Wilhelms, 1979). The concept of localized mixing is shown in Figure 1. A downward vertical jet composed of epilimnetic water transports generally high quality water downward into the hypolimnion. This jet is formed near the release structure and is designed to have adequate initial momentum to transport a quantity of epilimnetic water to the level of the hypolimnetic release outlet or well within the outlet withdrawal zone. A portion of the transported epilimnetic water will then be withdrawn from the reservoir along with a quantity of hypolimnetic water, thus diluting the hypolimnetic outflow and improving the release quality.

Design Information

Busnaina et al. (1981) have identified a number of design parameters which affect the effective dilution of the hypolimnetic release. However, a necessary condition for successful localized mixing is that the epilimnetic jet penetrate to the level of the release outlet or well within the outlet withdrawal zone. If the jet fails to penetrate to this level, the improvement of release quality will be lessened or negated. Further, penetration of the jet beyond the outlet represents a waste of energy which reduces the cost-effectiveness of the method. In certain cases, such as for bottom outlets, over-penetration may disturb bottom sediments and degrade rather than improve release quality. Holland (1983a) provides quantitative design information on the penetration of localized mixing jets. Moon et al. (1979) provide qualitative guidance on the expected dilution of a localized mixing jet of specified penetration. Holland (1983b) used the above references to design a hypothetical reservoir localized mixing system.

Applicability

Localized mixing systems are best suited for the enhancement of hypolimnetic releases that are not extremely dynamic. The systems have generally been used in the vicinity of low-level release outlets. However, the systems have much wider potential uses, such as in the enhancement of low-flow releases and leakage, destratification of coves and recreation areas, and, in tandem, the enhancement of moderate flows. The limitation of the system, however, is in its general inability to provide enhancement to large flows (such as hydropower flows).

OXYGENATION

Overview

Aeration using molecular oxygen (oxygenation) has been evaluated and/or employed in some prototype situations within the CE (i.e., Table Rock Reservoir (Weithmam et al. 1980); Clarks Hill Reservoir (Miller and Gallagher, 1980). The projects considering the use of this alternative have traditionally been hydropower facilities which cannot, by means of natural and/or other artificial means of reaeration, meet downstream release dissolved oxygen requirements. Pastorek, Lorenzen, and Ginn (1982) as well as Bohac et al. (1983) discuss a variety of existing and proposed oxygenation approaches. The most common oxygenation system being employed in the CE at present is a linear diffuser system such as the one being installed at Richard B. Russell Reservoir (U. S. Army Corps of Engineers, 1981).

Design Information

Holland and Tate (1984) summarize the work of Speece et al. (1978) at Clarks Hill Reservoir. The bulk of design information presently available for diffuser systems has come from these studies. The design of the Richard B. Russell Reservoir system under construction was predicated to a great degree upon the results of the Clarks Hill studies. Little general guidance on oxygenation systems exists. Holland and Tate (1984) and Pastorek, Lorenzen, and Ginn (1982) review the available literature on the multitude of site-specific oxygenation systems reported.

Applicability

The use of molecular oxygenation within the CE has generally been for hydropower release enhancement rather than reservoir rehabilitation. However, this latter alternative may be viable in certain cases. The primary limitation for the use of molecular oxygenation is cost-effectiveness. Speece et al. (1978) reported a potential oxygen cost of \$100/ton (O_2) excluding system capital costs at Clarks Hill Reservoir. This value, however, represented the expense of "trucked-in" O_2, a cost of which might be significantly reduced by construction of an on-site cryogenic plant (Speece et al., 1978). Pastorek, Lorenzen, and Ginn (1982) list additional concerns with oxygenation. However, in general, if the costs of molecular oxygen can be absorbed, the use of oxygenation may be an attractive enhancement alternative.

MULTILEVEL SELECTIVE WITHDRAWAL

Overview

As previously stated, stratification inhibits vertical mixing and affects various hydrodynamic processes within a reservoir. As a consequence, the quality of the water varies with its location in the reservoir. Furthermore, this variation is generally most pronounced vertically. With knowledge of the vertical distribution of temperature within a reservoir, a selective withdrawal outlet works (Figure 2) which provides the flexibility of withdrawing water of the desired quality from various strata in the lake can be designed.

Reservoirs can be operated to achieve in-lake objectives such as evacuating waters with low dissolved oxygen content from the bottom or releasing a density current of inflowing suspended sediment resulting from a storm in the upstream watershed. Many times, however, reservoirs are operated to meet downstream temperature objectives during the thermal stratification cycle. The design of these structures, then, involves three generalized parts: (a) the evaluation of the effectiveness of a given multilevel intake design in withdrawing releases of a specified quality from specific levels in a stratified reservoir; (b) the location of multiple intakes such that their positioning is "optimal" for maintenance of a given water quality objective over a given time frame; and (c) the optimal operation of the designed system for water quality objectives.

Design Information

The work of Bohan and Grace (1973) remains the basis from which the CE evaluates the selective withdrawal characteristics of a hydraulic structure. EWQOS-funded research on this topic has expanded the generality and applicability of the original Bohan and Grace work. These algorithms, which were developed from laboratory testing of generalized reservoir and structural geometry, can be used to evaluate the withdrawal characteristics of a wide range of withdrawal structures (weirs, spillways, water quality intakes) for any stable thermal stratification pattern. SELECT, a numerical procedure incorporating these algorithms is available on several CE supported computer systems.

For the computation of selective withdrawal characteristics for reservoir geometries which are relatively complex (due to the presence of islands, obstacles, intrusions, caves, structures), Thompson and Bernard (1984) report the development of the numerical model WESSEL. The model, a two-dimensional (2D) formulation, utilizes the attractiveness of boundary-fitted coordinate grid generation in order to solve dynamic, width- or depth-averaged flows with arbitrary and often highly irregular boundaries. An example of such a 2D-grid for an elevation view of the Taylorsville outlet works is given in Figure 3. Output from the model is in the form of velocity vector plots and flow pattern visualization. Investigation of WESSEL currently continues the exploration of additional uses of the model for hydrodynamic computations both internal and external to hydraulic structures.

The location of multilevel intakes for water quality maintenance has traditionally been done in an unsystematic, often tedious fashion. Holland (1982b) reported the development of a numerical approach coupling optimization and water quality simulation modeling capabilities which in a systematic, automated manner, determined (for a given set of conditions) the optimum number and elevations of intakes required to meet prescribed water quality objectives. This approach was employed to develop an optimal intake configuration design for the reformulated Cowanesque Lake (Holland 1982a). The utility of this approach, and its use, are described in a draft report by Dortch and Holland (1984).

Reservoirs with selective withdrawal capabilities may be facing project reformulation due to such changes as hydropower retrofitting, increased water supply requirements, and low-flow augmentation. Further, many projects with multilevel withdrawal facilities may be operating in less than an "optimum" manner for maintenance of water quality objectives year-round. Fontane, Labadie, and Loftis (1982) provide an excellent discussion of the optimal control of release quality through selective withdrawal. Their technique involves making decisions for multilevel port operations each day which anticipate future critical water quality operations. Poore and Loftis (1983) report the development of a numerical procedure which aids in the determination of effective multilevel outlet works operations for multiple water quality objectives. Both the Fontane, Labadie, and Loftis (1982) and the Poore and Loftis (1983) techniques are used in conjunction with water quality numerical models which simulate the expected conditions resulting from differing multilevel outlet works operations. These techniques, and the Dortch and Holland (1984) approach, have the potential to be highly useful for the planning and development of real-time operating criteria for water quality maintenance.

Applicability

Selective withdrawal structures are most appropriate for density stratified multipurpose reservoirs which have varying downstream and/or in-situ water quality objectives. The use of such systems is generally considered as a technically feasible alternative for water quality maintenance of most multipurpose reservoirs. However, the utility of selective withdrawal is lessened in well-mixed impoundments with little, if any, density stratification.

CONCLUSIONS

Research at WES as a part of the Environmental and Water Quality Operational Studies has developed a number of tools which will aid in the design and operation of reservoir and release water quality enhancement systems. These systems range in complexity and utility over the same general range as CE reservoirs. Thus, while one particular technique may be inappropriate for a given reservoir, a thorough investigation of the techniques presented herein should provide a first-order approximation to the design and/or operation of an efficient reservoir water quality enhancement technique. It is hoped that this paper has provided sufficient documentation for such an investigation. However, prior to any investigation of water quality enhancement techniques, a thorough understanding of the limnological and hydrodynamic attributes of a given reservoir should be achieved. Such an understanding generally provides the key not only to the extent of reservoir water quality problems, but, more importantly, to their effective solution.

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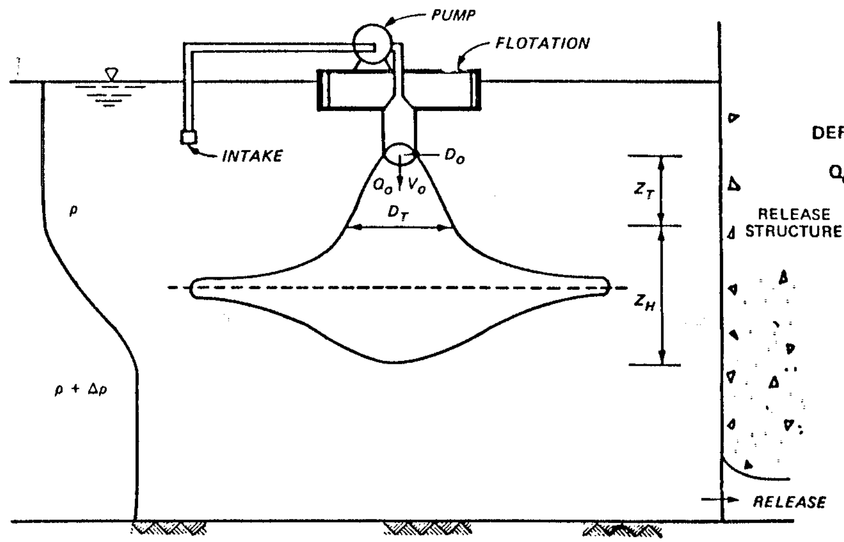

DEFINITION OF TERMS USED IN FIGURE 1:

Q_o, V_o, D_o = VOLUME FLUX, VELOCITY, DIAMETER OF JET AT THE OUTLET
D_T = DIAMETER OF JET AT THERMOCLINE
Z_T = DISTANCE FROM JET OUTLET TO TOP OF THERMOCLINE
Z_H = PENETRATION DEPTH BELOW TOP OF THERMOCLINE
ρ = EPILIMNETIC DENSITY
$\rho + \Delta\rho$ = HYPOLIMNETIC DENSITY
$\Delta\rho$ = ABSOLUTE DIFFERENCE BETWEEN EPILIMNETIC AND HYPOLIMNETIC DENSITIES

Figure 1. Schematic Representation of a Localized Mixing System

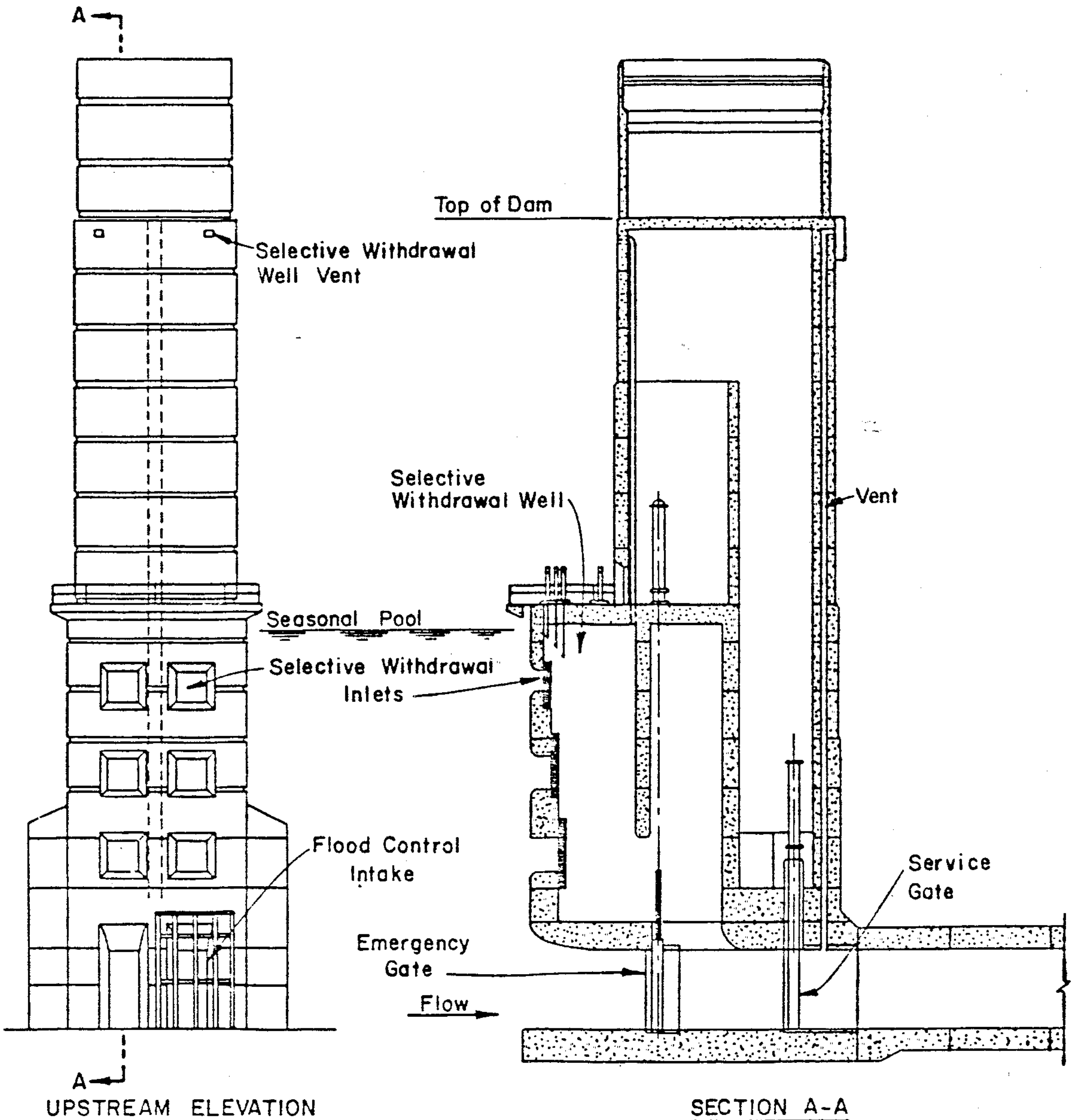

Figure 2. Example of selective withdrawal structure

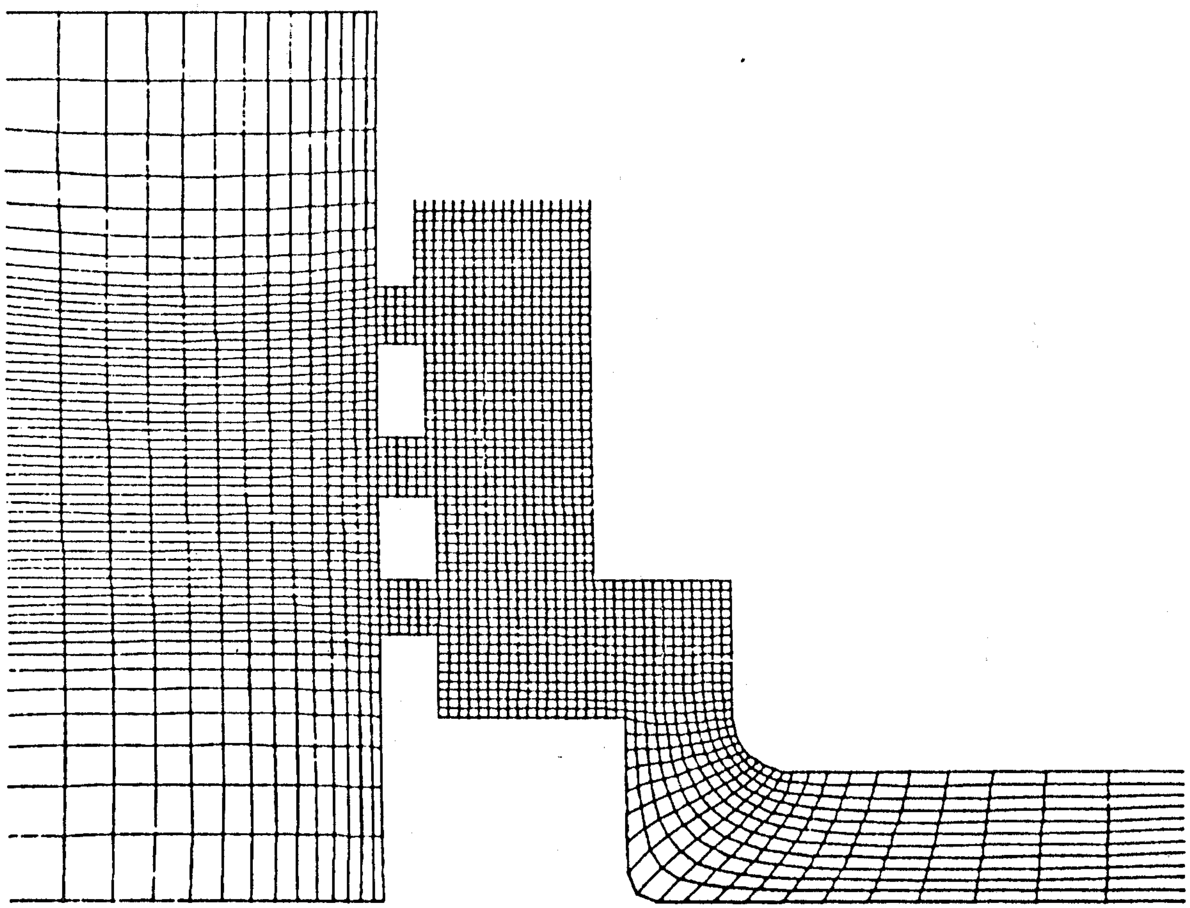

Figure 3. Two-Dimensional Grid Representation of the Taylorsville, KY Outlet Works

HYDROPOWER AND WATER QUALITY

by

Steven C. Wilhelms[1]

INTRODUCTION

Hydroelectric power generation has proven to be one of the most attractive energy sources available. Past energy shortages have spurred growth in the number of hydroelectric facilities and today, as never before, numerous existing and proposed sites for hydropower projects are being evaluated and developed. The unique advantages of hydropower over fossil fuel or nuclear-powered steam generation have significantly contributed to the enthusiasm for its use. Hydropower is presently meeting about 12 percent of our Nation's energy needs. Additionally, several thousand potential hydropower sites have been identified, and the attractive attributes of hydropower have resulted in evaluation, design, or construction of hydropower facilities for many of these sites.

From the standpoint of energy resource conservation, hydropower is very attractive because the energy source (water held in the reservoir) is renewable. Hydropower is also extremely flexible from an operational standpoint. The tremendous changes in power demand due to daily peaking and seasonal fluctuation dictate the need for a rapidly responding energy source. Hydropower generation can usually be stopped, started, or changed in a matter of minutes by simply controlling the flow rate of water through the turbine. This provides nearly optimum compatibility with peaking demand. If adequate water is available, hydropower can also be operated continuously to meet baseload power demand.

Hydropower is considered one of the cleanest major sources of electrical energy. However, the potential for adverse environmental impacts resulting from proposed project operations or to changes or additions at an existing project must be evaluated. Further, techniques that minimize or mitigate any damage to the environment must be developed. Some of these concerns are discussed in the following paragraphs, and potential solution or evaluation techniques are discussed later.

PROBLEMS AND CONCERNS

The most frequently cited adverse impact for proposed or existing hydropower projects is the release of water with a low dissolved oxygen (DO) content. Hydropower projects often withdraw water from deep in the upstream pool. Naturally occurring processes in the reservoir tend to reduce the DO in the lower levels of the pool; thus, low DO water may be withdrawn and released downstream during hydropower operations. Depending upon the severity of the DO deficiency in the release, it may be necessary to employ one or more techniques to increase the DO concentration in the releases. Alternatives to accomplish this will be discussed in later paragraphs.

The retrofit of an existing flood control or other non-power project with hydropower has produced a whole set of problems or concerns in addition to the potential for low DO releases. If, at the outlet structure, selective withdrawal

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capabilities exist, an add-on downstream turbine may completely eliminate selective withdrawal since downstream flow control usually results in a loss of flow control for the multi-level intakes in the upstream tower. The loss of selective withdrawal capabilities may dramatically affect release water quality parameters such as water temperature and dissolved oxygen. Additionally, the loss may also impact reservoir operation by limiting withdrawal flexibility for removing or flushing certain water from the reservoir.

In most tower-conduit-stilling basin type outlet works, significant reaeration occurs during flow through the structure. The open channel flow conditions and air entrainment in high velocity regions of flow promote oxygen transfer. With a downstream turbine and pressurized conduit, this reaeration is lost. Unless some techniques are employed to increase the release DO, severe degradation of downstream DO could result.

The far-field downstream area may also feel the effects of a loss of selective withdrawal and reaeration. Instead of starting with a relatively high DO concentration in the structure tailrace, the DO may be much lower. Should there be any oxygen demanding material in the release or added by a tributary downstream, a DO "sag" may result, potentially creating an extremely hazardous condition for aquatic life.

Research has indicated that the tailwater fishery at a flood control project is a result of fish passage from the upstream reservoir. An add-on hydropower project can greatly impact the tailwater fishery since there are few fish species that can survive passage through a turbine. If there is a highly used recreational or commercial fishery in the tailrace area, fish mortality due to turbine passage may be a significant problem that merits evaluation.

There are certainly many more potential concerns for water quality at existing, proposed, or add-on hydropower projects. The foregoing problems have been identified or encountered at existing projects.

POTENTIAL SOLUTIONS AND EVALUATION TECHNIQUES

As mentioned, the most frequently cited adverse impact for proposed or existing hydropower projects is the release of water with a relatively low dissolved oxygen content or the loss of reaeration with add-on hydropower. Under the Environmental and Water Quality Operational Studies, the Tennessee Valley Authority (TVA) contracted with the Waterways Experiment Station to conduct a literature review (Bohac, et al., 1983) identifying systems and techniques that mitigate or eliminate the problem of low DO hydropower releases.

Hydropower aeration/oxygenation systems may be grouped into three general areas: forebay, in-structure, and tailwater systems. Examples of techniques in each area are:

- Forebay Systems
  - Oxygen injection in hypolimnion or withdrawal zone
  - Pneumatic or hydraulic lake destratification
  - Localized mixing

- Tailwater Systems
  - Side stream pumping
  - Mechanical aeration
  - Reaeration structures, i.e., weirs, cascades, or U-tubes

- In-structure Systems
  - Penstock injection
  - Turbine injection/aspiration
  - Draft tube injection/aspiration

Forebay and Tailwater Systems

Forebay systems are designed to enhance the DO concentration of the intake water. Since the water quality usually does not degrade during travel through the hydropower plant, the release water quality is also improved. The major disadvantage of air/oxygen injection in the hypolimnion is that the volume of the hypolimnion is relatively large compared to the volume released during generation. This may necessitate a rather large aeration/oxygenation system (Speece, 1975). However, the oxygen-injection approach has the added advantage of enhancing the water quality of the reservoir and generally does not decrease the efficiency of the turbines (Speece, 1977; Merritt and Leggitt, 1981).

Pneumatic (Fast and Hulquist, 1982) or hydraulic destratification (Dortch, 1979) increases DO in the lower levels of the lake but has the disadvantage of mixing the entire reservoir and results in the loss of cold water for release which may not be desirable. Pneumatic destratification may also impact the dissolved nitrogen concentration in the pool.

Localized mixing (Garton and Rice, 1974; Garton and Jarrell, 1976; Dortch and Wilhelms, 1978) is designed to destratify the reservoir in the vicinity of the outlet. It may have more appropriate application for lower flows than for very large discharges. A downward vertical jet of epilimnetic water transports higher quality water into the withdrawal zone of the outlet in the hypolimnion (Holland, 1984). A portion of the epilimnetic water would be withdrawn along with hypolimnetic water, thereby diluting the hypolimnetic outflow and improving the release quality.

Tailwater systems in general have received little attention over the last decade in comparison to forebay and in-structure systems, although various tailwater aeration/oxygenation systems have been investigated both in Europe and the United States. The efficiency, operating cost, and capital investment vary considerably depending both on the project and respective techniques.

In-Structure Systems

In-structure aeration/oxygenation focused on turbine venting during the 1970's, although compressed air (or oxygen) injection was attempted. In venting systems, air (or oxygen) is aspirated into low-pressure regions that occur in the draft tube of many turbines. Part of the aspirated air (or oxygen) is absorbed as the gas is mixed with the discharge and as it travels down the draft tube.

Aspiration through existing venting or vacuum-breaker systems normally results in a relatively low DO enhancement. Most vacuum-breaker systems (Figure 1) were designed to allow only enough air into the turbine to prevent cavitation and reduce turbine vibration. Various constrictions often exist that limit air flow.

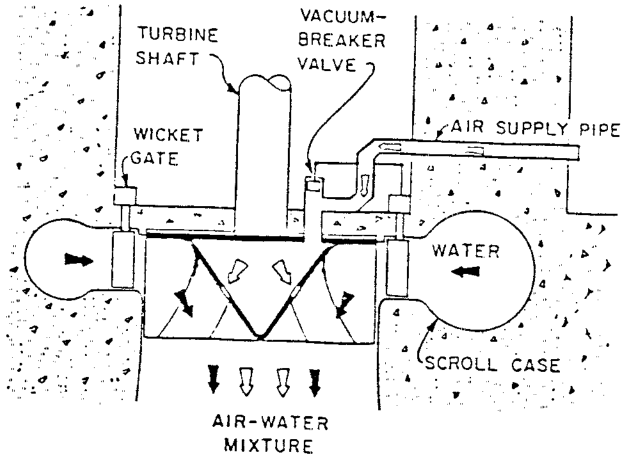
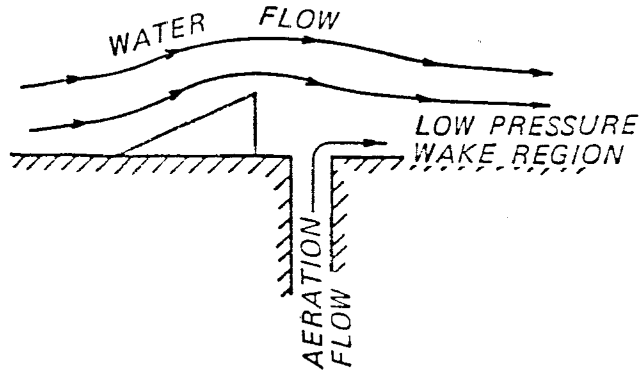


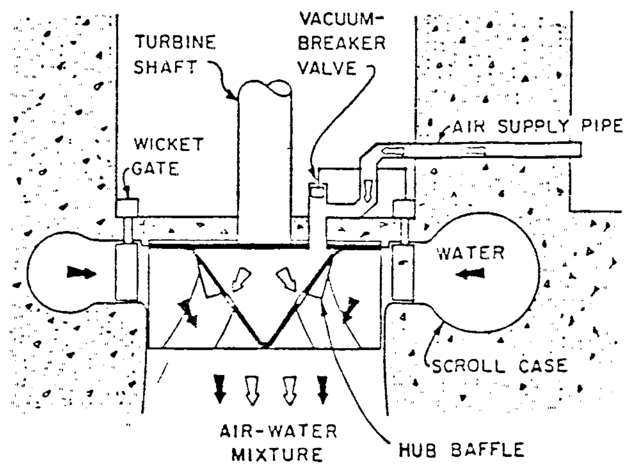
Figure 1: Vacuum-Breaker Venting System

Various modifications to existing venting systems have increased the air flow rate and enhanced gas transfer. In one instance, the air supply line was rerouted to bypass the vacuum-breaker valve. Deflector plates have also been used to increase the vacuum at points on or below the turbine wheel. The deflector plates, usually installed over existing air ports on the turbine, result in increased aspiration due to the lower pressure in their wake. A typical deflector design and deflector locations are indicated in Figure 2; however, a variety of geometries have been used.

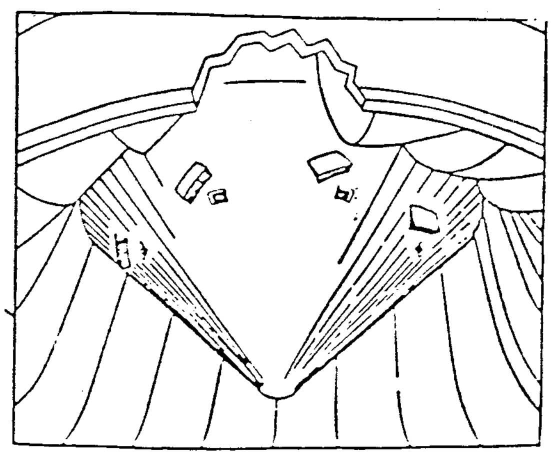
A manifold ring (Figure 3), attached to the periphery of the draft tube liner, has been used to create or enhance negative pressures in the draft tube. Vent holes in the ring are uniformly spaced and are sized appropriately to aspirate air or oxygen uniformly around the ring.



a. Deflector plate design



b. Location of hub baffles



c. Distribution of baffles on hub

Figure 2: Typical Deflector Design and Placement

Turbine venting has some disadvantages. Dissolved nitrogen concentrations may increase. Turbine efficiency and capacity are also usually reduced slightly. Turbine efficiency is reduced as a result of both the air flow and by the drag produced by hub or ring baffles. The magnitude of the losses depends upon the baffle geometry, air and water flow rate, and several other variables.

Historically, compressed air injection into flow regions at (or above) atmospheric pressure has not received significant attention. However, the TVA has recently evaluated and will continue to evaluate forced air injection to improve hydropower release quality.

Corps of Engineers field studies of turbine venting techniques were conducted at the Clarks Hill Reservoir project on the Savannah River (Wilhelms, 1983a). Three venting systems were investigated: (a) vacuum-breaker system, (b) large diameter bell-mouth air intake, and (c) a 25-hp blower for forced air injection. The "no-air" condition was tested as the base condition.

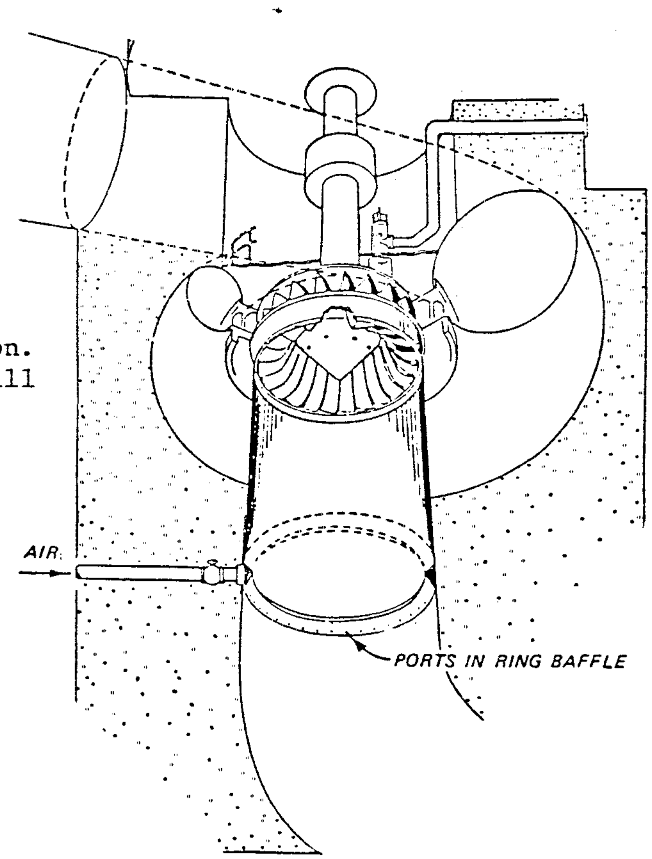


Figure 3: Draft Tube Baffle Ring

Data from the "no-air" conditions clearly indicated that there was a dissolved oxygen uptake due to turbulence in the tailrace area. When air was introduced to the flow, an additional increase in release DO was observed. It was concluded that two processes were at work causing gas transfer during vented hydropower releases. It was postulated that the energy available at the draft tube exit governed reaeration due to tailrace turbulence. It was also hypothesized that the air-flow-to-water-flow (Q_a/Q_w) ratio and the pressure-time history (Buck, et al., 1980) of flow through the draft tube and tailrace governed the oxygen uptake due to turbine venting. Figure 4 shows the conceptual relationship of these two processes. Their effects were mathematically described and then superimposed to form a numerical model (Wilhelms, 1983b) of oxygen uptake due to turbine venting.

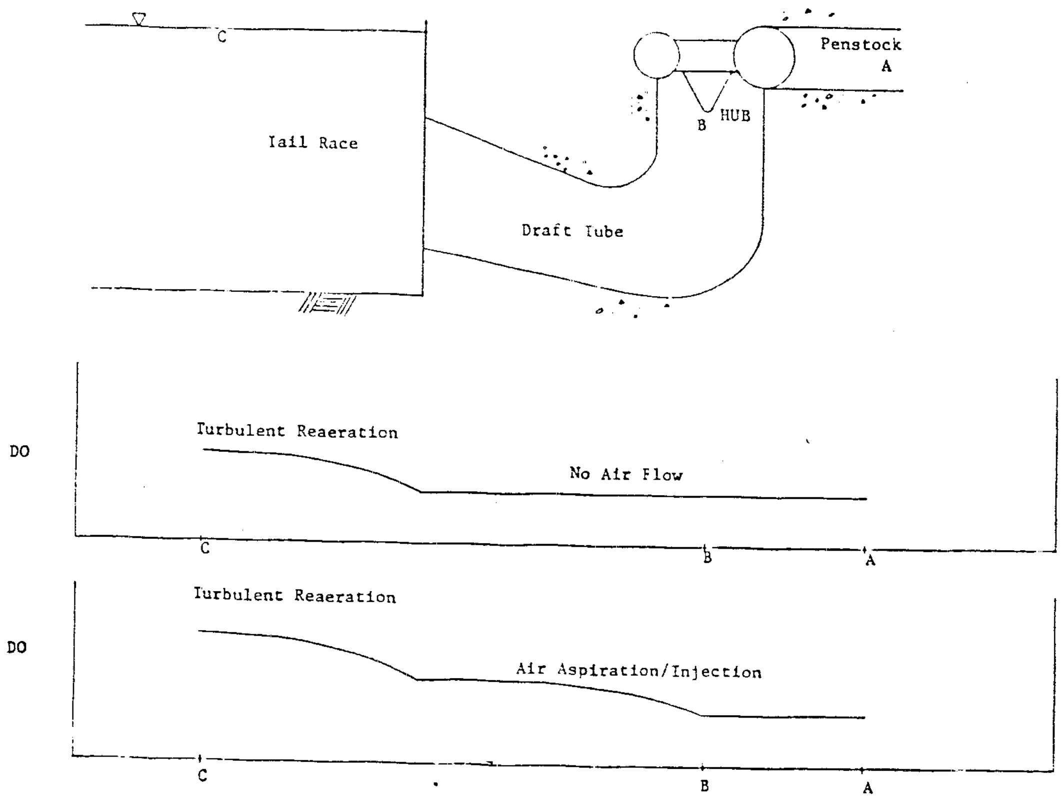


Figure 4: Schematic of Turbine Reaeration Processes

The model provided estimates of release DO with a standard error of estimate of 0.3 mg/l. Using observed DO data or coupled with a numerical water quality model, the DO uptake model can provide estimates of release DO if turbine venting were employed. Thus, environmental impacts and economic feasibility of alternatives may be evaluated. However, the coefficients for this model were developed with data from only the Clarks Hill project. It is reasonable to expect accurate predictions for hydropower facilities that are similar to Clarks Hill in size, geometry, and equipment. It must be recognized that, for significantly different hydropower plants, the accuracy of the predictions of this technique is limited. However, in cooperation with the Tennessee Valley Authority, verification of the model for several different hydropower projects is being accomplished. Once completed, this model should be a tool with applicability to a wide range of turbine sizes, geometries, and hydraulic conditions.

Additional Considerations for Hydropower Retrofit

All of the above alternatives may be applicable to add-on hydropower projects. However, another alternative exists that may receive great attention in the future. The development of upstream turbines could potentially eliminate some of the dissolved oxygen problems associated with downstream add-on facilities and loss of reaeration. For this case, the turbine and generator are located in the outlet works tower. Water is withdrawn from the reservoir, passed through the turbine, then discharged in the outlet works conduit. Some of the reaeration that occurs during flow through the conduit and stilling basin would remain intact. Upstream turbines when applicable may be very attractive in lieu of reaeration or oxygenation measures required to maintain a high DO release concentration.

Oxygen injection in the penstock may be an economically feasible alternative for downstream add-on hydropower. The extended travel time in the pressurized flood control conduit compared to the time of flow through an ordinary hydropower project may permit sufficient absorption to make molecular oxygen injection a more attractive alternative.

The loss of selective withdrawal capability resulting from hydropower retrofitting may be somewhat mitigated by blending in the outlet structure even without full flow control on each intake port. It may be possible to blend flows through an upper and lower port by controlling the relative flow areas of those ports. Research to address the feasibility of blending has been proposed.

Evaluation of the potential far-field downstream impacts of hydropower and retrofit operations is somewhat complex because of the many processes, physical and biochemical, that occur as water flows from the lake through the outlet works into the stream far below a project. This evaluation would require the combination of several modeling techniques to estimate the DO concentration at a point downstream. Either prototype data or predicted lake temperature, DO, and BOD profiles would be required as input to a selective withdrawal model for computation of withdrawal quality. If aeration occurs either in open channel flow through the structure or due to turbine venting, then an appropriate reaeration model must be used to estimate the DO concentration in the tailrace of the structure following this reaeration. A riverine model that accounts for reaeration and oxygen demand would have to be added to track the release to the point of interest. Some of these techniques exist and others are under development, but no single simplified model is currently available or being assembled for use in such fashion.

An observation made in fishery studies (Walburg, et al., 1983) of flood control project tailwaters was that the downstream fishery was a result of fish passage from the upstream pool. This could pose potential problems for hydropower and retrofit projects because of fish mortality due to passage through the turbines. It appears that the migration from upstream to downstream is relatively short-lived and occurs at particular times in response to hydraulic, hydrologic, and meteorological conditions. Thus, there may be operational techniques that could eliminate the hazard to the fishery. Structural alternatives may be feasible to prevent fish passage but this could prove detrimental to the tailwater fishery in the long-term. Past research efforts have identified some potential solutions; however, additional work is required to more accurately provide guidance on project operation or structural alternatives.

OBSERVATIONS AND REMARKS

The concerns discussed in the foregoing paragraphs are by no means a comprehensive list of potential or existing problems. However, these were most frequently cited in a recent survey. No single solution to a problem appears preferrable. In general, for Corps of Engineer impoundments, either forebay or in-structure aeration/oxygenation systems would be superior to tailwater aeration/oxygenation. Undoubtedly, the individuality of projects will cause numerous variations of these problems and probably preclude some solution techniques. However, the commonality of hydropower concerns dictates that nearly every solution has merit and warrants some level of evaluation.

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OVERVIEW OF CORPS ENVIRONMENTAL EFFECTS OF DREDGING PROGRAMS

by

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INTRODUCTION

Before the early 1970's, little was known of the environmental effects of dredging and dredged material disposal. As a result, the Congress of the United States recognized that there was no technical or scientific basis for regulating the disposal of dredged material. Consequently, proposed regulations would prove to be excessive and counterproductive. Therefore, the Congress directed the Corps of Engineers (Corps) to conduct a comprehensive research program to develop procedures for determining the environmental consequences of dredged material disposal and to develop new or improved methods of minimizing any adverse effects. The Corps was given the lead responsibility for conducting the research since, in the United States, the Corps is responsible for maintaining over 25,000 miles of waterways and 1000 harbors. In addition, Federal legislation requires the Corps to issue permits to regulate disposal of dredged material in waters of the U.S.

The Corps initiated the Dredged Material Research Program (DMRP) in 1973 and successfully completed the program in 1978 at a cost approaching $33 million. The DMRP was conducted by the US Army Engineer Waterways Experiment Station (WES).

The DMRP was designed to be applicable nationally with no major type of dredging activity, region, or environmental setting excluded. The program resulted in first-generation procedures for evaluating the physical, chemical, and biological impacts for a variety of disposal alternatives in water, on uplands, or in wetland areas. The program produced tested, cost-effective methods and guidelines for reducing the impacts of conventional disposal alternatives. At the same time, it demonstrated the viability and limits of new disposal alternatives, including the productive use of dredged material as a natural resource. The results of the DMRP provided the first definitive information on the impacts of dredged material disposal and have been used extensively by the Corps and the U.S. Environmental Protection Agency (EPA) to develop criteria for implementing regulations under federal legislation.

In addition to providing the data and information needed to develop criteria and guidelines, two major fundamental conclusions of the DMRP that are important to disposal management were reached. Studies conducted and experience gained in the years since the DMRP was completed support these conclusions. No single disposal alternative is most suited for a region or a type

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of project. Conversely, there is no single disposal alternative that can be dismissed as environmentally unsatisfactory due to potential impacts. In other words, from a technical standpoint there is no inherent effect or characteristic of an alternative disposal method that precludes its consideration before specific site assessment. This conclusion holds true for ocean disposal, confined disposal, or any other alternative.

To address a variety of environmental factors and considerations adequately, long-range regional planning is required if a lasting effective solution to disposal of dredged material is to be found. Through use of disposal management plans that consider project types, dredged material characteristics, disposal alternatives, and other factors, the best opportunity exists for maximum environmental protection at an acceptable cost.

DREDGING OPERATIONS TECHNICAL SUPPORT (DOTS) PROGRAM

It became apparent during the DMRP that it would be necessary to continue technology transfer of information after the completion of the program to derive maximum benefits. For this reason, the Corps established the DOTS Program, with responsibility for the program assigned to the Environmental Laboratory, WES. Through DOTS, advisory assistance is provided to all elements of the Corps in solving environmental problems associated with dredging and dredged material disposal. Since DOTS was formed in 1978, literally hundreds of requests for assistance from Corps elements have been addressed.

In addition to providing technical assistance, DOTS is responsible for continued monitoring of selected DMRP field sites to identify long-term trends and to verify and refine engineering and operational procedures developed during the DMRP. Monitoring is being conducted at open-water and habitat development sites. The habitat development sites include marsh creation, upland habitat development, and strip mine reclamation. Engineering procedures developed for the design of confined containment areas continue to be verified and refined. Included are methods to size containment areas to ensure that water quality is maintained over the weir and the given amount of material is held within the dikes. Management of sites through active dewatering is being evaluated as well as treatment of contaminated effluents. Until the LEDO Program began in October 1981, limited regulatory applied research was also conducted under DOTS; for example, first-generation biological testing procedures developed during DMRP for assessing potential impacts were refined. These tests provided the first definite means of predicting short-term impacts of disposal on marine and freshwater organisms. Emphasis is now being placed on developing methodologies for planning long-term disposal strategies.

CURRENT RESEARCH

The DMRP addressed and answered the critical environmental issues defined in the early 70's. Subsequent regulatory research efforts under DOTS provided much of the information used by the Corps and EPA in conducting dredging and disposal evaluations required by federal legislation. However, neither the DMRP nor the research conducted under DOTS addressed, to the level necessary, all questions on environmental impacts associated with dredging or dredged material disposal.

The need to continue research on the environmental consequences of dredged material disposal was articulated by the Corps during Congressional hearings in early 1980. The Congress, as a result of testimony received, expressed concern over the long-term effects of dredged material disposal in some instances. In addition, international agreements such as the London Dumping Convention require consideration of chronic or sublethal effects in the evaluation of dredged material disposal. Presently the state-of-the-art allows for accurate measurement of bioaccumulation. However, current evaluation protocol regards any bioaccumulation as an adverse impact because there is no sound basis for interpreting its ecological significance. The lack of defined and acceptable testing procedures that allow for an accurate assessment of the ecological significance of bioaccumulation has led to conflicts in interpretation of data among regulatory agencies and litigation on Federal projects.

To address the concerns expressed by the Congress, the Corps has recently initiated three programs: LEDO, Field Verification Program, and a work unit under another program that addresses methods of dredging contaminated sediments. All of this work is under the centralized management of the Environmental Effects of Dredging Programs (EEDP) at the Waterways Experiment Station.

Long-Term Effects of Dredging Operations (LEDO)

The objectives of LEDO are to provide new or improved state-of-the-art technology for predicting long-term environmental impacts of dredging operations and to improve or develop methods for minimizing any adverse impacts associated with dredged material disposal. Work is currently being conducted to determine the effects of aquatic disposal and the effects of terrestrial disposal. LEDO is planned as a continuing program since applied environmental research must address current problems and research priorities are subject to change.

Specific areas of research in LEDO include the following:

a. Bioaccumulation and biomagnifications in the aquatic environment. Establish the significance of bioaccumulation and biomagnifications of contaminants associated with the aquatic disposal of dredged material and develop or improve predictive techniques for bioaccumulation and biomagnification.

b. Minimize procedures to reduce adverse impacts. Field-test procedures that will eliminate or minimize adverse impacts of dredged material disposal. One possible procedure under study is capping contaminated material with noncontaminated material.

c. Upland plant and animal bioassays procedures. Improve first-generation plant and animal bioassays for predicting uptake of contaminants in wetland upland areas.

d. Water quality. Increase the understanding of the geochemical changes that occur with time in upland dredged material containment areas; develop or improve techniques for predicting contaminant concentrations in the effluent from these sites.

Study results will provide a much broadened state-of-the-art technical basis for the Corps' implementation of its environmental responsibilities under Federal legislation. Emphasis will be placed on presenting research results in the international technical and scientific literature as well as making results immediately available to the field through normal Corps channels.

Field Verification Program (FVP)

During planning of LEDO field studies with the Corps' New England Division, it became apparent that a unique set of circumstances existed in the New England region of the United States where three disposal alternatives could be evaluated at the same time (open-water disposal, upland disposal, and marsh creation). The Field Verification Program was established as a cooperative effort between the Corps and EPA to field-verify existing predictive testing procedures. Through the program, promising procedures already developed by the Corps along with techniques developed by EPA for nondredged materials will be applied to project conditions at Black Rock Harbor, Bridgeport, Connecticut, using dredged material from that single maintenance operation. Although the three disposal alternatives have been evaluated independently during the DMRP, these field studies will provide the first opportunity for direct comparison of the environmental consequences using the same dredged material under different disposal conditions.

The program's major areas of investigation include:

a. Bioaccumulation of contaminants by aquatic animals. Levels of bioaccumulation of selected contaminants over time, biological and physical factors affecting bioaccumulation, and variability of bioaccumulation predictions will be documented in the laboratory. Bioaccumulation will then be determined under field conditions and compared to laboratory predictions to verify the accuracy of the prediction methods.

b. Consequences of bioaccumulation in aquatic animals. Several physiological indices of biological health will be determined in organisms that have accumulated contaminants from dredged material. These indices, previously developed by EPA for use in nondredged material regulatory programs, will include scope for growth, benthic and reproductive effects, effects on enzyme systems, and histopathological parameters. The responses of aquatic animals to contaminants will first be determined in the laboratory to establish feasibility for assessing dredged material and correlation with bioaccumulation. Responses will then be verified in aquatic organisms exposed to contaminated sediments in the field.

c. Effects of aquatic disposal on community structures. Effects of contaminated dredged material disposal on community structures will be determined by measuring mortality, reproduction, and intrinsic rate of growth in selected populations within aquatic communities. These assessments will be documented in the laboratory and verified by monitoring in the field.

d. Effects of upland disposal on water quality. Laboratory tests for predicting effluent quality will be conducted on contaminated sediment prior to placement in a confined disposal area. The confined disposal area will be designed, operated, and managed to ensure optimum fill configuration for the field studies and evaluation of water quality effects. During filling operations, influent and effluent water quality parameters will be monitored extensively at selected stations within the disposal area. Following disposal, the quality of surface water runoff will be determined by collecting surface water samples from controlled simulation of rainfall. Monitoring wells will be placed around and within the disposal area, and groundwater samples taken before, during, and after filling.

e. Bioaccumulation of contaminants in upland and wetland plants. First-generation test procedures from DOTS and other studies will be verified at the field site. Saltmarsh plants will be grown under controlled wetland and upland conditions and analyzed for contaminant bioaccumulation. Field tests will be conducted to verify laboratory test results. Saltmarsh plants will be planted at the upland disposal facility at Black Rock Harbor and sampled each year to determine contaminant bioaccumulation.

f. Bioaccumulation of contaminants in upland and wetland animals. Existing upland and wetland animal bioassay test procedures developed in Europe will be verified in the field using selected upland animals (annelid worms) and wetland animals (snails).

Results of the Field Verification Program will provide both the Corps and EPA with documented and verified state-of-the-art procedures for complying with national regulatory requirements and international agreements. The study is scheduled to be completed in 6 years.

Dredging Contaminated Sediments

In the United States, much work has been conducted over the past 10 years on the effects of dredged material disposal. Little work has been done on the effects of the dredging operation because it was felt that the disposal operation would have the most significant impact. However, due to the need to dredge highly contaminated sediments, it became apparent that research was required to establish environmental parameters associated with conventional dredges as well as investigating and developing procedures and/or equipment to minimize adverse effects from the dredging operation. This study was incorporated into another major program in the Corps of Engineers (Improvement of Operation and Maintenance Techniques (IOMT) Program). As previously noted, this phase of the IOMT Program is being conducted under the general management of EEDP.

Existing data on the resuspension of sediments and contaminants will be collected on a national and international basis. In addition, field studies will be conducted at various sites where unconventional equipment is being used. Based on these data, guidelines will be developed for dredging highly contaminated sediments to minimize any adverse impacts.

SUMMARY

Prior to the 1970's little research had been conducted by the Corps of Engineers, or in fact by other agencies, on the environmental effects of dredging and dredged material disposal. Within the past 10 years, major research has been conducted and, as previously described, research is continuing. The Corps of Engineers now has, through the EEDP, a mechanism for providing technical assistance directly to the field and also a means to address high priority research items on a continuing basis. Since the research programs have recently been initiated, no results are available. Future papers will give results of these studies.

DREDGED MATERIAL DISPOSAL INTERFACING REQUIREMENTS OF THE CLEAN WATER ACT OF 1977 WITH THOSE OF THE RESOURCE CONSERVATION AND RECOVERY ACT

By

Robert J. Whiting[1]

INTRODUCTION

Let's put the title into a question. Where do the requirements of the Clean Water Act (CWA) and the Resource Conservation and Recovery Act (RCRA) interface in regard to dredged material disposal?

On the surface, it would appear that the answer could be relatively simple; i.e., that dredged material is to be regulated by RCRA, or that dredged material is to be regulated by the Clean Water Act or Ocean Dumping Act (CWA/ODA), or that dredged material is to be sequentially regulated first by CWA/ODA and then by RCRA.

Unfortunately, there does not appear to be an easy answer. There are proponents for each of these interpretations. Congress did not clearly delineate this issue when it drafted these laws, and as yet the issue has not been tested in the Federal courts.

It might be useful to look into the requirements and legislative history of RCRA.

BACKGROUND

The problem of hazardous waste spreading into the environment and ultimately endangering human health prompted the formulation of the Resource Conservation and Recovery Act of 1976, Public Law 94-580. The intent of this legislation was to identify hazardous wastes and provide management strategies that would protect the environment from widespread contamination and, ultimately, human adversities. The objective of RCRA pertaining to hazardous waste can be cited directly:

> ". . . to promote the protection of health and the environment and to conserve valuable material and energy resources by . . . regulating the treatment, storage, transportation and disposal of hazardous waste which have adverse effects on health and the environment. . . ."

[1]Chief, Project Evaluation Section, St. Paul District

The most serious issue of RCRA facing waste managers is the lack of flexibility and the assumption of a worst-case scenario of improper management. This has led to the overclassification of essentially all industrial solid wastes as potentially hazardous. Both new and existing facilities are subject to a highly structured and tightly monitored management system with limited regard for varying degrees of hazard or degrees of control, economic impacts, or site-by-site variations (Daniels 1977).

SOLID WASTE

A material cannot be classified as a hazardous waste unless it first is classified as a "solid waste." EPA defines "solid waste" as "any garbage, refuse, sludge or any other waste material that is not specifically excluded." The key phrase in this definition is "other waste material," which itself is further defined by Section 40 CFR 261.2(b). The definition of solid waste is in no way descriptive of the physical state of the material, in that solid, liquid, and contained gaseous materials are included.

The "status" of the waste is also important in the definition in that a solid waste includes any material which:

1. Is discarded, or being stored or treated prior to being discarded, or

2. Is sometimes discarded and has served its intended purpose or is a manufacturing or mining by-product.

At first, it would appear obvious that dredged material would be a "solid waste" until you examine the exclusions from the definition of the term. For example, a material is not defined as a "solid waste" if it is:

1. Burned as a fuel;

2. Domestic sewage and any other wastes that pass through a sewer system to a publicly owned sewage treatment system;

3. A discharge regulated by an NPDES permit covered under Section 402 of the CWA;

4. An irrigation return flow;

5. Materials covered under the Atomic Energy Act of 1954;

6. Material remaining after in-situ mining processes.

Exclusions (3) and (5) are noteworthy in that here RCRA has specifically given precedence to other existing laws in defining its jurisdiction.

"Hazardous waste" is defined as any solid waste that (a) exhibits one of the hazard characteristics defined in subpart C of 40 CFR part 261, or (b) is listed in subpart D of part 261, or (c) is a mixture of listed and unlisted waste.

There are four characteristics of hazardous wastes: ignitability, corrosivity, reactivity and toxicity. The definitions and testing procedures of these characteristics are specified in the regulations.

1. Ignitability

- Liquid with a flash point below 60° C (140° F).

- Nonliquid capable under standard temperature and pressure of causing fire through friction, absorption of moisture or spontaneous chemical changes, and, when ignited, burns so vigorously and persistently that it creates a hazard.

- Ignitable compressed gas as defined in Department of Transportation regulations.

- Oxidizer as defined in Department of Transportation regulations.

2. Corrosivity

- Aqueous material with pH less than or equal to 2 or greater than or equal to 12.5.

- Liquid which corrodes steel (SAE 1020) at a rate greater than 0.25 inch per year.

3. Reactivity

- It is normally unstable and readily undergoes violent change without detonating; or

- It reacts violently with water; or

- It forms potentially explosive mixtures with water, or when mixed with water, it generates toxic gases, vapors or fumes in a quantity sufficient to present a danger to human health or the environment; or

- It is capable of detonation or explosive reaction if it is subjected to a strong initiating source or if heated under confinement, or it is readily capable of detonation or explosive decomposition or reaction at standard temperature and pressure; or

- It is a forbidden explosive, Class A explosive, or Class B explosive, as defined by Department of Transportation regulations.

4. Toxicity

- A waste is considered extraction procedure ("EP") toxic if the USEPA-prescribed procedure (0.5N acetic acid, at pH 5, for 24 hours) yields an extract which contains constituents for which EPA has promulgated Primary Drinking Water Standards (under the Safe Drinking Water Act) at levels at least 100 times greater than the concentration allowed under the Drinking Water Standards. These standards cover 14 substances consisting of 8 metals and 6 common pesticides.

Contaminant	Maximum concentration milligrams per liter
Arsenic	5.0
Barium	100.0
Cadmium	1.0
Chromium	5.0
Lead	5.0
Mercury	0.2
Selenium	1.0
Silver	5.0
Endrin	0.02
Lindane	0.4
Methoxychlor	10.0
Toxaphene	0.5
2,4-D,(2,4-Dichlorophenoxyacetic acid)	10.0
2,4,5-TP Silvex (2,4,5-Trichlorophenoxypropionic acid)	1.0

Like the definition of "solid waste," the definition of "hazardous waste" also has exclusions. For example, solid waste is not hazardous waste if it is:

1. Household waste.

2. Agricultural waste used as a fertilizer.

3. Solid waste from extraction, beneficiation and processing of ores and minerals (including coal).

4. Fly ash, bottom ash, slag and flue gas emission control waste from fossil fuel combustion.

5. Drilling fluids associated with oil, gas and geothermal energy exploration, development and production.

6. One which has an extraction procedure (EP) toxic characteristic solely because of chromium, and nearly all chromium in the waste is trivalent.

7. Cement kiln dust waste.

8. Arsenical-treated wood wastes considered hazardous exclusively because of EP toxicity.

9. A combination of one or more of the following listed wastes and waste-water subject to regulation under the Clean Water Act:

   - carbon tetrachloride
   - tetrachloroethylene
   - trichloroethylene

where weekly solvent content does not exceed 1 ppm, or

   - methylene chloride
   - 1,1,1-trichloroethane
   - chlorobenzene
   - o-dichlorobenzene
   - cresols
   - cresylic acid
   - nitrobenzene
   - toluene
   - methyl ethyl ketone
   - carbon disulfide
   - isobutanol
   - pyridine
   - spent chloro fluorocarbon solvents

where weekly solvent content does not exceed 25 ppm, or

   - heat exchanger bundle cleaning sludge from the petroleum refining industry (K050), or

   - discarded chemicals listed in Section 261.33 arising from de minimis losses in manufacturing process, including minor leaks, personal shower discharges, and container rinsate, or

   - certain laboratory wastewater where annual average flow does not exceed one percent of total flow or such concentration does not exceed 1 ppm.

Exclusions from hazardous waste rules apply to special situations.

1. Small quantity exemption.

   - Generators of less than 1,000 kg per month (1 kg per month for section 261.33(e) wastes).

2. Reuse recycle exemption.

   - Totally or partial exempting if reused.

   - All hazardous waste, except listed wastes and sludges, exempt if beneficially used or recycled.

   - Listed wastes and sludges exempt from portions of hazardous waste rules when beneficially used or recycled.

   - Spent pickle liquor beneficially reused in NPDES wastewater treatment facility is exempt.

3. Raw material tank and pipeline settlings.

   - Hazardous waste generated in a product or raw material storage tank, pipeline, or process unit is exempt until it exits such unit, unless unit is surface impoundment or unless waste remains more than 90 days after unit ceases operation.

4. Laboratory samples.

   - Waste samples sent to lab for purpose of testing, stored before and after testing, and returned to collector after sampling are exempt as long as Department of Transportation requirements are met.

5. Empty containers.

   - Hazardous waste remaining in an "empty" container or "empty" liner is exempt.

   - Container is empty after removal of contents by ordinary practices and no more than one inch remains on bottom or no more than 3% by weight in containers up to 110 gallons or no more than 0.3% in larger containers.

6. PCB's.

   - PCB's are regulated under the Toxic Substances Control Act.

As previously mentioned, there are four characteristics of hazardous wastes; however, only the character of "toxicity" is of concern in the Corps dredging program.

Toxicity is certainly the most controversial hazardous waste characteristic. In common terminology, the word "toxicity" is often used to cover all tendencies of waste material to cause acute or chronic adverse health effects in persons exposed to the materials. In drafting the RCRA regulations, USEPA encountered great difficulty in developing precise workable methods that could be used to evaluate different wastes for this general definition. They found it impossible to come up with a test that measures carcinogenic, mutagenic, or tetratogenic effects on a broad scale. Therefore, USEPA chose a simpler method, the extraction procedure, which was designed to measure tendencies of wastes to generate a leachate with high concentrations of substances covered by existing drinking water standards.

The EP test has been severely criticized. The main criticisms have been that the test seems to evaluate what would happen with a scenario of mismanagement and that, in most real-world situations, it would not be possible to attain the acidic environment prescribed in the test. The American Society for Testing and Materials (ASTM) has been especially critical of the EP test. The USEPA has maintained its position by arguing that wastes should be classified by focusing on plausible mismanagement scenarios. The ASTM states that this approach appears to penalize the majority of waste managers for the sake of a few mismanagers.

A number of other concerns and criticisms still persist regarding the EP test, but the decisive factor in the debate over this procedure has been the absence of alternative testing methodologies that would appear superior. Accordingly, the ASTM has proposed alternative test procedures to the EP test. In addition to the new ASTM procedures, the Corps of Engineers, through its WES, has been actively involved in the development of methods to predict contaminant mobility and other environmental effects for dredged material.

The tests being developed consider the physiochemical environment at the disposal site. These tests will indicate the potential for adverse environmental effects resulting from any contaminant mobility from dredged material more appropriately than the EP test.

CONGRESSIONAL INTENT

The House Public Works Committee, when considering RCRA, divided the types of waste to be covered by RCRA into two categories:

1. Industrial by-products - solid waste generated from the industrial processes; and

2. Consumption by-products - solid waste, such as refuse, garbage, sludge, and other municipal wastes, generated by consumers.

Dredged material fits in neither of the above two categories.

The House did recognize that some "discarded material" (a term the House Public Works Committee preferred over "solid waste") is generated by water pollution abatement activity. However, the House was concerned only with externalities resulting from land disposal of sludge, not with land disposal of dredged material. Externalities resulting from implementation of Sections 404 and 103 were not addressed by the House and, therefore, probably were not considered to be of sufficient magnitude to warrant legislative action.

Unlike disposal of dredged material, disposal of solid waste has traditionally been a local concern. RCRA was intended to keep traditional functions intact. The legislative history is replete with statements on the prominent role State and local governments must perform in the regulation of solid waste disposal, and the Act itself reflects this attitude. But disposal of dredged material has always been a Corps regulated activity. It has never been a State or local concern, except when certain minor intrastate bodies of water not part of the navigable waterways system have been involved. Congress would have given the Corps a function if dredged material was to be dealt with under RCRA, as it has done in the ODA, CWA, and the Rivers and Harbors Act of 1970. Congress did give the Department of Commerce certain logical functions in RCRA; reason dictates that Congress would accordingly have given the Corps certain logical functions if dredged material was meant to be regulated by RCRA.

Statistics used by Congress to support the need for RCRA clearly do not include dredged materials. For example, the House Report discusses the following solid wastes in order of volume:

1. Mining waste - 1.8 billion tons per year.

2. Agricultural waste - 687 million tons per year.

3. Industrial waste - 200 million tons per year.

4. Municipal waste - 135 million tons per year.

If dredged material was included in this list, it would be the third largest amount of waste generated at approximately 450 million tons per year.

It is worth noting that none of the actual instances discussed in the House Report as support for RCRA concerned dredged material.

The Solid Waste Disposal Act of 1965, which RCRA revamped, did not deal with dredged material. The RCRA amendments have not disturbed the status quo in this respect.

If Congress had wanted to regulate dredged material in RCRA, it would have provided for land containment facilities, as it did for the Great

Lakes area in Section 123 of the Rivers and Harbors Act of 1970. The necessity of such a provision is demonstrated by the fact that, if RCRA were applied to dredged material, the quantity of such material normally dredged per year would overwhelm the capacity of existing approved RCRA disposal sites. Congress would not have so deviated from a proven successful and conventional course of providing for diked containment facilities for land disposal of dredged material.

Section 1004(27) of RCRA excludes substances regulated as point sources by Section 402 of CWA. If such exclusion did not exist, point source discharges would be subject to obvious dual regulation. Section 404 of the CWA, however, does not blatantly overlap with RCRA. More subtle overlaps are generally provided for in Section 1006(a), which disallows RCRA permit requirements inconsistent with permit requirements of ODA or CWA. In other words, if there is a conflict between RCRA and either ODA or CWA, ODA and CWA control. Such a conflict may exist whenever dredged material is disposed on land in such a manner that runoff or overflow will occur.

No provision exists in RCRA for applying Sections 404, 103, and RCRA sequentially. Congress would have explicity coordinated Section 404 with RCRA if it so intended.

IMPLICATIONS

What would be the impact of managing the Corps dredging program under RCRA?

To answer this question, WRSC-Dredging Division requested that the WES DOTS program estimate this economic impact. The summarized findings follow:

1. Out of an annual 300 million cy of dredged material, only about 10 percent would be considered unsuitable for conventional open-water disposal.

2. Based on the 1981 workload and an estimate of contaminated materials, approximately 15.5 million cy, or 7 percent of the 228 million cy dredged, would possibly be regulated under RCRA.

3. The minimum estimated annual costs to dispose of potentially contaminated dredged material under RCRA ranged from $442 million to $726 million. This would not include costs for dewatering the dredged material.

4. This amount of dredged material would quickly fill up existing RCRA landfill sites, leaving no place to put the truly hazardous industrial wastes.

Where does this leave us now?

PRESENT POLICY: RCRA does not apply to dredged material because:

1. Dredged material is not defined as a solid waste.

2. Dredged material deposited into waters of the United States is regulated by the CWA.

SPECIAL PROBLEMS:

1. On-land disposal - with and without return water, dewatered dredged material;

2. Ground water considerations;

3. Disposal of highly contaminated sediments;

4. When dredged material is used as cover material or co-disposed in sanitary landfill.

To help solve these problems, the WES DOTS staff this past August prepared a letter report entitled, "Management Strategy for Disposal of Dredged Material." This report provides appropriate testing and handling procedures for uncontaminated to highly contaminated dredged material.

In summation, I would like to state a few facts:

1. Several States are demanding RCRA testing and handling procedures for dredged material disposal;

2. USEPA has been silent in regulatory interpretation on this matter;

3. Congress did not clearly define this issue;

4. RCRA testing procedures are not appropriate for dredged material;

5. The regulation under RCRA of dredged material disposal would have a significant impact on our operations;

6. The technology and testing procedures have already been developed for handling dredged materials under the CWA/ODA. Adequate health and environmental safeguards are currently in place to regulate dredged material disposal operations.

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CAPPING AND CONTROLLED DISPOSAL OF CONTAMINATED DREDGED MATERIAL

by

James M. Mansky*

Bioassay/bioaccumulation testing results indicate that about 5% of proposed dredging projects consist of sediments which exhibit an unacceptable level of toxicity and/or bioaccumulation potential which would prohibit their unrestricted ocean disposal. Capping of these sediments with uncontaminated dredged material has been conducted in Long Island Sound and the Atlantic Ocean in order to isolate the contamination from the marine environment. Studies of these capping projects demonstrate the effectiveness of capping as a way to provide a barrier to chemical diffusion and prevent resuspension of contaminated dredged material. In addition, information is presented on a proposed study to investigate the use of depressions formed by sand mining in the bottom of New York Harbor for the disposal of dredged material with subsequent capping.

Introduction

Ocean disposal of dredged material is regulated under the Marine Protection, Research and Sanctuaries Act (MPRSA) of 1972. Under MPRSA, the U.S. Army Corps of Engineers was given permit authority over the ocean disposal of dredged material. In 1977, the U.S. Environmental Protection Agency promulgated regulations and criteria requiring state-of-the-art biological and chemical testing of sediment proposed for ocean disposal. Bioassay testing and bioaccumulation analyses have indicated that some (5% for New York Harbor) of the dredged material exhibits an unacceptable level of toxicity and/or bioaccumulation potential for polychlorinated biphenyls (PCBs), mercury or cadmium. For the 5% of dredging projects where the sediments exhibit unacceptable toxicity and/or bioaccumulation potential, these sediments are considered contaminated and are precluded from ocean disposal unless "special care measures" are taken to render the material harmless to the marine environment.

One special care measure which has been utilized by the Corps of Engineers in the northeastern United States is a procedure known as capping. Capping involves placing contaminated dredged material at a disposal site and subsequently covering over this deposit with clean dredged material. The capping process assumes: (1) that the cap serves as an effective barrier in sealing off the contaminated material from the overlying marine ecosystem and (2) that the capped deposit is physically stable.[1] This paper reviews the capping studies done by the Corps of Engineers in Long Island Sound and the New York Bight Apex.

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Long Island Sound Capping Study

Background

Dredging of Stamford and New Haven Harbors was required in 1979 in order to maintain the authorized depth of the federal channels to allow safe navigation. Bulk sediment analyses of the silty sediment from Stamford Harbor indicated that it was contaminated with heavy metals; New Haven Harbor sediments were less contaminated.[2] Since different areas within the New Haven Harbor project contained either mostly silt or mostly sand, the capping project was designed with two different mounds. About 37,800 cubic meters of contaminated silt from Stamford Harbor was point disposed and capped with 76,000 cubic meters of silt from New Haven Harbor. On the other mound, 26,000 cubic meters of silt from Stamford Harbor was point disposed and capped with 84,000 cubic meters of sand from New Haven Harbor.

Effectiveness as a Chemical Barrier

Copper (Cu) was selected as being representative of toxic heavy metals in the dredged material and received detailed analysis. Cu values for the Stamford Harbor sediment showed a variation in concentration between the two mounds. The sediment capped with sand had values between 400-500 ppm of Cu, while the silt capped site contained approximately 750 ppm of Cu. Samples of the surficial sediments after capping showed that Cu levels at the sand capped site were reduced from about 70 ppm to approximately 4 ppm. Cu levels at the silt cap were reduced to 70 ppm.[3]

Mussels (*Mytilus edulis*) were utilized as biological indicators of toxic metals released at the two capping sites. A variety of metals including Cd, Co, Cr, Hg, Ni, V, and Zn were analyzed from the exposed mussels. There was no statistical difference in metal uptake by mussels at the cap sites as compared to reference stations.

Both mounds were recolonized by benthic organisms within a year after capping occurred. The population at each cap site differed from each other and from that of the surrounding area. Since cap material was placed to a minimum depth of 0.5 m. over each mound, bioturbation of the contaminated sediment was precluded.[4]

Cap Stability

Disposal operations took place in approximately 20 meters of water. Stamford Harbor sediment, as well as the silt cap from New Haven Harbor, was dredged by clamshell dredge and disposed by scow. Dredging and disposal of the sand from New Haven Harbor was conducted with a hopper dredge. The surficial microtopography of the cap was greatly influenced by the type of dredging used. The mechanical dredging of cohesive silt from New Haven Harbor resulted in a rough, blocky surface texture, while the hydraulically dredged sand resulted in a smooth surface.[5]

In November 1979, a bathymetric survey of the silt capped site revealed that about 10,000 cubic meters of dredged material had been dispersed by a recent 10-year frequency storm. The dispersion of cap material did not expose any of the Stamford sediment.[6] Survey of the sand cap site 2000 meters north of the silt cap site showed no erosion. Explanation for the difference in erosion between the two caps can be attributed to the lower shear-stress which was developed over the smooth sand surface and which did not exceed the critical value. The storm wave conditions created a rough textured cap and the resulting increased shear-stress had a greater effect on the stability of the cap than any other factor, including the depth of the deposit, etc. The decisive factor affecting the stability of the cap was the shear-stress increased by the roughness and clumpy nature of the cap storm wave conditions, rather than the depth of the deposit, the strength of the currents or the cohesive nature of the sediment.[7] Bathymetric surveys of the two mounds have revealed that the cap is stable and that no additional erosion has occurred.

New York Bight Apex Capping Study

Background

In 1980, seven private permit applicants submitted testing data which showed their sediment was not toxic but that it did cause an unacceptable level of bioaccumulation of PCBs (6 projects) and cadmium (1 project). Ocean disposal was permitted because of the demonstrated need for the work and the lack of disposal alternatives, and capping was required because of the bioaccumulation. Disposal occurred in 27 meters of water at an approved location not previously used for dredged material disposal. The Bight Apex capping projects relied on the studies conducted in Long Island Sound and also involved additional monitoring studies.

After the 657,000 cubic meters of contaminated material were point disposed by scow, the cap was placed in two stages. First 224,000 cubic meters of clean, fine grained dredged material were deposited by scow to create an intermediate layer. Secondly, 1,173,000 cubic meters of sand were placed by hopper dredge. The intermediate layer of fine grained material prevented the release of contaminants at the time of impact.[8]

Effectiveness as a Chemical Barrier

In 1980, Attwell and Colwell (1981) determined that members of the genus Thermoactinomycetes which are found in large numbers in the sediment of New York Harbor were unable to grow under the conditions prevailing at the cap site. Microbiological examination of the cores collected at the dumpsite revealed that bacteria were contained below the cap and that large scale mixing of the contaminated dredged material and the overlaying sand did not occur.[9] (X-radiographic analysis of other cores taken at the capsite confirmed that minimal mixing took place between the sand cap and the underlying fine grained dredged material.)

An investigation was conducted by O'Connor (1981) on the sediments from the ten dredging projects involved in the Bight Apex capping study to evaluate chemical and physical differences within and between the sediments.[10] Analysis was performed on representative dredged material samples from the scows and on cores taken later through the capped mound. It was not possible to differentiate between the projects based on grain size and bulk sediment analyses of the fine grained dredged material. There was too much intraproject variation, and several projects were disposed concurrently which reduced the possibility of forming distinct disposal layers. A portion of a core taken from the cap indicated an area which exhibited the chemistry of a "unique" project (distinguished by its extremely high levels of copper, zinc and lead). X-radiographic data from the cores revealed that the average thickness of the sand layer was 1.08 meters.

A monitoring program utilizing mussels (*Mytilus edulis*), similar to that conducted in Long Island Sound, was established on the capped mound in the Bight Apex and at reference locations.[11] Bags containing 25 mussels each were retrieved from each station periodically and analyzed for metals, chlorinated hydrocarbons and petroleum hydrocarbons. Bioaccumulation analyses indicated that for all parameters, uptake was erratic and generally of such low levels that the results could not be statistically correlated to any variable.

Cap Stability

The Atlantic Oceanographic and Meteorological Laboratories of the National Oceanic and Atmospheric Administration conducted a study at the cap site to determine: (a) changes to the cap as a result of bottom currents and sediment transport over the winter of 1980-1981, (b) the bottom current velocities necessary to initiate erosion of the cap, and (c) the long-term probability of erosion by wind-generated waves.[12]

Field data was collected utilizing a sidescan sonar to determine the bottom microtopography. Results indicated that the sand placed by the hopper dredge formed a relatively smooth surface, similar to the sand cap in Long Island Sound. The small amount of sediment transport that took place over the winter resulted in a further smoothing of the surface which will tend to inhibit further transport. Field experiments to determine the threshold current velocity needed to initiate resuspension of the cap were conducted utilizing a sea-going flume (Seaflume). The Seaflume photographically recorded bottom sediment response to a systematic increase in flow velocity generated by a self-contained submersible pump and motor assembly. Freeland, et al.[13] found that the threshold shear velocities at the seabed interface ranged between 0.6 and 1.4 cm/sec. To determine the frequency of events which would generate sufficient shear velocity to erode the cap, two investigations were made. Two current velocity probes were deployed around the cap site to allow correlation of bottom shear velocity with wind and wave conditions on the ocean surface. Wave hindcasting, employing wave models and past wind and wave data collected from nearby weather stations, were compared to determine long-term cap stability. The data, as used in an erosion simulation model, indicated that approximately 1 centimeter of cap would erode each year. The COE conducted comparison of cores taken in August 1981 and July 1983 at various locations over the cap site. Insignificant changes in the thickness of the sand cap were revealed.

Additional Capping Studies

The Japanese government conducted a study of the overlaying of sand on top of highly organic sediment in Hiroshima Bay and compared the improvement in water quality to the dredging of highly organic sediment in Osaka Bay.[14] Studies conducted six months after capping in Hiroshima Bay showed that the capped areas had a reduced rate of nutrient release and a more diverse macrobenthos than was noted for the dredged areas of Osaka Bay.

To fill in information gaps concerning the long-term integrity of a capped deposit, the Corps of Engineers is conducting additional investigations into the areas of bioturbation and contaminant release, recolonization of capped mounds, and consolidation of capped dredged material deposits.

Subaqueous Borrow Pits

Subaqueous borrow pits are irregularly shaped depressions on the sea floor caused by sand and gravel mining, typically for construction material and beach replenishment. In an area within 160 kilometers of New York City, the demand for this material has been predicted to be about 10 million cubic meters per year[15]. Most sand mining in the New York Harbor area is confined to the Lower Bay which is composed primarily of sand and gravel[16].

The volume of existing borrow pits in the Lower Bay is about 23 million cubic meters[17]. The idea of using borrow pits as containment sites for dredged material is not new. It was suggested as early as 1973 by Carpenter[18]. Technology is available to carry out a disposal operation over a borrow pit. Volume 1 of the Mitre Report[19] identified the use of subaqueous borrow pits as feasible for large volumes of dredged material and a possible option for disposal of contaminated dredged material.

The filling of borrow pits is an environmentally beneficial dredged material disposal option since borrow pits are known to be subject to a high rate of fine-grained organic sediment deposition, with attendant adverse environmental impacts[20, 21, 22]. When disposal of fine-grained dredged material is combined with the placement of a layer of sand as a cap, the area can be restored to its original condition and productivity of the area should increase. The filling of borrow pits with fine-grained dredged material does have the unavoidable consequence of removing that immediate area from any future sand mining operation.

In order to evaluate subaqueous borrow pits as a dredged material disposal option, the New York District initiated a contract with the Marine Science Research Center at the State University of New York at Stony Brook (MSRC). MSRC had previously spent several years investigating the environmental effects of sand mining and filling of the borrow pits within the Lower Bay. Model studies were conducted and the results evaluated. It was determined that a demonstration project was required to obtain further information on the feasibility of combining the two operations.

Site selection consisted of an evaluation of the following factors:

1. Site must be accessible to barges for the disposal operation.

2. Site must be such that the pit is deep and large enough to contain the spread of the dredged material when it encounters the bottom.

3. Site must be outside areas susceptible to high wave and current energies.

4. Site must not be an area of high biological productivity.

5. Site must not be an area of current sand mining operation.

Evaluation of the above criteria revealed that no currently existing borrow pit was totally acceptable. Investigation was made of the idea of creating a borrow pit of ideal dimensions in an ideal location. This idea was dismissed because of the high cost and the disruption on an undisturbed area. It was determined that the best option available was to modify an existing borrow pit.

A borrow pit located in the center of the Lower Bay was chosen as the site for the Demonstration Project (Figure 1). The purpose of the Demonstration Project was to assess the physical stability and technical feasibility of capping dredged material within a subaqueous borrow pit. This borrow pit was originally dredged in the early 1970's to a depth of 28 meters below mean low water. Since that time, approximately 3.8 million cubic meters of dredged material have been disposed there, in an effort to alleviate the anoxic condition present in the bottom of the pit. Schwartz and Brinkhuis[23] have documented that a high rate of natural sediment deposition, low dissolved oxygen, and impoverished benthic fauna exist at this borrow pit. The sedimentation rate in the pit (on the order of 10 centimeters/year) was reported to be 100 times that of natural sedimentation rates in other estuaries, while no sediment was found to be accumulating on the shallower sandy bottom surrounding the pit. Due to the organic fraction of the sediment which exerts a high biological oxygen demand and has certain contaminants chemically bound to it, the borrow pit supports a benthic community of about 137 organisms per square meter while the surrounding area supports about 1100 organisms per square meter[17].

Because the remaining capacity of the borrow pit, 3.8 million cubic meters, was determined to be too large, it was decided to conduct the project in three phases. As proposed, phase I (Figure 2) of the Demonstration Project would involve the disposal of approximately 153,000 cubic meters of sand. The disposal of this material would accomplish two objectives. First, it would be used to determine the energy of the surge of the dredged material once it hits the pit bottom. Secondly, the material would be disposed so that it would form a three meter high berm across the southern portion of the borrow pit to create an isolated pocket consisting of 10 to 15 percent of the volume of the entire pit.

Phase II of the project would consist of the disposal of approximately 300,000 cubic meters of fine-grained sediment. This material would be non-toxic and uncontaminated sediment as demonstrated by bioassay and bioaccumulation testing. The phase II sediment would be placed behind the berm.

Phase III would involve the disposal of 230,000 cubic meters to cap the phase II material. This dredged sediment would be non-toxic and have no unacceptable bioaccumulation potential, and at least the top 0.3 meters would consist of clean sand the same grain size as the surrounding bottom. Upon completion of phase III, a monitoring program would be initiated to determine physical and chemical changes to the borrow pit and surrounding area.

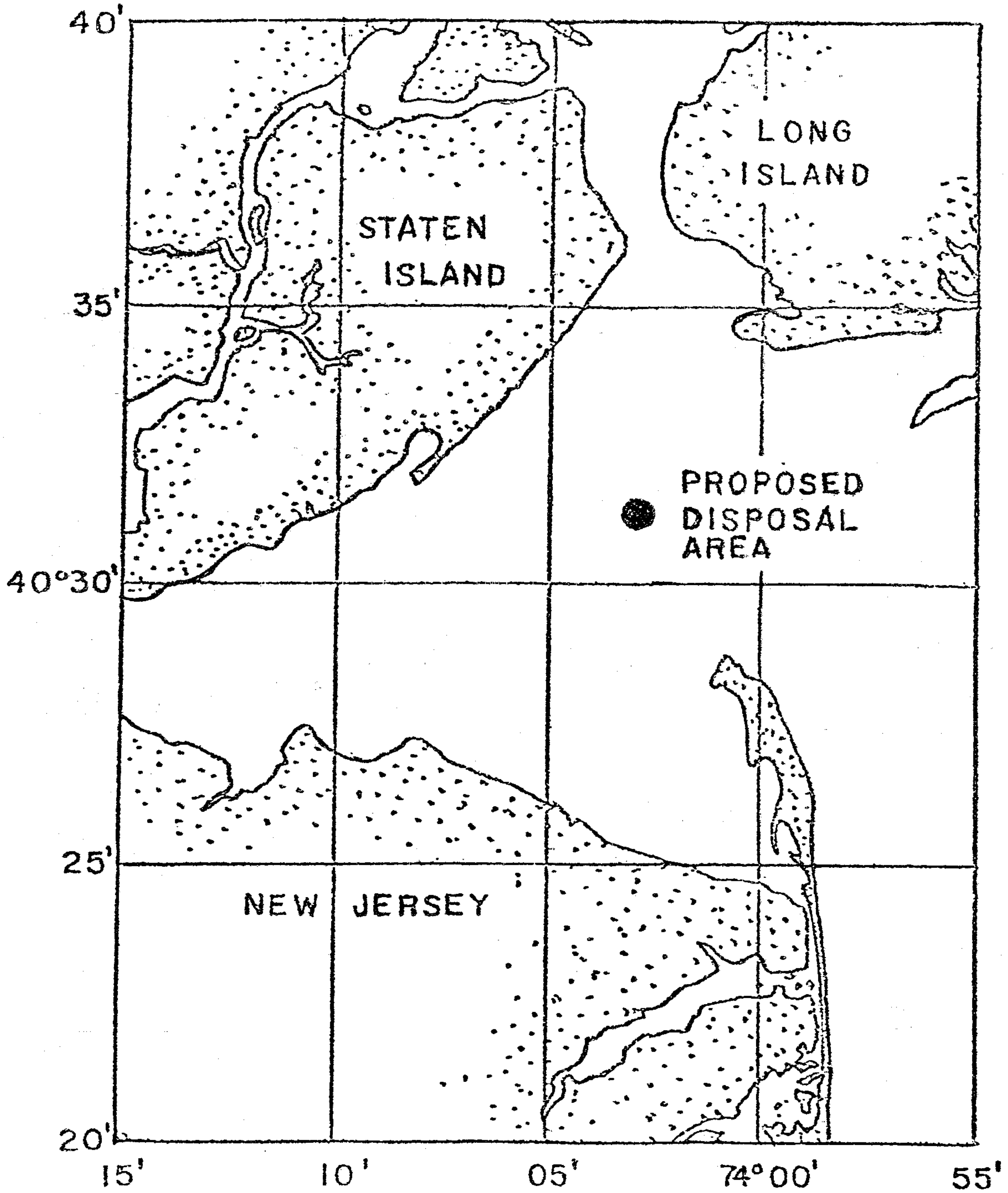

Figure 1. Site location of borrow pit to be used for Demonstration Project

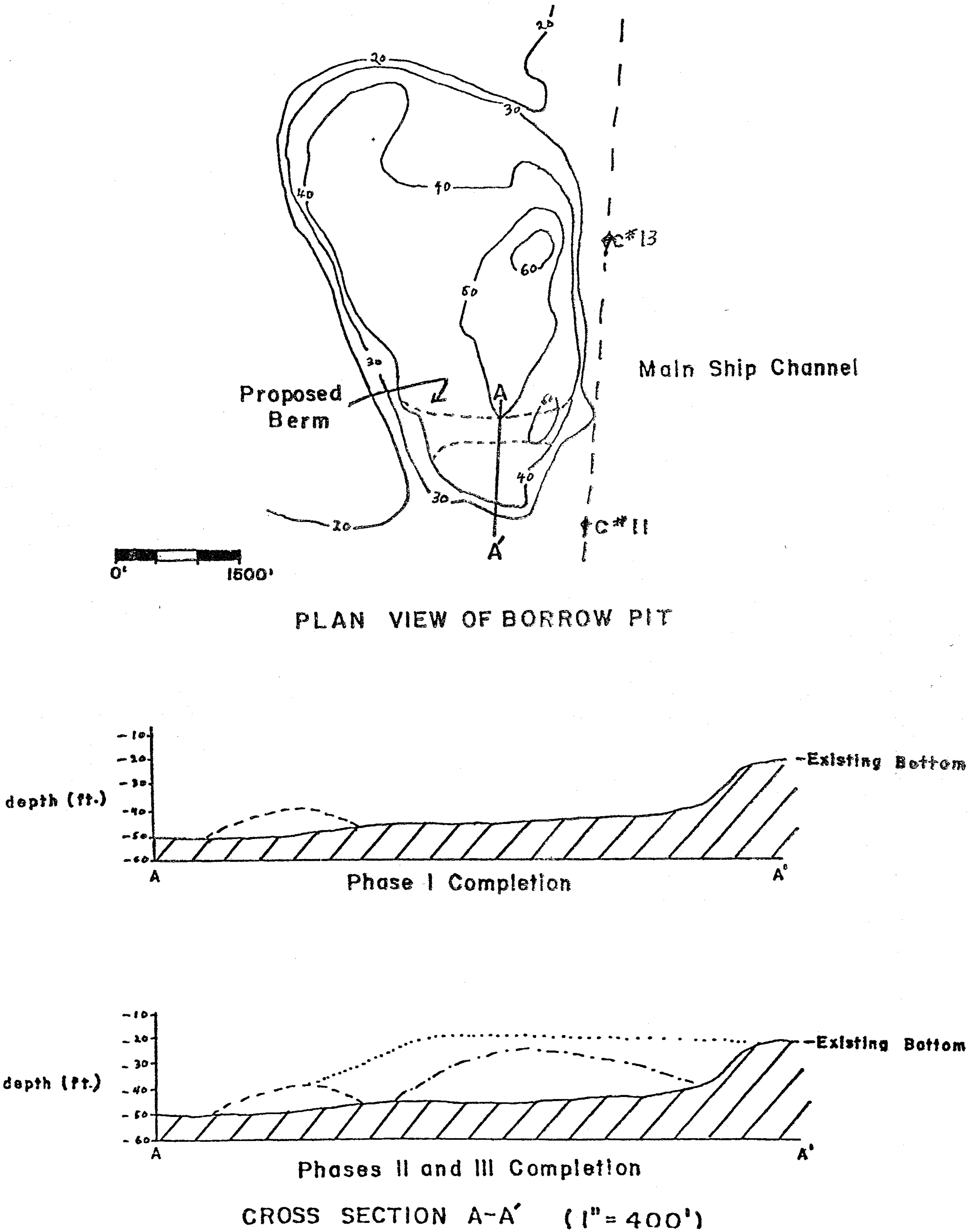

Figure 2. Proposed borrow pit Demonstration Project

During the month of December 1981, 167,000 cubic meters of sand was dredged from the Federal Navigation Project at Ambrose Channel for construction of the berm. Disposal was accomplished utilizing the Corps of Engineers hopper dredge Goethals. This created an isolated area with the dimensions 200 meters wide by 200 meters long with a maximum depth of 14 meters below mean low water.

Prior to the initiation of phase II, the Water Quality Certification issued by the New York State Department of Environmental Conservation authorizing the project was challenged in state court by a private environmental organization. It was the allegation of the environmental organization that the borrow pit proposed for the Demonstration Project was an area of high biological productivity and as such, an Environmental Impact Statement should have been prepared. At this time, the lawsuit has not been resolved. In the interim, the New York District Corps of Engineers has been funding two studies to further evaluate the biological productivity of the borrow pit and determine its relative importance compared to the rest of the harbor. Analysis of the results of these two studies is anticipated to be completed by the end of 1982.

Conclusions

The results of the capping experiments conducted to date demonstrate that with precision disposal, contaminated dredged material can be effectively isolated from the marine environment. A smooth contour mound has been demonstrated to be resistant to erosion for a period of at least 4 years. Periodic monitoring of the cap site allows detection of any breaching of the cap following major storms. The New York District Corps of Engineers has instituted a dredged material disposal management program for ocean disposal, whereby all dredged material originating from New York Harbor is point disposed at one off-shore location. When contaminated dredged material is disposed at this location, a quantity of clean dredged material is subsequently disposed at the same location to insure the isolation of the contaminated material from the marine environment. In addition, routine disposal of clean dredged material will act as a chemical barrier by adding additional layers of thickness which would have to be eroded before the contaminated dredged material would be exposed. This policy results in multiple layers of clean and contaminated dredged material with a substantial thickness of final cap of fine grained and sandy clean dredged material.

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OCEANOGRAPHIC STUDIES IN SUPPORT OF THE EPA DESIGNATION OF DEEP OCEAN DREDGED MATERIAL DISPOSAL SITES IN HAWAII

by

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ABSTRACT

In 1977 the US Environmental Protection Agency (EPA) published regulations to implement the Ocean Dumping Act of 1972 which spelled out procedures for Federal agencies and other applicants to use in obtaining EPA designation of specific deep ocean sites for dredged material disposal. In anticipation of the regulations, both the US Army Corps of Engineers, Pacific Ocean Division (headquartered in Honolulu) and the US Navy (Pearl Harbor) embarked on a series of baseline studies to obtain information for the site designation by EPA. The Corps has the responsibility to maintain safe navigation depths of Federal deep draft and shallow draft harbors in Hawaii. The Navy, of course, is responsible for maintaining the navigability of Pearl Harbor. Of all the harbors in Hawaii, only 6 generate sufficient quantities of dredged materials that cannot be handled by land disposal. These harbors are Honolulu (Oahu), Kahului (Maui), Hilo (Hawaii Island), Port Allen (Kauai), Nawiliwili (Kauai), and Pearl Harbor (Oahu). The Corps circulated an Environmental Impact Statement (EIS) in 1975 to address the impacts of maintenance dredging of these harbors in Hawaii.

In early 1976 the Corps initiated baseline studies at 11 candidate deep ocean disposal sites, at least 2 of which were situated near each of the five harbors falling under Corps responsibility. A full range of oceanographic, water quality, and ecological measurements and observations were taken, and that data formed the basis for narrowing down to 5 sites those areas that would be recommended to EPA for disposal operations in 1977, one near each harbor. The emphasis of the pre-disposal studies were on currents and benthic ecology since the primary impacts of disposal were expected to occur to the benthos. The other primary purpose of the pre-disposal studies were to select suitable sites for disposal operations, and EPA issued research dumping permits to the Corps in advance of the 1977 disposal operations. The Navy did a parallel set of studies at their Pearl Harbor disposal site, patterned after the Corps studies. Several reports were published that present the results of the pre-disposal studies.

In 1977, disposal impacts were monitored directly during maintenance dredging and disposal operations at the Honolulu (by the Corps) and at the Pearl Harbor (by the Navy) disposal sites. Surface vessels were again used as the basis for collecting data, and the emphasis was placed on monitoring the descent, dispersal, and settlement of the dredged materials and the subsequent short term response by benthic organisms. Sonic tracking of the descending plumes was also marginally successful in establishing descent and dispersal rates. Even though both disposal sites were in water depths approximating 200

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fathoms, data colleced during dumping operations indicated that much of the dredged material reached the bottom quickly (within 15 minutes to 4 hours). Much of the finer sized particles adhered to one another causing fine grained sediment to fall quickly to the bottom in chunks or clods. There was some evidence that bottom organisms (shrimps) were attracted to the bottom disposal areas. No evidence of significant impacts were reported in subsequent publications.

The final surveys in 1978 were conducted at all six sites, several months to a year after disposal operations, and most of the measurements taken were patterned after the pre-disposal techniques in order to achieve a better before-and-after disposal comparison. The results of these studies confirmed earlier observations that adverse impacts of disposal were minimal and benthos quickly recolonized the disposal sites. Again the results were published in reports.

On the basis of these studies the Corps recommended to EPA that 5 sites be designated for ocean disposal of dredged materials in 1979. The Navy likewise recommended designation of a site for Pearl Harbor. EPA hired a consulting firm to analyze the Corps and Navy reports and prepare an EIS for site designation. EPA later modified the recommendations slightly by designating one South Oahu site for use both by Honolulu Harbor and Pearl Harbor. The four other harbors each had a site designated for ocean disposal. EPA coordinated the EIS for site designation in 1980 and officially designated the 5 Hawaii deep ocean disposal sites in early 1981. These became the first sites designated for deep ocean disposal of dredged material in the US. The designation process identified the need for future monitoring of disposal operations, especially more data on water quality and long term response by benthic organisms.

The NOAA sponsored University of Hawaii Undersea Research Program (HURL) provided the opportunity to accomplish this monitoring in 1982-83 after 1982 maintenance dredging. The deep sea submersible provided a very successful and efficient procedure for assessing the long term impacts of ocean disposal of dredged material at the 3 sites within the sub's depth range (Hilo, Kahului, South Oahu). A total of 14 dives were awarded by NOAA for the research program covering 3 sites and results corroborated earlier conclusions on bottom currents, bottom geology, benthic ecology and distribution of dumped material *in situ*. NOAA also maintains National Undersea Research Programs with manned underwater facilities at several other locations outside of Hawaii which may be of value in conducting site designation studies.

INTRODUCTION

The Honolulu District of the Pacific Ocean Division is responsible for the navigational maintenance of all commercial deep draft harbors and roughly half of the shallow draft harbors in the State of Hawaii. Maintenance dredging of shallow draft (small boat) harbors in Hawaii does not generate sufficient quantities of dredged materials to justify ocean disposal,and upland disposal sites have been used exclusively. However, the maintenance dredging of deep draft harbors in Hawaii generates considerably more dredged material. Open land is scarce and expensive in Hawaii and designation of semi-permanent land disposal areas would be difficult to impossible to establish. Hence, ocean sites have been the choice for the disposal of dredged materials from Federally-maintained deep draft harbors in Hawaii.

There are seven deep draft harbors in the State with navigable depths in excess of 35 feet, and an eighth deep draft harbor, Barbers Point, is under construction at the southwest corner of Oahu. Honolulu Harbor is Oahu's main commercial port, and Pearl Harbor, located several miles west of Honolulu Harbor on Oahu's south shore,is a military harbor base under the responsibility of the Navy. The island of Hawaii has two deep draft harbors; Hilo Harbor is located on the eastern windward coast of the island near the town of Hilo while Kawaihae Harbor is situated on the north Kona or western coast. Maui Island has a single deep draft harbor, Kahului, located on the north coast, and Kauai Island has two deep draft harbors,one at Nawiliwili on the east coast and the other at Port Allen on the south coast of the island. All of the harbors except Kawaihae were situated near natural embayments at the mouth of river estuaries. Terrigenous sediments derived from weathered basaltic lavas are brought down to the coast by riverine flow and constitute the major source of sediment accumulating in the harbor basins and entrance channels. Reef carbonate sediments broken down into smaller particles by biological and mechanical erosion constitute the only other main source of accumulating sediments in the harbors. Kawaihae Harbor is the newest deep draft harbor in the State and was constructed in 1962. Fortuitously, Kawaihae Harbor is not located near any estuaries, and sediment is not accumulating appreciably in the harbor. Consequently, it has been dredged only once, in 1972 when 25,000 cubic yards were removed.

Of the six remaining Federal harbors requiring maintenance dredging, five are the Corps' responsibility and Pearl Harbor is the Navy's responsibility. The general procedure has been to dredge all harbors during a single cycle normally using one of the Corps' west coast hopper dredges, approximately every 5 years. The Navy takes advantage of the availability of the hopper dredges to dredge Pearl Harbor during each cycle. However, Pearl Harbor's rate of sediment accumulation is much higher than all of the Corps harbors put together, and maintenance dredging of Pearl occurs about twice as frequently as for the Corps harbors. The quantities of dredged materials generated during the 1977 dredging cycle are shown in Table 1. Note that over half of the material is derived from Pearl Harbor and approximately one-half of the remainder is generated from Honolulu Harbor. At the other extreme, Kahului Harbor generates only about 25,000 cubic yards every 10 years, and the low quantities justify that dredging of the harbor be accomplished less frequently.

METHODS

The passage of the National Environmental Policy Act in 1970 and the Marine Protection Research and Sanctuaries Act (Ocean Dumping Act) in 1972 prompted the Honolulu District to sponsor some preliminary field investigation and literature reviews on maintenance dredging and ocean disposal (Towill 1972a, b, Advanced Technology Center, 1973). The field studies consisted of observations of maintenance dredging in Honolulu and Nawiliwili Harbors and ocean disposal of the spoils at the EPA-designated interim sites. Later studies included chemical analyses of bottom sediments obtained from the harbors and collection of bottom photographs at the disposal sites. These studies and additional analyses culminated in the preparation and coordination of an EIS for maintenance dredging of Federal harbors in the State of Hawaii (Honolulu District, 1975).

In anticipation of the finalization of EPA regulations on ocean dumping, the Corps and the Navy, in consultation with the National Marine Fisheries Service (NMFS), the US Fish and Wildlife Service (FWS), the US Environmental Protection Agency (EPA), and the Hawaii State Departments of Health and Land and Natural Resources, decided to sponsor a set of more systematic oceanographic studies before, during and after the maintenance dredging operations in 1977.

The Corps initiated predisposal studies in late 1976 by entering into a contract with a consulting firm (Neighbor Island Consultants, 1977). The Navy awarded a contract to the University of Hawaii Environmental Center (1977a, b, 1978) to accomplish all of the Pearl Harbor investigations. The predisposal surveys were accomplished at 11 candidate sites for the five Corps harbors and at one candidate site for the one Navy harbor. The purpose of the predisposal studies was to evaluate and compare sites and select six environmentally feasible sites for recommendation to EPA as research disposal sites during the 1977 dredging operations. A full range of oceanographic data were collected utilizing surface oceanographic vessels, and considerable emphasis was placed on obtaining baseline data on circulation and benthic conditions. Techniques utilized included current studies, sediment analyses, underwater television, photographs, bottom trawls, sediment cores, bottom grab sample analyses, shrimp and fish trapping, water quality analyses and observations, and plankton tows.

In addition, the Corps requested assistance from the US Army Waterways Experiment Station to perform state-of-the-art mathematical modeling to simulate and predict the behavior of the dredged material to be dumped at the deep ocean sites. Based upon the results of all predisposal studies, EPA agreed to designate on a temporary basis five Corps disposal sites and one Navy site (one for each of the harbors) on a interim basis to facilitate ongoing research.

Both the Corps and Navy sponsored studies during 1977 disposal operations (Tetra Tech 1977, Univ. of Hawaii, 1977b) focused on monitoring the behavior of dumped material at the deep ocean sites and estimating the responses of benthic marine life to the dredged materials at the sites. These studies were performed only at the Corps' Honolulu disposal site and the Navy's Pearl Harbor disposal site. The techniques focused on water quality and sediment analyses during actual dumping of the research sites. In addition Tetra Tech (1977) successfully utilized sonic tracking of the sediment plumes during disposal and descent. Biological observations and trapping studies were also performed and photographs taken to estimate short term response of benthos to disposal activity.

After completion of 1977 dredging and disposal operations, the Corps and Navy sponsored a final set of oceanographic studies at the six sites utilized for dumping (Hawaii Planning Design and Research 1978; Univ. of Hawaii, 1978). The scope of these studies closely followed the predisposal baseline studies in order to facilitate comparison and quantify the impact of disposal operations. These final studies also integrated the results of previous studies into the analysis and recommendations for permanent site designations. On the basis of the research program, the Corps recommended to EPA that five deep ocean sites be designated for Honolulu, Hilo, Kahului, Port Allen and Nawiliwili harbors and the Navy recommended a sixth site for use near Pearl Harbor. The Corps was intending to prepare and coordinate an EIS for these designations, but EPA decided to hire its own consultant (Interstate Electronics Corp.) to review all data, make recommendations, and to prepare the site designation EIS for EPA in early 1979. The consultant concluded that additional study was not required and EPA agreed to go forward with an EIS. The final EIS was coordinated in late 1980, and EPA published in the *Federal Register* its decision to designate a total of five sites. A combined South Oahu disposal site was substituted for two separate sites to serve Honolulu and Pearl Harbors because the harbors were fairly close to one another. The EPA recommended that low-level monitoring activities be accomplished during subsequent disposal operations, particularly emphasizing long-term response and recovery rates by benthic organisms and additional water quality studies.

Establishment of the NOAA (National Oceanographic Atmospheric Administration) sponsored University of Hawaii National Undersea Research Program in Hawaii (HURL) in 1982 provided an opportunity for the Corps to collect additional oceanographic data at three of the designated disposal sites. NOAA granted the Corps four dives in Fiscal Year 1982 at the South site and ten dives in Fiscal Year 1983 at the South Oahu, Hilo, and Kahului sites using the deep diving manned submersible *Makali'i*. The 500- to 800-fathom depths for the two designated Kauai disposal sites were beyond the maximum depth limit of 200 fathoms for the submersible. Nevertheless the observations and data collected for the other three sites were invaluable and applicable to all sites. The *Makali'i* surveyed several miles of the ocean floor in the vicinity of the disposal sites, and a variety of data were collected including: color and black and white television footage, color still photographs, bottom sediment samples, water samples, biological specimens, and rock samples. The submersible was also equipped with accurate positioning instrumentation, depth recorder, current meter, continuously recording oxygen, pressure, temperature, and salinity meters, a mechanical arm and claw which could be manipulated, and a fish collecting device ("fish sucker"), voice recorders, viewing ports, floodlights and other scientific equipment. The successful use of the *Makali'i* provided the opportunity for the Corps to confirm and add upon earlier study results.

TABLE 1
TYPICAL MAINTENANCE DREDGING VOLUMES FOR FEDERAL DEEP DRAFT HARBORS IN THE STATE OF HAWAII, 1977-78 CYCLE

Harbor	Date	Volume		Frequency
Honolulu, Oahu	1977	456,923	cubic yards	Every 5 years
Hilo, Hawaii	1977	54,000	cubic yards	Every 10 years
Kahului, Maui	1977	24,329	cubic yards	Every 10 years
Nawiliwili, Kauai	1977	120,917	cubic yards	Every 5 years
Port Allen, Kauai	1977	141,891	cubic yards	Every 5 years
Pearl Harbor, Oahu	1977-78	1,917,140	cubic yards	Whenever required
TOTAL		2,715,200	cubic yards	

TABLE 2
CHARCTERISTICS OF THE EPA DESIGNATED DEEP OCEAN DISPOSAL SITES FOR DREDGED MATERIAL IN HAWAII

Name	Location Latitude/Longitude	Shape & Size	Distance Offshore	Water Depths
South Oahu[1]	21°15'10"N, 157°56'50"W	Rectangle, 2.0x2.6 km	6.1 km	400-475m
Nawiliwili	21°55'N, 159°17'W	Circle, 920m radius	7.4 km	840-1,120m
Port Allen	21°50'N, 159°35'W	Circle, 920m radius	7.0 km	1,460-1,610m
Kahului	21°04'42"N, 156°19'W	Circle, 920m radius	10.4 km	345-365m
Hilo	19°48'30"N, 154°58'30"W	Circle, 920m radius	8.3 km	330-340m

[1] to be used for both Pearl and Honolulu Harbor dredged materials

RESULTS

Earlier sponsored literature surveys and field studies (conducted before 1976) were not particularly useful in describing baseline conditions and assessing the impacts of ocean disposal at candidate deep ocean sites. Bottom photographs at the Honolulu site depicted the benthic environment to be barren of marine life and dominated by fine sediments. Heavy metal analyses of the harbor sediments indicated that the concentrations of some metals were high but comparable to the levels reported for other non-polluted volcanic (basaltic) rock and sediments. Field observations of dumping operations provided valuable insight into the dispersion and early descent of dredged materials at deep ocean sites off South Honolulu and Kauai, but the numerical calculations and "models" used to predict the long term fate of the materials were unrealistic. Although it was concluded that impacts to water column biota would be temporary and not significant, impacts to the benthic ecosystems could not be accurately predicted due to a lack of information about deep ocean environments in Hawaii. The literature reviews provided some information about shallower water ecosystems in Hawaii and deep sea ecosystems outside Hawaii, but the applicability to the candidate deep ocean sites and disposal operations for Hawaii was questionable.

Predisposal Studies

Oceanographic investigations between 1976-1978, primarily based on the use of surface research vessels, were successful in providing the information needed to describe baseline conditions and accurately assess the impacts of deep ocean disposal operations. Nevertheless, the open water conditions and the very deep water at the research disposal sites (see Table 2) rendered studies inefficient, expensive, and only partially productive.

The baseline (predisposal) studies were accomplished at 11 sites by the Corps researchers and one site by the Navy researchers in late 1976 to early 1977, and at least two sites were situated near each of the federal deep draft harbors. The emphasis of the studies was on bottom ecology and water currents. Current meter arrays were deployed and drogue studies were accomplished at each Corps study site for 1-3 day periods. Although the instrumentation was difficult to deploy and the data collection interval short, current data collected were adequate to document the moderate to strong current conditions at most sites. Although current reversals were noted at subsurface depths and during different tidal states, currents were generally unpredictable.

Bottom grab samples were sorted,and live and dead benthic assemblages were identified and analyzed. Although these studies documented that diverse populations of microbenthos inhabit the sites, the grab sampling technique was not appropriate to sample larger fixed and motile macrobenthos. As a consequence, bottom photographs, television and bottom trawls were accomplished in part to provide additional information on benthic and demersal organisms. The bottom photography indeed allowed additional fish and invertebrates to be observed but the television footage was not of sufficient quality to identify most organisms. Nevertheless, the television tapes provided considerable information on the bottom features and bathymetry at all sites. Where bottom trawling was successful, a number of benthic, demersal, and nektonic species were collected and identified although population size and relative abundance of the individual species could not be accurately calculated. However, the trawls were inoperable near rocky and steeply sloping bottom areas. The undocumented disposal of military munitions many years ago at the Honolulu study sites also hampered data collection efforts using the bottom trawls. Sediment cores were only partially successful because of the rocky nature of some of the bottom environments. Nevertheless, sediment samples were of sufficient size to undergo chemical and physical characterization, and analyses confirmed earlier hypotheses that all disposal site sediments were unpolluted.

The Waterways Experiment Station (1977) conducted numerical modelling studies for the Honolulu District in order to characterize the behavior of dredged material disposed of at each of the five open ocean study regions. Although their model studies predicted that only coarser sediments would reach the bottom at these deep sites, WES acknowledged that the model efforts and results may not be accurate because of the great depths involved.

During disposal studies - The focus of studies conducted during dredging and disposal operations in early 1977 for Honolulu and Pearl Harbor shifted towards tracking the dispersion and descent of the dumped dredged materials and documenting the short term response of marine life to the disposal activities. Discrete water sample profiles were taken in the plume at various time intervals to document water quality changes over time. However, the rapid descent of much of the disposed material reduced the value of these efforts. However, sonic tracking of the descending plume was successful and documented that much of the dredged material reached the bottom at a depth of 450m within 15 minutes and that most of the material reached the bottom within 1-4 hours, confirming earlier speculation that water column impacts during dumping are short term.

Bottom photographs and television documented the conspicuous presence of dredged sediments on the bottom, including clods of fine grained materials. The investigators calculated and concluded that most of the material - fine grained and coarser sediments - reached the bottom at the site despite the great depths involved and contrary to the predictions of earlier numerical model studies. A major factor was that much of the fine grained sediments were cohesive and descending as larger sized clods rather than as individual particles.

Bottom grab sampling enabled the investigators to map the distribution of dredged materials on the bottom following disposal operations by analyzing the chemical characteristics and benthic biological assemblages in each sample. Dredged material has a distinct geological and ecological signature that contrasts with the signature of the resident benthic organisms and sediments at the disposal sites. The grab sampling efforts indicated that some benthic marine life at the disposal sites were buried as expected and that most of the dredged material settled to the bottom within the boundaries of the interim disposal sites.

Bottom trapping studies were also conducted immediately before and after disposal operations. These studies tended to select for specific benthic organisms attracted to the bait in the wire mesh traps, namely bottom dwelling deep sea shrimp. An unexpected result of these studies was that disposal operations appeared to attract shrimp to the site of the dumped material rather than to cause the opposite behavior. However these observations cannot be extrapolated to the majority of the benthos at the sites (which were not attracted to nor caught in the traps).

Post disposal studies

The scope of the final phase of studies in late 1977 to early 1978 was similar to that of the predisposal surveys and were accomplished several months to a year after disposal operations were completed at the six deep ocean research disposal sites. These studies revealed that recovery and recolonization by benthic marine life in the disposal areas was conspicuous and rapid. No significant increases or decreases in benthic organism abundance was apparent when comparing grab sample biological data collected before and after disposal operations. Photographs and television coverage revealed that much of the dredged material, including fine grained materials, was still on the bottom in the vicinity of the disposal sites, especially off South Oahu. There was not any conspicuous evidence for significant redistribution of the dredged sediments on the bottom of the disposal sites. Dredged material was not located at some of the disposal sites where only small amounts of material were dredged from the respective harbors (especially Kahului). It was concluded that the zones of impact were too small to facilitate detection during the post disposal studies at these sites.

The overall conclusion offered by both the Corps and Navy sponsored investigators was that the 1977-78 disposal operations at the deep sea study sites did not result in any significant adverse impacts to the marine environment. On the basis of the study results, the Corps and Navy recommended to EPA that six permanent disposal sites be designated - one each for the federal harbors requiring periodic maintenance dredging and disposal of dredged materials. As noted earlier, EPA tentatively agreed to process a permit and circulate an EIS to address the permanent designation of deep ocean disposal sites in Hawaii for dredged materials.

Eventually in 1981 EPA designated five sites, including a combined South Oahu site for use by both Pearl and Honolulu Harbors and one each for the remaining four harbors. EPA recommended that additional environmental monitoring studies be conducted in conjunction with future disposal operations at the 5 designated sites. These include (1) dredged materials characterization studies, (2) dispersion studies, and (3) benthic studies. EPA believed that additional water quality and dispersion data were needed to identify where less-dense dredged materials will settle. In addition, EPA determined that more information on benthic biology recolonization at the disposal sites would be useful. Finally, EPA concluded that dredged material characterization studies could be accomplished as part of evaluations to determine the suitability of dredged materials for dumping.

In fact the characterization of dredged materials has occurred several times as part of the evaluation process to determine the suitability of dredged materials for dumping. In all cases since 1978, dredged material characterization studies have occurred only for material to be disposed at the South Oahu site. The characterization studies have included chemical assays, bioassays, and bioaccumulation studies. In all of the cases, the studies indicated that the materials to be dredged (primarily from Pearl Harbor, Honolulu Harbor, Ala Wai Canal) were suitable for ocean disposal at the site.

Hawaii Undersea Research Program Studies

During 1982-83, NOAA granted the Corps 14 deep sea submersible dives to use in evaluating the environmental impacts of dredged material disposal at the three deep ocean disposal sites within the depth range of the submersible *Makali'i*. These studies were concentrated at the South Oahu site with some dives also accomplished at the Kahului and Hilo disposal sites. These submersible surveys not only allowed the Corps to accomplish the monitoring studies earlier recommended by EPA in the site designation EIS, but also allowed the use of very efficient data gathering techniques previously unavailable to the Corps. Furthermore, NOAA funded the cost of all of the dives as part of the standard arrangement provided to facilitate use of the submersible for approved research projects.

The submersible surveys covered very long transects across the three disposal sites. Extremely valuable observations and data were collected on the long term fate of the dredged material, the bathymetry and geology of the disposal and adjacent sites, current and water quality conditions, and benthic, demersal and nektonic organisms.

Two of the most obvious discoveries during the dives were that fine grained dredged material was absent and that strong currents were occasionally present on the bottom. It was concluded that these currents over a period of months and years were able to winnow away previous accumulations of fine grained sediment deposits leaving only coarser dredged material deposits behind.

These residual deposits were very conspicuous at the South Oahu site-scattered over the ocean floor and atop elevated outcrops. Very little evidence of dredged material was seen at Kahului and Hilo where the disposal volumes were much smaller.

Another important discovery was that most of the bottom environment not affected by disposal operations were covered by sediment. Epibenthic organisms were generally scarce on the sediment deposits but were conspicuously attracted to hard surfaces and rocky outcrops, especially where vertical walls, ledges, and caves were present. Coarser dredged material deposits and metallic and plastic debris attracted a variety of benthos and fishes.

The observations, photographs, high quality color television video tapes, and biological and geological samples collected by the mechanical claw and "fish sucker" revealed a host of marine organisms previously undetected during the earlier oceanographic surveys using surface ships. A great variety of fish, worms, mollusks, corals, sponges, crabs, shrimps, tunicates, sea stars, sea cucumbers, sea urchins, and crinoids were reported during the dives, many for the first time in the Hawaiian Islands. Although the greatest abundance and diversity of marine life occurred on hard or rocky areas, sediment deposits dominated the bottom environments at all three sites surveyed, especially the South Oahu site.

The submersible dives greatly augmented our understanding of water current and water quality conditions at the sites. In particular, water currents were frequently strong and generally unpredictable except at South Oahu. Here bottom currents generally set to the west along the deep ocean slope of the island.

The dives also revealed that a considerable amount of man-made debris and trash littered the bottom of the three sites, especially South Oahu. Military munitions, motor vehicles, cable, pipes, scrap metal, cups, saucers, and aluminum cans were common observances. Often fish, crustaceans and mollusks were aggregated near larger pieces of junk.

DISCUSSION AND CONCLUSIONS

The results of oceanographic studies conducted at deep ocean sites in Hawaii between 1976-1983 have revealed that ocean disposal of dredged materials has not had any significant adverse or long term impact on the marine environment in Hawaii. Impacts to the water column are only short term, confined to a matter of hours for each individual dump. The major water column impact is the temporary increase in suspended sediments and turbidity. The small volumes of dredged materials, the infrequent interval for disposal operations, and the relatively clean nature of the dredged materials further reduces the overall magnitude of impacts.

Although the most significant impacts of disposal operations occurred to the benthic environment at the disposal sites, little long term adverse effects were noted and some beneficial effects may also have occurred. All of the deep ocean disposal sites in Hawaii are located in water depths in excess of 300m, and the benthic environment at these sites is well below the photic zone, thermocline, and surface mixed layer where most primary production is taking place. As a consequence, the primary sources of food to support the deep sea benthic ecosystems at the sites must directly or indirectly be carried down from shallow water, in the form of migrating fish and other nekton, drifting plankton, detritus, the carcasses of dead organisms, turbidity currents, etc. As a consequence, the biomass of the deep sea benthic communities is lower than what is observed in shallower water, lacking plants altogether. Furthermore, soft bottom communities appear less abundant and diverse than benthic communities and associated fish populations on more rocky surfaces. The great majority of the invertebrates appeared to be suspension and detrital feeders, and many of the fish supported by the benthos are carnivores. These types of communities appear to concentrate on hard surfaces because sessile forms can attach themselves and be elevated above the bottom where efficiency in capturing suspended food is improved. Also, bottom currents tend to be swifter near outcrops, further attracting suspension and associated species. However, rocky areas are rare, perhaps an order of magnitude less abundant than sediment dominated bottom environments.

In this context, the disposal of dredged material at these sites primarily affects soft bottom (sediment dominated) environments where communities are less developed. Although initially the dredged materials may bury benthic species, recovery and recolonization is rapid and some opportunistic species may be attracted to the dredged deposits where shallow water sources of detritus and other food resources may be available. Eventually, finer sediments adhering as clods are washed out of the deposits by currents, leaving behind only the coarser sediment and rock sized dredged materials. These deposits in turn add diversity to the bottom environment and attract fish and invertebrates which use the material for attachment and shelter. Thus the long term impact of dredged material disposal in these environments is minor, provided that the material is not toxic and that disposal operations are not occurring at a massive level.

The NOAA supported deep diving submersible used for some of the Hawaiian studies was an invaluable tool, ideally suited for environmental studies of deep ocean disposal of dredged material in Hawaii. The use of the submersible not only confirmed earlier observations and conclusions generated by traditional surface supported oceanographic techniques but added substantial new information and insights. The use of submersibles should be considered for the site characterization studies required for designation of deep ocean disposal sites elsewhere in the United States. Besides the National Undersea Research Program at the University of Hawaii, Honolulu, NOAA also sponsors similar manned underwater facilities at the following locations: West Indies Laboratory of Fairleigh Dickinson University, St. Croix, U.S. Virgin Islands; University of North Carolina at Wilmington; University of Southern California Marine Science Center at Avalon, Catalina Island, California; and the University of Connectict Avery Point Campus, Groton, Connecticut.

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EFFECTS OF CONTAMINATED DREDGED MATERIAL ON AQUATIC COMMUNITIES - DOCUMENTED LONG-TERM CHANGES

by

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INTRODUCTION

Dredging and aquatic disposal of dredged material is carried out by the U.S. Army Corps of Engineers or by private contractors authorized via permits issued by the Corps. In the early 1970's, it was clear that the technical and scientific knowledge was not available to insure that the disposal of dredged material was being carried out in an environmentally acceptable manner. The U.S. Congress, therefore, authorized the Dredged Material Research Program (DMRP) in 1970 to fill this gap in our knowledge and to identify areas requiring further study. The DMRP was initiated in 1973 and continued for five years.

Task 1A of the DMRP was the Aquatic Disposal Field Investigations. The purpose of this task was to study, in-depth, selected aquatic disposal operations. These field investigations were multidisciplinary in nature and included the evaluation of physical, chemical and biological aspects of aquatic disposal operations.

As will be discussed later, changes in physical and chemical parameters during aquatic disposal were minimal or were non-existent. Likewise organisms inhabiting the water column did not appear to be seriously impacted. However,

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the effects on the bottom-dwelling organisms were more pronounced. Consequently, further studies were conducted at two of the DMRP field sites to monitor long-term changes in the aquatic community.

The purpose of this paper is to briefly summarize the findings of the DMRP field studies and to report on the subsequent long-term monitoring investigations. However, it will be instructive to first point out why it is important to evaluate impacts on aquatic communities, how these studies are normally carried out and some of the problems associated with conducting such investigations.

The Importance and Value of Aquatic Communities

There are two interrelated reasons why one should be concerned with the potential impact of dredged material on aquatic communities. The first reason is economic. There are numerous aquatic species, primarily fish and shellfish, which are commercially important. The loss or the reduction in utilization of these biological resources would seriously impact not only the consumers, both also those people who obtain, process and market the final product.

The second reason why aquatic communities are important is ecological. We do not have, at present, the knowledge or predictive capability to accurately forecast what effects specific environmental perturbations will have on aquatic communities and subsequently on the ecosystem as a whole. Moreover, we are not able to interpret, in a quantitative fashion, the real significance of these changes which are observed in the communities structure and function.

The economic and ecological reasons for studying aquatic communities are interrelated. For example, a particular perturbation (e.g., disposal of dredged material) may have no direct or apparent effect on a commercially important species but may indirectly affect its survival potential by altering its natural habitat, by promoting species which prey upon it or by diminishing its food supply.

How Are Aquatic Communities Studied?

There are two primary approaches to study the potential effects of environmental perturbations on aquatic communities (Capazzo, 1981). In the first approach, the structure of the community is determined. Community structure is described both by the diversity of organisms (i.e., numbers of different species) and their relative abundance (i.e., numbers of individuals and their biomass). Determining community structure is the most common approach used to evaluate aquatic communities and potential environmental impacts.

The second approach involves an evaluation of community function. This would include processes such as energy flow and the recycling of nutrients through the biotic and abiotic portions in the ecosystem. Also included in this approach would be an evaluation of biological interactions such as the degree of competition for limited resources, predation and reproductive behavior. In summary, investigations of community function are generally asking "How does it work?" while studies of community structure are asking "What does it contain?"

Problems Encountered in Evaluating Aquatic Communities

The overriding problem encountered when examining aquatic communities is spatial and temporal variability. Aquatic communities are generally very dynamic and attempts to assess their structure and function is a formidable task. This variability can be partially overcome by intensifying the field sampling effort but this alternative is expensive and very labor-intensive.

This variability is such that it is exceedingly difficult to discriminate changes in communities due to a contaminant-related perturbation and those changes which are occurring naturally.

Another problem associated with studying aquatic communities is the lack of a generally accepted theoretical framework with which to interpret the data. This is especially true when examining community structure. For example, for many years it was accepted that the stability of aquatic communities was positively related to its diversity, or species richness. That is, if a community was composed of a wide variety of organisms, it was believed to be more stable and resistant to environmental perturbations than communities which were less diverse. This hypothesis has come under severe criticism because of increasing empirical evidence to the contrary (Boesch and Rosenberg, 1981).

Another problem encountered in many field evaluations of aquatic communities is the lack of a suitable control to compare with the potentially impacted area. In laboratory investigations, the control serves to detect any unknown or unforeseen variations which may occur during the investigation so that only the effects of the dependent variable being tested can be evaluated. Because the natural environment is so dynamic, it is generally not possible to have a true control treatment in the field. Instead, a reference site is designated and used to compare effects observed at the impacted area (e.g., aquatic disposal sites). Great care must be taken in selecting a suitable reference area which is as similar to the study site as possible (Cowell, 1978).

SUMMARY OF FINDINGS AT THE FIVE DMRP STUDY LOCATIONS

As described in the Introduction, selected aquatic disposal operations at several different locations were studied during the DMRP. Sites for these investigations were located in Eatons Neck, western Long Island Sound, New York; just offshore the Columbia River in Oregon; the Gulf of Mexico off Galveston, Texas; Elliot Bay off the Duwamish Waterway of Puget Sound, Washington; and Ashtabula Harbor, Ohio, in Lake Erie. In all but the first location, disposal operations occurred and were intensively studied during the DMRP.

It is not the purpose of this paper to review in detail the results of these field studies. Those results can be obtained elsewhere (see Wright, 1978 for summary and source of individual publications). Rather, the intent is to summarize the major findings which relate to impacts on the aquatic community. Those effects can be divided into those occurring in the water column and those associated with the bottom or benthic environment.

Water Column Effects

During the disposal operations at each of the DMRP study locations, minimial effects in the water column were observed. Those which were noted (i.e., elevated concentrations of phosphorous, ammonia and suspended solids) were of short duration (minutes to hours) and generally of low magnitude. At the Elliot Bay disposal site, water concentrations of polychlorinated biphenyls (PCB's) were observed in association with elevated suspended particulate matter. These elevated concentrations subsided to background levels within hours to days after disposal. With the exception of the above observation, no significant elevation of concentrations of contaminants (heavy metals, chlorinated hydrocarbons, oil and grease) in the water column was detected.

During some of the disposal operations, finfish migrated out of the area but returned within days after disposal was terminated, in numbers generally greater than before. No other effects on water column organisms were observed. This is not too surprising since changes in physical and chemical parameters were not observed or were of short duration.

Benthic Community Effects

At all of the disposal sites, the most readily observed effect was on the organisms associated with the bottom sediments. Not surprisingly, the structure of the benthic community, immediately following disposal, was diminished both in terms of diversity and abundance due to burial of animals. At all the field locations, animals had recolonized the disposal area within a few months after disposal. By the end of the DMRP, the structure of the benthic community at most of the disposal sites was not substantially different from that at the reference site. However, at several locations there were still some small but detectable differences between the disposal and reference area communities. Consequently, the question of long-term impacts on benthic communities was still not completely answered.

There was little evidence during the DMRP to suggest that benthic animals accumulated substantial amounts of contaminants from the dredged material. This observation was substantiated, in part, by laboratory studies conducted at the same time which showed that benthic animals accumulated considerably less contaminants from dredged material than was generally expected (DiSalvo et al., 1978; Neff et al., 1977). However, benthic animals at the Elliot Bay disposal site had slightly higher tissue concentrations of PCB's (2.0 ppm*) than did animals at the reference site (1.0 ppm). These tissue concentrations

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* ppm = parts per million

correlated very well with sediment concentrations and this correlation was observed throughout the DMRP studies.

SUMMARY OF FINDINGS AT THE TWO LONG-TERM MONITORING SITES

Field studies at the DMRP disposal locations showed that the primary biological effect was on the benthic community. Although sediments were soon repopulated at all the disposal sites, there were persistent differences in community structure at some of the disposal sites. In addition, there was still a concern that animals inhabiting disposal sites would accumulate contaminants, especially PCB's, to unacceptable levels in their tissues. Consequently, further long-term monitoring studies were conducted at two of the DMRP disposal locations, Ashtabula Harbor and Elliot Bay (Tatem, 1984).

The Ashtabula Harbor site was selected because it was the only freshwater DMRP location and because the disposed sediments were contaminated with heavy metals, especially mercury and cadmium. Elliot Bay, a saltwater environment, was selected for further study because the dredged material contained PCB's.

Ashtabula Harbor

Three years after disposal the community structure at both the reference and disposal locations had become very similar. The abundance and diversity were slightly, but not significantly, lower at the disposal location although meiofaunal diversity was greater than at the reference area. Similar animal groups of macrofauna (oligochaetes and bivalves) and meiofauna (copepods, nematodes, and annelids) were found at both sites. There were some differences detected at the species level of identification but these differences were believed to be due to more coarse-grained nature of the dredged material and not its chemical content.

This interpretation of observed differences due to the physical nature of the dredged material is supported by the chemical data. There was no difference in sediment concentrations of cadmium (4.8-5.3 ppm) and mercury (0.7-0.9 ppm) between the disposal and reference sites. Clams and worms from the reference area had slightly higher concentrations of cadmium (0.8-12.5 ppm) and mercury (0.9 ppm) than did animals from the disposal site which had tissue concentrations of 0.3-1.1 ppm cadmium and 0.5 ppm mercury.

Elliot Bay

Three years after disposal, the benthic community at the disposal site had become more similar to that at the reference area indicating further recovery since the end of the DMRP studies. Similar major animal groups (bivalves and annelids) dominated both the disposal and reference areas. However, the abundance and biomass of benthic animals were generally greater at the disposal site.

PCB's in sediments at the disposal site were similar to levels found during the DMRP studies and were still slightly higher (0.32-3.33 ppm) than in sediments at the reference area (0.16-1.75 ppm). Tissue concentrations of PCB's in benthic organisms correlated with sediment concentrations and were essentially unchanged from levels found during the DMRP. Animals at the disposal site contained 0.5-3.9 ppm PCB's while those at the reference location contained 0.2-1.1 ppm.

CURRENT INVESTIGATIONS

The Environmental Laboratory of the Waterways Experiment Station (WES) is presently conducting laboratory and field studies associated with a disposal operation which occurred in Central Long Island Sound, New York, in the spring of 1983. The five year program, called the Field Verification Program, is a large, multidisciplinary program designed, in part, to compare disposal alternatives in upland, intertidal and aquatic environments. Investigations for the aquatic disposal alternative are being conducted through WES by the E.P.A. Environmental Research Laboratory at Narragansett, Rhode Island.

One element of the aquatic study involves monitoring the benthic community which develops on the mound of dredged material. This is being accomplished by intensive sampling of the benthos immediately following disposal with samples collected on a quarterly basis thereafter. Concurrently, environmental contaminant concentrations will be determined in the disposed sediments as well as in the animals which eventually recolonize the mound.

Preliminary results show that very small annelid worms are present on the mound within a week following cessation of disposal (J. M. Germano, personal communication). These small worms are quickly succeeded by another larger type of polychaete worm. Data on contaminant concentrations are not yet available.

CONCLUSIONS

1. Aquatic disposal of dredged material has a minimal to non-existent impact on physical and chemical water quality parameters. No long-term effects on water column organisms have been observed.

2. Aquatic disposal of dredged material has the immediate impact of reducing diversity and abundance of benthic organisms. Animals recolonize mounds of contaminated dredged material rapidly.

3. Community structure at disposal sites becomes very similar to nearby reference areas within three years. Persistent differences which are observed are believed to be related more to geophysical differences in the dredged material than to the level of contaminants.

4. Investigations are currently underway to document short-term and long-term changes in benthic community structure at an uncapped mound of contaminated dredged material in Central Long Island Sound, New York.

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APPLICATION OF PHYSICOCHEMICAL ESTIMATION METHODS TO BIOACCUMULATION FROM CONTAMINATED SEDIMENTS. II. STEADY-STATE FROM SINGLE TIME-POINT OBSERVATIONS.

By

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ABSTRACT

In this investigation asiatic clams, Corbicula fluminea, and fathead minnows, Pimephales promelas, were simultaneously exposed to industrially contaminated river sediments. A simple kinetics model and physicochemical estimation methods were applied to bioaccumulation of PCB as total di- and total trichlorobiphenyls.

Results show that residues measured after a short exposure period together with elimination rate constants (k_2) estimated from octanol/water partition coefficients (K_{ow}) can be used to approximate steady-state nonequilibrium tissue concentrations (C_{ss}). The estimated C_{ss} values obtained using tissue residues (C_T) measured after seven days of exposure ($t = 7$), in the equation

$$C_{ss} = C_T/(1 - e^{-k_2 t})$$

agreed with C_{ss} values obtained using longer exposures and larger data sets at the $\alpha \le 0.01$ level.

Furthermore, normalization of residue data on tissue lipid content made comparisons of bioaccumulation between these dissimilar species possible.

INTRODUCTION

Persistent chemicals bioaccumulate to levels which are largely determined by their chemical activities in water and in tissue (Mackay, 1982). The time required to approach steady-state concentrations (C_{ss}) can range from a few hours to several weeks (Southworth et al., 1978; Southworth et al., 1979; Branson et al., 1975). Determinations of C_{ss} are usually made using time-sequence residue data, constant exposure concentrations and kinetic models (Neely et al., 1974; Branson et al., 1975; Southworth et al., 1978; Southworth et al., 1980; Bruggeman et al., 1981; Ellgehausen et al., 1980; Veith et al., 1979).

For the most refractory and persistent chemicals such as polychlorinated biphenyls (PCB), it is often possible to accurately predict C_{ss} based on the

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results of short-term exposures. For example, Neely et al. (1974) found that C_{ss} for 2,2',4,4'-tetrachlorobiphenyl in trout could be predicted based on time-sequence residue data measured over a five-day uptake period, although less than 0.15 of the theoretical C_{ss} was reached in that time.

Any procedure which predicts long-term effects from short exposures has limitations. Rate constants for both uptake and elimination of chemicals from organisms may change with length of exposure or with changing exposure zone concentrations through biological and physical processes. However, such methods are desirable because they cost much less and are more rapid than empirical assessments of C_{ss}.

Bioaccumulation from Sediments: Bioaccumulation of chemicals from contaminated sediments in natural systems has been inferred in a number of investigations (Duke et al. 1970; Nimmo et al. 1971; Nimmo et al. 1974; Young et al. 1977; Stainken and Rollwagen, 1981). However, direct contributions of contaminated sediment to the body burden of field-collected organisms are difficult to separate from contamination due to other sources such as food organisms, atmospheric input or terrestrially originating vectors.

A number of investigators, such as Halter and Johnson (1977), Courtney and Langston (1978, 1980), Fowler et al. (1978), Elder et al. (1979), Lyes (1979), Roesijadi et al. (1978), McLeese et al. (1980), Augenfeld et al. (1982) and Anderson et al. (1983), have spiked natural or artificial substrates with chemicals and analyzed residues in the tissues of organisms exposed to them. In some cases exposures were terminated after a few hours or days, but in most of those cited above an attempt was made to establish steady-state conditions. Some investigators reported concentrations in water as well as in sediments and tissues. Common observations in these reports are that concentrations of hydrophobic chemicals in exposure water are several orders of magnitude less than those in either tissue or sediment after a sufficient period of exposure and that concentrations in tissue approach the concentrations of chemical in the substrate within a factor of about 10, given sufficient time.

A few workers, e.g., Seelye et al. (1982) and Rubinstein et al. (1983), have measured residues in the tissues of organisms exposed in laboratory systems to field-collected contaminated sediments. Both of these investigations produced results that were qualitatively similar to results obtained in other work involving spiked sediments. The exposures of Seelye et al. (1982) lasted 10 days and demonstrated that PCB in the sediments was bioavailable, but no attempt was made to determine steady-state concentrations. Rubinstein et al. (1983) continued their exposures long after steady-state PCB concentrations were reached and found that PCB Bioaccumulation Factors (BAF) in polychaetes *Nereis virens* after 100 days in contaminated sediments were inversely correlated with the organic carbon content of the sediments ($r^2 = 0.76$).

These investigations provide evidence for the following points regarding bioaccumulation involving real-world sediments contaminated with PCB's:

(1) Uptake is demonstrable within 10 days.

(2) Ten days of exposure is probably insufficient to reach steady-state.

(3) Residue concentrations in tissue at steady-state approach those in the sediment within a factor of 10.

(4) Organic carbon content of sediments affects the proportional difference at steady-state between sediment and tissue residue concentrations.

Bioaccumulation and Organism Lipid: Stout (1980) and Schneider (1982) observed that tissue concentrations of hydrophobic chemicals in field-sampled fish could be directly correlated with lipid content. Geyer et al. (1982) suggested that bioaccumulation data should be standardized on lipid content rather than on fresh weight in aquatic organisms such as mussels and fish. Schneider (1982) also concluded that differences in BCF's of a chemical in different organisms were due to differences in "fat" content. "Fat" referred to the lipid pool of an organism exclusive of the structural and other more polar lipids. Mackay (1982) suggested that lipid is likely the dominant phase of bioconcentration of hydrophobic chemicals because of its relatively low phase activity coefficient (γ_i), and organisms of high lipid content should exhibit high bioconcentration values.

These works suggest that the tendency for a hydrophobic chemical to bioaccumulate can be viewed as being activity-specific rather than organism-specific. Given the same exposure opportunities organisms which are entirely dissimilar from a systematic biology view-point may, in fact, attain similar chemical residues if concentrations are expressed in terms of the relevant biochemical compartment in both. Unless the lipid content of an organism is very low, lipid appears to be the appropriate basis for normalization.

Kinetics: The principles, uses and limitations of kinetic models in bioaccumulation or bioconcentration evaluations were reviewed by Spacie and Hamelink (1982). Many workers, including Branson et al. (1975), Ernst (1977), Southworth et al. (1978), Southworth et al. (1979), McLeese et al. (1980), Bruggeman et al. (1981) and Eaton et al. (1983), have used a zero or first-order uptake, first-order elimination model analogous, when exposure concentration is constant, to the "Plateau Principle" of pharmacokinetics (Goldstein et al. 1974). The simplest form of this is a one-compartment, tissue-water model. For aquatic systems the model has been stated (Branson et al. 1975) as:

$$C_w \underset{k_2}{\overset{k_1}{\rightleftarrows}} C_T{}^{1}, \qquad (1)$$

and is expressed differentially as:

$$\frac{dC_T}{dt} = k_1 C_w - k_2 C_T, \qquad (2)$$

[1] C_T originally given as C_f, concentration in fish.

in which: C_T = chemical concentration in tissue,
C_w = chemical concentration in water,
t = time of exposure,
k_1 = uptake rate constant,
k_2 = elimination rate constant.

The model states that processes of uptake and elimination of chemicals in aquatic organisms take place simultaneously and proceed until a condition of steady-state exists between the two. The steady-state concentrations in the exposure zone and in the organism are directly proportional to the rates of uptake and elimination.

If C_w is constant or is assumed to be so over the period of exposure, equation (2) can be integrated as:

$$C_T = \frac{k_1 C_w}{k_2} (1 - e^{-k_2 t}). \tag{3}$$

Equation (3) can be used as a computational model for bioaccumulation, and the rate constants k_1 and k_2 can be calculated from time-sequence data.

Since equation (3) is exponential and first-order, as $t \rightarrow \infty$, $e^{-k_2 t} \rightarrow$ zero and:

$$\frac{k_1 C_w}{k_2} = C_{SS}. \tag{4}$$

Substituting C_{ss} in Eq. (3) and re-arranging the equation gives:

$$C_{ss} = C_T/(1 - e^{-k_2 t}). \tag{5}$$

Equation (5) states that under conditions of constant exposure a tissue concentration (C_T) at any time (t) can be used to estimate steady-state tissue concentration (C_{ss}) provided the elimination rate constant (k_2) is known or can be estimated for a chemical. The actual exposure zone concentration need not be known, but must be assumed as constant.

In addition, solution of equation (3) for time (t) results in:

$$t = -\ell n(1 - P)/k_2, \tag{6}$$

by which the time required to reach a proportion (P) of steady state may be calculated.

<u>Application</u>: Since 1977, a procedure developed jointly by the United States Environmental Protection Agency and the Corps of Engineers (USEPA/CE, 1977) has been used routinely in the USA to assess the potential for bioaccumulation of chemicals from sediment proposed for dredging. This procedure

involves exposure of organisms to sediment for 10 days in aquaria under flowing water conditions. Chemical analysis of the tissues of survivors at the end of this period is used to determine whether there is potential for xenobiotic substances to bioaccumulate from sediments.

Potential bioavailability, or lack of it, is demonstrated for sediment-associated chemicals in this test using a single time-point observation of contaminant concentration in tissue. This test does not consider the differing rates of approach to C_{ss} of bioaccumulating substances. Consequently, no information is provided regarding relationships between measured tissue residues and those that could result after prolonged exposure to the sediment. Residues that can be reported as expected steady-state are clearly more meaningful than levels reported without reference to kinetic processes. In this work we test the ability of observations made at a single time-point during exposure to predict steady-state residues in tissues using a kinetics model, time-sequence sampling and exposures lasting up to 28 days.

MATERIALS AND METHODS

Sediments: Industrially-contaminated river sediments were screened to remove debris ≥ 3.0 mm, homogenized and stored at 3-5°C until use.

Organisms: Adult clams, *Corbicula fluminea*, 20 mm shell length and juvenile fathead minnows, *Pimephales promelas*, were acclimated under flowing water at 23°C for at least 3 weeks before start of exposures.

Exposure System: A flow-through constant temperature exposure system (schematic, Figure 1) consisting of twenty-four, 75-litre aquaria was used. Water was replaced continuously at a rate equivalent to 90% in 16 hours. Twenty-five litre aliquots of each sediment homogenate tested were randomly assigned to three replicate aquaria.

Experimental Design: Three time-sequence sampling schedules were used each having a progressively longer interval between sample taking. A sampling at the end of seven days' exposure was common to all schedules. The intermediate-length schedule terminated at the estimated time[1] of $0.99(C_{ss})$ for dichlorobiphenyls (18 days). The longest exposures similarly terminated at the estimated time of $0.99(C_{ss})$ for trichlorobiphenyls (28 days).

The shortest exposure period was seven days and involved five samplings. These exposures are coded in the tables and in Figure 3 as LS-2, 5, 7, and 8 (low suspended particulates). The intermediate-length exposures contained seven samplings and are coded LS-1, 3, 4, and 6 (low suspended particulates). The longest exposures are coded HS-1 through HS-5 (high suspended particulates) and involved six samplings.

In all LS treatments fish and clams were exposed simultaneously. In these exposures clams were suspended in the water column in polyethylene mesh baskets and fish were allowed to swim freely. Tissue and water sampling was

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[1] Equation (6).

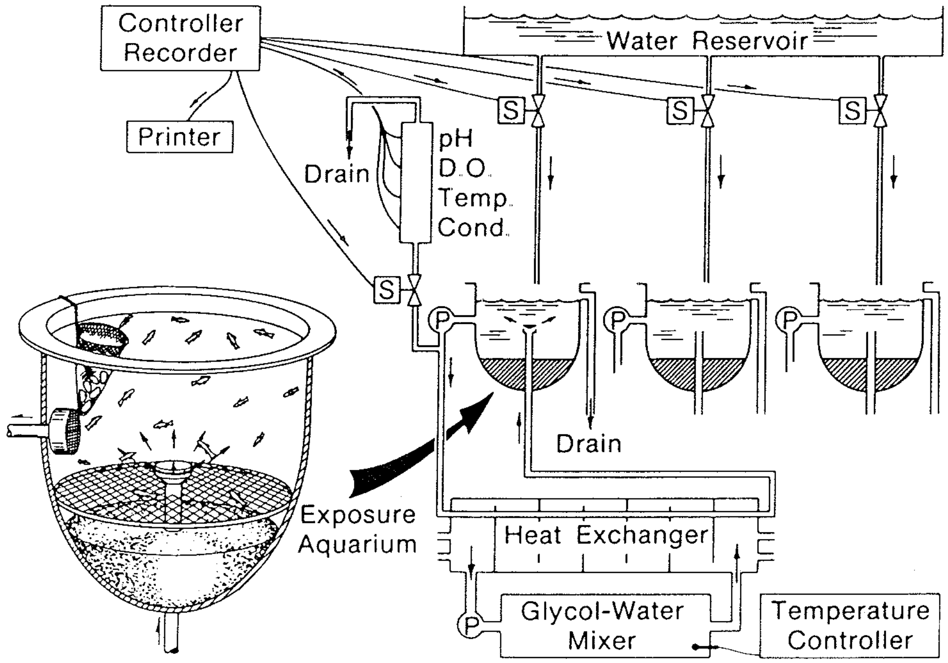

Figure 1. Flow-through exposure system.

not disruptive of the sediment deposits, and ambient suspended particulates (SP) were purposely kept at low concentrations. In the HS exposures, however, clams were allowed to burrow in the sediments and sampling produced periodic high SP concentrations. Fish were not included in HS exposures. Three replications of each exposure treatment were run simultaneously.

Physical Analyses: Sediment particle-size distributions were determined using a modified hydrometer method (Patrick, 1958). Suspended particulate (SP) concentrations in whole water samples were determined gravimetrically on retained ≥ 0.45-μm material pipetted directly from exposure aquaria.

Chemical Analyses: Sediment and tissue extracts were analyzed for PCB as total di- and total trichlorobiphenyls by GC/EC packed column methods using GC/MS for confirmation (US FDA,1977; US EPA,1978). Quantitation was by matching response and retention times of the samples with those of pure isomer standards. Tissues were also analyzed for percent total extractable lipids by gravimetry following whole organism (less shell, for clams) extraction with petroleum ether. Total organic carbon in sediments was analyzed by wet oxidation methods (Konrad et al. 1970; Gaudette et al. 1974).

Steady-State from the Time-Sequence Data: Data were fitted by an iterative procedure (Marquardt, 1963) to equation (3). The mean solution phase PCB water concentrations (C_w) listed in Table II and the lipid-normalized tissue data were used to calculate rate constants for uptake (k_1) and elimination (k_2). Bioconcentration factors (BCF) were calculated as the ratio of the rate constants. Steady-state PCB tissue concentrations (C_{ss}) were calculated as the product of BCF and $C_w S$.

Steady-State from Single Time-Point Data: *A priori* elimination rate constants (k_2') required for single time-point steady-state tissue concentrations using equation (5) and for time to a proportion of steady-state using equation (6) were obtained from the relationship reported by Spacie and Hamelink (1982)[1]:

$$\log k_2' = -.414 \log K_{ow} + 1.47, \quad r^2 = 0.90. \qquad (7)$$

Values of K_{ow} for diCBP and for triCBP isomer groups were estimated by regressing the means of High Pressure Liquid Chromatography (HPLC) - Generator Column determined values for pure isomers (Miller et al. 1983; Woodburn et al. 1983) on isomer chlorine number, $C\ell_n$. The equation obtained is:

$$\log K_{ow} = 0.40\ C\ell_n + 4.24, \quad r^2 = 0.96. \qquad (8)$$

RESULTS AND DISCUSSION

Sediments, characterized in Table I as the screened homogenates used in these investigations, were generally sandy but often contained an appreciable

Table I. Selected Characteristics of 13 Upper Hudson River Sediments.

	PCB, $\mu g\ g^{-1}$		Texture, percent	
Code	diCBP*	triCBP**	TOC%	Sand:Silt:Clay
HS-1	2.8	4.5	2.49	70:19:11
-2	48.	38.	2.17	64:22:14
-3	2.2	3.2	1.08	69:19:12
-4	11.	8.6	2.69	74:16:10
-5	22.	13.	1.94	43:38:19
LS-1	121.	40.	6.12	67:21:12
-2	26.	11.	10.4	62:20:18
-3	25.	13.	2.39	51:27:22
-4	246.	100.	7.18	75:05:20
-5	44.	21.	5.27	58:17:25
-6	1.1	2.1	--	83:05:13
-7	9.4	4.2	3.03	70:13:18
-8	47.	30.	5.85	05:54:41

[1]Reformatted from original. In this paper k_2' indicates estimated elimination rate constant using equation (7).

amount of silt and clay. Concentrations of total PCB spanned two orders of magnitude (~ 3 to ≥ 300 μg g^{-1}) and ranged in total organic carbon (TOC) content between ~ 1.0 and 10 percent. Relative proportions of the two PCB isomer groups showed a moderate correlation with TOC ($r^2 = 0.44$), diCBP being in greater abundance than triCBP as TOC increased.

Concentrations of suspended particulates (SP) and PCB in unfiltered water (C_wT) and in the solution phase (C_w) generally decreased over the periods of exposure as fines were removed from the aquaria by culture water flow-through. The overall mean concentrations and variances over the periods of exposure for these parameters are given in Table II. Desorption of the more soluble diCBP from particulates in the water was essentially complete. Somewhat less desorption of triCBP occurred, and the differences between C_w and C_wT for these isomers was significant at $\alpha \leq 0.05$ by paired t-test.

Evaluation of Bioaccumulation: Percent total extractable lipid in clam tissue (±95% C.I.) was 2.03 ± 0.10 and in fish 5.03 ± 0.25. We found that as duration of exposure increased, fish and clams became increasingly similar in their PCB residue levels when the data were normalized on a lipid basis. Figure 2 shows the relative responses of fish and clams exposed simultaneously to the same contaminated sediments. At the earliest samplings, fish contained diCBP residues about five times greater than those of clams when concentrations were expressed on a fresh weight basis and about 2 to 3 times greater on a lipid basis. As time progressed, diCBP concentrations in the two species became more similar and after 18 days were identical in the two when residues were calculated on a lipid basis. In the case of triCBP's, concentrations were statistically different ($\alpha \leq 0.05$) for clams and fish when expressed on a fresh weight basis but were the same on a lipid basis throughout exposure.

Calculated data using the simple kinetics model are shown in Table III, and uptake curves are drawn to the same scale for di- and triCBP in Figure 3 for all exposures. The effect on approach to apparent steady-state of an appreciably non-constant exposure zone concentration is evident in the responses of fish in exposures LS-1 and LS-4. In both of these the PCB content of sediments was unusually high, and the activity of the fish at the start of exposures caused high SP levels. The earliest tissue samples were taken at a time of high exposure zone concentration and did not diminish or increase markedly over the remaining exposure period. Clams exposed with the fish responded more slowly and reached apparent C_{ss} of diCBP at lower levels.

Since clams can temporarily isolate themselves from noxious conditions in the water by closing their valves and ceasing filtration, it is possible they avoided initial high PCB concentrations in the water caused by activity of the fish. The fish would have had no defense against exposure to PCB desorbing from the suspended material. Because PCB's depurate slowly, the initially high exposure of the fish caused them to remain at high levels of PCB residue during the remainder of the study.

Table II. Selected Characteristics of Exposure Water

Code	Dichlorobiphenyls, μg l^{-1}		Trichlorobiphenyls, μg l^{-1}		Suspended
	Solution Phase	Whole Water	Solution Phase	Whole Water	Particulates, mg l^{-1}
HS-1	1.3* (0.46)**	--	0.91 (0.27)	--	160. (160.)
-2	5.7 (4.2)	--	2.6 (1.7)	--	74. (98.)
-3	1.2 (0.81)	--	0.79 (0.53)	--	150. (190.)
-4	4.2 (2.4)	--	2.0 (0.94)	--	280. (280.)
-5	3.8 (1.5)	--	1.6 (0.67)	--	110. (107.)
LS-1	3.9 (2.5)	4.2 (2.8)	0.84 (0.45)	1.5 (1.1)	48. (39.)
-2	1.1 (0.50)	1.1 (0.53)	0.3 (0.15)	0.40 (0.22)	32. (33.)
-3	1.3 (0.69)	1.3 (0.69)	0.47 (0.30)	0.54 (0.33)	15. (6.8)
-4	9.4 (4.7)	10. (5.1)	1.9 (0.70)	2.6 (1.3)	28. (16.)
-5	1.5 (0.45)	1.5 (0.48)	0.44 (0.11)	0.58 (0.20)	37. (36.)
-6	0.31 (0.08)	0.32 (0.08)	0.19 (0.10)	0.19 (0.10)	6.4 (4.8)
-7	0.28 (0.06)	0.29 (0.06)	0.13 (0.09)	0.13 (0.09)	3.8 (1.8)
-8	1.7 (0.76)	1.8 (0.80)	0.67 (0.16)	1.00 (0.33)	93. (49.)

* Means of 36 observations
** Standard deviations

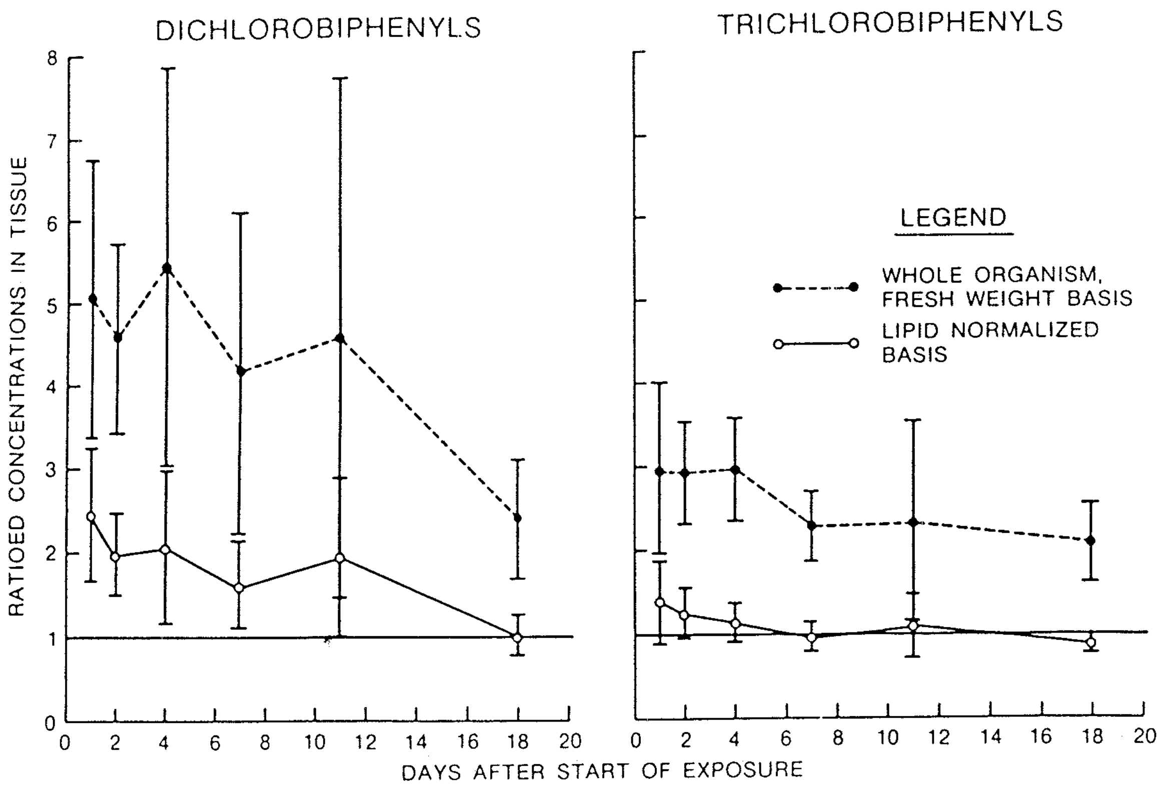

Figure 2. Ratio of PCB concentrations in clam and fish tissue expressed on lipid-normalized and on fresh, soft-tissue bases. Ratio = $[PCB]_{fish} \div [PCB]_{clam}$. Vertical bars and brackets represent 95% confidence intervals on the ratios.

Bioconcentration factors (Table III) were similar for triCBP regardless of exposure or organism. For diCBP, the high SP exposures resulted in higher BCF's than did the low SP exposures. This latter effect was probably the result of high initial exposure concentrations coupled with the higher water solubility of diCBP.

Evaluation of Single Time-Point Estimation: The assumption of constancy in the k_1C_w relationship permits use of k_2 to make estimations of steady-state tissue residue from concentrations measured after short exposures (equation 5). When experimental values of k_2 are not available they may be estimated. Neely (1979) used the bioenergetics-based model of Norstrom et al. (1976) to estimate rate constants for uptake and elimination of chemicals when the ventilation volumetric rate of fish and octanol/water partition coefficient (K_{ow}) of a chemical were known. The simpler relationship reported by Spacie and Hamelink (1982) and used here (equation 7) relies only on K_{ow} to obtain k_2'. This is recommended with the caution that use of thermodynamic relationships to estimate a process which takes place over time does not imply mechanisms of action. The empirical correlation is sufficiently strong, however, that it provides a usable simplification.

Table III. Calculated Data[1] for Clams and Fish Exposed to PCB Contaminated Sediments using Time-Sequence Sampled Residues and the Simple Kinetics Model. Steady-State PCB Concentrations in Tissue (C_{ss}) are Lipid-Normalized, $\mu g\ g^{-1}$.

	Dichlorobiphenyls			Trichlorobiphenyls		
Code	C_{ss}	log BCF	k_2 (C.I)	C_{ss}	log BCF	k_2 (C.I)
Clams, *Corbicula fluminea*						
HS-1	154	5.07	0.088 (0.09)	360	5.60	0.049 (0.04)
-2	1000	5.24	0.189 (0.12)	1070	5.61	0.099 (0.05)
-3	225	5.27	0.054 (0.07)	410	5.72	0.027 (0.07)
-4	416	5.00	0.200 (0.17)	531	5.42	0.077 (0.08)
-5	712	5.27	0.226 (0.13)	551	5.54	0.101 (0.05)
LS-1	200	4.71	1.68 (6.24)	387	5.66	0.127 (0.13)
-2	86	4.89	0.279 (0.37)	122	5.59	0.201 (0.33)
-3	83	4.81	0.646 (0.77)	241	5.71	0.141 (0.07)
-4	353	4.57	0.737 (0.72)	898	5.67	0.093 (0.07)
-5	115	4.88	0.721 (0.69)	204	5.67	0.230 (0.16)
Fish, *Pimephales promelas*						
LS-1	463	5.07	27.8 (3.14)	254	5.48	0.483 (0.32)
-2	51	4.67	1.19 (2.42)	73	5.37	0.316 (0.34)
-3	111	4.93	0.630 (0.37)	158	5.53	0.197 (0.16)
-4	933	5.00	30.7 (0.02)	735	5.59	0.611 (0.28)
-5	89	4.77	1.47 (1.50)	127	5.46	0.423 (0.19)
-6	35	5.05	31.5 N.C.	76	5.60	0.492 (0.29)
-7	18	4.81	0.371 (0.20)	21	5.21	0.196 (0.15)
-8	156	4.96	0.291 (0.046)	386	5.76	0.114 (0.23)

[1] Symbols defined in text.

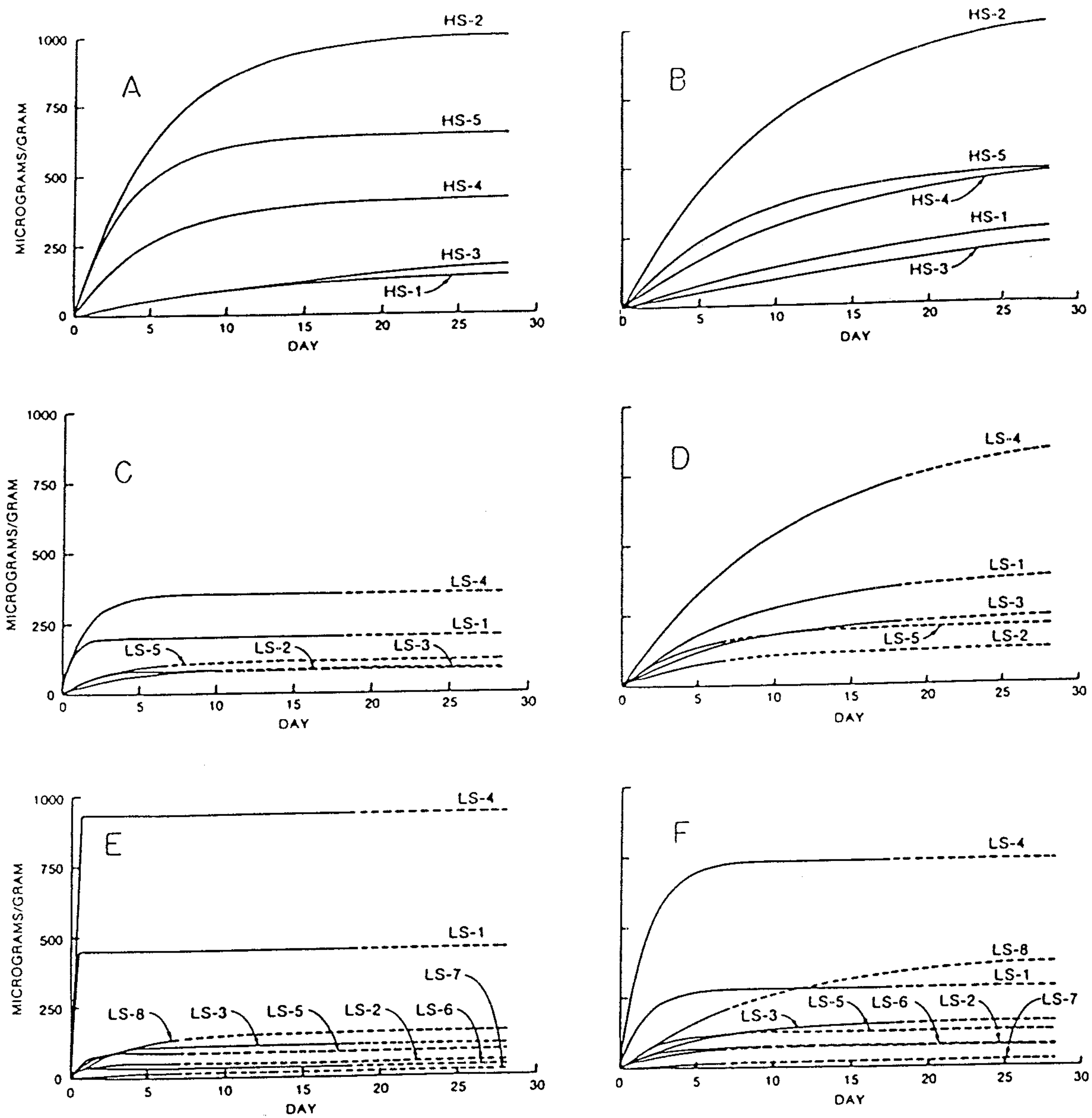

Figure 3. Uptake of PCB's in clams and fish. Tissue concentrations expressed on lipid-normalized basis. Codes HS are high and codes LS are low suspended particulate exposures. A, C, and E are diCBP and B, D, and F are triCBP concentrations. A, B, C, and D are clams, Corbicula fluminea, and E and F are fish, Pimephales promelas. All curves fitted using time-sequence data (means of three replicates, five to seven sample times) and the simple kinetics model. Dashed lines are projections beyond the measured data.

In order to evaluate steady-state tissue concentrations based on observations at a single sampling using equation (5), the following parameters were calculated:

C_{T7} : mean lipid-normalized tissue concentrations of di- or triCBP's in clams or fish measured after seven days of exposure.

C_{ss} : steady-state tissue concentrations determined by application of kinetics model, equation (3), to time-sequence data sets.

C_a : mean tissue concentrations measured at time corresponding to 99 percent of steady-state (18 days for diCBP; 28 days for triCBP) by equation (6) and k_2'.

C_{ss7}: steady-state tissue concentration estimations based on C_{T7} and k_2' using equation (5).

Correlation coefficients and symbols are shown in Table IV. All parameters correlate 0.91 or higher. Parameters are compared in Table V.

Table IV. Correlation Matrix: Evaluation of Single Time-Point Steady-State Estimates (Symbols identified in the text).

	C_a	C_{ss}	C_{ss7}
C_{T7}	0.926*	0.910	
C_a		0.942	0.933
C_{ss}			0.919

*Correlation coefficients, r.

The first comparison in Table V shows that tissue residues measured after seven days of exposure were not at steady-state. Tissue residues of the pooled di- and triCBP data were significantly lower at seven days than they were after 18 or 28 days of exposure.

The second comparison in Table V shows that single time-point measurements made at the times predicted for near steady-state tissue concentrations using k_2' (C_a) are not significantly different than the steady-state concentrations resulting from application of the simple kinetics model to the whole data set (C_{ss}).

Similarly, the third comparison in Table V shows that C_a does not differ significantly from the single time-point estimation of steady-state using equation (5) and measurements made at seven days of exposure (C_{ss7}).

Table V. Paired-t Statistics on Evaluation Parameters. Symbols identified in the text).

Pairs	D	S.D.	t	d.f.
1. C_{T7} and C_a	-125.0	140.0	-3.680**	16
2. C_a and C_{ss}	-14.0	100.0	-0.560 (N.S.)	16
3. C_a and C_{ss7}	33.0	114.0	1.20 (N.S.)	16
4. C_{ss} and C_{ss7}	20.0	118.0	1.00 (N.S.)	35

** Significant at $\alpha \leq 0.01$.

The fourth comparison shows that steady-state concentrations derived kinetically from time-sequence data (C_{ss}) do not differ significantly from the estimate (C_{ss7}). Thus, steady-state concentrations can in fact be estimated from observations made at a single time. The exposure period used here (seven days) is less than the current ten-day period of the sediment bioassay and substantially less than the time required to reach plateau in many of the cases illustrated in Figure 3.

CONCLUSIONS

This work has documented that steady-state concentrations of di- and triCBP, and by implication hydrophobic chemicals of similar persistence and partitioning, can be accurately estimated using a kinetics model and measurements of residue in tissue after a short exposure period during which steady-state was not reached. Linear estimation methods developed in aquatic exposures in which pure chemicals in solution constituted the exposure medium can be adapted to conditions in which industrially contaminated sediments are sources of mixed chemicals. We also find that lipid normalization of residue data facilitates interspecific comparisons of bioaccumulation of these chemicals in clams and fish. Expression of residue data on a lipid basis may similarly enable comparisons among other unrelated organisms.

Spacie and Hamelink (1982) suggested that the range over which octanol/water partitioning relationships are good indicators of bioconcentration of chemicals in fish is limited to log K_{ow} = 2-5. Our results indicate that for lipid-normalized data the applicability can be extended to clams, to systems involving deposited and suspended contaminated sediments, and the upper limit for predictions is at least marginally greater than log K_{ow} = 5.

If applied to current sediment bioassay and bioaccumulation practice, these relationships could allow more time- and cost-effective evaluations than are now being performed.

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TIOGA-HAMMOND LAKES: UNDER ICE HYDRODYNAMICS INFLUENCE ON WATER QUALITY CONTROL

by

Friedrich B. Juhle *

Tioga-Hammond Lakes are located just upstream of the confluence of the Tioga River and Crooked Creek near the town of Tioga in north central Pennsylvania. The Tioga Dam, 2,710 feet long and 140 feet high, blocks the Tioga River, and the Hammond Dam, 6,450 feet long and 122 feet high, blocks Crooked Creek. Both embankments tie into a low ridge that separates the adjoining watersheds. A short distance upstream of the dams, a man-made channel makes it possible for the two lakes to share a common flood control outlet works located through the Tioga embankment and a common emergency spillway around the Hammond embankment. Because of severe acid mine induced water quality problems in the Tioga watershed, a wier was built across the connecting channel to separate the good quality Hammond Lake from the degraded Tioga Lake. Flows from Hammond are released through a gate structure in the wier into Tioga Lake or through a small limited capacity (100 cfs) outlet through the Hammond Dam into Crooked Creek. During flood events, the lakes can rise above the wiers and become a single lake that is managed by the Tioga outlet works (Figure 1). Tioga Lake has a 280 square mile drainage area and a normal pool of 9,500 acre-feet. Hammond Lake has a 122 square mile drainage area and a normal pool of 8,500 acre-feet.

WATER QUALITY CHARACTERISTICS

The mainstream Tioga River flowing into Tioga Lake is generally severely degraded by acid mine wastes. These mine wastes cause the pH to range from <4.0 to 6.3. Typically, the pH is near 4.3 when it enters the lake. Fortunately, Mill Creek, a high quality alkaline stream, flows into an arm of Tioga Lake. This alkaline tributary effectively neutralizes the acidic component of the Tioga River, and as a result, the main body of the Tioga Lake sustains a pH near 6.0 most of the time and sustains a limited fishery.

The neutralization capacity of Mill Creek is enhanced by the generally shallow nature of Tioga Lake (maximum depth 45 feet), limited stratification due to the management of the selective withdrawal system in the Tioga outlet tower, and the substantial wind mixing that takes place in the project.

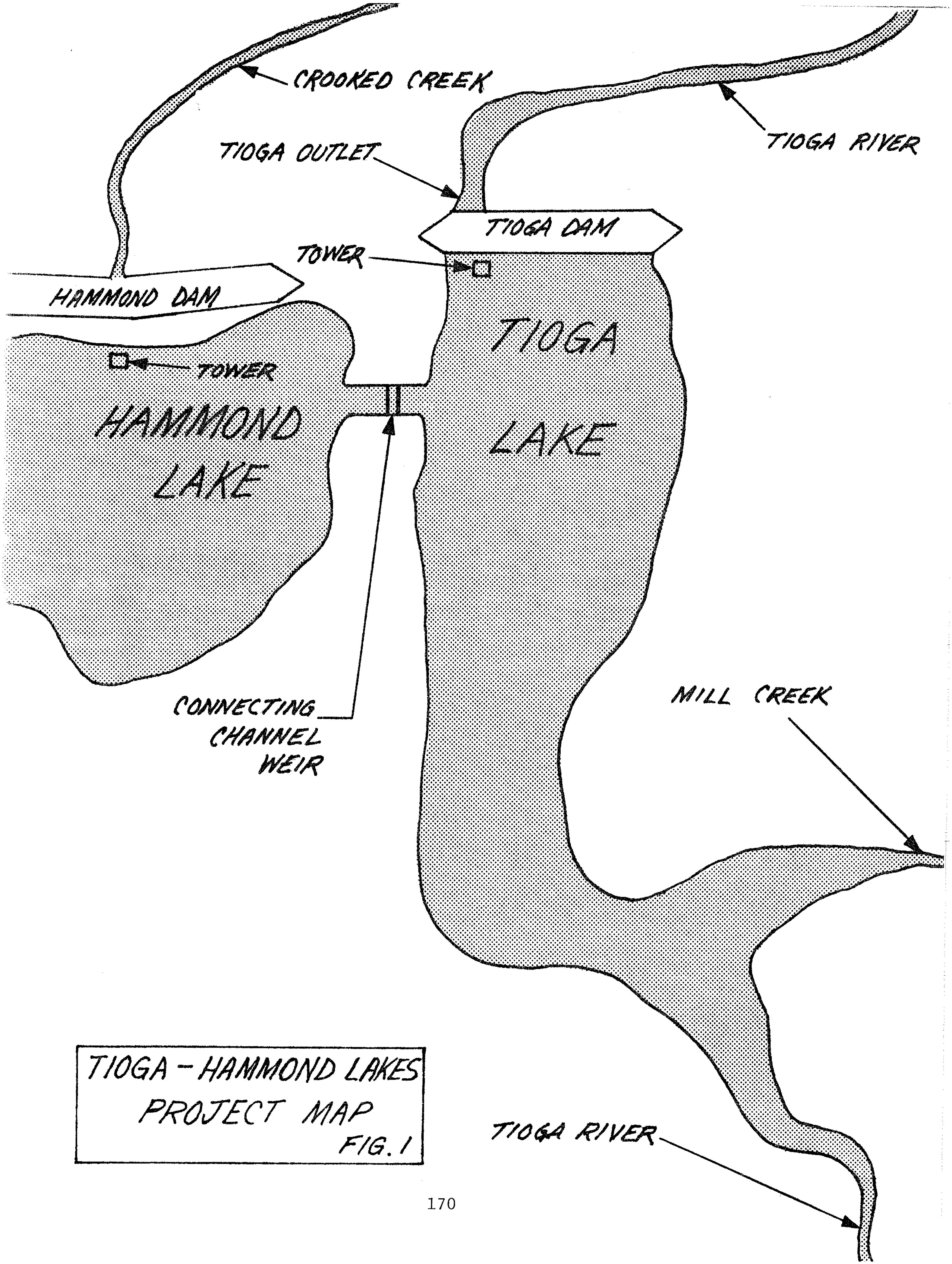
It is the operating objective of the Tioga project to sustain a reasonable water quality in the lake so that a continuously adequate quality for fishery habitat can be sustained downstream. From the years of study before the project's completion, it was concluded that the most difficult time for water quality management would be at the end of a prolonged drought. The worst quality in the river has traditionally been associated with low flow periods. Thus, a prolonged drought would fill the lake with poor quality water. Indeed, prolonged low flow periods have caused reduced lake quality but quality has consistently remained at levels well above survival levels throughout the summer and fall low flow periods. The real problems have occurred after ice formation in winter. These problems were totally unexpected.

*Chief, Water Quality Control Section, Baltimore District

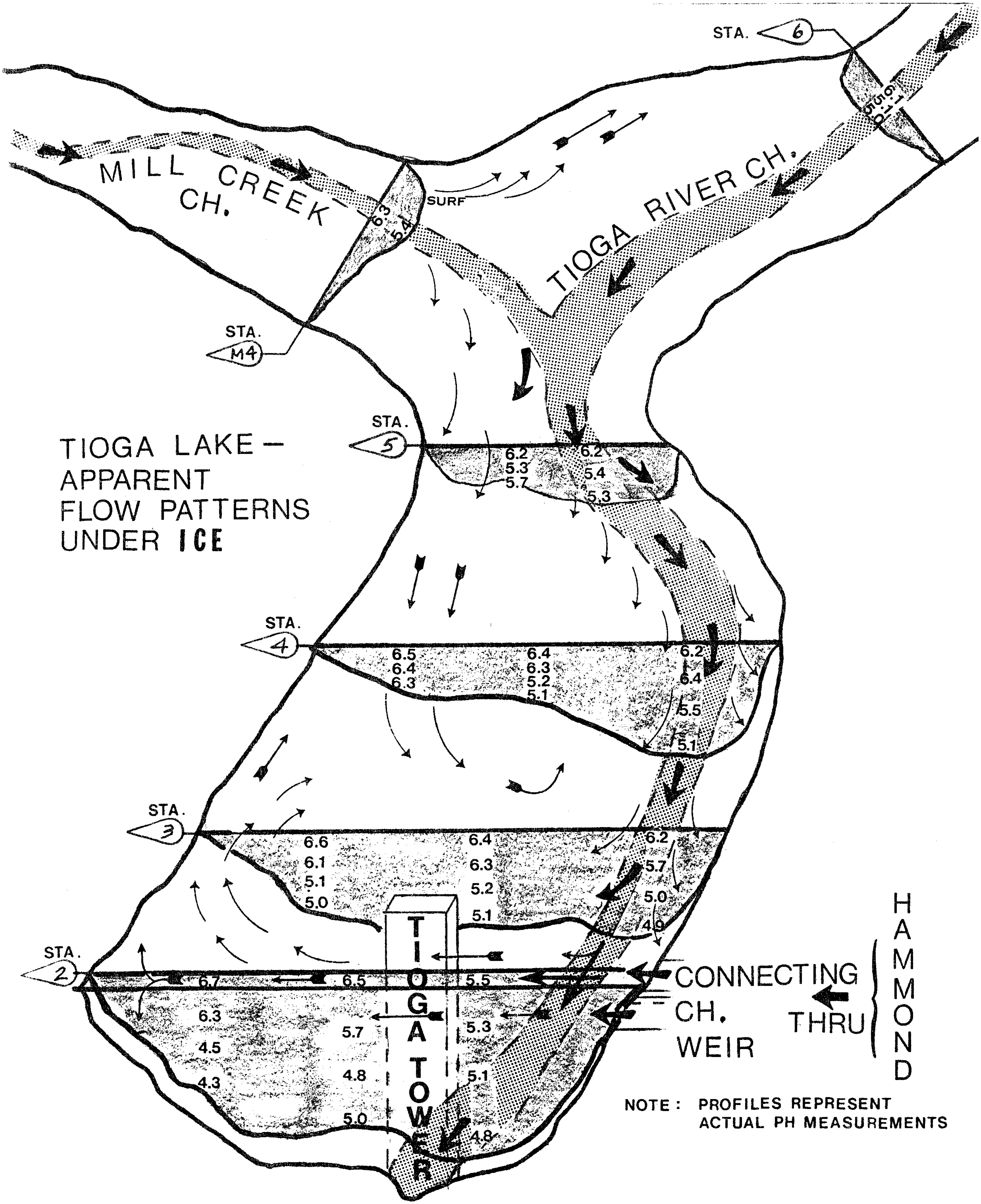
Prior to freeze-up, the lake is completely destratified. A few days after ice forms, the water quality, at depth, begins to decline. This lower quality water accumulates higher and higher toward the surface of the lake. Inflow quality is not significantly changed before or after ice cover. With the increasing accumulation of low quality water, the lower selective withdrawal ports must be closed or partially closed to maintain the downstream quality. As a result of closing the lower ports, the accumulation of low quality water accelerates.

To alleviate the acidic buildup of the Tioga Lake, Hammond water is released into Tioga through the connecting channel's gated wier. This Hammond release into Tioga forms a jet that travels across Tioga Lake and pools on the right bank, essentially out of reach of the Tioga outlet tower, which is located at the end of the Tioga embankment (Figure 1). It appears that with the formation of ice on Tioga Lake, the mainstem Tioga inflow short-circuits through the lake to the intake tower. This flow temporarily blocks the alkaline Mill Creek inflow to the right side of the lake, where it accumulates with Hammond releases (Figure 2). This blockage of alkaline water results in a severely degraded condition along the right side of Tioga Lake and causes acid induced fish kills on the Tioga River downstream of the project. Typically, the fish kills occur when inflows are high (first winter runoff). At this time, the highly acidic hypolimnetic Tioga water apparently surges up the face of the Tioga tower rendering selective withdrawal useless and causing outflow quality to collapse. After a short period of severely depressed quality, the quality at the tower may soar to the point where no acidic water can be found for a short time, then it returns to acidic again. This apparent seiching may continue for several cycles. Once the spring thaw melts the ice, water quality conditions quickly return to normal and operation for quality control is no longer so difficult. It appears that the denial of wind mixing by ice on the lake's surface may be the cause for the situation just described.

This problem was presented to the Corps' Committee on Water Quality. The Committee report is an appendix to this report.



TIOGA - HAMMOND LAKES
PROJECT MAP
FIG. 1


STA. 6
6.1
5.1
5.0
MILL CREEK CH.
SURF
6.3
5.4
TIOGA RIVER CH.
STA. M4
STA. 5
TIOGA LAKE – APPARENT FLOW PATTERNS UNDER ICE
6.2
5.3
5.7
6.2
5.4
5.3
STA. 4
6.5
6.4
6.3
6.4
6.3
5.2
5.1
6.2
6.4
5.5
5.1
STA. 3
6.6
6.1
5.1
5.0
6.4
6.3
5.2
5.1
6.2
5.7
5.0
4.9
STA. 2
6.7
6.5
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6.3
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4.3
5.7
4.8
5.0
5.3
5.1
4.8
TIOGA TOWER
CONNECTING CH. WEIR
THRU
HAMMOND
NOTE: PROFILES REPRESENT ACTUAL PH MEASUREMENTS

FIG. 2

Tioga - Hammond Lakes
Evaluation of Water Quality Problems

1. In response to a request from Baltimore District, five members of the Committee on Water-Quality met with NAB personnel on 14-15 June 1983 to assist the District in evaluation of water quality problems at the Tioga - Hammond Lakes project. The following is a summary of major conclusions and recommendations of this effort.

2. Background. The Tioga - Hammond Lakes project is located in northeastern Pennsylvania. The project consists of two lakes, Tioga Lake on the Tioga River and Hammond Lake on Crooked Creek, which are connected by means of a channel with a control structure. Runoff from the Crooked Creek watershed above Hammond Lake is of good quality and is mildly alkaline in nature. Runoff from the Tioga River watershed above Tioga Lake is adversely impacted at times by acid mine drainage. This acid inflow is believed to have been the cause of minor fish kills in Tioga Lake, following ice formation, during each winter season since the project was completed.

3. Discussion. The group's initial efforts to evaluate the causes of the fish kills included a review of the existing water quality data base. It was determined that insufficient data were available to firmly establish causes of the problem or to make recommendations for specific solutions. The remaining discussion then was devoted to general causes and effects, evaluation of data and study needs, as well as possible remedial measures after additional information was developed.

4. Conclusions and Recommendations. Based on the discussions, the following observations and suggestions are offered.

a. It was concluded that except for periodic under-the-ice events, existing lake regulation appears to be very closely approaching reliable year-round control of acid pollution from the tributary basin of Tioga Lake. While there have been close calls, the District is very effectively protecting the quality of the Tioga Dam outflow, both above and below the confluence of Crooked Creek, without any additional aggravation of the acid pollution problems within Tioga Lake.

b. Since the internal hydrodynamics of Tioga Lake during the under-the-ice acid pollution events are poorly understood and appear to be both very complex and pertinent to the problem, it is suggested that NAB make a major commitment to documenting these conditions. From the previous experiences of the District in this respect, such documentation will likely require sampling at very close time intervals and depth increments for perhaps as long as a month after initial ice formation at Tioga Lake. While all pertinent parameters should be monitored, conservative parameters such as sulfates and specific conductance may be particularly useful indices of internal flows and should be emphasized. The surveys should include sampling at a number of transect stations along the entire length of the lake.

c. Specific recommendations for additional sampling are:

1) Weekly profiles along the vertical near the Tioga outlet works are an essential minimal monitoring effort. Three points (bottom, mid-depth, top) are not enough. Five or more points will give much more useful information.

2) Inflow quality parameters at sufficient frequency to do balance calculations are needed for the Tioga inflow, Mill Creek inflow, Hammond discharge into the Tioga pool, and outflows from the project. Temperature, pH, alkalinity, sulfates and conductivity are the most important. Density differences are influencing flow patterns, so more attention should be devoted to an accurate determination of these variations.

3) A period of detailed monitoring, at least two weeks, is recommended. This effort would be most beneficial if there were a runoff event midway during the monitoring so that time for mixing could be determined for use in arriving at a flood flow regulation strategy. The monitoring should include a cross section in line with the connecting channel, a longitudinal cross section up the center line of the reservoir and a cross section opposite Mill Creek. Again, density profiles would answer some uncertainties on what is going on. It would also be helpful to repeat some profiles at the outlet to determine if seiching is occurring.

d. In addition to lake monitoring, the District should take a closer look at variations in the intensity of acid mine drainage pollution at its source with flow and season. An improved understanding of these relationships could have predictive value that would be useful in regulation of the projects and the data developed may eventually facilitate and help justify reclamation.

e. Because of logistical problems, personnel constraints and unique methodologies involved in such a program, NAB may wish to contract the monitoring to an organization such as CRREL which specializes in this type of work. After completion of this program, it should be possible to more effectively analyze the problem and make definitive recommendations.

5. As a short term measure, it appears that detaining the under-the-ice acid slugs in Tioga Lake for somewhat longer time periods, could improve mixing and neutralization as well as generally improve the ability to control the problem. This would require a periodic compromise of flood control storage in Tioga Lake. However, considering the relatively short time periods and small volumes of storage involved, along with the typical timing of the events during lower flow and low flood risk periods, this detention should be a feasible and acceptable interim control measure.

Earl E. Eiker

Earl E. Eiker
Chairman, Committee on Water Quality

PAINTED ROCK DAM: WATER QUALITY OF AN INTERMITTENT DESERT RESERVOIR

By

Bruce C. Beach[1]

INTRODUCTION

Every project is unique, but some projects are more unique than others. One project paramount in its uniqueness is Painted Rock Dam, in Arizona. The large size and ephemeral nature of the impoundment have created some unique water quality issues at the project.

THE PHYSICAL PROJECT

Painted Rock Dam is situated on the Gila River in southwestern Arizona, near the town of Gila Bend, 120 miles upstream of the confluence with the Colorado River. It is a single purpose flood control project. Capacity at the crest of the ungated spillway is 2,500,000 acre-feet. Surface area at the spillway crest is 53,000 acres. Depth from spillway crest to outlet sill is 131 feet. Selective releases are not possible. No hydropower facilities exist at the project.

The Borrow Pit Lake, created during excavation of material for the embankment, is located about one-quarter mile downstream of the outlet works. Groundwater fed when releases are not being made, the 200 surface acre, 15-feet deep lake is the only permanent body of water within 100 miles. The site of a State Park, the Borrow Pit Lake is a significant source of water based recreation.

HYDROLOGIC ASPECTS OF THE PROJECT

Although the drainage area is over 50,000 square miles, inflow is ephemeral. Most of the watershed is semi-arid, with average annual precipitation ranging from 4 inches in the lower deserts to 30 inches in the mountains. The semi-arid nature of the watershed and the effect of upstream irrigation projects combine to cause flood years to be separated by several years of little or no flow. Average annual runoff is 200,000 acre-feet and median annual runoff is effectively zero. Figure 1 depicts the annual runoff record for the years 1905 to 1983. Because Painted Rock Dam is a single purpose flood control project, the reservoir is fully drained after each impoundment. Each impoundment typically lasts for several months to several years in duration, with little or no inflow occurring between floods, other than a meager bank return.

WATER QUALITY ISSUES

There are three main water quality issues of concern at Painted Rock Dam: salinity, nutrients, and other contaminants.

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SALINITY

The major water quality issue at Painted Rock Dam is the effect project operation has on the salinity of the Colorado River. Downstream of Painted Rock Dam and adjacent to the Gila River lies the 60,000-acre Wellton-Mohawk Irrigation and Drainage District (WMIDD). The District is underlain by highly saline groundwater. When flows on the Gila River occur, the saline groundwater body is recharged,and this causes significant crop losses when the saline water rises into the root zone. To alleviate this, the WMIDD operates relief wells in an attempt to keep the saline water below the root zone. Prior to 1973, WMIDD discharged this highly saline wastewater into the Gila River. This action increased the salinity of Mexico's source of irrigation for 140,000 acres of crop land. In 1973, Minute 242 of the Water Treaty with Mexico was negotiated. It stipulates that the salinity of Colorado River water entering Mexico at Morelos Dam be no more than 115±30 mg/l greater than the Colorado River at Imperial Dam, upstream of the Gila River confluence.

The Colorado River Basin Salinity Control Act was passed in 1973 to facilitate implementation of Minute 242. Among other things, it authorized construction of a drain that now carries the saline wastewater from WMIDD through Mexico to the Gulf of California, bypassing the Colorado River. Figure 2 depicts the relationship between the drain and the Gila and Colorado Rivers. Salinity can still be a factor in water control decisions, although the use of the drain has significantly reduced the effect of project operation on the Colorado River. For instance, flows high in salinity occur on the Gila River when groundwater discharges into the Gila River during periods of falling river stages.

Figure 3 demonstrates the effect the operation of Painted Rock Dam has on Colorado River salinity subsequent to the construction of the salinity drain. Volume of impoundment in acre-feet, outflow in cfs, and salinity at four sites are depicted for the years 1978 through 1981. Salinity, reported as total dissolved solids, is shown at Painted Rock Dam, Gila River at mouth, Colorado River at Imperial Dam, which is upstream of the Gila River confluence, and Colorado River at Northerly International Boundary (NIB) which is downstream of the confluence. The difference between the two Colorado River sites is applicable to Minute 242 requirements.

Notice that the salinity of the outflow was relatively low during the duration of the impoundment, but quite high after draining the reservoir in November 1980. The source of the high salinity water after draining the reservoir is bank return. Salinity of the Gila River at mouth is high prior to the start of significant releases but drops to a level slightly above that of Painted Rock Dam after December 1978. Salinity of the Gila River at mouth rose rapidly after significant releases from Painted Rock Dam ceased. The source of high salinity waters was groundwater flowing into the Gila River during falling river stages. Salinity of the Colorado River at NIB was higher than the Colorado River at Imperial Dam for the period January 1978 through January 1979. Salinity levels then reversed themselves through November 1980, with a spike of high salinity occurring in February 1980 in conjunction with a temporary reduction in outflow from the dam. In November 1980 the salinity levels of the two locations on the Colorado River again reversed themselves with the NIB site higher than at Imperial Dam.

Due to the high desert temperature and an average annual precipitation of six inches, effective reservoir evaporation averages six feet per year. Consequently, salinity of the impoundment can be significantly increased due to the effects of concentration due to evaporation. Figure 4 shows how salinity of the reservoir increased during one impoundment.

NUTRIENTS

Another significant water quality issue at Painted Rock Dam is the presence of high levels of nutrients. Large amounts of hydrogen sulfide gas generation are associated with high nutrient loading. Hydrogen sulfide gas concentrations have approached toxic levels, and have caused extensive damage to electronic equipment in the control house and increased corrosion rates of the gates and concrete of the outlet works. The hydrogen sulfide gas generation apparently occurs when the reservoir is stratified and water is released from the anoxic hypolimnion.

Massive fish kills have occurred in conjunction with the final draining of the reservoir. Twice, an estimated two million fish in the Borrow Pit Lake and one million fish in the reservoir have died (Arizona Game and Fish, Department, Personal Communication). Exact cause of the fish kills has not been identified, but significant water quality parameters that may have been partly or wholly responsible include low dissolved oxygen, rapid temperature change, high suspended solids, high pH, and high ammonia levels. These water quality problems were intensified by extreme overcrowding of the fish community at the time of the fish kill.

Exact sources of nutrients have not been identified, but the most likely source of nutrients is plant growth which occurs in the reservoir lands between impoundments. An exotic plant, the tamarisk, has colonized much of the reservoir bottom lands. It is a very fast growing halophyte and can withstand long periods of inundation. Consequently, every impoundment at Painted Rock Dam is analogous to a first filling at other projects.

CONTAMINANTS

The third water quality issue at Painted Rock Dam is the presence of other contaminants. In addition to salinity, DDT metabolites, iron, manganese, and boron meet the definition of a contaminant (U.S. Army Corps of Engineers, 1983). High levels of DDT metabolites, found in fish flesh at the project and in the watershed, is attributed to continued illegal use of the pesticide (Arizona Department of Health Services, 1983). High levels of iron and manganese occur when the reservoir is stratified or high suspended sediment levels occur. High boron levels are attributed by the author to concentration due to evaporation and have occurred only during insignificant flow rates.

ACTIONS TAKEN TO ADDRESS WATER QUALITY ISSUES

Certain actions have been taken to address water quality issues at Painted Rock Dam and downstream. The most significant action was the construction of the saline water drain previously described. Another action taken was the establishment of the water quality goal of draining the reservoir as fast as possible. This reduces the concentration of contaminants due to evaporation and reduces the amount of hydrogen sulfide gas generation by reducing the duration of impoundment.

In response to low levels of dissolved oxygen observed in the Borrow Pit Lake, an experimental program of surging the outflow was initiated. Normally, turbulence in the outlet works is sufficient to reaerate the outflow before it reaches the Borrow Pit Lake. However, insufficient reaeration occurred when releases were reduced at mid-summer of 1983 due to flooding on the Colorado River. Figure 5 depicts the effect the surging had on dissolved oxygen levels in the Borrow Pit Lake. Dissolved oxygen, outflow, and daily wind speed are plotted for a six day period. A diurnal variation of dissolved oxygen is clearly demonstrated, with little or no dissolved oxygen observed at night, and a day time peak near saturation levels for the very warm water. It should be noted that a dissolved oxygen peak was observed on the first day of the experiment when no surging took place. This leads to the conclusion that the elevated dissolved oxygen levels are due to photosynthesis and wind action, rather than turbulence in the outlet works. One recommendation is to repeat the experiment with a continuous recording of wind speed and sunlight intensity and to perform the surging at night, to better differentiate between the various sources of dissolved oxygen (Arizona Department of Health Services, Personal Communication).

Another water control action, suggested by the Committee on Water Quality, is managing the bottom 30,000 acre-feet of the reservoir to protect the fishery in the Borrow Pit Lake (Anthony, Personal Communication). The final 'dregs' of the reservoir are very poor in quality, with high suspended sediments, high dissolved solids and high nutrient levels. It is at this time that the large fish kills occur. The plan is to reduce the outflow to limit the rate that sediments and nutrients are flushed into the Borrow Pit Lake, to below a rate that the lake can assimilate.

Managing the bottom 30,000-acre feet for water quality has been attempted twice, once in December 1980 and once in October 1983. The December 1980 event still resulted in a massive fish kill, partly because an inaccurate storage elevation curve caused a more rapid draining of the reservoir than was intended. In October 1983 a similar attempt was successful at preventing a fish kill, not because of reduced loading of the Borrow Pit Lake, but because a flood brought an additional 500,000 acre feet of water into the reservoir before the water quality pool was fully drained.

Another action which is contemplated, but has not yet been implemented, would be the installation of stop logs in one of the bays of the outlet works. When the reservoir is drawn down below 30,000 acre-feet, water would be forced to weir over the top of the stop logs, drawing water off the top of the reservoir. This is expected to increase dissolved oxygen and reduce suspended solids in the Borrow Pit Lake. A screen or net may be installed to reduce the number of fish flushed from the reservoir to reduce overcrowding in the Borrow Pit Lake.

SUMMARY

Some unique actions have been taken at Painted Rock Dam to address the unique water quality issues encountered there. Some have been successful, some have not. However, progress has been made. Experience has shown that even at a single purpose flood control reservoir that is empty most of the time, water quality issues can be appreciable enough in magnitude to influence project operation and to require considerable attention.

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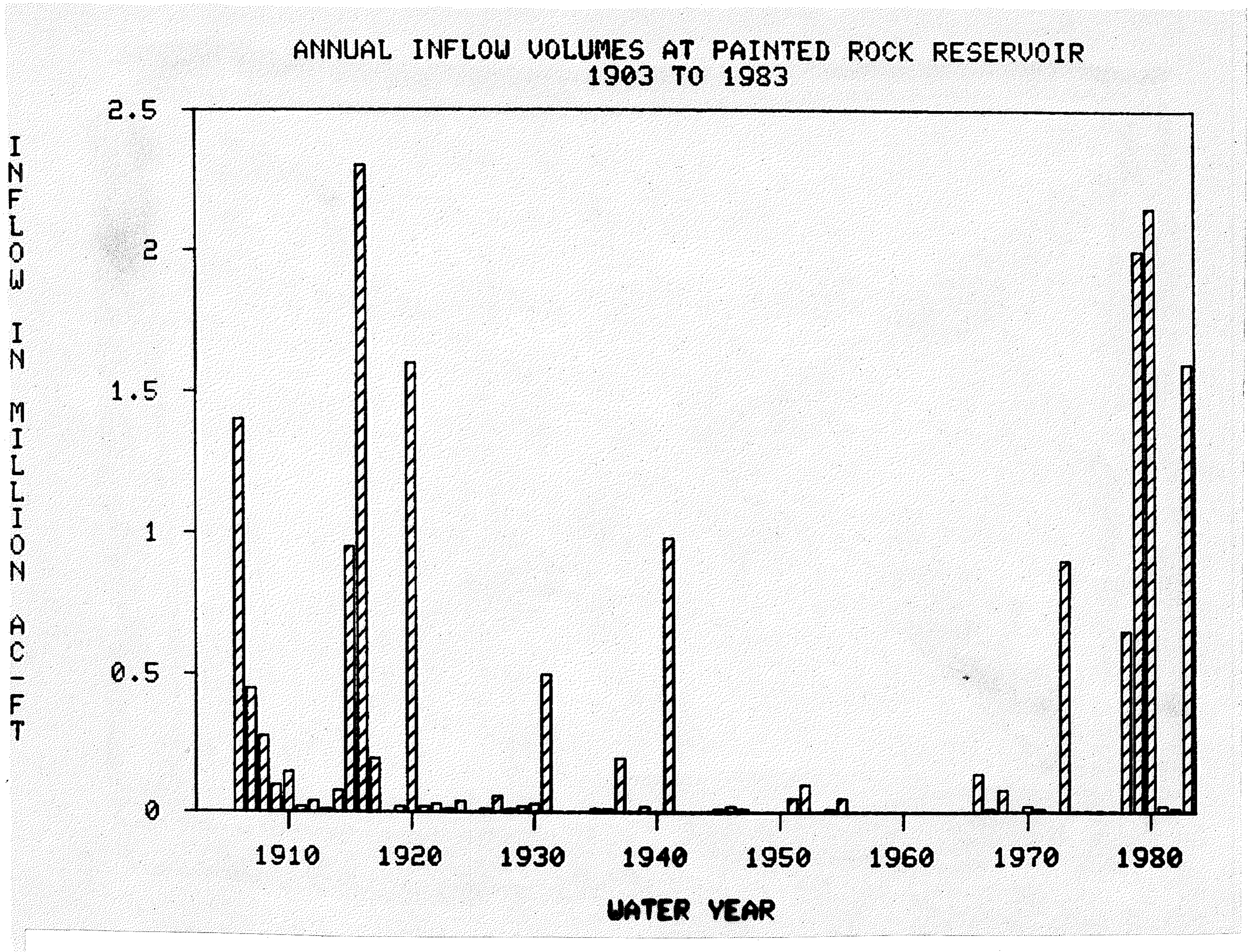

Figure 1. Annual Inflow Volume to Painted Rock Reservoir for the period 1903 to 1983.

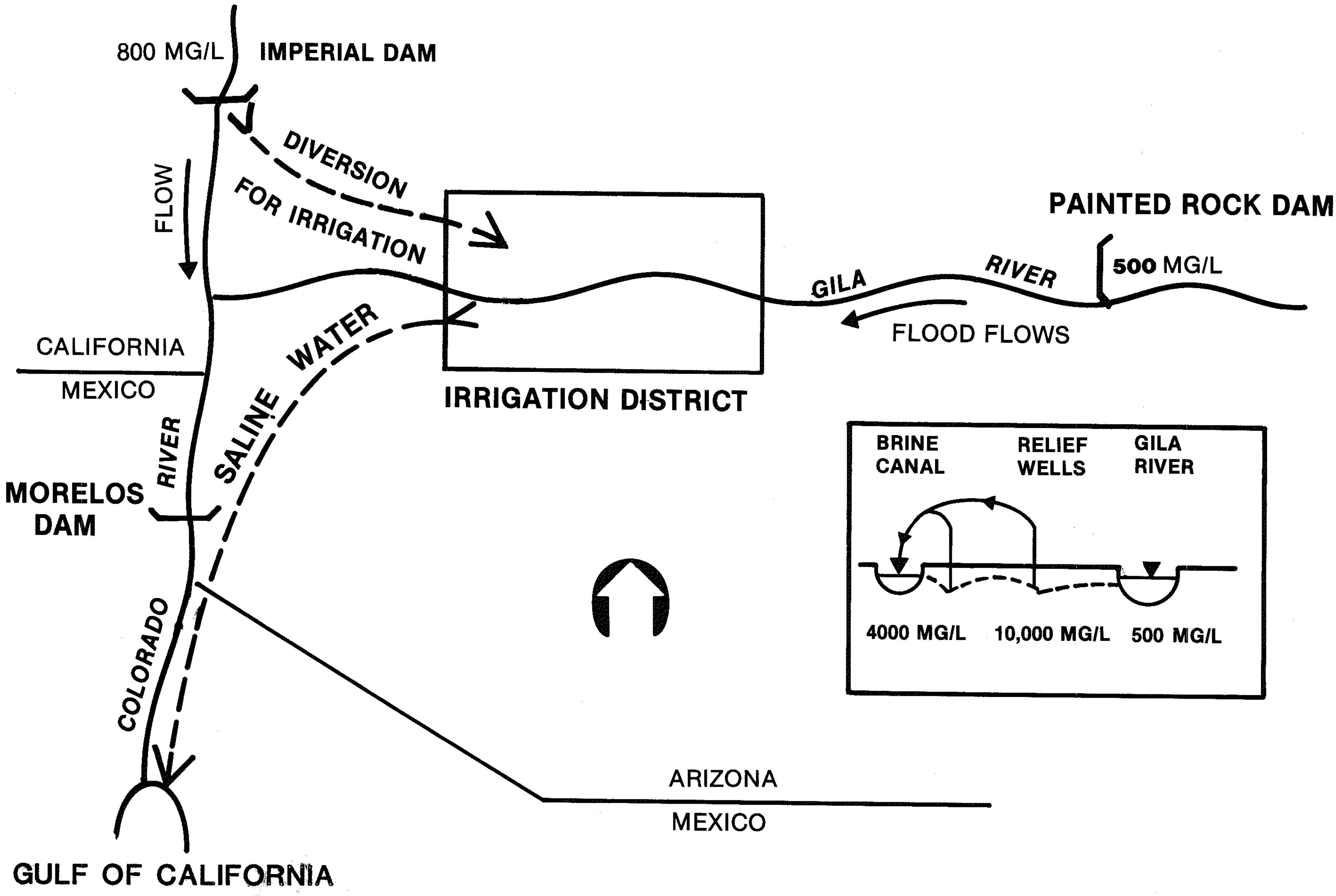

Figure 2. Salinity drain, Wellton Mohawk Irrigation and Drainage District, U.S. and Mexican sources of irrigation water downstream of Painted Rock Dam. Typical salinity values shown.

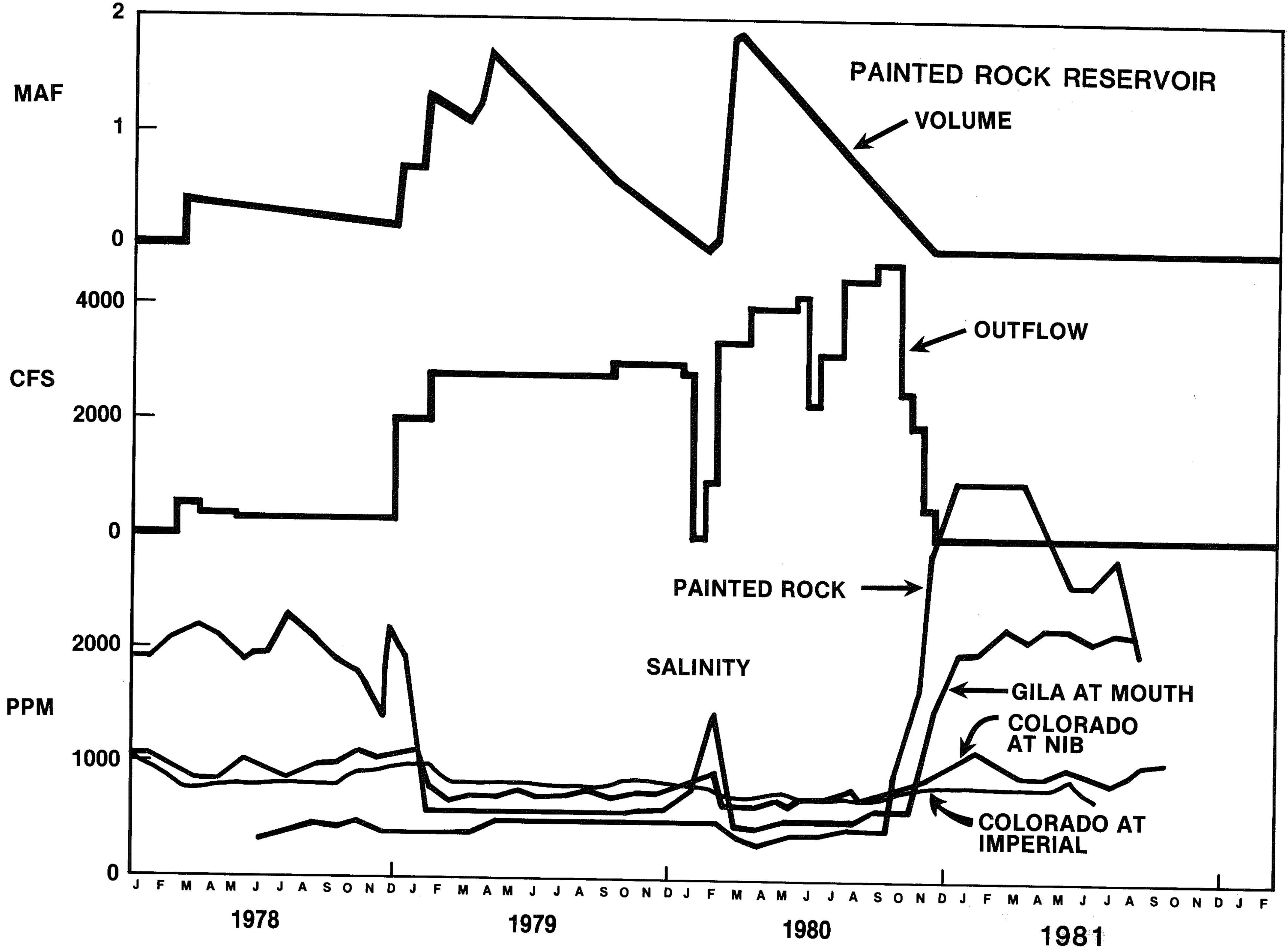

Figure 3. Effects of Painted Rock Dam operation on Colorado River salinity subsequent to construction of salinity drain.

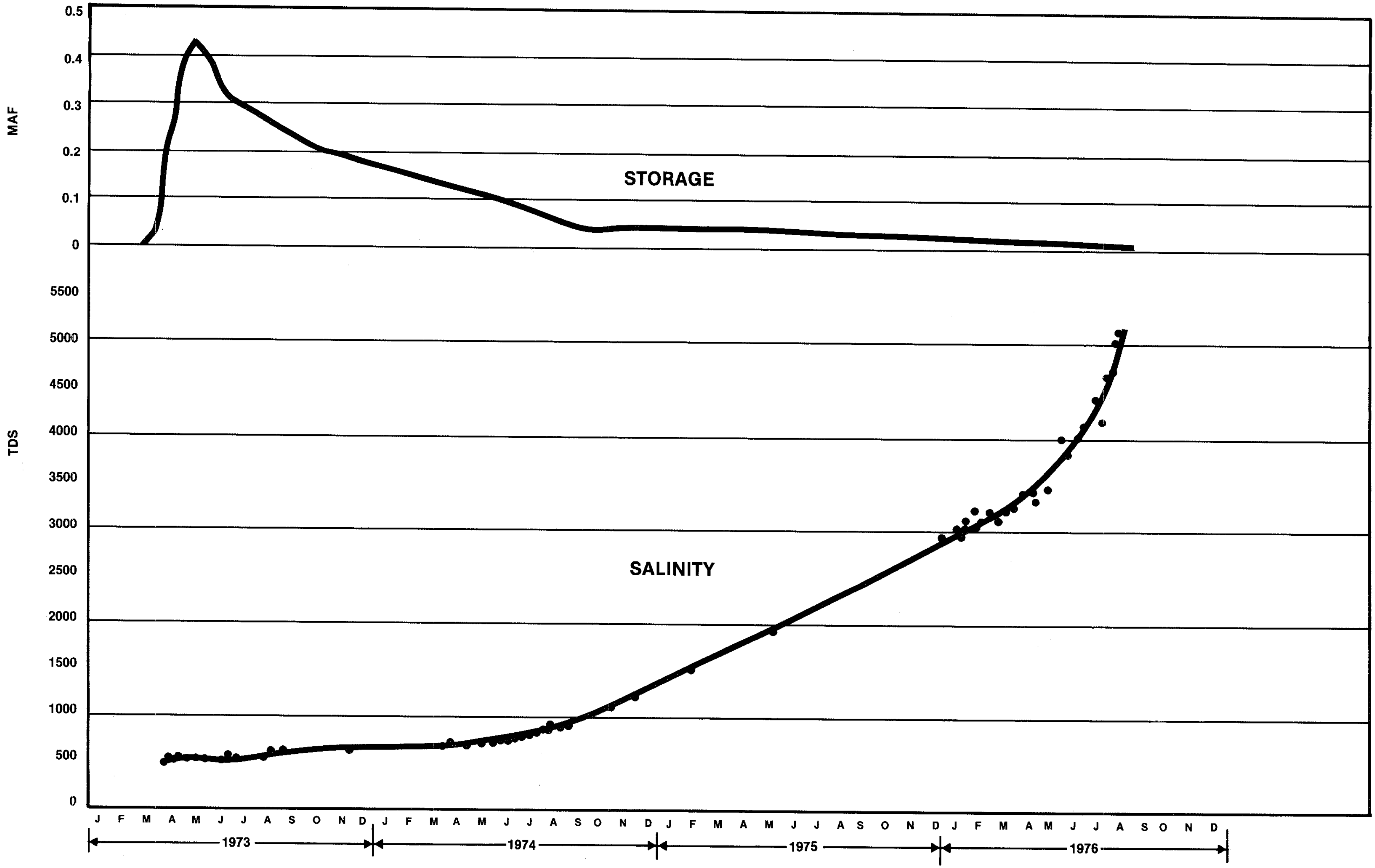

Figure 4. Effects of evaporation on salinity of Painted Rock Dam during a long duration impoundment.

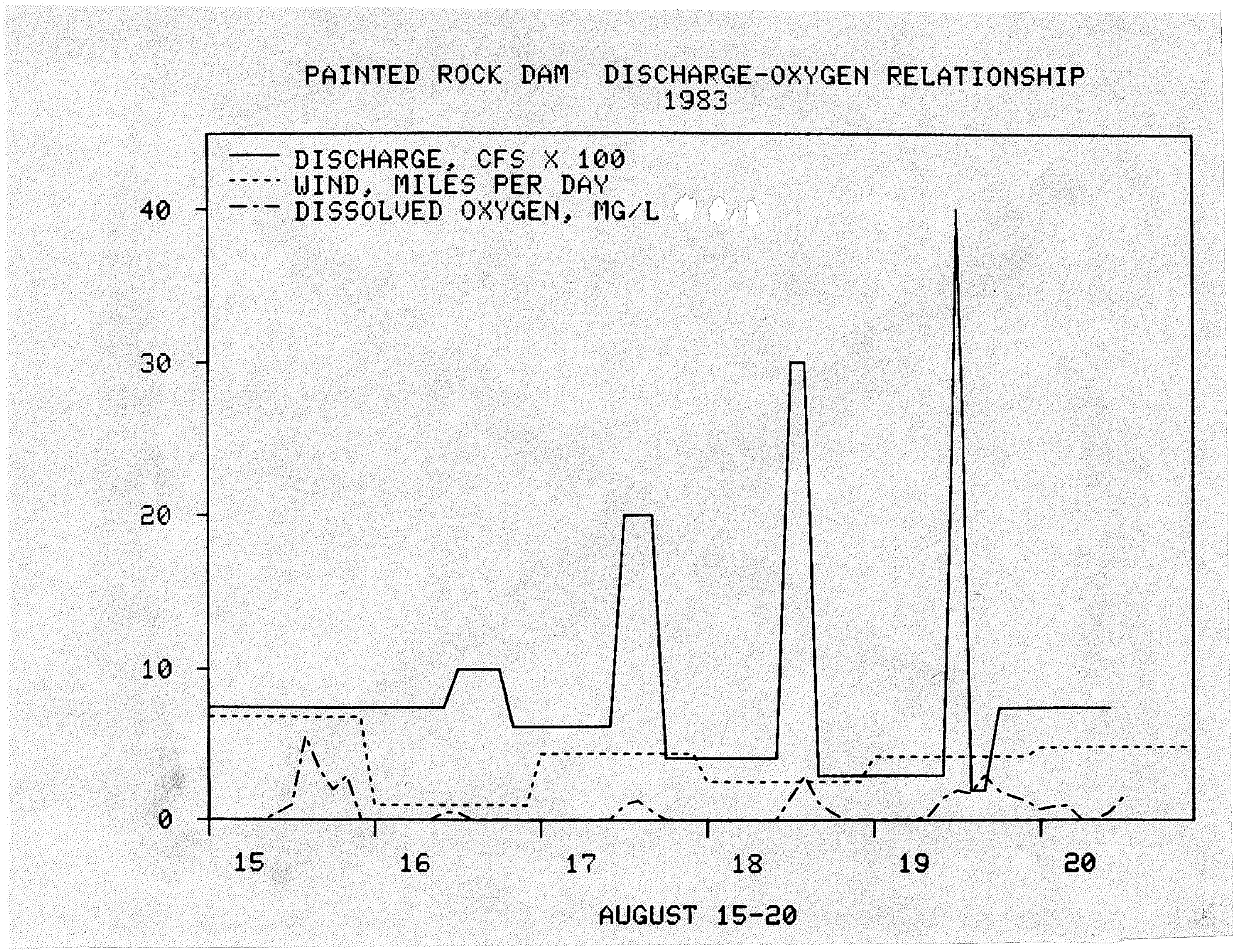

Figure 5. Dissolved oxygen levels in the Borrow Pit Lake compared to average daily wind speed and outflow from Painted Rock Dam.

RICHARD B. RUSSELL DAM AND LAKE OXYGEN INJECTION SYSTEM

By James W. Gallagher, Jr.[1/]

INTRODUCTION

The Savannah District, U.S. Army Corps of Engineers, is currently constructing the Richard B. Russell Dam and Lake project on the Savannah River between Georgia and South Carolina. The Richard B. Russell Damsite is on the Savannah River 37 miles above Clarks Hill Dam and 30 miles below Hartwell Dam. Clarks Hill and Hartwell are both Corps of Engineers projects.

When completed, the Richard B. Russell Dam and Lake will be a multipurpose project designed to provide hydropower, some flood control, recreation, and has a potential for water supply. The dam will consist of a 195-foot high, 1,900-foot long concrete gravity structure flanked by two earth embankments. The project is designed as a peaking powerplant with an installed capacity of 600 megawatts. The powerhouse will contain four 75-megawatt conventional units and four 75-megawatt pump units. This installation will make the project one of the largest Corps of Engineers' hydropower facilities in the nation. During periods of maximum generation the plant will release about 60,000 CFS. During maximum pumpback operation, 30,000 CFS will be pumped from Clarks Hill Lake back into Russell. The average daily release will be over 3,500 CFS. At maximum power pool the Russell Lake will cover 26,650 acres and impound 1,026,000 acre-feet of water.

Work on the dam began in January 1976 with construction of the upstream and downstream cofferdams, and the river diversion channel around the damsite. Based on total contract awards to date, the project is 87 percent complete. The concrete dam and earth embankment are finished. Work on the powerhouse began in September 1981. Power-on-line for the first of the four conventional units is scheduled for August 1984. Power-on-line for the first of the four pump storage units is scheduled for May 1989. The current project cost estimate is 529 million dollars.

INTERAGENCY INVOLVEMENT

During the early planning stages of the Russell project, a major concern of the State and Federal agencies was that the project comply with State water quality standards. The State of Georgia was particularly emphatic on this point, and as a result, the cost sharing agreement between the State of Georgia and the Federal Government for development of the project's recreational areas includes the stipulation that the operation of the project will meet State water quality standards. Besides this commitment to Georgia, a commitment to provide 6 PPM dissolved oxygen in the releases from the reservoir is also explicitly stated in the Statement of Findings for the project filed pursuant to Section 404 of the Federal Water Pollution Control Act Amendments of 1972.

In July 1972, the Georgia Department of Natural Resources requested the formation of a technical committee to analyze the water quality matters relating

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to the Russell project. Milo Churchill, former Chief of TVA's Water Quality Branch, served on the committee along with representatives from the Corps of Engineers and the State of Georgia. The objective of the committee was to evaluate the thermal and dissolved oxygen characteristics of the Russell project as an integral part of the Hartwell-Clarks Hill reservoir system including the following specifics:

a. Maintenance of Federal and State water quality standards.

b. Maintenance of a coldwater fishery in a 10-mile reach downstream from Hartwell Dam.

c. Development of a warm and cold water fishery within Russell Lake.

d. Maintenance of a warm and cold water fishery within Clarks Hill Lake.

With these objectives established, physical and mathematical modeling were conducted to determine whether or not the objectives could be met. The physical model determined travel time, level and thickness of inflows, entrainment and pumpback currents which were then input into a mathematical model which determined the dissolved oxygen and temperature regimes in the lakes and in the hydropower releases.

In its final report, the committee observed that the water quality objectives could be met with the artificial addition of oxygen. Several methods of adding oxygen were then investigated including surface aerators, diffused air injection, spillway aeration, penstock air injection, multilevel penstock intakes, submerged weirs, oxygen injection into the penstocks, side stream oxygenation, localized destratification, pulsed oxygen injection through porous diffusers into the lake at the face of the dam, and continuous oxygen injection through porous diffusers into the lake at a point several days travel time upstream of the dam. With the high oxygen and low temperature constraints, continuous oxygen injection with an on-site Government-owned cryogenic plant was identified as the most feasible alternative. Continuous oxygen injection is favored over pulsed oxygen injection because it avoids the high capital and operating costs associated with liquifying and storing gaseous oxygen.

FIELD TESTS

Between 1975 and 1980, the Savannah District, through contracts with Dr. Richard Speece of Drexel University, conducted field tests of an oxygen injection system at Clarks Hill Lake. As a first step, a small scale system capable of providing sufficient oxygen for the discharge of one turbine was installed adjacent to the dam face at Clarks Hill and operated in a pulsed mode by Speece, et al,[1] in the summer of 1975. This made it possible to rapidly monitor the oxygen level in the discharge and determine the oxygen absorption efficiency immediately.

Three oxygen diffuser racks were constructed. Basically, each rack was a pipe frame which served as a structural support for the individual diffusers, as well as a manifold to distribute the oxygen from the supply hose to each diffuser. The diffuser plates were porous carborundum having an area of 1 square foot each. Ten diffusers per rack were located 10 feet on center. The standard permeability of the diffusers was 2 feet per minute at 2 inches of water pressure. The bubble size generated under these conditions was approximately 2 mm in diameter. The diffuser loading rate was 2,400 pounds of oxygen per diffuser (1 $ft.^2$) per day at 140 feet of submergence. This is equivalent to about 4 actual cubic feet per minute per square foot of diffuser at this depth and oxygen loading rate.

The three diffuser racks were positioned 10 feet from one intake. A bridle of 1/4-inch cable was attached to the rack to facilitate placement. A 1-inch flexible hose supplied gaseous oxygen to each rack from a liquid oxygen tank located on the dam face. Oxygen was metered to each rack and the dissolved oxygen content was monitored at a point in the draft tube beneath the turbine wheel. Lowered oxygen absorption efficiencies were noted when discharges were made from the turbine with its intake 10 feet from the racks. However, marked improvement in oxygen absorption efficiency was noted when discharges were made from the turbine with its intake 120 feet from the racks. It was concluded that maximum oxygen absorption efficiencies were obtained by insuring that the bubbles in the rising plume were not swept prematurely into the penstocks. This required that the diffuser racks be located at least 100 feet horizontally from the intake. Dissolved oxygen concentrations of 6-8 ppm with oxygen absorption efficiencies of 85 percent were obtained with the diffuser racks located at least 100 feet from the intake.

It was concluded from these tests that it was technically feasible to dissolve oxygen in a pulsed mode that was matched to the water discharge rate. However, as mentioned earlier, the recommended method is continuous oxygen injection at an upstream point in the lake rather than pulsed oxygen injection at the face of the dam. Pulsed injection of oxygen to match the water discharge rate involves matching the peaking discharge pattern which normally occurs less than 12 hours each week day and even less on weekends. With on-site cryogenic oxygen being produced in the gaseous state at a uniform rate, compression and storage would need to be provided to match the production with the usage rate. This would increase the capital costs of the oxygen production facility. Therefore, it was decided that field tests should be conducted to evaluate the feasibility of continuous injection into a diffuser system located approximately 1 mile upstream of the dam.

The field tests of the continuous injection system began the next summer[2]. The tests were divided into three phases. Phase I was an evaluation of the oxygen absorption efficiency of various diffusers. The experimental system constructed on the face of the dam consisted of an 11-foot diameter bubble collection hood with off-gas totalizer and analyzer. The diffuser to be tested was lowered to the bottom of the lake. Oxygen was fed to the diffuser at rates of 0.3, 0.6, 1.0, 2.0, and 3.0 actual feet per minute. The bubble collection hood was positioned above the diffuser at heights of 10, 50, and 130 feet. The gas collected from the diffuser through the hood was measured and analyzed for percent of oxygen. In this way, the oxygen absorption efficiency of each diffuser was determined. In this test, diffusers with a standard permeability of 0.5 to 2.0 fpm were identified as the optimum diffusers.

Phase II involved tests of the diffuser racks to determine the elevation in the water column at which the oxygenated water would come to equilibrium. The three diffuser racks from the pulsed injection test previously described, along with two racks constructed to the same specifications, were located about 300 feet in front of the penstock intakes. The racks were aligned roughly parallel to the dam about 60 feet apart. A manifold was mounted on a raft and distributed vaporized liquid oxygen to the racks from the liquid oxygen storage tank located on the face of the dam. Oxygen was injected at rates varying from 57 to 100 pounds per minute, and temperature and dissolved oxygen profiles were taken in the forebay while monitors on the seven draft tubes recorded the dissolved oxygen variations in the turbines. The data revealed that 80 percent of the oxygen remained in the hypolimnion, accessible for downstream release, when injected at these rates.

Phase III of the study involved installation of nine diffuser racks at a location approximately 1 mile upstream of the dam in water approximately 130 feet deep. Due to the availability of the diffusers, 10.0-fpm diffusers were installed in these racks rather than diffusers in the 0.5 to 2.0 fpm range identified as optimum in the Phase I study. The diffuser racks were placed in a semicircular pattern with a spacing of about 100 feet on center. Again, vaporized liquid oxygen was fed from a liquid oxygen storage tank through a manifold mounted on a raft to the oxygen diffuser racks. Oxygen was injected at a rate of 100 tons/day which would have been sufficient to provide a dissolved oxygen concentration of 6 ppm in the average discharge from Clarks Hill Dam. Temperature and dissolved oxygen concentrations were recorded at fixed sampling stations within the lake between the dam and the diffuser rack location, and also 1/4 mile upstream from the diffuser racks where background profiles were obtained. The automatic dissolved oxygen monitoring system which sampled water passing through the turbines was also operated during this phase of study. Oxygen was injected continuously for a period of 8 days after which time the lake began to destratify, terminating the test. The highest dissolved oxygen concentration recorded in the turbines was 4.1 ppm which occured about 6 days after oxygen injection started. The background dissolved oxygen before oxygen injection commenced was 0.5 to 0.8 ppm. Only about 30 to 40 percent of the oxygen that was injected appeared to eventually reach the turbines. The low oxygen absorption efficiency was due to two factors. First, as stated earlier, the diffusers on the rack were not the most efficient as determined in Phase I of the study. Second, the close semicircular spacing of the diffuser racks and high injection rates per diffuser caused localized destratification in the vicinity of the diffuser racks which resulted in the dissolved oxygen-rich water coming to equilibrium in the upper level of the lake where it was unavailable for dissolved oxygen enrichment of the turbine discharges. It was determined that improvements in the performance of the oxygen injection system could be realized by lowering the injection rate per diffuser by quadrupling the number of diffusers per rack, equipping the racks with the optimum 2 fpm diffusers, and spreading the racks across the lake cross section.

These improvements were made to the system and field tests were conducted in the summer of 1977[3]. The nine racks were fitted with 40 square feet of diffusers of 2 fpm standard permeability. The racks were placed across the lake cross section one mile upstream from the dam and spaced approximately 300 feet apart with the first rack located approximately 1,200 feet from shore. The manifold from the test of the previous year was eliminated and a 3-inch diameter oxygen supply hose was floated on the surface of the lake and guyed with anchors. This supply hose delivered vaporized liquid oxygen from the storage tanks on the shore to the diffuser racks via 1-inch diameter hoses. Oxygen was injected continuously for 30 days at a rate of 100 tons/day, and dissolved oxygen and temperature were monitored in the lake and the turbines. During this period of oxygen injection, dissolved oxygen concentrations of 4 to 5 ppm were maintained with an absorption efficiency of 50 percent. Although this represented an improvement over the results from the previous year, the goal of 6 ppm dissolved oxygen was still not achieved and the absorption efficiency was still unacceptable. Although the lake did not destratify in the vicinity of the racks, pumping of the oxygenated water occurred causing it to reach the surface where it warmed and returned to an intermediate layer generally above the turbine withdrawal zone. It was determined that the pumping was due to the four-sided diffuser configuration of the racks and that the pumping could be eliminated by installing baffles over the diffuser racks to deflect the oxygenated plume or by employing a linear diffuser configuration.

The results of the 1977 test led to the small scale tests performed between July and September 1978 to evaluate the effects of plume deflectors, loading rate, and diffuser configuration on the performance of the oxygenation system(4). During these tests, oxygen was injected each day for an 8-hour period from 12 midnight to 8 a.m. through a "ribbon" diffuser 40 feet long and 1 foot wide located perpendicular to the face of the dam. To identify the zone where the oxygen-enriched water came to equilibrium, dissolved oxygen profiles were taken at 0, 100, 200, and 400 feet from the diffuser rack in the morning before the turbines began discharging. To compare the effectiveness of this modified diffuser configuration with the four-sided square configuration used in the 1977 test, a diffuser rack used in the 1977 study was also installed and operated at the face of the dam. The oxygen loading rates applied to the diffusers were 125, 250, 375, 500, and 2,000 pounds per square foot of diffuser per day. At the beginning of the tests, it was discovered that the 250 and 500 pound per square foot per day loading rates were optimum; therefore, the majority of the testing was conducted in this range. Three different baffle configurations were evaluated: a 1-inch diameter PVC pipe grid with 1/2-inch clear spacing, 1 1/2-inch wood slats with 1 1/2-inch clear spacing, and 6-inch wood slats with 1-inch clear spacing. All of the plume baffles consisted of a frame 6 feet wide and 40 feet long and were evaulated at 25, 50, and 75 feet above the diffuser. Primarily, the data collected from these tests indicated that the linear diffuser configuration is superior to the four-sided diffuser configuration in depositing the oxygenated water in the withdrawal zone. The most effective plume deflector was the PVC configuration. However, generally the effects of the baffles on the behavior of the oxygenated plume were minimal.

RICHARD B. RUSSELL OXYGEN INJECTION SYSTEM

The oxygen injection system under construction at the Richard B. Russell project is described in the Richard B. Russell Dam and Lake Design Memorandum 35 and Supplement No. 1 (5 and 6). The system at Russell will have a continuous injection system located 1 mile upstream of the Russell damsite but will also have supplemental injection capability at the face of the dam to be used during periods of higher than normal releases and unusually high dissolved oxygen deficits. Gaseous oxygen will be supplied from a facility on the lakeshore.

The continuous system will involve constructing a distribution pipe from the oxygen facility to two parallel diffuser pipes suspended 5 feet off the lake bottom. The two diffuser pipes will be over 1,600 feet long and will be spaced 100 feet apart. For the pulsed system, a main distribution line will extend from the oxygen supply site to the top of the dam. Additionally, eight feeder pipes will extend from the main distribution line down the face of the dam and then be connected to the diffuser lines between each intake perpendicular to the face of the dam. Each feeder pipe for the pulsed system will be equipped with a motorized control valve which will allow operation of any combination of pulsed diffuser lines. Power and telemetry for the motorized control valves will be provided from the oxygen supply facility by means of underground cables.

The diffuser piping will consist of an 8-inch center manifold pipe made of fiberglass reinforced plastic. Flotation will be attached to the manifold pipe to provide a constant positive buoyancy to the system during shutdown. Vertical and horizontal alignment of the system will be secured by guying the manifold pipe to concrete anchor blocks on the lake bottom with stainless steel cables. Flanged to the manifold pipe will be 20-feet sections of 4-inch schedule 80 PVC diffuser pipe. The diffusers will be spaced 1 foot apart along this pipe. A control orifice will be installed in each flanged connection between the manifold pipe and the diffuser pipe to insure proper flow distribution through this system. The diffusers will be 7 inches in diameter and are made of silica glass bonded together with an organic binder. The diffusers will have a standard permeability of 2 FPM.

Based on the best estimates currently available on the expected dissolved oxygen content of the reservoir releases, the expected daily discharge from the project and the expected oxygen absorption efficiency, approximately 5,500 tons of oxygen would have to be added annually at a maximum rate of 150 tons/day to meet the downstream dissolved oxygen objective of 6 PPM in the hydropower discharges that would occur 90 percent of the time. For the first few years of operation, oxygen will be purchased from commercial suppliers, stored in liquid oxygen storage tanks on the site, and then vaporized as needed. Although ultimate plans are to install an onsite oxygen production facility, purchased liquid oxygen will be used initially to gain detailed data on project performance and oxygen requirements. After this period, we will be in a better position to determine both the desirability of an onsite production facility and the type of facility that will be most economical to operate.

The estimated cost of the oxygenation system is approximately 2.6 million dollars: $490,000 for the oxygen storage facility and 2.1 million dollars for the distribution and diffusing system. The cost of purchasing liquid oxygen has been budgeted at $100/ton. The diffusing system is currently under construction and the contract for the oxygen storage facility is scheduled to be awarded in February 1983 so that the complete oxygen injection system will be ready for operation during the first stratification period beginning in May 1984.

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MOBILITY OF TOXIC MATERIALS IN SURFACE RUNOFF AND INTO PLANTS

By

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INTRODUCTION

Various aspects of water quality have been discussed as it is related to Corps projects. This paper will give some insight to the how's and why's of the water quality observed at dredged material disposal sites. The most important factor that influences water quality and mobility of contaminants from dredged material into the environment is changes in the physicochemical nature of the dredged material. These physicochemical changes that occur after dredged material is placed in a disposal environment are the keys to predicting contaminant mobility. Environmentally acceptable management of dredged material can be formulated and implemented only after due consideration has been given to the physicochemical nature of the disposal site environment.

In relation to aquatic disposal of dredged material, very little change has been observed in the physicochemical nature of dredged material placed in an aquatic environment. That is to say, a reduced sediment is removed from the bottom of a waterway and placed under water in a similar reduced environment. Under these disposal conditions contaminant mobility at the aquatic disposal site will be very similar to that occurring in the waterway.

Disposal of that same sediment in an upland environment, however, can have drastically different effects on the mobility of contaminants. This paper will discuss what happens to contaminants (heavy metals) in dredged material placed in an upland disposal environment and cite two examples of test results that will emphasize the importance of the physicochemical nature of dredged material.

SURFACE RUNOFF RESEARCH

First let me discuss some of the research being conducted at the WES under the FVP that addresses the quantification of contaminant mobility in surface runoff when contaminated dredged material is placed in an upland environment (Lee and Skogerboe, 1983). Dredged material was collected from Black Rock Harbor, Bridgeport, Conn. The reduced dredged material was placed in a soil bed lysimeter at a depth of 25 cm. A simulated acid rainfall event of 5 cm/hr for 30 min was applied to the soil bed using the WES rainfall simulator. Surface runoff water was sampled through the rainfall event. Water samples were analyzed for conductivity, pH, suspended solids and contaminants. The dredged material was allowed to dry and another simulated rainfall event was applied as before. This procedure was repeated until the dredged material dried to 5% moisture.

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Results

Suspended solids concentrations in surface runoff water ranged from 10,057 to 46,326 mg/ℓ when the rainfall event was applied to the initially wet sediment (58 percent soil moisture). A crust formed on the surface of the sediment as the drying process occurred. Eventually large cracks formed randomly across the soil bed lysimeter. Suspended solids concentrations in surface runoff water decreased significantly to a range of 252 to 627 mg/ℓ when sediment moisture reached 5 percent. It was necessary to apply the rainfall event for 60 minutes in order to obtain a constant rate of surface runoff after the cracks filled with water. The conductivity of the surface runoff water increased slightly from a range of 540-1,510 to 4,380-6,200 μmho/cm after sediment drying. The pH of the surface runoff initially was 8.5-8.6 for the wet sediment and decreased to pH 6.3-6.8 when the sediment had dried to 5 percent moisture.

As the amount of suspended solids in the surface runoff decreased with sediment drying, a concomitant decrease was observed in the total nitric acid digestible heavy metals in unfiltered water samples (Table I). The most significant observations in this study were those of dissolved heavy metals found in surface runoff after sediment drying. While the surface runoff from the wet sediment contained high concentrations of heavy metals in the unfiltered water samples, an extremely small amount of these metals were in the dissolved or soluble form. For example, unfiltered runoff water contained 800 or more μg/ℓ total Cd for the initial wet sediment. Only 10.5 μg/ℓ of dissolved Cd was found in these runoff water samples. The remainder of the Cd was sediment and particulate adsorbed and essentially insoluble. However, a significant increase in the dissolved form of the metals Cd, Zn, Cu, Ni, and Mn was observed after the sediment had dried out as shown under filtered runoff in Table I.

Essentially all of the Cd observed in the surface runoff from dried sediment was soluble or dissolved Cd. These results agree well with the published literature on the effects of sediment redox and oxidation on the transformation and solubility of heavy metals (Gambrell et al. 1977; Gambrell et al. 1978; Folsom, Lee and Bates, 1981). The other heavy metals Cr, Pb, As, and Hg did not show any change in dissolved concentrations in surface runoff upon sediment drying. Soluble iron showed a decrease in surface runoff after the sediment dried out.

As the sediment dries out, suspended solids concentrations decrease in surface runoff; however, concentrations of soluble Cd, Zn, Cu, Ni, and Mn increase in surface runoff water. These results reflect the physicochemical changes occurring when the dredged material dries out.

PLANT BIOASSAY RESEARCH

A second example which also illustrates the importance of physicochemical changes in dredged material on contaminant mobility is a solid-phase plant bioassay test of the same dredged material (Folsom and Lee, 1983). In this

TABLE I. Effect of drying on surface runoff water conductivity (μmho/cm), pH, and concentrations of heavy metals (μg/ℓ) from sediment placed in an upland disposal environment.

Parameter	Simulated Rain	Sediment Moisture Initial - 58%	Sediment Moisture Dry - 5%
		Unfiltered	
Conductivity	4	540-1,510	4,380-6,200
pH	4.2	8.5-8.6	6.3-6.8
Cd	2.4	790-1,830*	95-165
Zn	<50	35,400-91,800	950-1,830
Cu	12	67,200-168,400	920-1,860
Ni	<10	4,600-11,600	150-450
Mn	<6	7,820-18,300	280-460
Cr	<10	38,400-106,900	600-1,050
Pb	<10	9,430-26,400	200-300
Fe	67	157,000-2,045,000	11,000-21,000
As	<5	78-210	<5-6
Hg	<1	6-128	0.7-22.7
		Filtered	
Cd	2.4	2.0-10.5	95-140
Zn	<50	<50-77	290-580
Cu	12	37-72	96-154
Ni	<10	11-27	80-120
Mn	<6	22-57	148-204
Cr	<10	13-25	<10
Pb	<10	<10-13	<10
Fe	67	116-298	30-70
As	<5	<5-11	<5
Hg	<1	<1	<1

* Each range consists of at least 9 surface runoff water samples collected during the storm event.

test a plant is used as an indicator of contaminant mobility and bioavailability. One portion of the dredged material from Black Rock Harbor was placed in the double bucket apparatus of the plant bioassay. The reduced dredged material was maintained in a flooded reduced condition by keeping a 5 cm depth of water over the dredged material throughout the test. Another portion of the dredged material was allowed to air dry in drying flats in the greenhouse. When air dried, the dredged material was ground into a coarse powder and placed in the double bucket apparatus. Water was added to the dredged material only to meet the needs of the index plant grown in it. After 120 days of growth the plants were harvested, weighed and analyzed for contaminants.

Results

Sediment physical and chemical parameters are listed in Table II. Total amounts and DTPA extractable heavy metal concentrations are listed in Table III. The total sediment content of heavy metals (Table III) was typical for that of contaminated saltwater sediments with the exception of copper which was much greater (Folsom, Lee and Bates, 1981).

TABLE II: Selected Physical and Chemical Parameters of Black Rock Harbor Sediment

Organic Matter, %	18.7
Salinity, ppt	25.3
Conductivity, dS/m	35.7
$CaCO_3$ equivalent, %	1.0
pH wet	7.6
pH reconstituted air-dried	6.6
Oil and Grease, mg/g	5.3
Total Sulfur, %	1.3

TABLE III: Total Acid Digestible and DTPA Extractable Concentrations of Heavy Metals in Black Rock Harbor Sediment

Heavy Metal	Concentration, μg/g			
	Total Acid Digestible	DTPA Extractable		
		Original Sediment		Washed Sediment
	Original Sediment	Flooded	Upland	Upland
Zn	1264	1.73	765	1017
Cd	16.7	<0.0005	22.0	24.7
Cu	2377	<0.005	701	235
Cr	1346	0.18	1.45	1.62
Pb	330	0.01	14.3	23.2

DTPA extractable heavy metals from the BRH sediment is also listed in Table III. The data show that air-drying resulted in increased heavy metal extractability. Washing the sediment before air-drying had only a slight increased effect on DTPA extractability of the heavy metals. The DTPA data would predict plant uptake of heavy metals to be greater from the air-dried, upland sediment compared to the original, flooded sediment.

Contents of the heavy metals in the index plant, *S. alterniflora*, grown in original sediment under both flooded and upland conditions and in upland, washed BRH sediment, are presented in Table IV.

TABLE IV: Plant Content of Heavy Metals in Leaf Tissue of *S. alterniflora* Grown in Sediment from Black Rock Harbor

Heavy Metal	Concentration, μg/g		
	Original Sediment		Washed Sediment
	Flooded	Upland*	Upland
Zn	13.0	219	341
Cd	0.04	0.91	4.65
Cu	3.77	18.7	36.2
Cr	0.01	0.93	2.79
Pb	0.39	1.53	2.53

* Only one replicate supported plant growth.

Spartina alterniflora grew well and had low heavy metal contents under the flooded conditions. These results are typical for contaminated saltwater sediment placed under a flooded condition and compared well with levels observed in natural saltmarshes (Simmers et al. 1981). *Spartina alterniflora* did not grow well in the original, air-dried (upland) sediment; only one plant of one replicate survived. Decreased plant growth resulting in increased metal content could explain the elevated heavy metal content (Table IV) of the plant. Plants grown in washed sediment under an upland condition grew much better than the unwashed upland condition. However, heavy metal content of the plants was much greater compared to that of plants grown in the flooded condition. This same effect has been shown by Folsom and Lee (1981) to occur with freshwater plants grown in freshwater sediments under flooded and upland disposal environments. Apparently, once the saltwater sediments are washed free of excess salt and plant growth occurs, the air drying process results in increased availability of heavy metals. Removing excess salt from the sediments by washing simulates the natural salt leaching process and can be used in a modified Saltwater Solid-Phase Plant Bioassay to predict contaminant mobility into plants on upland saltwater dredged material disposal sites.

CONCLUSIONS

These test results illustrate clearly the effects of physicochemical changes on contaminant mobility from dredged material. Management of contaminated dredged material can be accomplished in an environmentally acceptable manner if consideration is given to the physicochemical changes that might occur at a dredged material disposal site. Once these are known, appropriate management strategies can be formulated and implemented to minimize contaminant mobility.

One management strategy that could be employed to reduce soluble metal concentrations in surface runoff and in plants is to keep the sediment wet to prevent sediment oxidation. Another strategy could be the surface application of limestone to maintain a pH of neutrality and to adsorb and precipitate soluble metals from runoff water. A third strategy could be to cover the contaminated sediment with a cleaner sediment so that surface runoff waters will not carry contaminants to the discharge weir.

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APPLICATION OF WATER QUALITY MODELING TO ESTUARIES

By

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INTRODUCTION

Estuaries such as San Francisco Bay and Delta are extremely complex hydraulic systems. The general public and the decision-makers are demanding more precise and accurate predictions on the impacts associated with water resources development. Not only do we have difficulty evaluating what is occurring in models and how the model relates to the prototype, we have difficulties in the pure analysis of data from the models. This paper presents information on the technical upgrading of the San Francisco Bay-Delta physical model and discusses several key points which the author feels must be considered in model test program design.

SAN FRANCISCO BAY-DELTA MODEL

The San Francisco Bay-Delta model is a physical, hydraulic model which includes about seventeen miles of the Pacific Ocean and all reaches subject to tidal influence in San Francisco Bay and the delta of the Sacramento and San Joaquin Rivers. The original model of the ocean and bay was constructed in 1956-57. The delta portion was added to the model in 1966-69. The scales on the model are 1:1000 horizontal, 1:100 vertical and 1:100 time. The model covers an area about the size of two football fields.

Data collection on the model had been a manual operation. Water level was determined by use of point gages with a reading from a vernier. Salinity was determined from water samples collected by squeezing and releasing rubber bulbs to suction the water from a predetermined depth. Samples were then analyzed on a Beckman meter. Velocity was measured by counting the revolutions of a pigmy meter. Direction was determined by observing the flow of dye. All of the data was processed manually before any analysis could be started. The system was manpower intensive, especially with 24-hour operation over a four or five day period.

TEST DESIGN PROBLEMS

As the demand for more precision and accuracy increased, confusion and variation in data led to a decrease in model credibility. The problems arose because of the combination of the data collection method and several factors in the design of test programs.

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Test programs have been, and are now, set up as formal, rigid programs emphasizing data as data and not evaluation. Very few opportunities are presented to "play" with the model as a basis for designing a test program. Quite often test results were presented only as numbers without any evaluation of what the numbers were saying. Without this evaluation, translation to the prototype cannot be given.

The emphasis in test programs was on quantitative data collection with little or no acknowledgement of qualitative observations to compliment the data for evaluation. The manpower intensity of testing definitely was a drain on the qualitative observations.

Test programs emphasized detailed data collection which did not consider macro and micro changes (by location and time) that influence the water flow. To develop a test program, to include both macro and micro influences, some information on what is occurring in the system must be known either from "playing" with the model, from previous tests on the model or from observations of the prototype.

Opportunities to broaden an information base are generally ignored by funding and time constraints to restrict the testing only to the problem at hand. Often times model tests are redone because original scopes were too restricted. Other tests can benefit by broadening the information base of the whole system.

MODEL UPGRADE

In February 1983, the operation of the first stage of the upgrade was initiated. The first stage was the automation of the data collection.

Data Acquisition System

Data acquisition is now accomplished by a Hewlett Packard (HP) 1000/65 minicomputer. The system functions as an on-line process control system, outputting control signals, collecting data, processing data, and providing on-line monitoring of model operations. System capabilities also include generating report quality plots, narratives, and tables. Back-ups of all programs and raw data are on magnetic tape.

The Hewlett Packard system combines real-time data gathering capability with a secure multi-user operating environment. Events or processes requiring real-time response are given guaranteed priority in scheduling and execution of programs. The data acquisition task is given highest priority, with all other users sharing the remaining computational resources on either a priority or time-slice basis.

The minicomputer system is made up of the HP 1000/65 mainframe containing the central processing unit, the operating system, system memory, and the necessary interfaces with all peripheral devices. The system supports Fortran, Pascal, and Basic high level languages, along with all software needed to create, edit, compile, load and execute programs. Peripheral devices on the system include a 16-megabyte disc drive for system operations programs, a 64-megabyte disc drive for all user programs and data, 9-track magnetic tape drive for data/program backups, and four control processors for signal input/output. Other devices include a matrix line printer, letter quality daisywheel printer, graphics plotter, remote terminals and system control.

Model test data are acquired by the Hewlett Packard Model 2250 and 2240 measurement and control processors. The HP 2250 and 2240 send and receive all signals through special function cards which allow analog input, analog output, digital input, digital output, and various counting functions. All model data collection (and in the future, tide control) is accomplished by the HP 2250. Data are obtained from all stations at approximately 22-second intervals, stored in a data buffer until requested by the HP 1000, then stored on magnetic disc. The HP 2240 measurement and control units are also used to collect discrete sample analysis data from Beckman Conductivity Meters for the HP 1000. These data are also placed on disc files.

Model Instrumentation

Model parameters currently acquired by the HP 1000 system include water surface elevation, X-Y components of current velocity, electrical conductivity and temperature of the water. The conductivity/temperature probes used at the Bay Model are manufactured by the Montedoro-Whitney Corporation. They utilize a miniature sensor to minimize disturbances to the flow field. The sensor consists of two electrodes excited with a constant AC voltage. The water path between the electrodes acts like a variable resistance dependent on the value of the water conductivity. The signal is output to the front LED display panel and to the computer. Simultaneously, measurements of water temperature are obtained via a linearized thermistor. A software program converts temperature and conductivity into salinity in parts per thousand (ppt) for the calibration curve on each instrument.

The water velocity probes were also manufactured by the Montedoro-Whitney Corporation. These sensors operate on the principle in that water flow distorts an electromagnetic field set up by the electrodes and yields a measure of the flow intensity. Transaxial placement of two sets of electrodes gives a measure of the flow in two directions. Output is the magnitude of the north/south and east/west components of flow velocity. Vector addition is used to obtain the resultant magnitude and direction of the tidal flow.

Water level changes in the model are recorded using a device that converts distance into DC voltage. The electrical capacitance formed between the probe and the water surface is held constant. Internal electrodes maintain this constant capacitance so that a change in the water surface elevation appears as a DC voltage change at the output.

Data Analysis Software

Software is available to retrieve data, convert raw data to prototype equivalent and compute basic statistics such as mean, standard deviation, percent deviation, minimum/maximum values and lunar day averages. Future statistical analysis routines will include harmonic and spectral analysis, analysis of variance, and a variety of curve fitting programs.

Plotting software has been developed to give a visual presentation of salinity, flow velocity and water level time histories for all stations. Other software includes storage and retrieval of discrete sample data, word processing and comparisons of data from different tests.

Tide Generation

The second stage upgrade involves conversion of the tide generator which is controlled by mechanical cams to control by the minicomputer in February 1984. The mechanical system is restricted to repetitive tides. The computer controls will permit not only the repetitive tides but also lunar, solar and storm variations. Computer analysis of three water level detectors for the ocean boundary will guide the computer in tweaking the operation of the valves. The tide generator works on a continuous flow system which includes a 75hp pump forcing water into the model on a valve controlled line and gravity outflow on a slide gate controlled line. The new system not only allows for the variation in tides but also gives tighter controls and reduced operation costs.

TEST DESIGNS

Four studies will be used to illustrate program test design as presented in the aforementioned problems.

The repeatability study was the first study to utilize the new data acquisition system. The study consisted of three sets of seven tests in steady state conditions. With statistics applied to the massive amount of data, the model was shown to repeat and that, at some stations, changes as low as 10 ppm salinity could be detected. The tests generally ran 270 lunar days with 40 data points collected during each lunar day. The instruments provide an essentially continuous record of boundary and station conditions.

Any problems or variations can be evaluated in terms of other factors. As an example small variations in the tide operation were analyzed in terms of a positive or negative salinity wave occurring in the system. Regression analysis gives us the expected salinity variance, if any, as associated with small changes in the tides.

In the Fall of 1983, tests were conducted for the Federal Emergency Management Administration to verify previous tests conducted in the Fall of 1982. The purpose of the tests was to determine salinity changes associated with the failure of delta islands. In some of the tests, unexpected decreases in salinity occurred. The secondary data on water levels were necessary to provide credence to the tests. The large open area of the island flooding reduced the amplitude and the phasing of the tides in the complex area of sloughs in the delta. This study illustrates the necessity of secondary data and analysis of micro changes in both area and time.

In December 1983, the model was called on to assist the Coast Guard in a body search. The critical factor in the test was the interface area between the saltwater and the high delta outflow freshwater penetrating from the north bay into the south bay. No previous data was available from the model or the field. Extensive observations, however, had been made from the downtown office overlooking the central bay. Based on the field observations, drogues were introduced about six inches apart at the lower high water interface. If the body had been lost in the freshwater, it would travel north of Alcatraz Island, outside the Golden Gate, and into the north eddies. In the saltwater it would move south of Alcatraz Island, outside the Golden Gate, and into the south eddies. Building a broad base of information has to be a long term goal in the operation of the model.

In January 1984, detailed current studies were made in the central bay just inside the Golden Gate. Three different tides were run, each with two different levels of freshwater flows. The density of the stations monitered was increased by mounting eight instruments for four stations each with top and bottom probes. The package of probes was relocated each two lunar cycles to grid the area. To interpret the data, colorful jet balls on the surface and on the bottom were placed in the model. A video camera was lifted into the rafters to record, not only the video of the general circulation patterns of the balls, but also the audio of the computer announcement of time and the comments by the investigators on the movement of the balls both within and outside of the study area. The video tape will be used to analyze the hundreds of rose plots, present study findings to groups, and broaden the information base on the central bay system.

CONCLUSION

The upgrade of the San Francisco Bay-Delta Model was a necessary event for the continuous use of the model. As the years continue, more and more is going to be expected from the model. The upgrade by itself is not the total answer for the model. Evaluation of test results must be emphasized not only for understanding the data but also to translate the results to the prototype.

ACKNOWLEDGEMENT

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PROBLEMS ASSOCIATED WITH THE OPERATION OF THE TIDE GATE - SEDIMENT BASIN PROJECT, SAVANNAH HARBOR, SAVANNAH GEORGIA

by Gary V. Mauldin[1]

INTRODUCTION

Savannah Harbor is located on the South Atlantic Coast, partly in South Carolina and partly in Georgia. The inner harbor is formed by the lower 21.3 miles of the Savannah River. The Savannah River's headwater tributaries rise in the Blue Ridge Mountains of North Carolina and unite to form the main stream which flows 297 miles to the city of Savannah. The Savannah River is tidal for approximately 50 miles, from its mouth to Ebenezer Landing (Figure 1).

When James Oglethorpe first sailed up the Savannah River in 1733, the inner harbor was approximately 10 feet deep at low water and was in a state of equilibrium with respect to the amount of sediment entering the system and the ability of the tides and currents to carry the material to the ocean. However, with the advancement of steam powered, ocean going cargo vessels, ships became larger and larger and required deeper and deeper channels.

The Corps of Engineers' first attempt at providing a deeper channel in Savannah Harbor began in 1826 when Congress appropriated $50,000 to remove obstructions from the channel. Later attempts by the Corps consisted of removing snags and obstructions, closing side channels to force more water through the main channel and constructing dikes to provide greater depths over shoal areas. These improvements provided a channel depth of approximately 20 feet.

Ships continued to grow larger and by the early 1920's it became necessary to dredge the inner harbor to a depth of 26 feet to provide adequate channel depth. Further deepening of the channel occurred in 1939, 1953, and 1969, when the inner harbor was dredged to 30 feet, 34 feet, and 38 feet, respectively.

RESULTS OF HARBOR IMPROVEMENTS

The present channel depth of 38 feet, which is about 28 feet deeper than the "equilibrium" depth, has resulted in tremendous growth in commercial usage of the river. As of 1981, the latest year figures are available, the port of Savannah was the third largest port in the Southeast behind Norfolk and Jacksonville, with an annual tonnage of 12,707,864.

Not all of the impacts caused by harbor improvements were beneficial. Shoaling in the inner harbor increased from 2 1/2 million cubic yards per year to 7 1/2 million cubic yards per year. Further, the location that most of this shoaling occurred moved upstream away from the larger disposal areas.

Savannah Harbor, being a partially mixed estuary, has a salt water - fresh water interface, commonly called a salt water wedge. At this salt water wedge a chemical reaction occurs which changes the calcium clays suspended in the fresh water to sodium clays which tend to flocculate and settle to the bottom. This reaction causes approximately 5 million cubic yards of silty shoal material to settle in the harbor annually. Prior to the channel improvements of 1953, the salt water - fresh water interface was located approximately 5 miles downstream

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of downtown Savannah, where adequate disposal areas were nearby. After deepening the channel to 34 feet in 1953, the salt water - fresh water interface moved upstream to the vicinity of downtown Savannah away from adequate disposal areas.

Studies were conducted during the early 1960's to develop plans for reducing the rate of shoaling in Front River. The result of these studies revealed two alternatives for reducing shoaling: either divert the flow of fresh water away from Front River thereby eliminating the source of the shoal material, or move the salt water - fresh water interface downstream to an area with nearby disposal areas. The first alternative was determined to be impractical due to the potential adverse environmental impacts. Alternative 2, moving the salt water - fresh water interface, was then studied in greater detail by the Waterways Experiment Station and a recommended plan was developed. The major features of this plan consist of a sediment basin, 2 miles long, 600 feet wide, and 40 feet deep; a tide gate structure with 14 gates, 43 feet wide and 22 feet high; and a drainage canal connecting Back River with Front River (Figure 2). Construction of the project began in 1970 and was completed in May 1977.

The tide gates are free-swinging, tidally influenced. During flood tide, the force of the incoming water opens the gates and they remain open until high water slack, when the weight of the gates cause them to close. The water in Back River, downstream of the tide gate structure, slows significantly, causing the suspended material in the water column to settle out in the sediment basin. The water trapped upstream of the tide gates must travel through the drainage canal connecting Back River with Front River and ebb out Front River. This additional volume of water in Front River increases the water velocity which prevents the development of major shoals in the navigation channel.

RESULTS OF TIDE GATE - SEDIMENT BASIN OPERATION

Since May 1977, when the tide gate structure was completed and placed into operation, the Tide Gate - Sediment Basin project has operated as designed. Over 4 million cubic yards of sediment, approximately 55 percent of the total inner harbor shoaling, is now deposited in the sediment basin. Because of nearby disposal areas, dredging in the sediment basin instead of in Front River cuts the cost of dredging by approximately 80% for a savings of $1 million dollars or more annually.

The sediment basin has operated as designed in removing suspended solids from the water column up to the point the sediment basin becomes 40 percent full. The trapping efficiency of the basin drops significantly after the basin becomes 60 percent full. Maintenance dredging of the sediment basin is scheduled twice yearly to maintain a relatively clean basin and high trapping efficiency.

The Tide Gate - Sediment Basin project has increased the maximum ebb velocity in Front River from 5.5 feet per second to 6.5 feet per second, which has practically eliminated buildup of shoal material in this reach.

PROBLEMS ASSOCIATED WITH OPERATION OF THE TIDE GATE - SEDIMENT BASIN PROJECT

The success of the Tide Gate - Sediment Basin project has not been without problems and controversies. Problems identified during the design, construction, and operation of the project are the following:

- Increased salinity levels upstream of the tide gates
- Decreased striped bass spawning grounds
- Increased salinity levels in the shallow groundwater upstream of the tide gate structure
- Increased water velocity in Front River
- Increased duration in high water slack
- Increased height in low water slack

a. Increased Salinity Upstream of Tide Gate Structure. Located approximately 5 miles upstream of the tide gate structure is the Savannah National Wildlife Refuge (SNWR). The refuge includes 26,000 acres with 3,000 acres of fresh water impoundments managed for migratory water fowl. The availability of fresh water is essential for proper management of these impoundments. During the design of the Tide Gate - Sediment Basin project, higher salinity levels upstream of the tide gate structure were predicted and mitigative measures constructed. Among these were a series of canals and control works to ensure that adequate fresh water would be available to the SNWR and to private landowners located along Back River. Specifically, these mitigative measures consisted of 4 miles of channel improvements in Little Back River, 8 1/2 miles of fresh water canals, 11 water control structures, and 8 water distribution structures (Figure 2). The salinity levels at the intake to the fresh water canals are monitored continuously with a salinity recorder to insure the availability of adequate fresh water for use by the SNWR and the private landowners.

b. Reduction of Spawning Areas in Back River. The Savannah River is the major source of brood fish for the Richmond Hill Fish Hatchery which supplies striped bass for stocking reservoirs throughout Georgia. Both the Game and Fish Division of the Georgia Department of Natural Resources and the U.S. Fish and Wildlife Service (USFWS) have expressed concern that operation of the tide gate structure adversely affects the striped bass spawn. According to the USWFS Habitat Suitability Index Model,[1] traditional striped bass spawning grounds are reduced 50 percent due to increased salinity levels in Back River with the tide gate structure in operation. However, studies conducted by Dr. Richard G. Dudley of the University of Georgia[2] indicates that spawning still occurs in Back River but the spawning ground has shifted upstream.

Also, there are concerns that operation of the tide gate structure exposes young striped bass eggs to higher salinity concentrations than without the tide gates operating. The scientific literature concerning the effect of salinity on striped bass egg survival is somewhat inconclusive. However, the striped bass is a rather adaptable fish and different races of striped bass have different requirements.

Another concern is that the operation of the tide gate structure exposes more eggs to industrial pollution in Front River by flushing the eggs out of Back River through the drainage canal. Operation of the tide gate structure may, however, reduce the number of young eggs exposed to industrial discharges since spawning occurs farther upstream. During operation of the structure, eggs passing through Front River would be older eggs, or larvae, and least susceptible to adverse impacts from toxics.

c. Increased Salinity in Shallow Groundwater. To monitor the shallow groundwater in the vicinity of the Tide Gate - Sediment Basin project, ten pairs of shallow (less than 25 feet deep) groundwater wells were established in 1972 and have been monitored monthly (Figure 3). To date, the data indicates no major increase in the salinity levels of the shallow groundwater from operation of the tide gate structure. This is likely due to the low permeability of the surrounding soil and the twice daily change in the hydraulic gradient caused by the semidiurnal tidal cycle.

d. Increased Water Velocity in Front River. The increased velocity in Front River caused by the added volume of water being diverted from Back River into Front River scours the channel bottom preventing any accumulation of shoal material. However, these higher velocities have also aggravated scouring of the river banks especially in the lower reaches of Front River. The area around Fort Jackson, located at the junction of Front and Back Rivers, required the construction of a retaining wall and riprap protection after installation of the tide gate structure. The bank along the Corps of Engineers Depot has also experienced significant scouring.

e. Time of High Water Slack Extended. When the tide gates close at high water slack, the water trapped upstream must flow through the drainage canal connecting Back River with Front River and ebb out Front River. This causes a delay in the beginning of ebb tide which extends the duration of high water slack upstream of the tide gate structure. During 1983 extreme high tides caused overtopping of several private landowners' dikes. The longer high water slack duration caused by the tide gate structure increased the volume of water that overtops the dikes. The Corps of Engineers is currently involved in a study to determine if this problem can be eliminated.

f. Increased Low Water Slack Heights. During the design process, concerns were raised on whether proper drainage of the surrounding areas would still be available after the installation of the Tide Gate - Sediment Basin project. Experience to date indicates this anticipated drainage problem has not occurred.

CONCLUSIONS

The Tide Gate - Sediment Basin project has been in operation since May 1977 and has operated as efficiently as predicted. Dredging 4 million cubic yards of shoal material annually from the sediment basin with nearby disposal areas instead of dredging this material from the navigation channel in Front River saves approximately $1 million dollars annually. The major environmental problems associated with construction and operation of the project were predicted in the design phase and effectively mitigated. There are a few minor problems discussed earlier that we continue to monitor for compliance.

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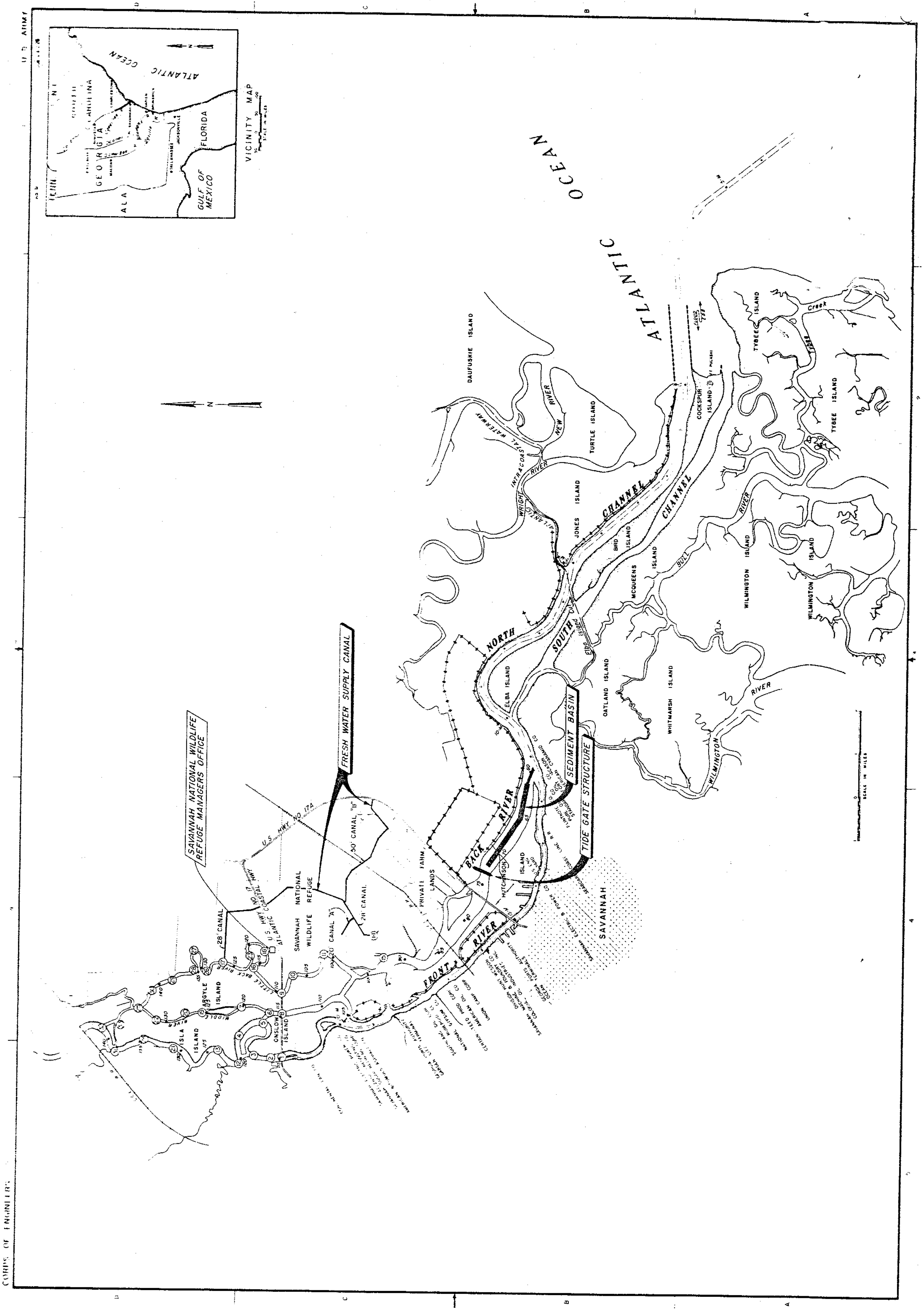
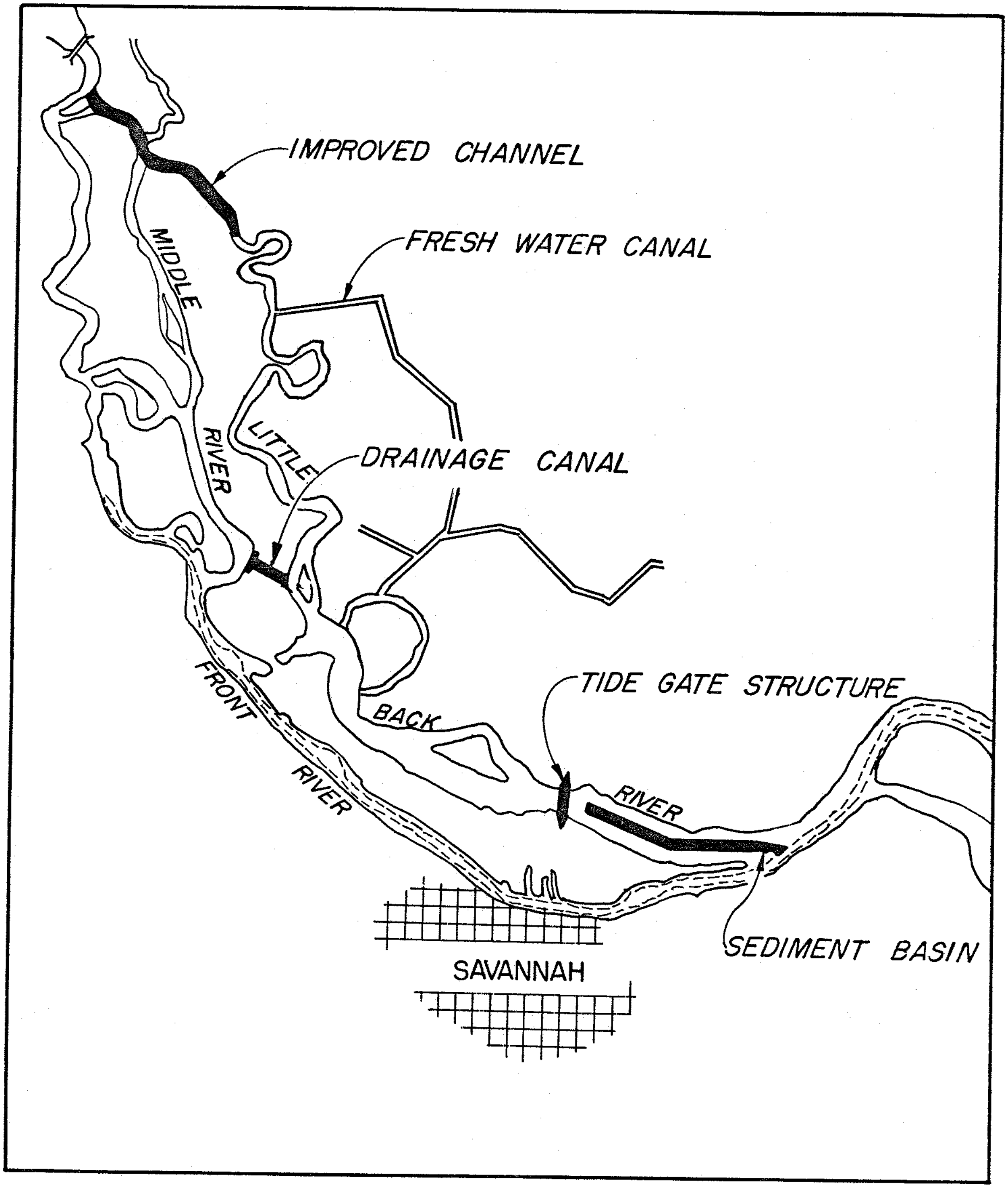


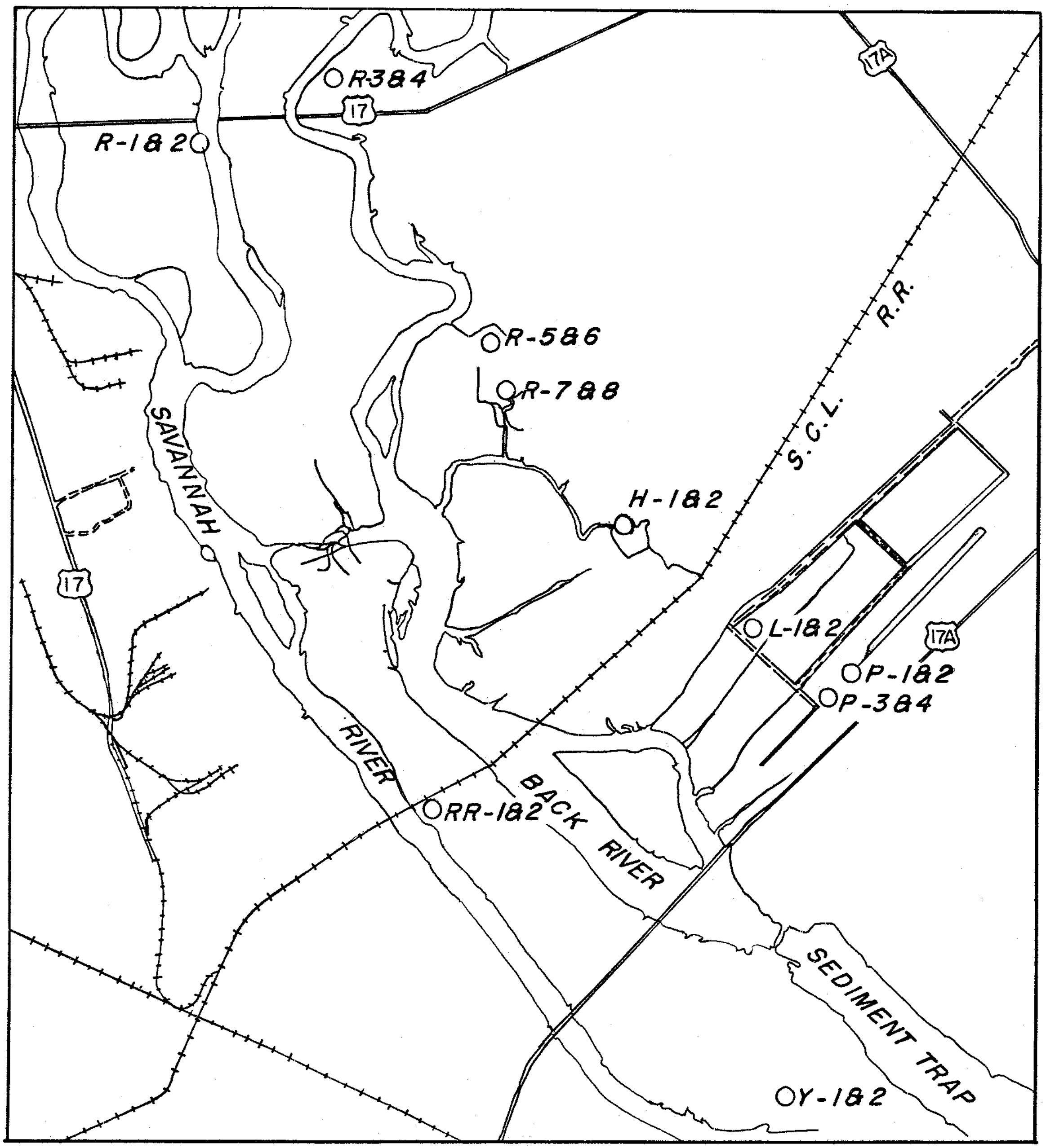
U.S. ARMY
CORPS OF ENGINEERS
VICINITY MAP
ATLANTIC OCEAN
GEORGIA
FLORIDA
ALA
GULF OF MEXICO
ATLANTIC OCEAN
SAVANNAH NATIONAL WILDLIFE REFUGE MANAGERS OFFICE
FRESH WATER SUPPLY CANAL
SEDIMENT BASIN
TIDE GATE STRUCTURE
NORTH CHANNEL
SOUTH CHANNEL
BACK RIVER
FRONT RIVER
LITTLE BACK RIVER
MIDDLE RIVER
NEW RIVER
WRIGHT RIVER
BULL RIVER
WILMINGTON RIVER
INTRACOASTAL WATERWAY
SAVANNAH
SAVANNAH NATIONAL WILDLIFE REFUGE
U.S. HWY NO 17 ATLANTIC COASTAL HWY
U.S. HWY NO 17A
PRIVATE FARM LANDS
28' CANAL
DAUFUSKIE ISLAND
TURTLE ISLAND
JONES ISLAND
ELBA ISLAND
BIRD ISLAND
MCQUEENS ISLAND
OATLAND ISLAND
WHITMARSH ISLAND
WILMINGTON ISLAND
COCKSPUR ISLAND
TYBEE ISLAND
HUTCHINSON ISLAND
ARGYLE ISLAND
ONSLOW ISLAND

Figure 1



MAJOR FEATURES OF TIDE GATE
SEDIMENT BASIN PROJECT

Figure 2



SHALLOW GROUNDWATER WELLS
SAVANNAH HARBOR AND VICINITY

Figure 3

CHEMICAL AND BIOACCUMULATION STUDIES AT TIMES BEACH DREDGE DISPOSAL AREA, BUFFALO, NEW YORK

By Richard P. Leonard*

ACKNOWLEDGEMENT

The design, field sampling, greenhouse studies, and chemical analyses for the plant, animal, and chemical investigations described in this paper were performed by personnel from the environmental laboratory of the Corps of Engineers Waterways Experiment Station (WES). Special appreciation is extended to Dr. John Simmers who was the principal WES investigator for the Times Beach studies.

INTRODUCTION

In order to maintain safe and economical navigation for the commercial shipping industry using the Buffalo Harbor, navigation channels are dredged on an annual basis. This material is principally fine-grained erosional sediments comprised mainly of silts and clays. Because the drainage basin providing sediments to the harbor channels receives appreciable influents from various industries (chemical, steel, petroleum refining) as well as agricultural runoff, the sediments contain significant organic and inorganic contaminants and nutrients (i.e., oil and grease, heavy metals, nitrogen, phosphorus, pesticide residues).

Prior to 1967, dredge sediments from Buffalo Harbor were dumped into an open lake disposal area located offshore of Bethlehem Steel Company near Smokes Creek. As a result of extensive study of dredge disposal problems in the Great Lakes and concern regarding the effect of dredge disposal on aquatic life, the Army Corps of Engineers decided to build confined disposal facilities (CDF's) for sediments dredged from polluted harbors, including Buffalo Harbor.

One such area was constructed in 1971 near the Coast Guard Station at the mouth of the Buffalo River. This 46-acre area, known as Times Beach, received dredge material from 1972 to 1976. This area has been only partially filled because of the recognized great ecological significance, particularly for birds, and value of the aquatic/terrestrial plant landscape which has evolved successionally over time. Approximately 550,000 cubic yards of dredge material had been placed in this facility at the time disposal was discontinued.

At the present time, approximately one-half of the 46-acre site is shallow open water with depths ranging from 1 to 7 feet. Although there is exchange of water between the lake and the disposal area through the permeable dikes, there is no direct discharge from the disposal area to the lake, or direct entry from the lake. About half of the remaining 23 acres is occupied by wetland vegetation including cattails, rice cutgrass, purple loosestrife and minor inclusions of sedges and rushes. More highly elevated portions of the

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site are occupied by trees, shrubs, and herbaceous vegetation including cottonwood, dogwood, golden rod and ragweed.

Although the Times Beach disposal area is observed to have lush vegetative growth and is commonly visited by many species of migratory waterfowl, possible impacts of the polluted sediments on the site ecosystem was not evaluated prior to this study. The purpose of this study, therefore, was to conduct limited ecosystem studies to ascertain if the contaminants contained in sediments disposed at the Times Beach confined disposal facility migrate to the water column or are biomagnified in the terrestrial food chain.

METHODS AND MATERIALS

This study involved the investigation of 4 types of media: sediments, plants, earthworms, and water. Methods of investigation for each of these media are discussed individually.

Sediments - Since this study focused on the possible movement of contaminants from Times Beach dredge sediments into water and the extent to which these contaminants could be taken up by plants and earthworms, bulk chemical analyses of sediments was necessary to establish baseline contaminant levels and to select chemical parameters for measurement in the other media.

Sediments were sampled from the same transects where the field plant uptake studies were conducted and, therefore, represented the chemical nature of the test sediments in the plant uptake investigations. Sediments were sampled to a depth of 6 inches along the transects. A total of 16 samples were taken, stored under ice and shipped to the Army Corps of Engineers, Waterways Experiment Station (WES) for chemical analyses.

Plant Uptake Studies - The objective of this phase of the study was to determine if plant uptake of contaminants from the Times Beach dredge disposal material was greater than uptake from that measured from natural wetland soils in the Lake Erie Basin. To this end, yellow nut sedge (Cyperus esculentus) was planted in sediments at the Times Beach site, allowed to grow to maximum vegetative growth (45 days), harvested, and shipped to WES for chemical analysis. Four transects consisting of 8 embedded buckets per transect (four of test sediment and four of an uncontaminated WES reference soil) were arranged across the disposal site from a wet-flooded situation in cattails to the drier elevated upland containing rice cutgrass and shrubs. Only above ground vegetation was harvested. Screening was placed around and on top of the planted sedge to keep large insects (grasshoppers) and animals out while the planted sedge plants were becoming established.

Earthworm Studies - The Times Beach dredged material used in these studies was collected from the locations of the plant uptake studies. Studies were conducted in buckets at the WES Environmental Laboratory. Thirty grams of common "red wiggler" earthworms (Eisenia foetida) were placed in each bucket and kept in a growth chamber for 28 days. The technique employed was patterned after that of the European Economic Commission developed by Dr. C. A. Edwards. Sixteen buckets of dredged material were used as well as

4 buckets of a WES reference soil. The reference soil was used only as an index to evaluate the environmental conditions in the growth chamber. Prior to chemical analysis, the worms were washed and placed on moist paper toweling to depurate for 24 hours. Contaminant concentrations in the worms exposed to the dredged material were compared to concentrations in the worms from the same lot from the supplier prior to exposure to the dredged material, and to concentrations in the dredged material.

Water Quality Studies - Samples of water were obtained from four locations within the open water of the confined dredge disposal area. Samples were taken at mid-depth with a ponar sampler, stored under ice, and shipped to the WES for chemical analyses. Analyses were conducted on the total water samples and after filtration through a 45-micron filter. This was done to ascertain if suspended matter in the water column contained pollutants to a significantly higher degree than that dissolved in the water column.

RESULTS AND DISCUSSION

The results of the testing and experimental programs will be discussed for each environmental medium and comparisons made between the media. Table 1 compares the levels of metals found in the sediment to those found in the open water. EPA maximum allowable levels of contaminants for drinking water are also given in Table 1. Table 1 shows that although there are elevated levels of metals in the sediments, little solubilization into the water column has occurred . The concentrations of metals found in the dike water were well below maximum allowable levels under the Safe Water Drinking Act.

Table 2 gives the levels of organics found in the sediments and water. Although a variety of organics (phthalates, PCB's, polynuclear aromatic hydrocarbons (PAH's)) were present in the sediments, none of these organics were found in the water at the detection limits given in Table 2.

Sediment/Plant Comparisons - Table 3 compares the levels of metals found in the sediments to average and maximum concentrations found in sedge grown in the sediments. Metals found in the sediments were also found in the plants, with the exception of mercury which was not detected in the sedge. The levels of metals found in sedge were much less than the levels in the sediments. The soil to average plant concentration ratios for the various metals were as follows:

Zinc	-	23:1
Cadmium	-	11:1
Copper	-	23:1
Iron	-	65:1
Manganese	-	3:1
Arsenic	-	14:1
Mercury	-	>960:1
Nickel	-	11:1
Chromium	-	23:1
Lead	-	72:1

The U.S. Army Corps of Engineers Waterways Experiment Station (WES) measured metal levels in plants of a number of natural Great Lakes wetlands in 1981 (Reference 1) Table 4 compares levels found in Cyperus (sedge) from natural wetlands to levels found in Cyperus growth in Times Beach dredge sediment. The ratios of mean metal levels in Times Beach Cyperus to mean metal levels in natural Cyperus are also given. The data indicates that uptake of some metals including arsenic, cadmium, chromium, and iron may be greater in Cyperus grown in Times Beach sediments. Although the arsenic ratio is high, researchers on the natural Cyperus levels state that there may have been significant volatilization losses of arsenic during chemical analyses of natural plant material giving lower values than actually present. The data indicates that there are no significant differences in uptake of copper, lead, manganese, mercury, nickel or zinc.

The only organic pollutant found in plants was Bis (2-Ethylhexyl) Phthalate.

Table 1 - Comparison of Metal Levels in Sediment and Water at Times Beach

Metal	Average Sediment Concentration (ug/g)	Maximum Water Concentation (ug/g)	Maximum Allowable Levels, National Drinking Water Standards (ug/g)
Zinc	1,283	<0.05	5.0
Cadmium	11.9	0.005	0.01
Copper	251	0.020	1.0
Arsenic	22.7	<0.005	0.05
Mercury	4.8	<0.0008	0.002
Nickel	55	0.008	-
Chromium	332	0.002	0.05
Lead	497	0.007	0.05

- No National Standard

Table 2 - Organic Compounds Found in Times Beach Sediments and Water

Name	Sediment Maximum Concentration	Water Concentration
	(ug/g)	(ug/g)
Bis (2-Ethylhexyl) Phthalate	5.5	<0.01
AROCLOR 1242	1.0	<0.05
AROCLOR 1254	2.5	<0.05
Aniline	2.8	<0.01
1-Amino-Napthalene	4.1	<0.01
N-Benzyl-N-Ethyl-Aniline	7.0	<0.01
4,4-Methylene Bis (N,N-Dimethyl-Aniline)	1.4	<0.01
P,P-Benzylidene Bis (N,N-Dimethyl-Aniline)	4.7	<0.01
Benzo-(a)-Pyrene	96	<0.01
1,2-Dichlorobenzene	9.8	<0.01
1,3-Dichlorobenzene	9.5	<0.01
1,4-Dichlorobenzene	22	<0.01
Napthalene	20	<0.01
Phenanthrene	15	<0.01
Anthracene	13	<0.01
Fluoranthene	24	<0.01
Pyrene	27	<0.01
Benzo-(a)-Anthracene	23	<0.01
Chrysene	26	<0.01

Table 3 - Comparison of Metal Levels in Sediment and Sedge, Times Beach

Metal	Average Sediment Concentration (ug/g)	Average Plant Concentration (ug/g)	Maximum Plant Concentration (ug/g)
Zinc	1,283	55.5	84
Cadmium	11.9	1.04	1.5
Copper	251	11	19
Arsenic	22.7	1.54	4.2
Mercury	4.8	<0.005	<0.005
Chromium	332	17	11
Nickel	55	4.9	38
Lead	497	6.9	18

Table 4 - Comparison of Metal Levels in Cyperus from Natural Great Lakes Wetlands and Times Beach Cyperus

	Cyperus Times Beach		Cyperus Great Lakes		Ratio Times Beach/ Great Lakes
	Mean (ug/g)	Max. (ug/g)	Mean (ug/g)	Max. (ug/g)	
As	1.54	4.22	<0.026	0.70	>59
Cd	1.04	1.53	0.35	3.50	3
Cr	17	38	2.40	33.2	7.1
Cu	11	19	8.4	27	1.3
Pb	6.9	17.9	6.4	85.2	1.1
Fe	983	1,751	166	1,873	6.6
Mn	213	359	158	551	1.3
Hg	<0.005	<0.005	<0.015	2.0	<0.03
Ni	4.9	10.6	3.5	14.1	1.14
Zn	55.5	83.8	79	317	0.7

This compound was also detected in plants grown on uncontaminated control sediment. It is suspected that phthalate which is used as a plasticizer leached from the plastic containers.

Sediment/Worm Comparisons - None of the organics found in Times Beach sediments were found in earthworms grown in the sediment for a 28-day period. This indicates that either the worms did not extract these compounds from the sediment or metabolized them. Table 5 compares concentrations of metals found in Times Beach sediment to earthworms which were incubated in the sediment for a 28-day period. Levels of metals found in the earthworms before incubation are also given. Except for cadmium, levels of the metals in the sediments were much higher than in the worms cultured in the sediments. Average sediment to worm concentration ratios were as follows:

Zinc	-	5.6:1
Cadmium	-	0.9:1
Copper	-	17:1
Arsenic	-	>143:1
Mercury	-	> 8:1
Lead	-	200:1

Table 6 compares maximum metal levels in worms before and after the 28-day incubation period. From Table 6 it may be observed that for the six metals analyzed, levels in the worms before incubation in the Times Beach sediment were higher than after incubation. It was thought that perhaps the worms used in the study contained higher than normal metal levels. However Helmke and his associates (Reference 2) found similar levels of metals in natural worm populations, especially cadmium.

Table 5 - Comparison of Metal Levels in Sediment and Incubated Earthworms, Times Beach

Metal	Average Sediment Concentration (ug/g)	Average Worm Concentration after 28 Days (ug/g)
Zinc	638	114
Cadmium	3.3	3.8
Copper	153	8.8
Arsenic	57	<0.4
Mercury	2.4	<0.3
Lead	300	1.5

Table 6 - Comparison of Maximum Metal Levels in Worms Before and after Placement in Times Beach Sediment

Metal	Before Placement (ug/g)	After Placement (28 Days) (ug/g)
Zinc	153	122
Cadmium	11	4.3
Copper	36	14
Arsenic	5.7	<0.4
Mercury	1.5	<0.3
Lead	12	2.6

CONCLUSION AND FUTURE STUDIES

The data gathered from the Times Beach studies to date indicate that although there are high levels of organic pollutants contained in Times Beach dredge disposal sediments, there is no significant accumulation of these organics in plant material (i.e., sedge) or earthworms grown in the sediment. The data indicates that the heavy metals arsenic, cadmium, chromium and iron may accumulate in sedge grown in the sediments at levels above those found in natural wetlands in the Great Lakes Region. There appears to be no significant differences in plant uptakes of copper, lead, manganese, mercury, nickel or zinc. There was no accumulation of heavy metals in earthworms above original whole body concentrations after 28 days of incubation in Times Beach sediment.

Although there are elevated concentrations of heavy metal and organic pollutants in Times Beach sediments, there is little transfer to the open water within the disposal area as evidenced by the water quality data. The presence of individual organic pollutants at concentrations less than 10 to 50 parts per billion in the water would not be detected at the analytical detection limits achieved. Water quality for heavy metals exceeded national drinking water standards.

It is known that some fish can bioconcentrate organic pollutants from water by orders of magnitude. For that reason, fish indigenous to the contained water at Times Beach have been sampled and are presently being analyzed for those organic contaminants (and heavy metals) found at elevated concentrations in the sediments.

Additionally, natural vegetation including reed, cattails, sedge, and aquatic plants have been sampled from the area and are being analyzed for heavy metals. Since the seeds of sedge are an important source of food for birds and waterfowl, they have been sampled and analyzed individually. Trends of metals accumulation will be compared to those observed in the sedge pot studies discussed previously.

Earthworms indigenous to the Times Beach disposal area and a natural wetland in the Buffalo vicinity have been collected by WES Personnel and will be analyzed for organic and metal pollutants. It is of particular interest to compare concentrations of pollutants in the natural wetland worms to those of worms from the disposal area.

REFERENCES

1. Simmers, John W. et al."Field Survey of Heavey Metal Uptake by Naturally Occurring Saltwater and Freshwater Marsh Plants," Technical Report EL-81-5, Environmental Laboratory, U.S. Army Engineers Waterways Experiment Station, June 1981.

2. Helmke, P.A. et al. "Effects of Soil-Applied Sewage Sludge on Concentrations of Elements in Earthworms," J. Environmental Quality, Volume 8, Number 3, 1979.

APPENDIX

Information papers distributed on the Mount St. Helens
and the Bonneville Dam Field Trips

US Army Corps of Engineers
Portland District

Information Paper

Date: 14 October 1983

MOUNT ST. HELENS ERUPTION RESPONSE

The May 18, 1980, eruption of Mount St. Helens devastated 150 square miles, mostly prime timber land, north of the mountain. An estimated three billion cubic yards of material were deposited in a 17-mile avalanche flow in the upper North Fork of the Toutle River. Another 50 million cubic yards (mcy) filled the upper four miles of the Toutle South Fork. An estimated 150 to 200 mcy of volcanic and landslide material was deposited in the river channels in the 70-mile course down the Toutle and Cowlitz rivers and into the Columbia River.

The Army Corps of Engineers' mission was to restore the river channels for navigation and flood control.

The Columbia River

Fifty mcy of sediment material were deposited in the Columbia River near the Cowlitz River confluence, reducing the Columbia navigation channel's normal 40-foot depth to 14 feet. Thirty-one ships were stranded in upstream ports and 50 others enroute were halted or diverted to other ports.

Corps and contract dredges cleared a restricted channel within 5 days and the channel was fully restored by late November.

Approximately 39 mcy of material has been removed to date to restore and maintain the Columbia channel. An estimated 29 mcy have been deposited in the Columbia and the mouth of the Cowlitz since October 1981.

The Cowlitz River

Sediment that filled the Cowlitz River reduced the channel's previous flow capacity at Castle Rock, Washington, from 76,000 cubic feet per second (cfs) to 13,000 cfs. This created the probability of major flooding in the next winter season at Castle Rock, Lexington, Kelso, and Longview, with a total population of about 45,000. Emergency flood control action began to excavate the Cowltiz channel and improve existing levees, and the channel was restored to a minimum flow capacity of 50,000 cfs.

At the populated communites 14,700 feet of levee was upgraded and 21,400 feet of new levee constructed, initially providing 500-year levels of flood protection.

Due to average rainfall condition and the emergency action taken, no major flooding has occurred on the Cowlitz River since the eruption. A total of about 56 mcy of sediment was removed from the Cowlitz River through May 1981.

In early November 1982, preliminary results of the Corps' sediment study indicated that flood protection for leveed areas along the lower Cowlitz River was less than previously estimated. In order to provide adequate flood protection, ten miles of levee were raised at Lexington, Castle Rock, Kelso and Longview. The cost of these temporary short-term flood protection measures totalled $3.1 million.

The Technical Task Group on Sedimentation preliminary report is complete and the final report is under review.

Contracts have been awarded and work is underway to remove about three mcy from the Cowlitz River in the vicinity of Castle Rock. This work is designed to remove sediment that deposited last winter and to restore 100-year flood protection to Castle Rock.

The Toutle River

The recovery program on the Toutle River system was designed to prevent flooding on the Cowlitz River downstream by curtailing sediment transport from the two forks of the Toutle River. The major source of this erodible sediment is the 17-mile avalanche flow in the upper Toutle North Fork.

To address the immediate problem in the first winter after the eruption, sediment retention projects were undertaken on the Toutle system. Two debris retaining structures were constructed. The Toutle North Fork structure, at the toe of the avalanche flow, is 6,100 feet long (two sections) and 43 feet high, with an impoundment capacity of six mcy. The Toutle South Fork structure, 23 miles above the Toutle River mouth, was 600 feet long and 20 feet high with an impoundment capacity of 600,000 cubic yards. A total 11.4 mcy of sediment was excavated from the impoundments (9.5 mcy at North Fork, 1.9 mcy at South Fork) before maintenance ended in September 1981. The first three miles of the Toutle River channel and eight sedimentation basins upstream were excavated. A total of 7.5 mcy of sediment were removed before excavation ended in May 1981.

The North Fork structure has been breached several times since essential dredging behind the structure was terminated in 1980. The structure has continued to hold back 10 to 12 mcy of material, preventing it from moving downstream. Work to stabilize a large breach on the southern portion was accomplished in November 1982. The South Fork structure was removed in the Fall, 1982, allowing fish to again pass upstream to spawn.

Dredging to remove approximately three mcy of material from the sediment stabilization basin in the lower Toutle River began January 17, 1983. This dredging, in combination with the emergency levee raising, provided 100-year flood protection early in the winter season. Levels of flood protection had deteriorated to 10 year at Castle Rock by March.

Lake Hazards

The Toutle North Fork avalanche flow blocked a number of small tributaries, creating new lakes in unstable terrain near the mountain. These lakes posed a flash flood hazard with winter rains threatening to overtop and breach their embankments. To protect areas downstream, outlet channels were excavated at several of the lakes. Three large ponds were channelized prior to the 1980-81 winter, and revetted outlet channels were constructed at the larger, more hazardous impoundments, Coldwater and South Castle Creek lakes in summer and fall of 1981. Since the creation of the National Volcanic Monument, the Forest Service is preparing to monitor these lakes.

Spirit Lake

Spirit Lake is located at the base of Mount St. Helens near the headwaters of the North Fork Toutle River. Spirit Lake was significantly altered when Mount St. Helens erupted on 18 May 1980 carrying a massive debris avalanche and forming a ridge of volcanic material up to 600 feet deep at the lake's outlet to the North Fork Toutle.

A U.S. Forest Service Task Force report, published in July 1982, stated that the natural dam barrier at Spirit Lake is unstable and an uncontrolled breach would release lake waters and cause flooding downstream.

On August 19, 1982, President Reagan declared a state of emergency, activating the Federal Emergency Management Agency to coordinate federal response. FEMA requested that the Corps develop an interim solution to stabilize the lake over the winter. Barge-mounted pumps began operating November 5, 1982, pumping water from the lake through 3,450 feet of pipe across the debris "plug" to a stilling basin, and from there to the North Fork of the Toutle River. The contract was completed 31 July and pumping stopped. The pumps and engines were overhauled and pumping resumed on 22 September under a negotiated contract with Harder Mechanical Inc., the original contractor. Cost for the pumping facility was approximately $9 million and the negotiated contract for overhaul of the pumps and next year's pumping is for $2,964,000.

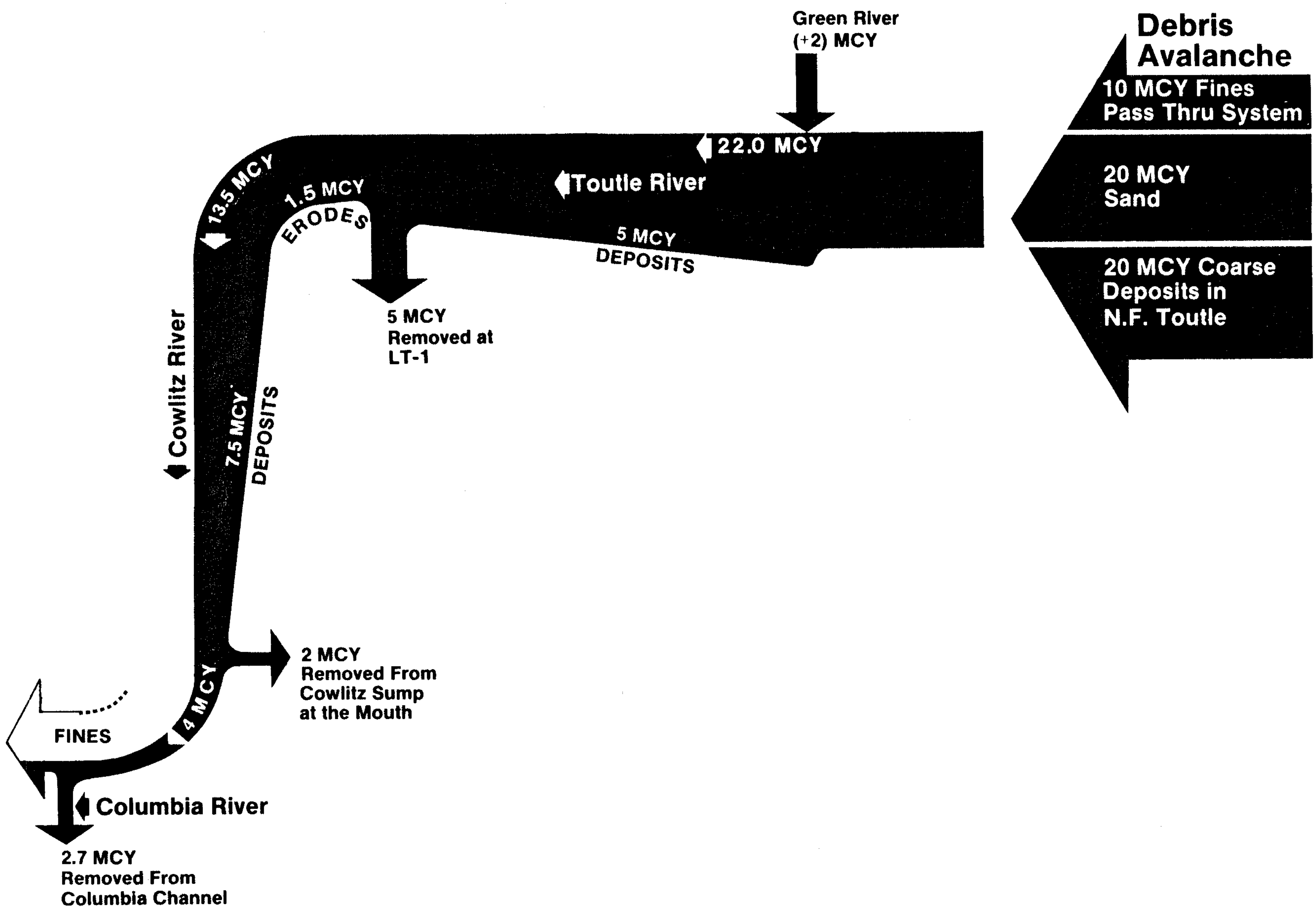
Potential long-term solutions to the problem at Spirit Lake are being considered in the Comprehensive Plan directed by the President.

Cowlitz/Toutle Funding

Congress in June 1980 appropriated $170 million in PL-99 funds for the Cowlitz and Toutle, and $45 million in O&M funds for the Columbia. The total allotment for the Columbia has since risen to $56 million. The Office, Chief of Engineers, has contributed $87 million in PL-99 funds since June 1980, plus $300,000 in PL-99 funds for stabilization of the North Fork debris structure. In May 1982, President Reagan directed the Corps to prepare a comprehensive plan for long-term flood control and navigation maintenance. This 18-month study will be completed in November 1983.

Total allocations for Mount St. Helens work, through mid-May 1983, are approximately $315 million (including comprehensive plan and Forest Service funds). "Jobs Bill" funds of $17.55 million have been allocated for work at Spirit Lake, dredging and levee rehabilitation on the Cowlitz and Toutle rivers. Four contracts totaling $13,133,900 were awarded to remove about 3,500,000 cubic yards of material from the Cowlitz River and about 4,500,000 cubic yards of material from LT-1. The dredging and levee rehabilitation on the Cowlitz River will be completed by early November. Dredging at LT-1 will continue through the winter to late March 1984.

FY 84 PROJECTED SEDIMENT MOVEMENT



THE BONNEVILLE FISHWAYS

The Bonneville Fishways include both fish ladders and fish elevators or locks. The elevators are no longer in use, since the fish definitely prefer the fish ladders.

The ladders consist of three principal parts, the collection systems or entrances, the stepped ladder sections, and the exits, together with various auxiliaries.

Collection System

There are three collection systems, each consisting of several entrances. Each collection system is equipped with the necessary equipment to compensate for changes in tailwater elevation, which may exceed 40 feet during the year. Each collection system is also provided with facilities to add auxiliary water to maintain desirable flow velocities as the tailwater elevations change. The auxiliary water systems also serve the lower and entrance ends of all ladders.

The powerhouse collection system or entrance is a 17-foot channel extending the full length of the downstream or tailrace side of the powerhouse. There are 65 entrances provided, but only 22 are used. It is equipped with 22 diffusion chambers and valves for the introduction of auxiliary water as desired. Only 21 are used for normal operating conditions.

Fish enter the collection system through submerged orifices. Orifices are 12 square feet in area, and the velocity of water discharged through them is approximately eight feet per second. After entering, fish are attracted by a linear flow of about two to three feet per second, which they ascend to the first stage of the fish ladder.

The two main dam entrances are identical. They consist of three vertical slot entrances each. Entrance velocities are about eight feet per second. The auxiliary water system is different in design, but in principle is very similar to that of the powerhouse collection system auxiliary system.

Ladder Sections

The fish ladder sections are conventional pool-type ladders and have been the primary means of passing fish upstream over Bonneville Dam since the spring of 1938. The ladders are built on a 16 to 1 ratio, rising one foot for each 16 feet of length. Pools are formed by weirs, either fixed or adjustable. All weirs are numbered, the number referring to the elevation of the weir crest in feet above mean sea level.

The fixed weirs are used in the lower parts of the ladders, from tailwater elevations up to and including weir 68. These weirs consist of a four-foot concrete section topped by a two-foot timber extension consisting of five sections of 6 x 6-inch timbers. The fixed weirs in both branches of the Bradford Island ladder have additional timber extensions, dividing the weir crests into two open sections. All fixed weirs are provided with one submerged 2 x 2-foot orifice in each end of each weir.

The exit-control section is located upstream of the counting station exit pool. The upstream end and the east two-thirds of the fish ladder is the exit-control section. The width of the exit-control section, 27.5 feet, has been used to integrate with the piers in the existing ladder and provide space for the make-up water system. The exit-control section provides fish passage from the fixed elevation counting station to the 7 foot range (70.0 to 77.0) forebay fluctuation. The exit-control section will also be able to operate with little modification throughout forebay elevation range of 70.0 to 82.5 feet.

The exit-control section consists of vertical fixed baffles with full-height slots, bleed-off diffusers, and add-in-diffusers. The verticle fixed baffles, located between the counting station exit-pool and the forebay, by developing nearly equal water-surface-drops through the 18 baffle slots. Each of the five bleed-off diffusers, located in pools 4,6,8,10 and 12 decreases the discharge in the successive slots downstream where pools are shallower. This provides a more uniform slot-width, a more uniform turbulence level in each pool, more uniform water-surface drops through all the slots, and more uniform flow conditions at all forebay levels. Add-in diffusers are located in pool 2 and in the counting station exit pool. They are intended to increase the discharge in the downstream end of the control section and in the counting station exit pool, at low forebay levels, thus encouraging fish away from the counting station and into the exit-control section in order to minimize the number of fish dropping back past the counting window.

<u>Identification and Counting</u>

All fish passing through the exit are identified and counted by species. This is done by specially trained women stationed in small counting houses beside the counting board. They must be able to identify about 35 species of fish at a glance, estimate the size of salmon and note various tags used by fishery agencies studying migration patterns. So far, there is no satisfactory substitute for these counters.

Counters work eight-hour shifts, counting 50 minutes of each hour, and resting 10 minutes of each hour. In March and November, fish are counted on only one eight-hour shift per day. In April through October, counting is done 16 hours per day. Very few fish use the ladders in December through February, and no counting is done. All major maintenance work is done in the latter period.

Five species of so called Pacific Salmon, all classified under the generic name: Oncorhynchus (or the hooked nose genus), migrate upstream beyond Bonneville Dam. After entering fresh water these salmon cease to feed.

In addition, the Steelhead Trout, cousin to the Atlantic Salmon, also spawn in the tributaries to the Columbia River.

All Pacific Salmon and Steelhead Trout are Anadromous in that they are spawned in fresh water, then go to the ocean to grow.

Jack Salmon are precocious male Chinook, Blueback, Silver and possibly Chum Salmon which return to fresh water after a shorter than normal period in the ocean. Some jacks may stay in salt water only a few months, others will stay a year or longer and return as small under mature individuals to attempt to fertilize the eggs of the much larger females. When this occurs the majority of the progeny will be of normal size. The British and Canadian term for Jacks is Grilse. Numerous Chinooks never go to the ocean, but at sexual maturity the sperm from these 8-inch long males have been used experimentally to fertilize the eggs of a 45 pound female. Approximately 11 per cent of the Chinook Salmon count are Jacks.

All Salmon of the genus Oncorhynchus die after spawning, but this is not necessarily true of the genus Salmo, the Steelhead Trout of the Pacific coast and the Atlantic Salmon. However, the vigors of the spawning migration causes a mortality of 85 to 90 per cent.

Anglers esteem the fighting prowess of the Steelhead and spend thousands of dollars outfitting for and seeking to catch "the leaper." Individuals weighing up to 36 pounds have been taken, but the average size is smaller. This is the rainbow trout which goes to the ocean. It spends a considerable part of its life in the sea, entering the rivers in the third, fourth, fifth years after two or more summers in salt water, for the first spawning.

As many as 260,000 of these graceful fish are counted at Bonneville Dam fish ladders each year en route to the spawning grounds.

In 1871 the U. S. Bureau of Fisheries brought several thousand small Shad across the continent by railroad from the Atlantic seaboard and liberated these juveniles in the Sacramento River. The experiment was successful and by 1876 numerous Shad had spread northward via the ocean and entered the Columbia River.

These members of the Herring family have been taken by gill-netters, fishwheels, and dipnetters since that time.

Before The Dalles Dam was raised a few Shad penetrated as far as McNary Dam.

Alosa, the generic name of our Shad, has spread to Alaska and to Southern California. Both sexes ascend into fresh water where the female may release 150,000 eggs in wild dashes at the water surface with several males in frantic pursuit. Weights of 13½ pounds and ages up to 7 years have been recorded.

The fish is too bony to have great commercial value as a food fish on the Pacific coast. However, the roe (egg clusters) is valuable as a table delicacy and commercial fishermen now take tons of the eggs each year. Neither male nor female die as a result of the spawning act, but a heavy mortality occurs naturally during the Shad migration season. Only since 1960 has the species become abundant. Apparently The Dalles Dam, completed in 1957, created optimum spawning conditions and we now count up to 617,000 Shad in June and July. Before 1960 the average count was 15,000 fish, it is now 66,000.

Lampreys, commonly but erroneously called Eels, are blood sucking parasites or predators that attach, by use of their round sucking mouth, to the host fishes. With their three rasping teeth they chew a hole into the tissues of the host and feed upon the blood thereof. But like the Salmon, they cease to feed when they enter into fresh water. However, when landlocked they can adapt to fresh water existence and then may feed upon trout or other fishes, as in the Great Lakes and several small lakes in Klamath County.

At spawning time the females lift enough pebbles from the bottom of the stream to make a shallow depression. The eggs are then deposited and fertilized in the shallow nest. When the eggs hatch the young Elvers are called ammocoetes. These burrow into the mud along the margins of the streams where they feed upon vegetable material. Here they stay and grow for 1 or 2 years or perhaps longer before going to the ocean. These larvae differ in many structural respects from the adults. Although Yankees are usually too squeamish to eat them, Lamprey have been considered a food delicacy for centuries in Europe and were considered the diet of Kings. They are a primary food of the Great Sturgeon of the Western Rivers.

THE AVERAGE NUMBER OF FISH COUNTED AT BONNEVILLE DAM FROM 1938 UNTIL 1982

The following are monthly averages of the most common anadromous fish counted at Bonneville Dam. Fish are now counted from March 15 until November 15. The following numbers represent a 45 year period from 1938 until 1982.

MONTH	CHINOOK SALMON (King)	SOCKEYE SALMON (Blueback)	COHO SALMON (Silver)	STEELHEAD TROUT	SHAD
March 15th-30th	890			957	
April	52,072			3,191	
May	41,101	175		2,302	4,433
June	33,140	30,033		4,795	132,654
July	25,620	56,383	92	47,811	98,873
August	33,245	868	3,088	42,923	1,852
September	163,738	56	22,539	36,724	77
October	5,044		2,741	2,429	9
November 1st-15th	309		388	468	
45-year average	355,174	87,532	28,854	141,742	237,898