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Water Quality '96
Proceedings of the 11th Seminar

26 February-01 March 1996
Seattle, Washington

WES

Prepared for Headquarters, U.S. Army Corps of Engineers

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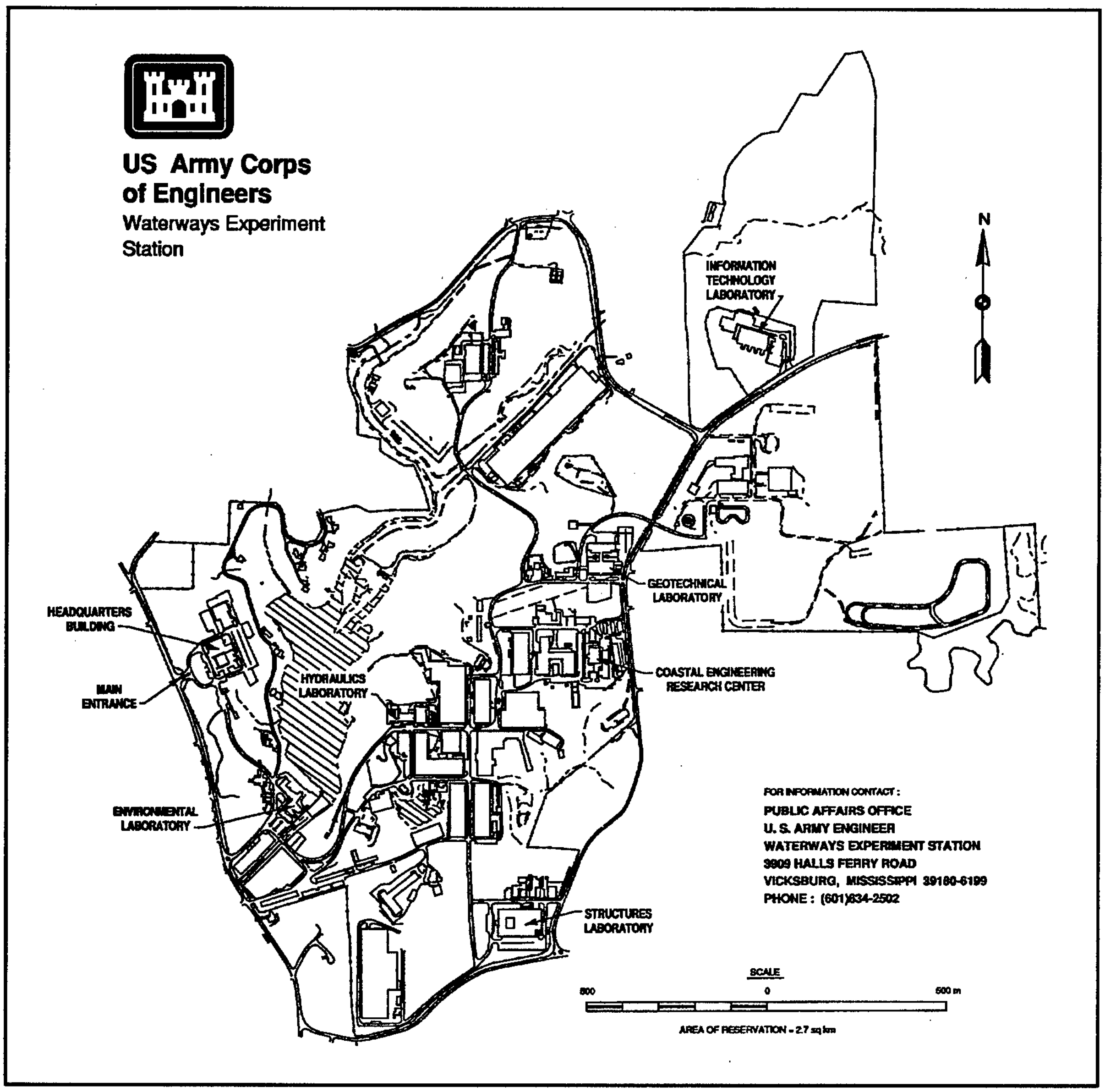

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Contents

Preface

The U.S. Army Corps of Engineers (CE) 11th Seminar on Water Quality was held in Seattle, Washington, 26 February - 1 March 1996. The seminar was co-sponsored by the CE Committee on Water Quality (CWQ) and the U.S. Army Engineer Waterways Experiment Station (WES). Planning and organizational activities for the seminar were the responsibility of Mr. Robert C. Gunkel, Assistant Manager, Water Quality Research Program, Environmental Laboratory (EL), WES. Ms. Billie F. Skinner, EL, provided assistance. The cooperation of Mr. Bolyvong Tanovan, U.S. Army Engineer Division, North Pacific, is appreciated.

Activities leading to the publication of this report were coordinated by Mr. Gunkel and Ms. Skinner. Activities were carried out under the general supervision of Dr. John W. Barko, Director (Ecology), Environmental Modeling, Simulation, and Assessment Center, EL. Dr. John W. Keeley was Director, EL. Mr. Frederick B. Juhle was Chairman, CWQ, for the Headquarters, CE.

At the time of publication of this report, Director of WES was Dr. Robert W. Whalin. Commander was COL Bruce K. Howard, EN.

Agenda

Tuesday, 27 February

General Session

Chair: Pete Juhle, Headquarters U.S. Army Corps of Engineers (HQUSACE)

8:00 a.m. **Call to Order and Announcements**
Pete Juhle, HQUSACE

8:05 a.m. **Welcome**
COL Donald T. Wynn, Seattle District (NPS)

8:15 a.m. **Comments by Chairman, Committee on Water Quality**
Pete Juhle, HQUSACE

8:30 a.m. **Headquarters Overview**
Pete Juhle, HQUSACE

8:40 a.m. **Research and Development Update**
Bill Roper, HQUSACE

8:55 a.m. **Keynote Address**
Donna Darm, National Marine Fisheries Service

9:25 a.m. **Dredging Operations Technical Support Update**
Bob Engler, Waterways Experiment Station (WES)

9:40 a.m. **Break**

Session 1: Statistical and Analytical Tools

Chair: Bolyvong Tanovan, North Pacific Division (NPD)

10:10 a.m. **Power Analysis--Are Your Tests a Waste of Money?**
Joan U. Clarke, WES

10:30 a.m. **Progress in the Development of a Rapid, Low-Cost Assay for Dioxins**
Victor A. McFarland and Michael E. Honeycutt, WES

10:50 a.m. **Progress in the Development of In Vitro Assays for the Assessment of Sediment Genotoxicity**
Michael E. Honeycutt and Victor A. McFarland, WES

11:10 a.m. **Automated Windows Programming for Matrix Organization and Isopleth Graphic Display of Reservoir Limnologic Data**
Robert J. Molnar and Michael Koryak, Pittsburgh District

11:30 a.m. **Implementation of the New ORD Water Quality Database**
Joseph E. Svirbely, Ohio River Division (ORD)

11:50 a.m. **Lunch**

Session 2: Ecosystem Management, Part I

Chair: Dave Buelow, ORD

1:10 p.m. **Predictive Submersed Aquatic Vegetation Model for Chesapeake Bay**
Carl F. Cerco, WES

1:30 p.m. **Eutrophication of San Juan Bay and Estuary, Puerto Rico: Management Issues and Initial Assessment**
Robert H. Kennedy, WES

1:50 p.m. **Planning Versus Rehabilitation: Formulation of an Integrated Environmental Equation**
Julie Conaway, NPD

2:10 p.m. **Sluice/House Unit Releases for Aquatic Habitat Improvement Downstream of Buford Dam**
Jerry Jones and John E. Zediak, Mobile District (SAM)

2:30 p.m. **The Evolution of Ecosystem Management Policies**
Lynn R. Martin, Institute for Water Resources

2:50 p.m. **Break**

Session 2: Ecosystem Management, Part II

Chair: Dave Buelow, ORD

3:20 p.m. **In-Reservoir Effects of Water-Level Drawdowns at Dworshak Reservoir, Idaho**
Ronald E. Wierenga, Steve T. J. Juul, Jamey T. Tielens, and William H. Funk, Washington State University

3:40 p.m. **Temporal Trends in the Limnology of Lower Snake River Reservoirs, Washington**
Gregory L. Geist, Steve T. J. Juul, Jamey T. Tielens, and William H. Funk, Washington State University

4:00 p.m. **Twentymile Creek Habitat Restoration Project**
Howard D. Danley, Michael J. Eubanks, and Cecil L. Jernigan, Jr., SAM

4:20 p.m. **Late-Summer Temperature and Dissolved Oxygen Dynamics in the Tailrace of a Southeastern Hydropower Reservoir**
William E. Jabour, John H. Hains, Joe H. Carroll, WES, and Mark S. Satterfield, DynTel Corp.

4:40 p.m. **Heterogeneity and Variability in Tailwater Quality**
John J. Hains, William E. Jabour, WES, Michael C. Vorwerk, DynTel Corp., and Joe H. Carroll, WES

5:30 p.m. **Poster Session and Reception**

Wednesday, 28 February

Session 3: Erosion, Sediment Resuspension, and Sedimentation

Chair: Mike Lee, Pacific Ocean Division (POD)

8:00 a.m. **Quantifying Sedimentation Using a Three-Dimensional Numerical Sedimentation - Model**
Brad R. Hall, WES

8:20 a.m. **Effect of Wave-Induced Resuspension of Fine Sediment on Water Quality in Nearshore Zone**
T. M. Parchure and W. H. McAnally, Jr., WES

8:40 a.m. **Shoreline Erosion--Management Techniques and Practices**
John L. Andersen, Omaha District

9:00 a.m. **Sediment Resuspension and Export in a Shallow Minnesota River Impoundment**
William F. James and John W. Barko, WES

9:20 a.m. **Demonstration Erosion Control Program Impact to Water Quality in the Coldwater River Watershed**
David R. Johnson, Vicksburg District

9:40 a.m. **Break**

Session 4: Environmental Quality, Part I

Chair: Gary Mauldin, South Atlantic Division (SAD)

10:10 a.m. **Water Quality Studies in America's Greatest Wetland**
Richard Punnett, Jacksonville District (SAJ)

10:30 a.m. **Capabilities of the Groundwater Modeling System**
Jeffery P. Holland, WES

10:50 a.m. **Predicting the Removal of Pollutants by Wetlands**
Mark S. Dortch, WES

11:10 a.m. **Passaic River Flood Diversion Tunnel Water Quality Model Study**
Barry W. Bunch, WES

11:30 a.m. **Simulation of Richard B. Russell and J. Strom Thurmond Reservoirs for Pump-Storage Using CE-QUAL-W2**
Dorothy H. Tillman, WES

11:50 a.m. **Lunch**

Session 4: Environmental Quality, Part II

Chair: Gary Mauldin, SAD

1:10 p.m. **Effects of Aquatic Plants on Water Quality**
John W. Barko and William F. James, WES

1:30 p.m. **Water Quality Evaluation for In-Lake Enhancement Techniques at Alamo Lake, Arizona**
Steven L. Ashby and John L. Myers, WES

1:50 p.m. **Application of Department of Defense Groundwater Modeling System to Columbus Air Force Site, Columbus, Mississippi**
Mansour Zakikhani, WES

2:10 p.m. **Determination of Potential Site-Specific and Cumulative Water Quality Effects of Multiple Flood Control Dams in the Red River of the North Basin**
Dennis Holme, St. Paul District, and Willis Mattison, Minnesota Pollution Control Agency

2:30 p.m. **Water Quality Monitoring for Sea Turtle Migration During Dredging Operations**
Burnell Thibodeaux, New Orleans District (LMN)

2:50 p.m. **Break**

Session 4: Environmental Quality, Part III

Chair: Gary Mauldin, SAD

3:20 p.m. **Water Quality Constraints to Implementing Endangered Species Act on the Columbia River**
Bruce Glabau, NPD

3:40 p.m. **Hydrology, Environment, and the Shared Use of Water**
Nathaniel D. McClure IV and Diane I. Findley, SAM

4:00 p.m. **Applicability of the General Conformity Rule to Corps of Engineers Projects**
Johnna J. Potthoff and Linda M. Sorn, Chicago District

4:20 p.m. **Our CE-QUAL-W2 Experience**
Lisa E. Barnese-Walz, Louisville Distirct

Thursday, 29 February (Concurrent Sessions)

Session 5: Dissolved Gas Issues, Part I: Monitoring and Measurements

Chair: Bolyvong Tanovan, NPD

8:00 a.m. **Total Dissolved Gas Studies on the Lower Columbia: Instrument Performance Evaluation**
Nicole A. Flint, Portland District (NPP)

8:20 a.m. **Evaluation of Total Dissolved Gas Continuous Sensor at Tims Ford Hydro Plant**
Helen G. Rucker and John M. Higgins, Tennessee Valley Authority

8:40 a.m. **Dissolved Gas Measurement Uncertainty**
Darrin Geldert and John Gulliver, University of Minnesota

9:00 a.m. **Remote Monitoring of Total Dissolved Gas: Design, Installation, and Verification of Remote Monitoring Systems**
John W. Lemons, DynTel Corp., Joe Carroll, WES, and Michael Vorwerk, DynTel Corp.

9:20 a.m. **Statistical Considerations of Total Dissolved Gas Sampling on the Columbia River**
Michael Vorwerk, DynTel Corp., Joe Carroll, WES, and John Lemons, DynTel Corp.

Session 5: Dissolved Gas Issues, Part II: Data Collection

Chair: Kimberly Fodrea, NPP

10:10 a.m. **Total Dissolved Gas Abatement Study on the Clearwater River, Idaho**
Richard T. Ramian, William H. Funk, Steve T. J. Juul, Washington State University; Russell Heaton and Thomas Miller, NPW

10:30 a.m. **Total Dissolved Gas Studies on the Lower Columbia: An Overview**
Faith E. Ruffing, NPP

10:50 a.m. **Filtering Dissolved Gas Data**
Steve Wilhelms and Mike Schneider, WES

11:10 a.m. **Total Dissolved Gas Studies on the Lower Columbia: Transect Studies**
Faith E. Ruffing, Nicole A. Flint, and George A. Kalli, NPP

11:30 a.m. **Use of Methane Gas as a Surrogate Tracer for Reaeration-Rate Measurements at Navigation Structures in the Ohio River Basin**
George Kincaid, Huntington District (ORH), Kimberly Miller, U.S. Geological Survey (USGS), William Cremeans, Vincent Marchese, and Richard Meyer, ORH

Session 5: Dissolved Gas Issues, Part III: Analysis and Modeling

Chair: Steve Wilhelms, WES

1:10 p.m. **Prediction of Dissolved Gas Supersaturation Downstream of Hydraulic Structures**
Darrin Geldert and John Gulliver, University of Minnesota

1:30 p.m. **Dissolved Oxygen and Nitrogen Degassing Dynamics in the Lower Columbia and Lower Snake Rivers**
Joe H. Carroll, WES, and Nicole Flint, NPP

1:50 p.m. **Predicting Dissolved Gas with Neural Network Models**
Bolyvong Tanovan, NPD

2:10 p.m. **Modeling of Dissolved Nitrogen in the Columbia and Snake Rivers**
Mike Schneider and Steve Wilhelms, WES

2:30 p.m. **An Empirically Based Numerical Model for Computing Aerator Effectiveness in Reservoirs**
Robert S. Bernard, WES

2:50 p.m. **Total Dissolved Gas Distribution Data Collection Methods: Modelers' Needs**
Thomas D. Miller, Chris A. Pinney, and Russell D. Heaton, NPW

Session 6: Dredging-Related Issues, Part I

Chair: Boni Bigornia, South Pacific Division (SPD)

8:00 a.m. **Assessing the Impact of Channel Deepening in Delaware Bay Through Numerical Modeling**
K. W. Kim and B. H. Johnson, WES

8:20 a.m. **Effects of In-Water Disposal of Dredged Material on Fish and Benthic Communities in Lower Granite Reservoir, Snake River, Idaho-Washington**
David H. Bennett, University of Idaho; Teri Barila and Chris Pinney, Walla Walla District (NPW)

8:40 a.m. **Changes in Larval Fish Abundance Associated with In-Water Disposal of Dredged Material in Lower Granite Reservoir, Idaho-Washington**
Thomas J. Dresser, Jr., and David H. Bennett, University of Idaho

9:00 a.m. **Trends in Resident Fish Abundance Associated with Use of Dredged Material for Fish Habitat Enhancement**
Steve R. Chipps, David H. Bennett, and Thomas J. Dresser, Jr., University of Idaho

9:20 a.m. **Dredged Material Disposal and State Water Quality Standards**
Thomas D. Wright, WES, and Joseph R. Wilson, HQUSACE

9:40 a.m. **Break**

Session 6: Dredging-Related Issues, Part II

Chair: Boni Bigornia, SPD

10:10 a.m. **Technical Considerations for Sediment Quality Criteria**
James M. Brannon and Victor A. McFarland, WES

10:30 a.m. **Beneficial Uses of Dredged Material: The Wilmington Offshore Fisheries Enhancement Structure Wilmington Harbor Ocean Bar, North Carolina**
Philip M. Payonk, Wilmington District

10:50 a.m. **The Puget Sound Dredged Disposal Analysis Program: An Interagency Approach to Sediment Management**
Stephanie Stirling, Therese Littleton, David Kendall, John Malek, Justine Barton, Rachel Friedman-Thomas, and Ted Benson, NPS

11:10 a.m. **Evaluation of PCB and PAH Concentrations in Two Great Lakes Dredged Material Disposal Facilities**
David W. Bowman, Detroit District (NCE), James M. Brannon, WES, and Stuart A. Batterman, University of Michigan

11:50 a.m. **Lunch**

Session 7: Contaminant Issues

Chair: Jan Miller, North Central Division (NCD)

1:10 p.m. **Natural Attenuation of Explosives in Groundwater**
Judith C. Pennington, WES, and Ted Ruff, Army Environmental Center

1:30 p.m. **Air Sparging/Soil Vapor Extraction Remediation, Landfill 4, Fort Lewis, Washington**
William Goss, NPS

1:50 p.m. **Remediation of Explosives-Contaminated Groundwater at Umatilla Army Depot Superfund Site, Oregon**
Michael M. Easterly and Richard E. Smith, NPS

2:10 p.m. **Naturally Occurring Radioactive Materials**
Cheryl G. Peyton, LMN

2:30 p.m. **Trends of PCB, PCDD, and PCDF Federal Navigation Channel Sediments Saginaw River, Michigan**
M. Pamela Horner and Florence K. Bissell, NCE

2:50 p.m. **A Modified Solid-Phase Bioassay for Geotextile Confined Contaminated Sediments**
Glenn R. Schuster, SAJ

3:10 p.m. **Break**

3:40 p.m. **Wrap Up Session**

5:00 p.m. **Adjourn Water Quality '96**

Poster Session Presentations

Development of a Corps of Engineers Comprehensive Watershed Water Quality Model
Pat Deliman, WES

Water Quality and Contaminant Modeling
Toni Schneider and Dorothy H. Tillman, WES

A Rainfall Simulator/Lysimeter System for Predicting Surface Runoff Water Quality
Richard A. Price, Charles R. Lee, and John G. Skogerboe, WES

Contaminants Database Running on the Contaminants Bulletin Board System
Charles H. Lutz and Victor A. McFarland, WES

TNT Transformation Under Controlled Eh and pH Conditions
Cynthia B. Price, James M. Brannon, WES, and Charolett A. Hayes, AScI

An Integrated, Coordinated Study of the Effects of Dredged Material on the Marine Environment - Mamala Bay Example
Kathleen A. Dadey, POD, Michael E. Torresan, USGS, and Allan Ota, U.S. Environmental Protection Agency

The What, When, and Where Behind Data Collection for a Water Quality Math Model
Ruth Leonard, Kansas City District

Low-Cost Portable Integrated Hydroacoustic/GPS System for Underwater Characterization and Mapping
Bruce M. Sabol, WES

The Use of Computer-Based Information Systems for Technology Transfer Activities
Lavon Jeffers, DynTel Corp.

Demonstration of a Basin-Wide Water Quality Database
Robert H. Kennedy, WES

Analyzing Transport on the Snake River
Mike Schneider, Laurin Yates, and Steve Wilhelms, WES

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Conversion Factors, Non-SI to SI Units of Measurement

Non-SI units of measurement used in this report can be converted to SI units as follows:

Multiply	By	To Obtain
acres	4,046.873	square meters
acre-feet	1,233.489	cubic meters
cubic feet	0.02831685	cubic meters
cubic yards	0.7645549	cubic meters
degrees (angle)	0.01745329	radians
Fahrenheit degrees	5/9	Celsius[1]
fathoms	1.8288	meters
feet	0.3048	meters
gallons (U.S. liquid)	3.785412	liters
inches	2.54	centimeters
miles (U.S. statue)	1.609347	kilometers
pounds (mass)	0.4535924	kilograms
pounds (force) per square inch	6.894757	kilopascals
square miles	2.589998	square kilometer
tons (2,000 pounds, mass)	907.1847	kilograms

[1] To obtain Celsius (C) temperature readings from Fahrenheit (F) readings, use the following formula: $C = (5/9)(F - 32)$.

Introduction

The Corps of Engineers Seminar on Water Quality is held biennially to provide for professional presentation of water quality activities. This seminar is attended by representatives of the Headquarters, U.S. Army Corps of Engineers (CE); CE Division and District offices; CE Laboratories and Field Operating Activities; other Federal agencies; State agencies; universities; and private industry.

The overall objective of this biennial seminar is to provide an opportunity for individuals involved in water quality and water control to present and discuss their findings. Topic areas included statistical and analytical tools; ecosystem management; erosion, sediment resuspension, and sedimentation; environmental quality; dissolved gas issues; dredging-related issues; and contaminant issues.

The contents of this report include the presentations of Water Quality '96, the Corps of Engineers 11th Seminar on Water Quality, held in Seattle, Washington, 26 February - 1 March 1996. Abstracts have been substituted for papers not provided.

Power Analysis—Are Your Tests a Waste of Money?

by
Joan U. Clark[1]

Introduction

The Dredged Material Testing Manuals (U.S. Environmental Protection Agency (USEPA)/U.S. Army Corps of Engineers (USACE) 1991, 1994a) specify biological testing in Tier III of a four-tiered testing scheme to evaluate dredged material proposed for open-water disposal. Biological test results are analyzed statistically, and the resulting inferences can form an important component of decision making concerning dredging and disposal alternatives. Recommended procedures for statistical analysis are provided in Appendix D of the Inland Testing Manual (USEPA/USACE 1994a). Supplementary information on the effects of nonideal data in these statistical procedures is given in a companion Applications Guide (Clarke and Brandon, In Preparation).

The statistical analyses for Tier III test a null hypothesis that the average population response of organisms exposed to the dredged sediment does not differ from that of organisms exposed to a reference sediment with respect to the biological test end point. The alternative hypothesis is that response to the dredged sediment is worse (e.g., lower survival, higher contaminant bioaccumulation) than response to the reference sediment in terms of the test end point. Statistical hypothesis test outcomes and their possible consequences in dredged sediment evaluations are diagramed in Table 1. When the null hypothesis is in fact true, environmental degradation as measured by the biological test end points should not occur regardless of the statistical test outcome. However, rejecting the null hypothesis can result in possible unnecessary increased cost if subsequent management

Table 1
Statistical Test Conclusions and Consequences for Dredged Sediment Evaluation Biological Testing

Statistical Test Conclusion		True State of Nature	
		Null Hypothesis True	Null Hypothesis False
Null Hypothesis True (do not reject)	Result	Correct	Type II error
	Probability	1 - α (= confidence)	β
	Consequences	No environmental degradation Cost containment	Possible environmental degradation Cost containment
Null Hypothesis False (reject)	Result	Type I error	Correct
	Probability	α	1 - β (= power)
	Consequences	No environmental degradation Possible unnecessary increased cost	No environmental degradation Possible necessary increased cost

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

decisions require more expensive alternatives than unrestricted open-water disposal. Rejecting a true null hypothesis is known as Type I error. The rate or probability of Type I error, α, is the significance level of the test, and $1 - \alpha$ is the confidence level of the test. The term α is usually prespecified to some nominal value; 0.05 has become conventional in biological testing. However, there is no compelling reason to require that $\alpha = 0.05$ for environmental evaluations, as will be shown in this paper.

When the null hypothesis is in fact false and the statistical test rejects it, environmental degradation should not occur; but the resulting management decision may require necessary increased cost for more expensive disposal options. On the other hand, if the statistical test does not reject a false null hypothesis, a Type II error has occurred. As a consequence, environmental degradation is possible if unrestricted disposal is permitted and the biological communities at the disposal site are adversely impacted. The probability of Type II error is β, and $1 - \beta$, the ability of the statistical test to identify true differences as significant, is known as the power of the test.

Thus, for environmental protection in dredged sediment evaluations, it is important to consider Type II error rather than simply restricting Type I error. Requiring that $\alpha = 0.05$, for example, may help to avoid unnecessarily expensive disposal alternatives, but it can also result in statistical tests that have insufficient power to detect adverse biological effects. Papers emphasizing statistical power in environmental assessments have begun to appear in recent years (Alldredge 1987; Fairweather 1991; Green 1989; Mapstone 1995; Millard and Lettenmaier 1986; Muller and Benignus 1992; Peterman 1990). However, it is still often the case that little thought is given to statistical power either in planning experiments or after the tests have been conducted. The unfortunate result can be experiments of such low power that detectable differences are of an exceedingly large magnitude, or conversely, tests of such high power that statistical significance is attributed to small, biologically trivial differences. In either case, the tests are essentially a waste of money.

Power analysis is a tool for determining whether a statistical hypothesis test has the ability to detect meaningful differences between treatments. For the standard tests recommended in the Inland Testing Manual (two-sample *t*-test or Least Significant Difference (LSD) test), power analysis uses a simple formula to relate α, β, sample size, variability of the data, and the amount of difference between sample means that the test can detect as significant (i.e., effect size). Power analysis can be used to determine any one of the above parameters given an estimate or specification of the others. Power analysis is best used during experimental design to determine the sample size needed to detect a biologically meaningful effect size within acceptable Type I and Type II error rates. Careful consideration must be given to the magnitude of difference in test end points that constitutes a biologically meaningful effect size for a specific test in a specific environment. Power analysis will be demonstrated in this paper with example calculations from dredging project bioaccumulation and toxicity data. Effect sizes used in these examples are for illustration only and do not necessarily indicate biologically meaningful differences.

Calculating Power

Power for a two-sample *t*-test is calculated using the following formula:

$$t_{1-\beta,\nu} = \frac{\sqrt{n}\ d}{\sqrt{2}\ s} - t_{1-\alpha,\nu} \tag{1}$$

where:

n = number of replicates

d = effect size

s = pooled standard deviation

$\nu = 2(n - 1)$ degrees of freedom

When sample sizes are unequal, n is the average number of replicates. The t value for desired confidence $1 - \alpha$ may be obtained from a t table or by a computerized t quantile function such as TINV in SAS (SAS Institute, Inc. 1988:100) or IDF.T in SPSS (SPSS, Inc. 1994:50). Solving the equation gives a t value for power $1 - \beta$ and ν degrees of freedom. Power is then determined using a computerized t probability function such as PROBT in SAS (SAS Institute, Inc. 1988:86) or CDF.T in SPSS (SPSS, Inc. 1994:50). SAS program statements for calculating power are listed in Appendix A of Clarke and Brandon (In Preparation).

Equation 1 may be generalized for the LSD test by substituting for s the root mean square error (rmse) term from analysis of variance. Degrees of freedom $\nu = k\,(n - 1)$ where k is the number of treatments included in the LSD test.

From Equation 1 it can be seen that power will increase as a result of the following:

- Sample size is increased.
- Effect size is increased.
- Variability in the data decreases.
- Confidence level is decreased (i.e., α is increased).

Power analysis can be conducted a priori for a specified sample size, effect size, and confidence level given an estimate of the variability in the test end point. If an estimate of variability is not available, the effect size can be specified in terms of the standard deviation. For example, one might wish to be able to detect a difference in treatment means of one standard deviation. In this case, the d/s term in Equation 1 reduces to 1.

Bioaccumulation Example

A Corps District biologist has conducted Tier III 28-day bioaccumulation tests using the clam *Macoma nasuta* to assess contaminant uptake from a dredged sediment compared with contaminant uptake from a reference sediment. Based on available data, the biologist wishes to determine the power of a t-test to detect a 50-percent increase in mercury concentration in clams exposed to the dredged sediment compared with mean mercury bioaccumulation from a reference sediment, using $\alpha = 0.05$ and pooled standard deviation $s = 0.059$. If mean mercury bioaccumulation from the reference sediment is 0.114 μg/g, then effect size d is 50 percent of 0.114, or 0.057 μg/g. When $n = 5$, degrees of freedom $\nu = 2(5 - 1) = 8$ and $t_{0.95,8} = 1.86$. Then,

$$t_{1-\beta,\nu} = \frac{\sqrt{5}\ (0.057)}{\sqrt{2}\ (0.059)} - 1.86 = -0.332$$

Using a probability function to solve for $1 - \beta$ when $t_{1-\beta,8} = -0.332$, $1 - \beta$ = power = 0.37. Power of 0.37 is so low that such a test would be a waste of money. The biologist would do better to flip a coin since a coin toss has a probability of 0.50 of "detecting" a true significant difference.

One way to increase power is to increase sample size. Figure 1 displays power of a t-test for the example described above using several sample sizes ranging from 5 to 30 replicates. If power of 0.95 is desired, sample size of 25 is required for this example. Because of the cost of biological testing and subsequent chemical analyses, such a large sample size will rarely if ever be feasible.

As sample size is usually limited by fiscal or investigator may have little control over samplevariability. Another way is to increase the amount of difference in test end points that will

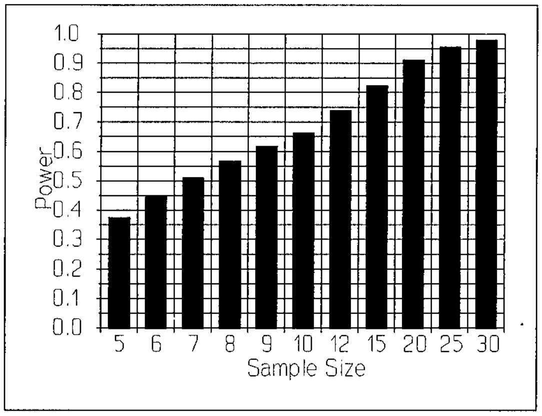

Figure 1. Power of a two-sample t-test for various sample sizes using parameters from bioaccumulation example data

be considered statistically significant. In general, there is no consensus on what effect size is biologically meaningful for each test. Figure 2 displays power of a *t*-test for the example described above using two sample sizes and effect sizes ranging from 50- to 300-percent increase above the reference mean mercury bioaccumulation. When $n = 5$, reasonably high power (>0.80) is not achieved until effect size logistical constraints, other ways of increasing statistical power should be considered. One way is to decrease the standard deviation, but the is a 100-percent increase above the reference mean.

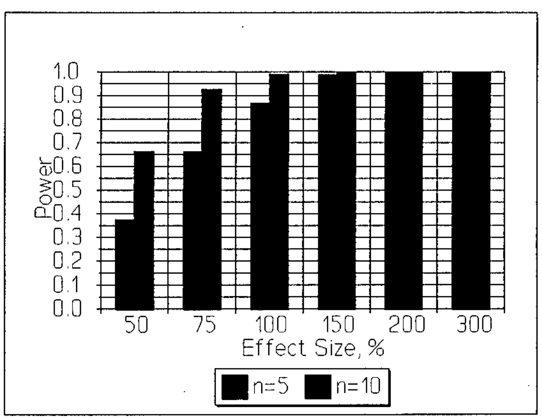

Figure 2. Power of a two-sample t-test for various effect sizes as percent increase above reference mean mercury bioaccumulation, using parameters from bioaccumulation example data

When $n = 10$, power >0.90 is achieved with a 75-percent increase above the reference mean. Based on the example data parameters, the *t*-test will always detect as significant an increase of 200 percent or more above the reference mean for $n \geq 5$.

The final way to increase the power of a *t*-test is to increase the significance level of the test. As α increases, $t_{1-\alpha,\nu}$ in Equation 1 decreases. Figure 3 displays power of a

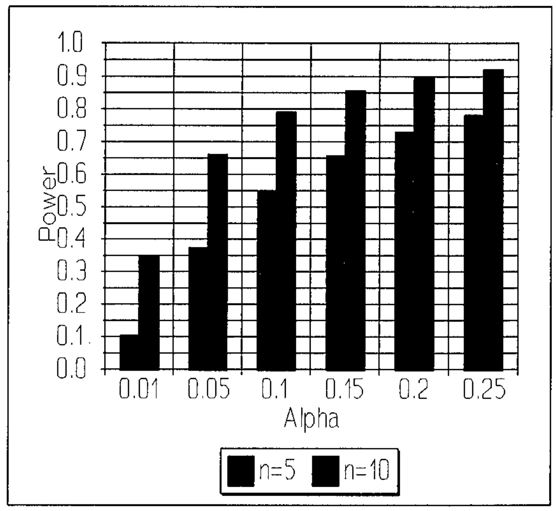

Figure 3. Power of a two-sample t-test for various α levels using parameters from bioaccumulation example data

t-test for the example described above using two sample sizes and α ranging from 0.01 to 0.25 (i.e., confidence levels decreasing from 0.99 to 0.75). When α is a very stringent 0.01 and $n = 5$, power is only 0.10. At the typical $\alpha = 0.05$, power is not very high even for sample size of 10. Power is about the same as the confidence level (i.e., Types I and II errorrates are approximately equal) when $\alpha = 0.25$ and $n = 5$, or when $\alpha = 0.15$ and $n = 10$. If environmental protection is considered to be as important as avoiding unnecessary expense in dredging projects, it makes sense to give equal weight to Types I and II error rates in statistical hypothesis testing. Thus, the risk of environmental degradation would balance the risk of unnecessary cost. A table providing approximately balanced Types I and II error rates for a t-test using various sample sizes and relative effect sizes is available in Clarke and Brandon (In Preparation: Table 4). For example, if one wishes to detect a difference equal to two standard deviations when $n = 5$, using $\alpha = 0.10$ in a t-test will result in $\beta = 0.06$. In other words, the confidence level will be 90 percent and power will be 0.94. In a situation where environmental protection is considered more important than avoiding unnecessary cost, the biologist should design experiments in which power is given priority over confidence.

Benthic Toxicity Example

A Corps District biologist has data from a 10-day benthic toxicity test in which the amphipod *Hyalella azteca* was exposed to three dredged sediments and a reference sediment. Mean reference survival proportion was 0.857, and mean survival proportions from the dredged sediments were 0.900, 0.700, and 0.929. The arcsine-square root transformed survival proportions were used in an LSD test according to the statistical protocols in USEPA/USACE (1994a), and no significant decreases in survival were found in any of the dredged sediments compared with the reference. The rmse for the transformed data was 0.266 and sample size was seven replicates per treatment. The biologist wants to know the power of the LSD test to detect the difference between mean reference survival and the lowest dredged sediment survival (i.e., d = 0.857 - 0.7 = 0.157; d for arcsine-square root transformed data = 0.217). For $n = 7$ and $k = 4$ treatments, $\nu = 4(7 - 1) = 24$ and $t_{0.95,24} = 1.711$. Substituting into Equation 1,

$$t_{1-\beta,\nu} = \frac{\sqrt{7}\ (0.217)}{\sqrt{2}\ (0.266)} - 1.711 = -0.187$$

Using a probability function to solve for $1 - \beta$ when $t_{1-\beta,24} = -0.187$, $1 - \beta$ = power = 0.43. The power of this test is too low to be able to detect the observed difference as significant. To detect the observed difference with power of 0.95 would require $n > 30$ when $\alpha = 0.05$.

The minimum detectable difference for this LSD test, when power = 0.5 and $t_{1-\beta,\nu} = 0$, is 0.244 for arcsine-square root transformed data. That is, the LSD test could detect a dredged sediment mean survival proportion of 0.651 (after backtransforming) as significantly lower than the mean reference survival proportion in half of all comparisons of samples from the two populations.

Power Analysis for Great Lakes Tier III Testing Protocols

The USEPA Great Lakes National Program Office and the USACE North Central Division have jointly developed guidance for Great Lakes dredged material testing and evaluation (USEPA/USACE 1994b). Biological tests that may be performed in the dredged material evaluation include toxicity tests using the midge *Chironomus tentans* (10-day growth and survival) the cladocerans *Daphnia magna* or *Ceriodaphnia dubia* (48-hr survival), and the amphipod *Hyalella azteca* (10-day survival); and a 28-day contaminant bioaccumulation test using the oligochaete *Lumbriculus variegatus*. Available data for all but the *Ceriodaphnia* test were collected and subjected to power analysis. The data are summarized in Table 2.

Root mean square errors for use in power analysis were obtained from analysis of variance for each test, which also served to statistically remove variability attributable to differences in dredging project locations. Power was calculated for effect sizes of 20-, 50-, and 100-percent change from the mean reference response, using $n = 5$ and $\alpha = 0.05$ (Figure 4).

For all of the tests, power to detect a 20-percent change from the mean reference response is low, ranging from <0.10 for *Lumbriculus* PCB bioaccumulation to 0.38 for *Hyalella* survival. If a 20-percent decrease in survival compared with the reference were considered a biologically meaningful effect size for a toxicity test, then these tests generally would not have enough power to detect that

Table 2 Summary of Data for Power Analysis of Biological Tests Included in Great Lakes Dredged Material Testing and Evaluation Manual			
Test	**Total Number of Observations**	**Mean Reference Response**	**Root Mean Square Error**
Chironomus tentans 10-day growth	139	1.096 mg dry wt.	0.255
Chironomus tentans 10-day survival	228	0.841 survival proportion	0.281[a]
Daphnia magna 48-hr survival	76	0.965 survival proportion	0.462[a]
Hyalella azteca 10-day survival	292	0.931 survival proportion	0.271[a]
Lumbriculus variegatus 280-day bioaccumulation	57	90.32 ng/g wet wt. total polychlorinated biphenyls (PCB)	98.67

[a] Using arcsine-square root data transformation.

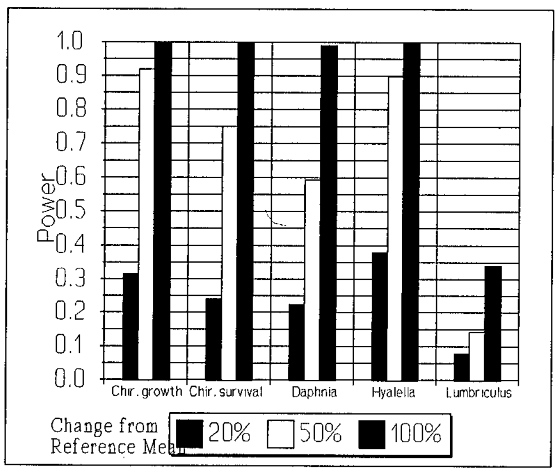

Figure 4. Power of a two-sample t-test to detect effect sizes of 20-, 50-, and 100-percent change from mean reference response when n = 5 and α = 0.05 (see Table 2 for description of biological tests)

difference when sample size is five replicates and $\alpha = 0.05$. The data indicate high power (~0.90) to detect a 50-percent decrease in *Chironomus* growth or *Hyalella* survival. Power to detect a 100-percent decrease is ~1.0 for the growth and all three survival tests. The *Lumbriculus* bioaccumulation test, on the other hand, has low power to detect even a 100-percent increase in PCB bioaccumulation compared with the reference. To obtain power of 0.95 when $n = 5$ and $\alpha = 0.05$ would require an effect size of 260 percent in the bioaccumulation test.

The data used in these power analyses were obtained from several Great Lakes locations. For a given effect size, the power of each test can differ substantially among locations when locations are analyzed individually rather than combined. An extreme example is the power of the *Chironomus* growth test, which ranges from 0.18 for Ashtabula Harbor data to 0.98 for Michigan City data, when d is 20 percent of the mean reference response, $n = 5$, and $\alpha = 0.05$.

The relative performance of the various tests can be judged by looking at the effect sizes that can be detected at a specific power. Table 3 shows effect sizes as percent of the mean reference response when power = 0.5 (i.e., the minimum significant difference) and when power = 0.95. It may not be reasonable to

Table 3
Two-Sample *t*-Test Effect Sizes for Biological Tests Included in Great Lakes Dredged Material Testing and Evaluation Manual; α = 0.05 and n = 5

Test	Effect Size as Percent of Mean Reference Response	
	Power = 0.50	Power = 0.95
Chironomus tentans 10-day growth	30	60
Chironomus tentans 10-day survival	28[a]	56[a]
Daphnia magna 48-hr survival	39[a]	78[a]
Hyalella azteca 10-day survival	24[a]	48[a]
Lumbriculus variegatus 28-day PCB bioaccumulation	129	258

[a] Using arcsine-square root data transformation.

expect that a 20-percent decrease from referencemean survival or growth can be detected as significant when sample size is 5 and α = 0.05, because the minimum significant difference for each of these tests exceeds 20 percent. Rather, effect sizes of about 50 to 80 percent of the reference mean are what can be detected with high power. If a 20-percent decrease in survival or growth is considered biologically meaningful, it may be necessary to increase n to 8 or 10 replicates per treatment and increase α to 0.20 or 0.25.

The *Lumbriculus* bioaccumulation data, while more limited than the toxicity test data, suggest that larger effect sizes can be expected for a bioaccumulation test than for a toxicity test. Because of the variability typical of contaminant concentration data, it is unlikely that a 20-percent increase in bioaccumulation above the reference mean would be statistically detectable. Such a small increase in mean contaminant concentration might have little biological significance even if it were statistically significant. Power analyses conducted on several dredging project bioaccumulation data sets with a variety of contaminants suggest that effect sizes detectable with high power when n = 5 and α = 0.05 are 50 to 100 percent of the reference mean for metals and 300 to 1000 percent of the reference mean for organics such as PCBs and polynuclear aromatic hydrocarbons. Although these effect sizes seem high, it may not be unreasonable to consider an order of magnitude difference necessary for significance in some cases, especially when the data are highly variable or when most of the reference data are near or below detection limit.

Summary

Statistical power analysis is an important tool for determining the efficacy of statistical tests to detect biologically meaningful differences among treatments. Power formulas are relatively simple for the statistical procedures (*t*-test and LSD test) recommended in the Inland Testing Manual. Power analysis should be conducted as part of experimental design to determine the sample size required to detect a meaningful effect size at a desired power and confidence level, or conversely, to determine the effect size that can be detected given a limited sample size. Such a priori analysis can prevent waste of time and money on poorly designed or insufficiently replicated experiments. Power analysis can also be conducted a posteriori, especially given a nonsignificant statistical result, to determine what effect size could be detected as significant. Careful consideration should be given to what constitutes a biologically meaningful effect size for each test and to the relative importance of Types I and II error rates. Increasing the acceptable Type I error rate above the traditional 0.05 may be desirable to increase the power of the statistical test to a reasonable level.

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Progress in the Development of a Rapid, Low Cost Assay for Dioxins

by
Victor A. McFarland[1] *and Michael E. Honeycutt*[1]

When dioxins are suspected of being present in dredged sediments the costs of analytical chemistry required for ecological evaluations in the permitting process can be exorbitant. Additionally, monitoring to assess performance in remediation, confinement, and treatment activities for high risk dredged material containing dioxins also relies on analytical chemistry and is excessively expensive. A significant portion of the high cost of testing could be eliminated if faster and cheaper methods were available to identify and measure the levels of dioxins in soils and sediments.

The Aquatic Contaminants Team (ACT), CEWES-ES-F, is developing procedures for testing dredged sediments for genotoxicity. One of the procedures is an assay that uses cultured mammalian cells to detect the induction of marker enzymes that signal the interaction of dioxins and related planar compounds with genetic material. The strongest response in the assay is to 2,3,7,8-TCDD--the most potent dioxin congener. The response is linear with concentration and is very sensitive. By refining this assay to include standardization against 2,3,7,8-TCDD and by using appropriate clean-up and extraction techniques it is possible to develop a low-cost alternative to GC/MS analysis of dioxins. The sensitivity of the assay is potentially on a par with high resolution GC/MS analysis. Results can be reported numerically as dioxin toxic equivalents (TEQ) in units of concentration and used similarly to GC/MS analyses.

This work was proposed and accepted for funding in the Dredging Operations Environmental Research (DOER) Program as the number one priority work unit in the Contaminated Sediments Focus Area. However, DOER has been rejected for funding by Congress. Still, the potential for cost-reduction amounting to at least one order of magnitude (about \$200/sample for the assay as opposed to about \$2000/sample for GC/MS) is so great that the ACT is proceeding with development of the assay to the extent possible with available resources. This paper reports the progress we have made to date.

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Progress in the Development of In Vitro Assays for the Assessment of Sediment Genotoxicity

by
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Introduction

A great number of the contaminants typically found in dredged material are toxic to exposed organisms through effects on DNA (deoxyribonucleic acid). Such effects are usually the result of low-level chronic exposures. These effects can result in reproductive failure of organisms, impaired growth and development of offspring, and tumors, often cancerous, in vertebrates. Collectively, such effects are called "genotoxicity" and result from damage to the genome of a (somatic or germ) cell. The damage is heritable, i.e., passed on to future cell generations upon duplication of the affected cells.

Although tests of sediment genotoxicity are not routinely applied in regulatory contexts, the potential for their requirement in special circumstances is implied by the language of Public Law. For example, Section 103 of the Marine Protection, Research, and Sanctuaries Act of 1972 (PL 92-532), which regulates disposal of dredged material in coastal regions, specifically prohibits open-water disposal in other than trace amounts of "known carcinogens, mutagens, or teratogens or materials suspected to be carcinogens, mutagens, or teratogens by responsible scientific opinion." In addition, the emphasis in environmental toxicology over the last decade has increasingly shifted away from the catastrophic end point (death of individual organisms in acute exposures) to chronic and sublethal effects having the long-range potential to seriously affect the viability of populations of organisms. If they are to be relevant, risk assessments involving environmental contamination must take this into account.

For this reason, research sponsored by the Long-term Effects of Dredging Operations (LEDO) Program (Work Unit # 32726) aimed at developing a methodology for testing the genotoxic potential of contaminated sediments to aquatic organisms is ongoing at the U.S. Army Engineer Waterways Experiment Station (WES). Progress on one aspect of that research, in vitro assays for assessing sediment genotoxicity, is summarized in this paper.

Approach

A tiered approach is being developed in which a battery of mechanistically related rapid low-cost assays are applied initially. Based on the results of these assays, decisions can be made regarding whether more definitive tests are necessary at higher tiers in the evaluation. The assays are based on the U.S. Environmental Protection Agency Health Effects Research Laboratory approach for assessing the genotoxic potential of chemicals to rodents and humans (Kitchin, Brown, and Kulkarni 1994).

The battery contains two types of tests: assays to assess damage to DNA and assays to assess nongenotoxic adjuncts of DNA damage. The rationale for selecting these two types of assays lies in the knowledge that cancer and other results of DNA damage are multistage events

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requiring alterations in protein synthesis and cell development and function. For example, the development of cancer involves processes known as initiation and promotion. Initiation can be simply defined as damage to DNA, also known as mutation. A mutation occurs when a DNA nucleotide is chemically modified, deleted, or substituted. Certain environmental contaminants act as mutagens in that they covalently bind to DNA nucleotides, chemically modifying the DNA forming a DNA adduct. The cell contains DNA repair enzymes that can repair mutations under normal circumstances. When the organism is exposed to an excessively high level of a mutagen, the DNA repair enzymes may not be able to repair all of the mutations or may misrepair some mutations by deleting the nucleotide rather than replacing it, or by substituting a wrong nucleotide for the mutated one. Depending on the location of the mutation, the number of mutations, and whether or not the mutation is repaired by the cellular DNA repair enzymes, a mutation may progress to tumor formation or cancer in the organism. The stage of cancer development following initiation is promotion, in which the initiated cell is altered to allow reproduction of the cell, passing the "defect" on to daughter cells.

In order to adequately assess dredged material genotoxic potential, the ability of sediment contamination to cause DNA damage and its subsequent effects on exposed organisms must be ascertained. Even if analytical chemistry were capable of identifying and quantitating all of the genotoxic agents present in a sediment, an assessment of genotoxic potential could not be made with analytical data alone because contaminants interact in unpredictable ways. The toxicological approach involves the use of a battery of biomarker-based in vitro assays on sediment extracts in the first level, or tier, of testing. These assays assess the potential for DNA damage and the subsequent biochemical and molecular changes that lead to tumor formation and other adverse somatic effects. The second level of testing is in vivo testing, which involves exposing fish to the dredged material and assessing genotoxic effects, thereby incorporating bioavailability of sediment-associated contaminants.

In Vitro Testing for Mutagenicity

In vitro testing utilizes two basic types of assays for mutagenicity. Bacterial assays (Ames assay and Mutatox) are designed to detect the presence of mutagenic compounds in a sample, while a second type of assay (alkaline unwinding) is used to determine whether an exposed living cell has experienced mutations.

Ames Assay

The Ames assay utilizes strains of a bacterium, *Salmonella typhimurium*, that have been purposely mutated so that they cannot produce the amino acid histidine, which is required for survival (Ames et al. 1973). In order to grow colonies of these bacteria, they must be cultured on media containing sufficient levels of histidine. For the assay, the bacteria are incubated with the test compound on culture media containing trace levels of histidine and are checked for colony formation. If the test compound is mutagenic, the genetically altered bacteria will reverse mutate, or revert, back to the "wild type" and be able to synthesize their own histidine. Thus, bacterial colonies growing on the histidine-deficient media indicate the presence of mutagens in the growth medium. Chemicals that are mutagenic in the Ames assay are usually carcinogenic in life-cycle rodent bioassays (Bogan 1995).

Mutatox

Mutatox is a relatively new, easy to perform proprietary assay that utilizes a dark mutant of the bacterial strain, *Photobacterium phosphoreum*, which normally bioluminesce (like fireflies) (Kwan et al. 1990). These dark mutants of *P. phosphoreum*, similarly to the *S. typhimurium* tester strains used in the Ames assay, revert to the wild type in the presence of mutagens. The mutation causes *P. phosphoreum* mutagens. The mutation causes *P. phosphoreum* to bioluminesce, and the light produced is easily measured with a luminometer.

A characteristic of prokaryotes such as *S. typhimurium* and *P. phosphoreum* is that, unlike almost all vertebrate and invertebrate species, they do not contain cytochrome P450. Cytochrome P450 is a group of enzymes located primarily in metabolically active tissues such as the liver, spleen, kidney, lungs, etc., that function in steroid metabolism and also metabolize xenobiotic compounds such as polycyclic aromatic hydrocarbons (PAHs) (Sipes and Gandolfi 1991). Many environmental contaminants, such as PAHs, are not genotoxic in their original chemical form, but can be biotransformed (metabolically activated) to a reactive chemical form by cytochrome P450 enzymes. The Ames assay and Mutatox can be used to distinguish between contaminants that require bioactivation and those that do not by inclusion of a rat liver preparation (S9) containing cytochrome P450 with the bacteria and test extract. The Ames assay is typically performed both with and without exogenous metabolic enzymes to differentiate direct-acting mutagens from promutagens, or those that must be metabolized for activity.

Alkaline Unwinding Assays

Alkaline unwinding assays are used to indirectly measure DNA adduct formation by determining the number of strand breaks that occur (Bradley and Dysart 1985). The DNA from exposed cells or organisms is isolated and subjected to alkaline conditions ($pH > 10.5$) by the addition of a strong base. The alkalinity simultaneously causes the DNA to break at the sites of most DNA adducts and causes the DNA to unwind from its normal double-stranded configuration to the two single DNA strands. Higher numbers of DNA adducts cause higher numbers of DNA strand breaks which, in turn, causes a faster rate of double stranded DNA unwinding as compared with normal DNA. The rate of DNA unwinding is calculated and can be expressed as number of adducts per milligram of DNA.

In Vitro Testing for Nongenotoxic Effects

The in vitro testing battery also utilizes tests of nongenotoxic effects on adjunct systems. These assays include cytochrome P450 induction, glutathione fluctuations, ornithine decarboxylase activity, oxidative stress, and cytotoxicity.

Cytochrome P450

Exposure to certain groups of compounds including polyhalogenated dibenzo-*p*-dioxins and dibenzofurans, polychlorinated biphenyls, and PAHs induces the formation of specific cytochrome P450 enzymes that are normally not present or present in very low quantities. This enzyme induction is believed to have a promotional effect on initiated cells, in that it stimulates cells that have been predisposed, or initiated, to cancer formation to become cancerous. Induction of cytochrome P450 also leads to a higher rate of xenobiotic metabolism and elimination, which is generally beneficial to the organism. However, in the case of some of the PAHs and other chemicals, metabolism leads to activation of the parent compound to a reactive metabolite that can form adducts with DNA causing mutations.

Knowledge of the mechanism and effects of cytochrome P450 induction can provide tools for screening sediment contamination. Monitoring cytochrome P450 levels in organisms exposed to sediments either in the laboratory or in the field can give an index (biomarker) of contaminant exposure. Cytochrome P450 induction is also the basis for a bioanalytical assay for dioxins, the EROD (7-Ethoxyresorufin-O-deethylase) induction assay (McFarland and Honeycutt, In Preparation).

Glutathione

Glutathione (γ-glutamylcysteinylglycine), or GSH, is a small peptide that functions as the major defense against electrophilic compounds in most organisms (Meister and Anderson 1983). Electrophiles, often formed during xenobiotic metabolism, bind to DNA and thereby cause mutations. For example, an activated metabolite of benzo[a]pyrene (a carcinogenic PAH) is an electrophile. The reduced form of glutathione, GSH, protects from electrophilic attack by acting as a reducing agent. GSH reduces the electrophile, becoming oxidized GSSG, two glutathione molecules linked by an interchain disulfide (-S-S-) bond. Levels of reduced (GSH) and oxidized (GSSG) glutathione are important indicators of contaminant exposure, as an organism (or cell) may be depleted of glutathione upon exposure to such compounds, leaving it vulnerable to an increased rate of mutation.

Ornithine Decarboxylase

Ornithine decarboxylase (ODC) is an enzyme indicative of cellular proliferation, signaling possible exposure to a cancer promoter (Pegg 1988). ODC removes a carboxyl (-COOH) group from ornithine, a derivative of the amino acid arginine, to form putrescine, the initial product in the polyamine biosynthetic pathway. Polyamines are normally present at very low levels in quiescent cells, but are elevated many fold during periods of active cell division. To assay for ODC activity, livers from exposed organisms are isolated and prepared using ultracentrifugation. Ornithine having a radiolabeled carboxyl group is incubated with the enzyme preparation and the metabolized radiolabeled carboxyl group, which is liberated as a gas (CO_2), is trapped and quantitated. The more ^{14}C-CO_2 liberated, the greater the indication of promotion.

Oxidative Stress

While oxygen (O_2) is essential for all multicellular organisms, some forms of oxygen produced during the metabolism of oxygen, i.e., superoxide anion radicals ($O_2\cdot^-$), hydroxyl radicals ($OH\cdot$), and hydrogen peroxide (H_2O_2), are highly reactive (Yu 1994). These oxyradicals can upset the reduction-oxidation (redox) potential of the cell, leading to a highly oxidizing environment within the cell. The redox potential of cells is normally tightly controlled by cellular antioxidant defense mechanisms. These mechanisms include the enzymes superoxide dismutase, catalase, and glutathione peroxidase, as well as vitamins (β-carotene, α-tocopherol, retinoic acid, and ascorbic acid) and glutathione. Cellular antioxidant defense mechanisms can be overwhelmed by xenobiotic chemicals that induce oxidative stress, exemplified by paraquat and quinones. Consequently, genomic function can be impaired due to alterations in DNA, such as the formation of 8-hydroxydeoxyguanosine adducts. Enzyme function may also be inactivated, and cell membranes may be disrupted due to protein and lipid peroxidation, ultimately resulting in cell death.

Oxidative stress is evaluated by measuring either the oxygen radicals themselves, oxy-adducts such as 8-hydroxydeoxyguanosine, or the induction of oxidative stress defense mechanisms such as glutathione, superoxide dismutase, catalase, and glutathione peroxidase.

Cytotoxicity

Subcellular biochemical changes such as those described above can also lead to cytotoxicity, or cell death. For example, the effect of DNA damage depends on the function of the area of the DNA to which the damage occurred and whether or not the damage is repaired. Damage to nonsense regions of DNA (regions that do not code for a particular protein) will usually not have an effect on the function of the cell, but may alter replication of DNA during cell division. In this case, daughter cells may not be formed, leading to the death of the parent cell. Damage to genes that code for functional proteins (enzymes) or structural proteins necessary for cell viability would also cause cell death. Since DNA damage occurs in a seemingly random manner to cells within an organ, not all of the cells with DNA damage would be expected to die. Some cells with DNA damage may be initiated while others die. Thus cytotoxicity may be used as an indirect determinant of genotoxicant exposure.

Research Focus

All of these biomarkers can be measured in vitro, and when used together, they provide a short-term means of predicting long-term effects, e.g., carcinogenicity. The Aquatic Contaminants Team of the WES Environmental Laboratory is currently developing and validating the in vitro assays that have been described in this paper. Sediments that are to be screened are extracted and prepared as for GC/MS analysis. Cultured cells are dosed with the sediment extracts and are then incubated for an appropriate length of time. After incubation, the cells are assayed. The assays utilize two types of cultured cells, H4IIE cells and Chinese hamster ovary (CHO) cells. H4IIE is an "immortal" or continuous rat liver hepatoma cell line that contains cytochrome P450. The CHO cell line is a continuous cell line that does not contain cytochrome P450. This distinction is important because many invertebrate aquatic species do not possess well-developed cytochrome P450 systems. Thus, using both cell types gives a better indication of risk to all aquatic species than does using only one type. Additionally, because some chemicals such as the PAHs must be metabolically activated in order to exert their genotoxic effect, the use of both types of cell lines can discriminate their presence or absence.

To keep the assays rapid, inexpensive, and sensitive, multiwell fluorescence plate reader technology is being used as the basic developmental methodology whenever possible. For example, in the cytotoxicity test, H4IIE or CHO cells are plated in 96-well plates and incubated overnight. The cells are then spiked with sample extracts or chemical standards and incubated an additional 24 hr. At the end of the 24-hr exposure period, the culture medium is removed from the wells using a microplate washer. Buffer containing calcein AM is added to the cells. Calcein AM is absorbed by live cells, fluorescing green at 530 nm. The cells are read in the fluorescence plate reader, and cytotoxicity is expressed as percent viability. Similar techniques are being applied to most of the other assays in the in vitro genotoxicity testing battery. The results of these tests can then be compared with a matrix of the effects of known genotoxic compounds. Matrix comparisons enable interpretation of the biomarker data in terms of the effects of model chemicals having known modes of action.

In vitro testing serves to identify potentially genotoxic dredged material but does not yield information concerning bioavailability of the contaminants in the sediments. Though methods are continually being refined to predict contaminant levels in aquatic organisms (McFarland et al., In Preparation), the genotoxic potential of dredged material must be evaluated for individual sediments on a case-by-case basis. For this purpose, in vivo assays will be developed to test those dredged sediments for which in vitro testing indicates a genotoxic potential. The development of in vivo genotoxicity testing methods has not yet begun.

Regulatory practices are increasingly being framed in the context of risk assessment. The assessment of risk from environmental chemicals cannot be done accurately based on acute toxic responses alone. Long-term chronic exposures resulting in effects on growth and reproduction in whole organisms and extrapolation of such effects to populations is still in development. Even when such tests are available, their utility will be limited by high cost and diminishing resources for regulatory implementation. Additionally, many of the contaminants in sediments are genotoxic and may not be detected by chronic laboratory exposures. Risk assessments that do not include the potential for genotoxic effects when that potential exists are inaccurate. The work described here is intended to address the need for less costly and more mechanistically interpretable ways to provide the basic data on which accurate risk assessments can be conducted.

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Automated Windows Programming for Matrix Organization and Isopleth Graphic Display of Reservoir Limnologic Data

by

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Introduction

Isometric plotting of reservoir water temperature, dissolved oxygen, iron, manganese, hydrogen sulfide, chlorophyll, algae, and other water quality parameter variations within time and depth (elevation) matrixes is an effective method for the interpretation and display of limnologic data. Isopleth plotting of limnologic data is a very powerful analysis tool because it allows for the simultaneous organization of large quantities of data with three variables (depth, time, and parameter concentration). Utilization of this graphics tool simplifies the demonstration of complex concepts, site specific interactions, and limnologic processes for both technical and non-technical audiences. One isopleth plot can replace dozens or more depth verses concentration graphs, while also extrapolating reservoir conditions for time periods where there are no actual field data. This technique is a particularly useful tool for operating hydropower projects and selective withdraw intakes to achieve downstream water quality objectives. However, constructing matrixes by hand can be a time consuming and tedious task. This paper introduces a very user friendly, and flexible windows driven program to quickly and easily create simple numeric matrixes, line isometric plots, color band isopleth contour plots, and other graphics.

Program Description

Figure 1 is an example of a limnologic data matrix. This particular example presents water temperature data in degrees Celsius from Stonewall Jackson Lake which is located in central West Virginia. These data were collected at a minimum of five-foot depth increments at monthly time intervals between May and December, for one year during the season of summer thermal stratification. A list of the raw data, sorted and ranked by concentration, which is used in the raw data matrix plot, appears to the right of the matrix plot under the "samples" heading. Also, the maximum, minimum, and mean values of this data set are presented above the matrix plot The x-axis is time and the y-axis is elevation. The wavy solid line above the columns is the surface elevation of the reservoir.

The program first interpolates vertically along 600 y-axis units, then horizontally on a default six-inch wide x-axis populated with 400 units to develop a fine grid definition. A grid of 240,000 points makes a postscript file which can be interpreted by a postscript printer, which then develops an isometric bank water temperature contour plot (Figure 2). It is possible to select any temperature or concentration contour

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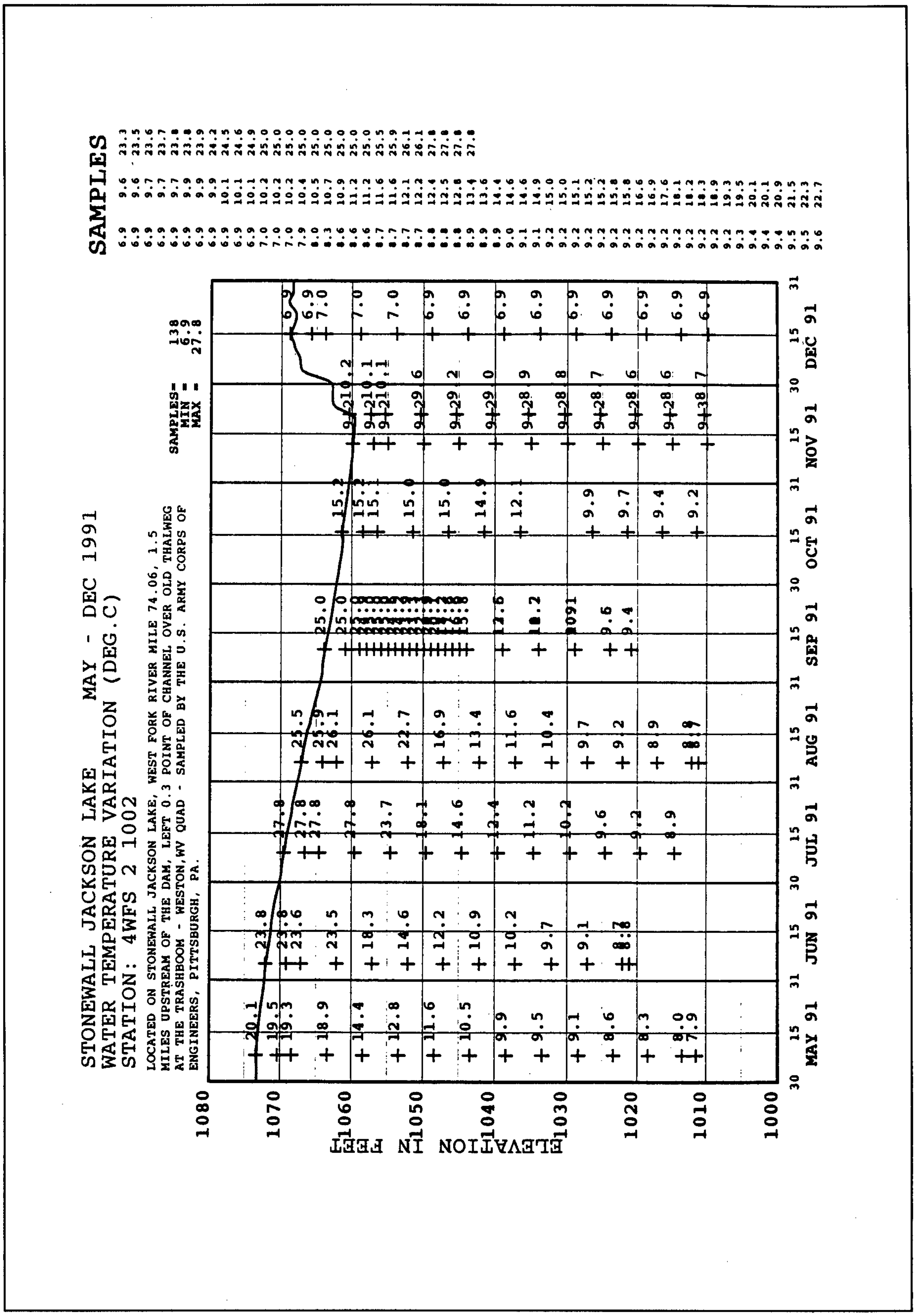

STONEWALL JACKSON LAKE MAY - DEC 1991
WATER TEMPERATURE VARIATION (DEG.C)
STATION: 4WFS 2 1002
LOCATED ON STONEWALL JACKSON LAKE, WEST FORK RIVER MILE 74.06, 1.5
MILES UPSTREAM OF THE DAM, LEFT 0.3 POINT OF CHANNEL OVER OLD THALWEG
AT THE TRASHBOOM - WESTON,WV QUAD - SAMPLED BY THE U.S. ARMY CORPS OF
ENGINEERS, PITTSBURGH, PA.
SAMPLES-
MIN =
MAX =
138
6.9
27.8
ELEVATION IN FEET
MAY 91
JUN 91
JUL 91
AUG 91
SEP 91
OCT 91
NOV 91
DEC 91
SAMPLES

Figure 1.

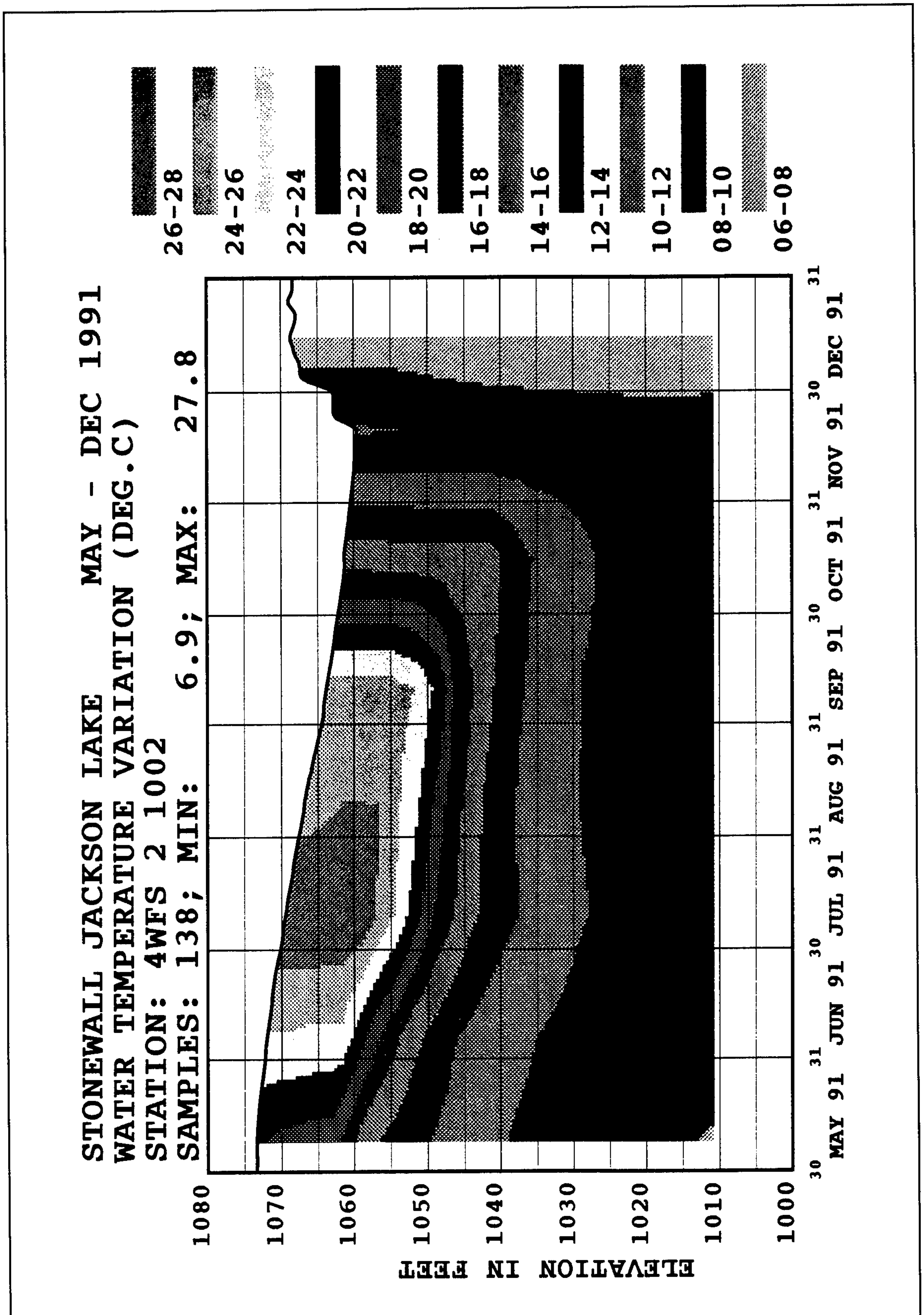

STONEWALL JACKSON LAKE MAY - DEC 1991
WATER TEMPERATURE VARIATION (DEG.C)
STATION: 4WFS 2 1002
SAMPLES: 138; MIN: 6.9; MAX: 27.8
26-28
24-26
22-24
20-22
18-20
16-18
14-16
12-14
10-12
08-10
06-08
ELEVATION IN FEET
1080
1070
1060
1050
1040
1030
1020
1010
1000
30
MAY 91
31
JUN 91
30
JUL 91
31
AUG 91
31
SEP 91
30
OCT 91
31
NOV 91
30
DEC 91
31

Figure 2.

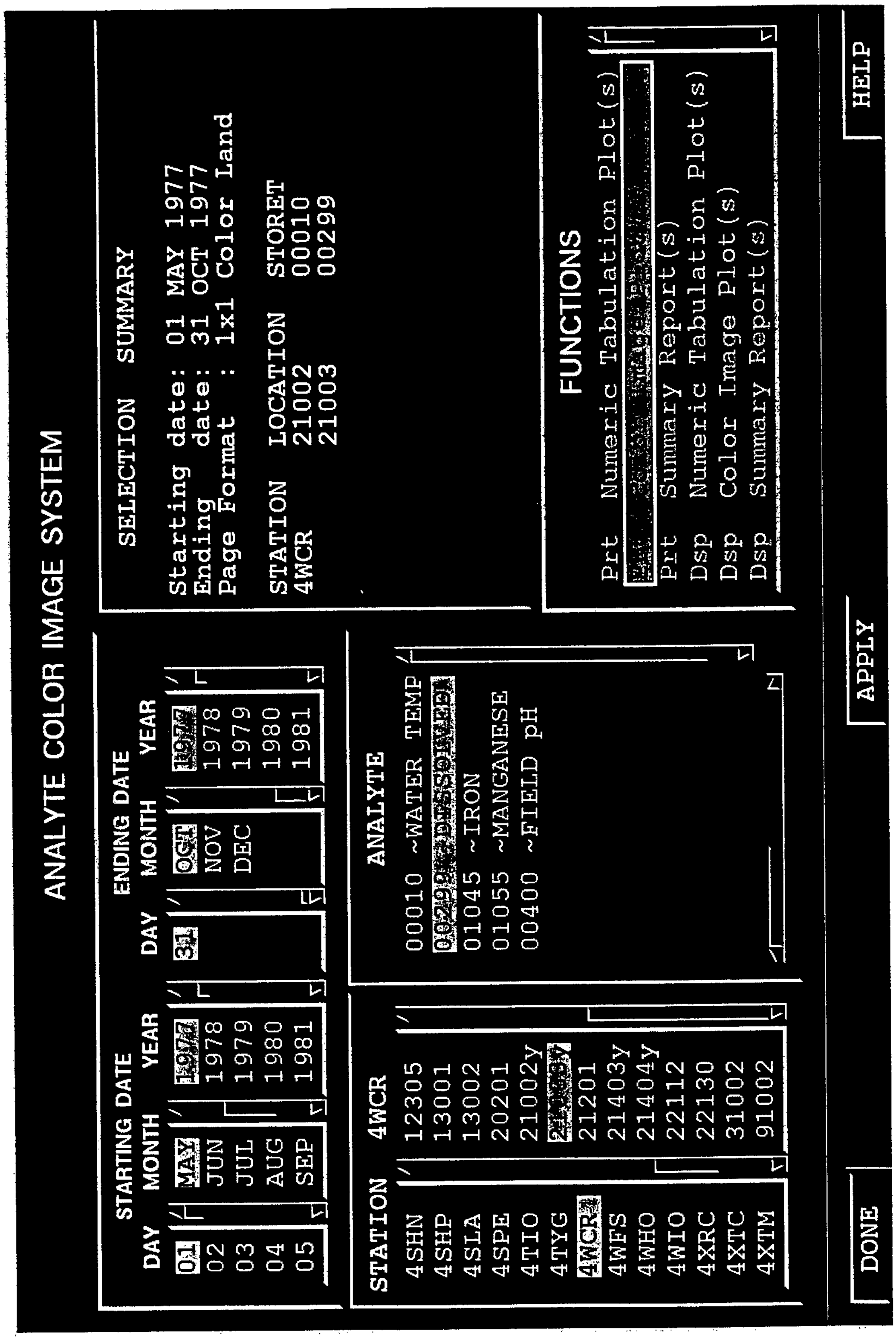
intervals you desire. For the Figure 2 example I have chosen 2 degrees Celsius water temperature band intervals. For parameters such as iron and manganese where summer stratification season vertical concentration gradients may involve orders of magnitude variation, logarithmic or other convenient scales may by selected.

Figure 3 shows the program's Graphic Users Interface, which basically involves selecting the data set for the isopleth by setting up only four windows to produce a product. In the first window, the time range of the data set, which is usually a season is selected by using a mouse. Location is selected in the second window by first selecting a reservoir project and then selecting a reservoir project sampling station from an associated selection box. Parameter or analyte is selected in the third scrolled window screen. Our data is in EPA STORET format, but it is not necessary for the user to know where the data is stored or how it is formated to use this program. In the fourth window, data ranges for contour lines are selected and colors are assigned using the raw data matrix plot and, in particular, the list of sorted and ranked data which appears to the right of the matrix (Figure 1) as references. When assigning contour colors for water temperature, our experience suggests that concept recognition and plot interpretation by audiences unfamiliar with this type of graphics can be expedited by using a visible light spectrum color sequence, where red is used to represent the warmest water temperature observed in an impoundment over the length of a season, and blue represent the coldest. For most other parameters, the spectrum color sequence also usually proves effective, with red being used for undesirably high or low concentrations of dissolved oxygen, hydrogen sulfide, metals, nutrients, chlorophyll, etc. If the use of color plates causes reproduction problems, as it does in this seminar proceedings, non-color, differentially shaded isometric contour plots can also be generated (Figure 2).

After setting up the four windows, simply press the apply button which examines the selected screen configuration and produces either a raw data matrix plot or a color isometric contour plot. By default, the size of each plot is six by six inches and titling is totally automatic. However, the size of the plots can be enlarged or reduced to image multiple plots on one page for comparison. For instance, in Figure 4, four different parameters are plotted for the same station on one page.

Summary

This data organization and graphics program was successfully and very satisfactorily customized to meet District needs. Although it was only developed several months ago, we have already been able to make applications to most of our sixteen reservoir projects. While now essentially finished and operational, it could be further improved and/or expanded to be of more widespread use. A further conversion from Unix to Windows 95 would be appropriate. If time and funds were available, this isopleth plotting program could be modified to three dimensional graphics or possibly even animated graphics applications.



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01 MAY 1977
02 JUN 1978
03 JUL 1979
04 AUG 1980
05 SEP 1981
ENDING DATE
DAY MONTH YEAR
31 OCT 1977
NOV 1978
DEC 1979
1980
1981
SELECTION SUMMARY
Starting date: 01 MAY 1977
Ending date: 31 OCT 1977
Page Format : 1x1 Color Land
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4WCR 21002 00010
21003 00299
STATION 4WCR
4SHN 12305
4SHP 13001
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4SPE 20201
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4WFS 21403y
4WHO 21404y
4WIO 22112
4XRC 22130
4XTC 31002
4XTM 91002
ANALYTE
00010 ~WATER TEMP
01045 ~IRON
01055 ~MANGANESE
00400 ~FIELD pH
FUNCTIONS
Prt Numeric Tabulation Plot(s)
Prt Summary Report(s)
Dsp Numeric Tabulation Plot(s)
Dsp Color Image Plot(s)
Dsp Summary Report(s)
DONE
APPLY
HELP

Figure 3.

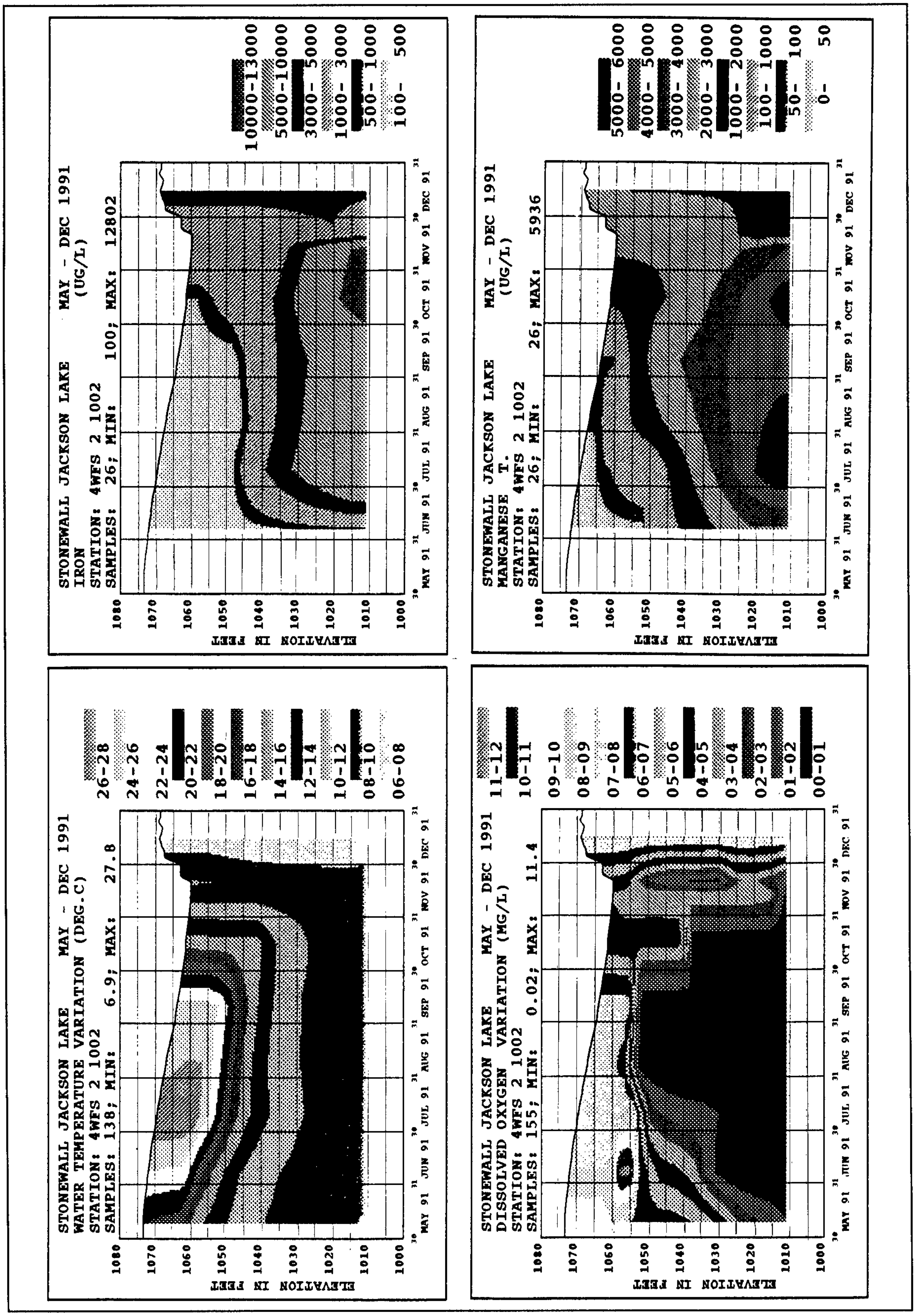
STONEWALL JACKSON LAKE MAY - DEC 1991
WATER TEMPERATURE VARIATION (DEG.C)
STATION: 4WFS 2 1002
SAMPLES: 138; MIN: 6.9; MAX: 27.8
STONEWALL JACKSON LAKE MAY - DEC 1991
IRON (UG/L)
STATION: 4WFS 2 1002
SAMPLES: 26; MIN: 100; MAX: 12802
STONEWALL JACKSON LAKE MAY - DEC 1991
DISSOLVED OXYGEN VARIATION (MG/L)
STATION: 4WFS 2 1002
SAMPLES: 155; MIN: 0.02; MAX: 11.4
STONEWALL JACKSON LAKE MAY - DEC 1991
MANGANESE T. (UG/L)
STATION: 4WFS 2 1002
SAMPLES: 26; MIN: 26; MAX: 5936
ELEVATION IN FEET

Figure 4.

Implementation of the New Ohio River Division Water Quality Database

by
Joseph E. Svirbely[1]

In 1994, a decision was reached division wide to implement a new Water Quality database. The recommended ORACLE database is the Corps standard database. EQUIS is a front end program that allows ORACLE to be efficiently accessed and queried. It was developed by the Hydrologic Engineering Center with ORDO and ORH as development sites. EQUIS has several advantages including quality assurance and sample tracking capability, and allows cost management of testing. The final design involves ORH-IM hosting the system with district and division access through the CEAP network. Training was given for the four districts at Huntington. Support will be given to ORD and each district in its own offices to fine tune the system to meet each district's way of doing business. The final system will yield a database located in one district but virtually transparent and with each district's customized flavor.

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Submersed Aquatic Vegetation Component for CE-QUAL-ICM Eutrophication Model

by
Carl F. Cerco[1]

Introduction

Submersed aquatic vegetation (SAV) is a valuable component of coastal and estuarine ecosystems. SAV provides habitat and nourishment to living resources of commercial and recreational value. In recent decades, SAV has been on the decline worldwide. The decline has been linked to cultural eutrophication (nutrient enrichment) of coastal waters. The nutrients stimulate algae that dwell in the water column as phytoplankton and on SAV stems and leaves as epiphytes. The algae diminish light available to the SAV. Without sufficient solar radiation, SAV beds decline or disappear entirely.

Chesapeake Bay is an example of a estuarine system with endangered SAV. Between 1965 and 1984, SAV coverage in the bay declined by 75 percent (Figure 1). Although a partial recovery has occurred, SAV coverage is still only half what it was 30 years ago. The recovery has been qualitatively linked to control of nutrients released to the bay. Nutrient controls have diminished algae and increased light penetration to SAV beds, thereby stimulating a partial SAV recovery.

Recently, a state-of-the-art eutrophication model package was applied to the bay (Cerco and Cole 1994). Primary focus of the application was to evaluate effects of nutrient controls on bottom-water anoxia. Effects of management activities on living resources were addressed indirectly through comparison of model results with "living-resources criteria." For SAV, light attenuation computed by the CE-QUAL-ICM eutrophication model was compared with maximum attenuation tolerated by healthy SAV.

Addressing living resources through computation of living-resources criteria has its limits. An approach that directly quantifies living resources is preferable to one that merely states if conditions are favorable or not. Moreover, living resources interact with their environment to produce effects that cannot be quantified through application of criteria. As part of a series of refinements to the Chesapeake Bay eutrophication model, living resources (zooplankton, benthos, SAV) are being incorporated into the model as state variables. The present paper describes the formulation of the SAV component and presents initial model results.

Model Formulation

The model builds on principles established by Wetzel and Neckles (1986) and Kemp, Boynton, and Hermann (1995). Three state variables are computed:

Shoots (aboveground biomass).
Roots (belowground biomass).
Epiphytes (attached algae).

Each state variable is computed as a density, e.g., gm shoot carbon per square meter of bottom area. The area of the SAV bed is computed also, as a fraction of the total area available to SAV.

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

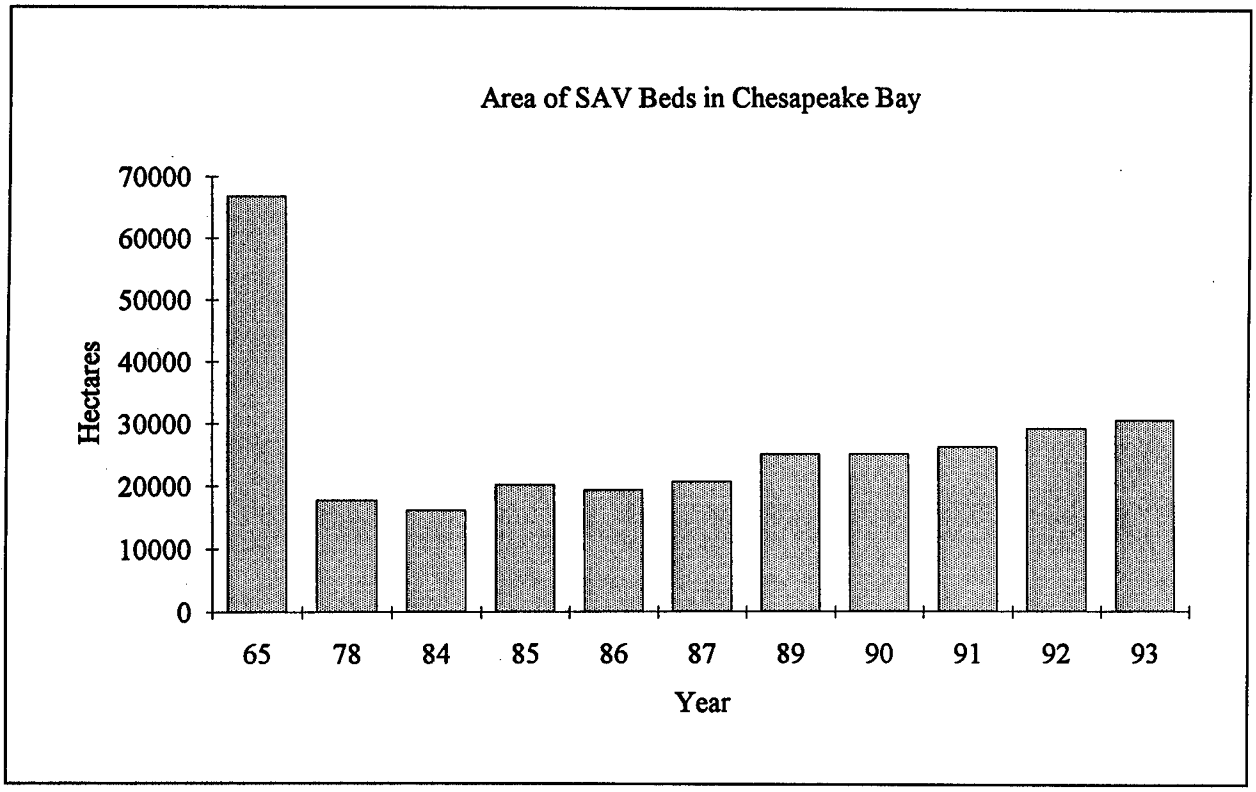


Figure 1. Submersed aquatic vegetation abundance in Chesapeake Bay, 1965-1993

Shoots

The basic relation governing shoot biomass is as follows:

$$\frac{d}{dt} C\,SH = (1 - Fpsr\,f(T))\,P\,C\,SH \qquad (1)$$

where

SH = shoot biomass, gm C m^{-2}

C = coverage, m^2

P = net primary production, day^{-1}

$Fpsr$ = fraction of primary production transferred from shoots to roots

$f(T)$ = temperature function

Individual relations for density and area are obtained through application of the chain rule and by separating the result into two equations:

$$\frac{d\,SH}{dt} = FD\,(1 - Fpsr)\,P\,SH \qquad (2)$$

$$\frac{d\,C}{dt} = (1 - FD)\,(1 - Fpsr)\,P\,C \qquad (3)$$

in which FD = fraction of total production that goes to increased/decreased density. The fraction is determined by an optimization algorithm that determines whether production is optimized through spatial expansion of the bed or through increasing shoot density within the bed.

Roots

The basic relation governing root biomass is as follows:

$$\frac{d}{dt} C\,RT = Fpsr\,f(T)\,P\,C\,SH - R\,C\,RT \qquad (4)$$

where

RT = root biomass, gm C m^{-2}

R = root respiration, day^{-1}

Epiphytes

The basic relation governing epiphyte biomass is as follows:

$$\frac{d}{dt} C\,EP\,SH = \qquad (5)$$
$$(Gep - Rep - PR - SL)\,C\,EP\,SH$$

where

EP = epiphyte density, gm epiphyte C gm^{-1} shoot C

Gep = epiphyte growth rate, day^{-1}

Rep = epiphyte respiration rate, day^{-1}

PR = predation on epiphytes, day^{-1}

SL = sloughing rate of SAV shoots, day^{-1}

Primary Production

Net primary production is the balance between growth, respiration, and sloughing of leaves. All of these are computed as functions of environmental variables including temperature, solar radiation, light attenuation (Figure 2), and nutrients (Figure 3) in the water and sediments.

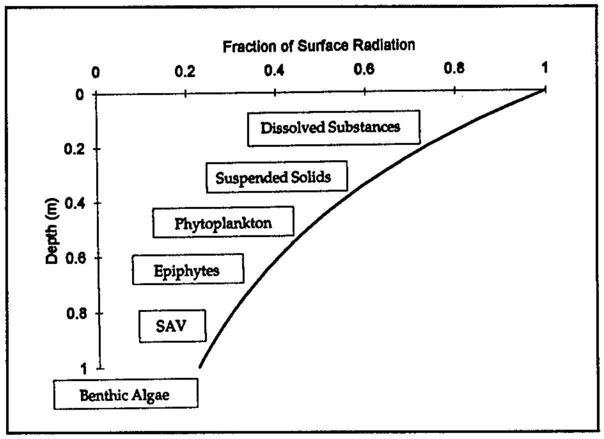


Figure 2. Factors affecting light attenuation simulated in model

Application to Chesapeake Bay

Estuaries are transition regions between rivers and the sea. Consequently, the estuarine environment varies from a freshwater regime to a marine one. A myriad of SAV species exist in Chesapeake Bay that reflect the varying environment. The approach adopted for the bay applies one model formulation to SAV throughout the bay, but allows for variations in model parameters in different regions of the bay. Three characteristic regions have been defined, based on ambient salinity. Within each salinity region, a characteristic SAV species or community has been defined. Model parameters within each region are specified based on the characteristic species. In the high-salinity region (>20 ppt), the target species is *Zostera marina*. In the moderate salinity region (10 to 20 ppt), a mixed community of *Potamageton perfoliatus* and *Ruppia maritima* is targeted. In the low salinity region (<10 ppt), a mixed community is also targeted consisting of *Myriophyllum spicatum*, *Vallisneria americana*, and *Hydrilla verticillata*.

High-Salinity Regions

Initial application of the model of *Zostera marina* was based on a data set assembled from

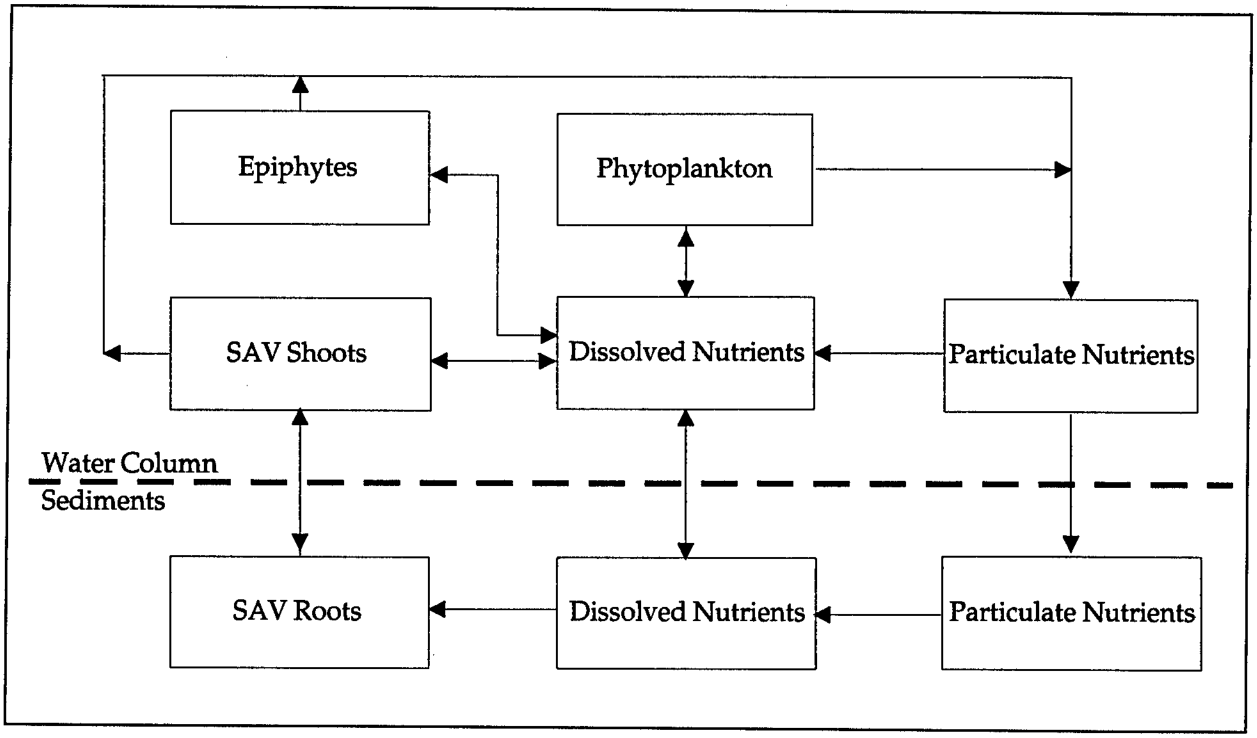


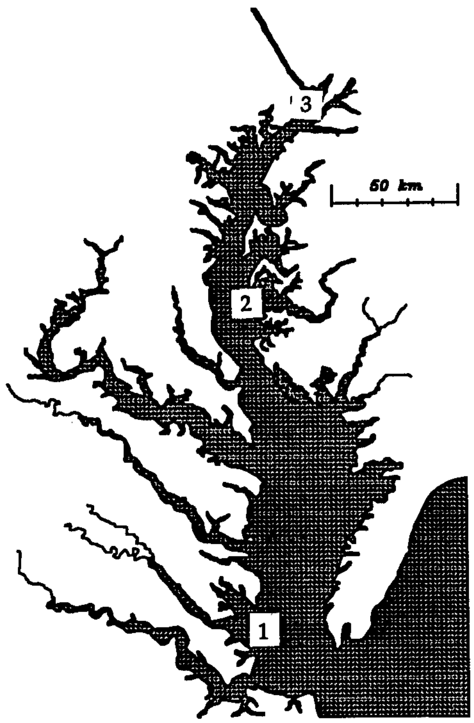
Figure 3. Nutrient cycling as simulated in model

a number of sources (Penhale 1977; Orth and Moore 1986; Buzzelli 1991; Wetzel and Penhale 1983; Moore et al. 1995). Data collection centered around the mouth of the York River estuary (Figure 4). Daily mean forcing functions (temperature, solar radiation, light extinction) were derived from the data (Figure 5). Nutrient cycling was not considered in the initial model application. The model was run, using 1/10 day time steps, until a repetitive annual SAV cycle was obtained. The annual cycle of shoot and root biomass computed by the model provided excellent agreement with the assembled data (Figure 6). Epiphyte data for comparison with the model was sparse, such that meaningful monthly means could not be computed. Magnitude and range of observations and model were in agreement.

Moderate-Salinity Regions

Initial application of the model with parameterization for moderate-salinity communities was to a data set collected in mesocosms (Kemp et al. 1983) characteristic of conditions in the central bay. The intent of the mesocosm experiment was to examine possible causes of the observed decline of SAV. Eight mesocosms (27 by 13 by 1.2 m) were constructed on the shores of the Choptank River (Figure 4). The mesocosms were stocked with ambient SAV and flushed with river water. After 2 years, nutrient levels in the ponds were manipulated, and the effect of nutrient levels on SAV was observed. Initial model application was to the control ponds that represented ambient conditions. This was a full application of the model and included computation of light extinction, phytoplankton abundance, nutrient cycling in the water and sediments, and SAV biomass.

Both the observations and the model (Figure 7) indicate that phytoplankton, epiphytes, and SAV rapidly remove available nutrients from the water column during the summer peak growth period. The sawtooth fluctuations in computed concentration represent flushing operations that temporarily restored the mesocosms to ambient river conditions. Light extinction, computed from calculated suspended solids and



1 = York River Site
2 = Choptank River Site
3 = Susquehanna Flats Site

Figure 4. Temperature (A), solar radiation (B), and light extinction (C) used in simulation of Zostera marina

chlorophyll concentrations, is well represented in the model. Temperature and solar radiation, imposed as boundary conditions, combined with computed nutrient concentrations and light extinction, allow for accurate computation of SAV abundance.

Low-Salinity Regions

Simulation of low-salinity SAV communities is based on a data set (Moore et al. 1995)

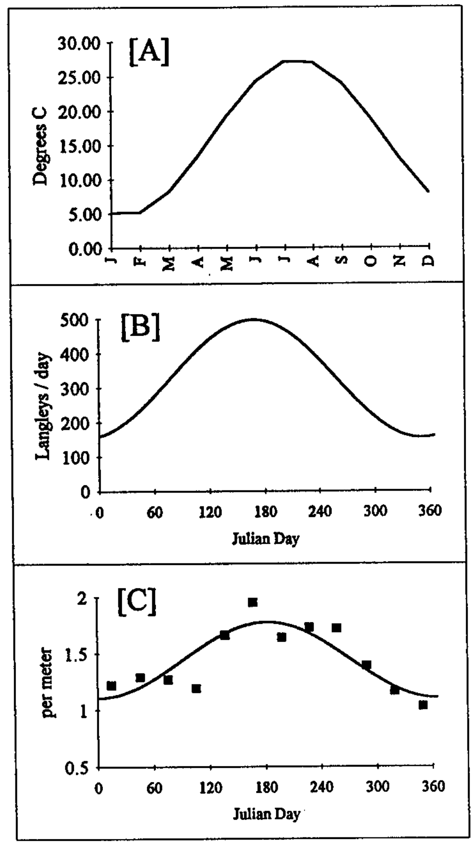


Figure 5. Chesapeake Bay showing model validation sites

collected in the Susquehanna Flats (Figure 4). Parameter evaluation is underway at present. Hence, results are not available at this time.

Where Do We Go From Here?

Results establish that one model formulation, supplied with different parameter values, can simulate SAV abundance in varying communities. Two additional validations of the model remain. The first is simulation of the low-salinity SAV community. The second is simulation of the response of SAV to nutrient loading, as demonstrated in the mesocosm studies. When the validations are completed, the SAV model will be applied in a multiyear simulation of SAV baywide.

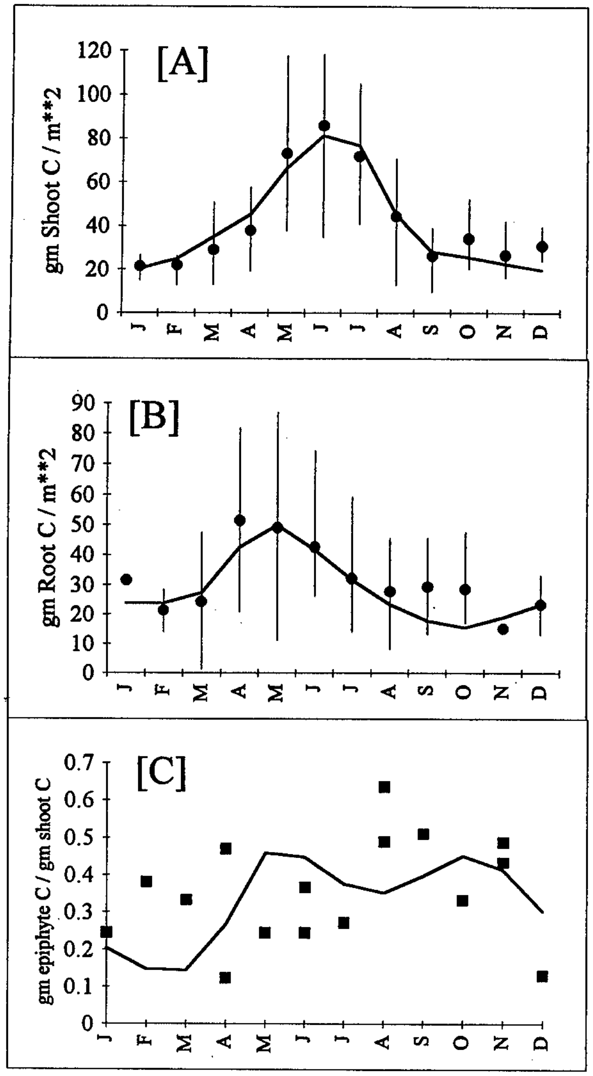


Figure 6. Observed and computed Zostera shoot density (A), root density (B), and epiphyte density (C)

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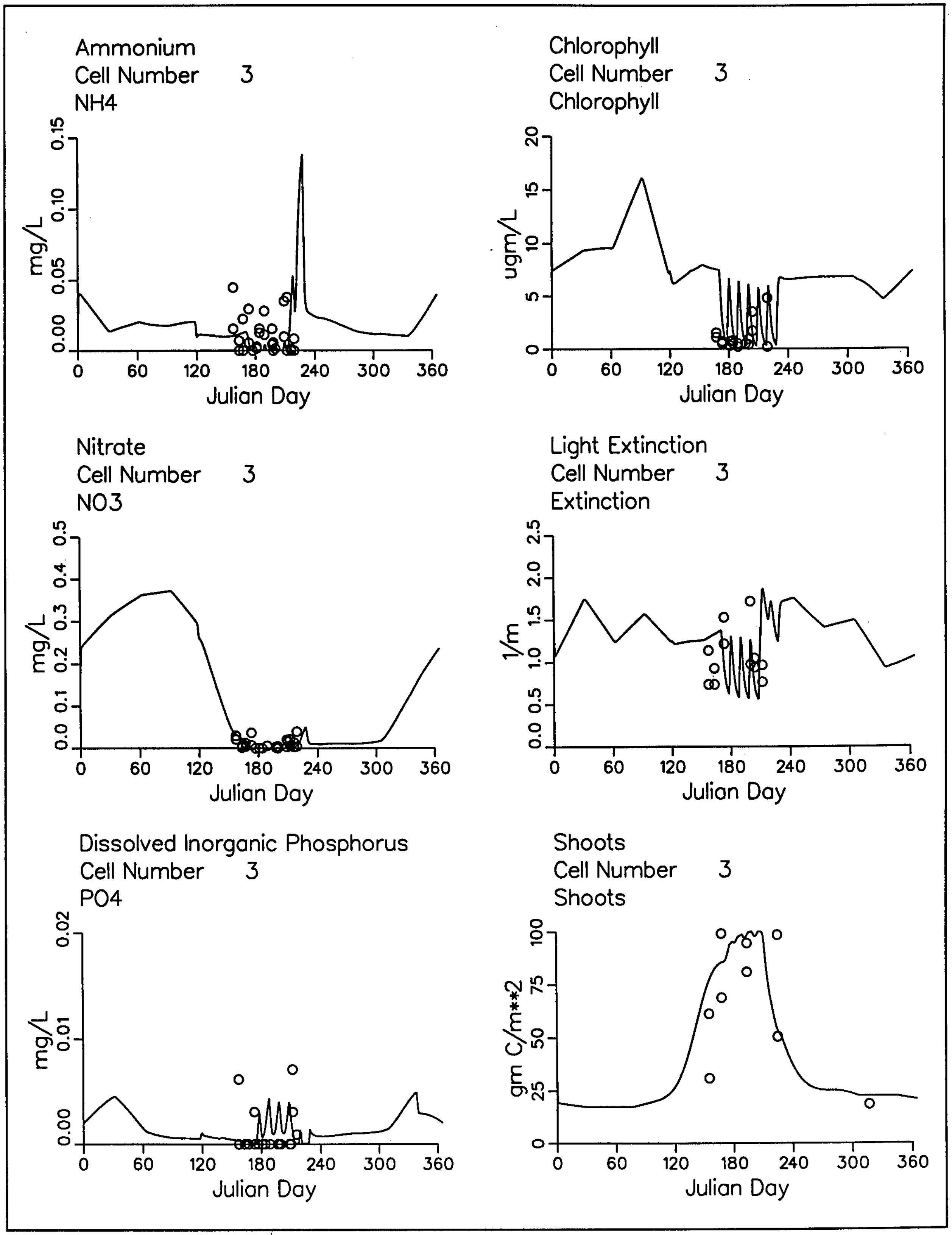

Figure 7. Simulation of Choptank River mesocosms

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Eutrophication of San Juan Bay and Estuary, Puerto Rico: Management Issues and Initial Assessment

by
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Introduction

Estuaries are productive and biologically diverse ecosystems. Since they occur at the narrow interface between freshwater drainage systems and the ocean, such ecosystems are also valuable sites for human activity and, thus, are often the locus of environmental impact. Included among these impacts is eutrophication resulting from the excessive influx of nutrients due to anthropogenic influences. Commonly-cited symptoms of eutrophication in coastal areas are reduced biological diversity, frequent blooms of phytoplankton, disease in commercially important fish and shellfish, reduced dissolved oxygen concentrations in bottom waters, and declining water clarity (Nixon 1993, Meybeck et al. 1990).

San Juan Bay and Estuary is one of 21 estuarine systems included in United States Environmental Protection Agency's National Bay and Estuary Program (NEP; United States Environmental Protection Agency 1993). The NEP was established in 1987 as part of the Clean Water Act to protect and restore estuaries while supporting economic and recreational activities. Principal environmental issues in the San Juan Bay and Estuary include loss of habitat, accumulation of debris and sediment, eutrophication, and toxic and pathogenic contamination. The design of efforts to ameliorate these and related problems will be based, in part, on the development and application of coupled mathematical models describing both hydrodynamics and water quality conditions for the bay and estuary system. Reported here are results of efforts to (1) characrterize current water quality conditions throughout the system, (2) identify critical environmental processes, and (3) provide data for validation of the water quality model.

Site Description

San Juan Bay and Estuary is an interconnected system of bays, channels and lagoons within the metropolitan area of San Juan in northeastern Puerto Rico (Figure 1). This area, which includes the municipalities of Cataño, Guaynabo, San Juan, Carolina, Loiza, and Toa Baja, has a total area of approximately 220 km^2 and a population of nearly 1 million people. Urban and residential development are the primary landuses in the drainage area. San Juan Bay is an international port servicing both cargo and touring vessels, and is Puerto Rico's major port of entry for industrial and domestic materials.

Major features of the bay and estuary system include San Juan Bay (ca. 7 km^2) to the west

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[3] San Juan Bay and Estuary Program, San Juan, PR.

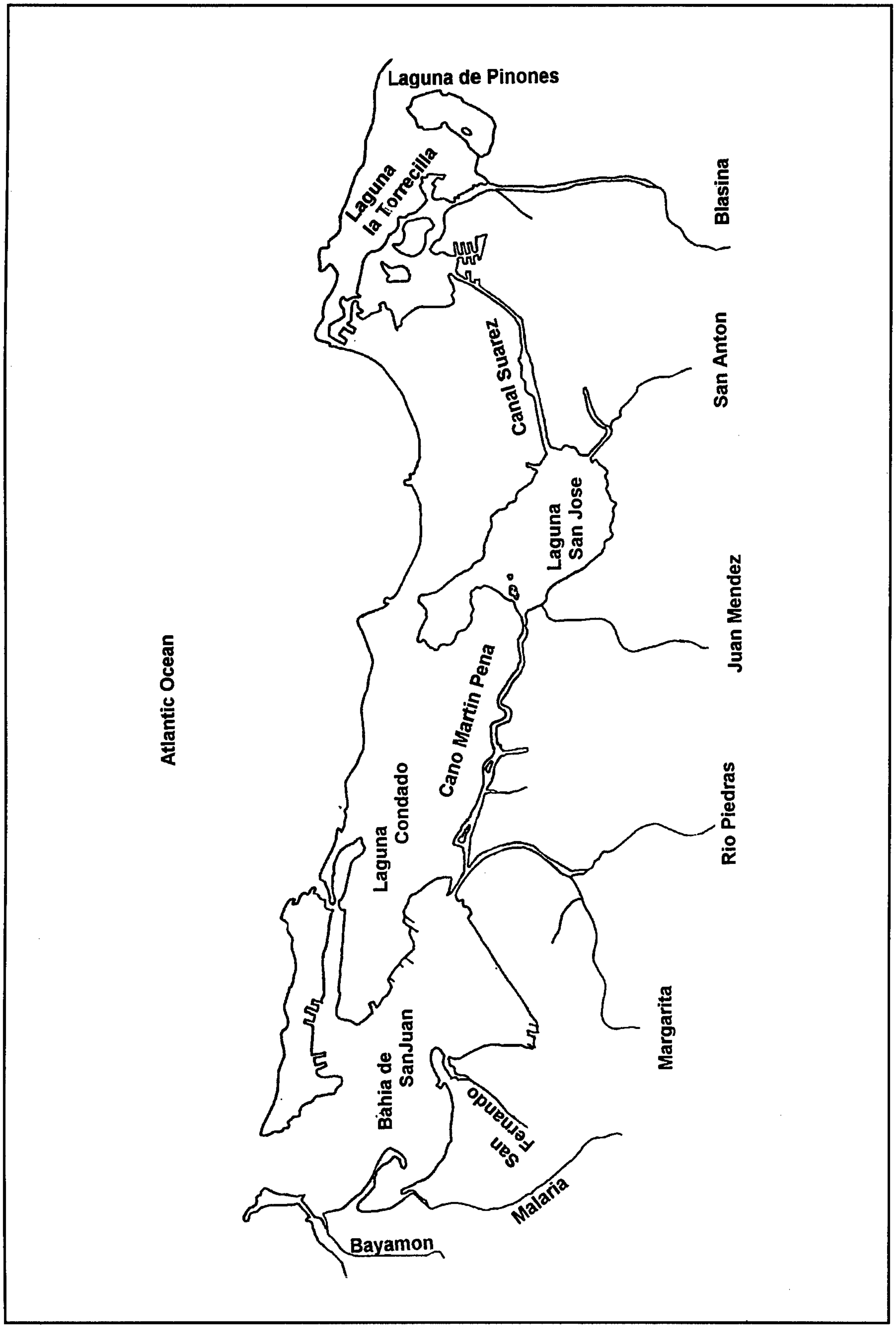

Figure 1. The San Juan Bay and Estuary system, San Juan, PR

and Laguna Torrecilla (2.5 km^2) to the east, both of which are open to the Atlantic Ocean, and Laguna San Josè (4.6 km^2), a shallow (mean depth = 1.5 m), centrally-located brackish lake connected to Laguna Torrecilla and San Juan Bay by Suarez and Martin Peña Canals, respectively. Laguna Piñones is located east of Laguna Torrecilla and within the State Forest of Piñones, an extensive (40 km^2) mangrove forest. While tidal exchanges in San Juan Bay occur across a deep, 0.9-km boundary at the mouth of the bay, those for Laguna Torrecilla are restricted by the narrow outlet at Boca de Cangrejos. Exchanges between Laguna San Josè and San Juan Bay are currently limited by shoaling and debris accumulation in the Martin Peña Canal.

During the last decade, municipal sewage was diverted to a regional treatment plant located outside the watershed and treated waste is now discharged through an offshore outfall. However, faulty sewage lines, numerous direct discharges of household wastes to nearby streams and ditches, and the lack of any sewage collection system in several heavily populated neighborhoods results in continued high rates of nutrient and pathogen export to the bay and estuary system. Such exports are particularly high following rainfall events due to washoff and the impacts of combined sewer overflows.

Methods

Twenty five sampling sites were established throughout the bay and estuary system. Stations were sampled at 2-week intervals during the study period (26 June to 24 August 1995) as a means to characterize water quality conditions in surface waters and at selected depths. In situ measurements of temperature, pH, specific conductivity or salinity, and dissolved oxygen (DO) concentration were taken at 1-m intervals at sampling sites having depths less than 10 m and at 2-m intervals at sites with depths greater than 10 m. Only near-surface (0.5 m), mid-depth, and near-bottom (0.5 m from bottom) measurements were taken at ocean sampling stations. Secchi disk transparency and water column depth were recorded at all sites on each sample date. Percent oxygen saturation was determined based on observed dissolved oxygen concentration, and salinity and temperature (Parsons et al. 1984).

Discrete water samples were taken at mid-depth for all sampling sites with depths less than 3 m, and near-surface (0.5 m) and near-bottom (0.5 m from bottom) for stations with depths greater than 3 m. Samples were retained in the dark on ice until returned to a field laboratory for sample preparation prior to shipment. Sample preparation and laboratory analyses for nutrients, solids and pigments followed standard methods (American Public Health Association 1992).

Sampling sites were established on 9 tributaries to the system as a means to assess material loadings. These sites were also sampled at 2-week intervals; however, routine sampling was discontinued after the second round of sampling. Tributary sampling for the remainder of the study period was restricted to Rio Piedras and Quebrada Juan Mendez, tributaries to Martin Peña Canal and Laguna San Josè, respectively. Samples were collected during and following selected rainfall events as a means to characterize storm-related material loading. Hourly and daily flows for Rio Piedras were obtained from the U.S. Geologic Survey gage located near Hato Rey and 1 km downstream from the water sampling location. Flows for Quebrada Juan Mendez were estimated at the time of sampling based on channel width and depth, and flow velocity measured with a rotating bucket flow meter.

Water samples were collected from a representative location across the stream section with a hand-held sampler. Samples for storm events were collected throughout the rising and falling portion of the hydrograph when possible. Samples were prepared, stored, and shiped as described above for surface water samples. Because of variable holding times, chemical analyses of storm water samples were limited to total nutrient concentrations of filtered and unfiltered samples.

Results

Nutrient concentrations for samples collected at nine stream sites on dates when flows were minimal (i.e., during routine sampling in July), were high and indicative of impacts due to urban point and non-point source influences (Table 1). Total phosphorus concentrations ranged from 185 to 1,940 $\mu g/\ell$ and had a median value of 527 $\mu gP/\ell$; on average, dissolved phosphorus accounted for 51 percent of the total phosphorus concentration. Total nitrogen ranged from 420 to 5,820 $\mu gN/\ell$ (median = 2, 010 $\mu gN/\ell$).

Changes in nutrient concentrations were observed during storm-related runoff events. In general, total and particulate phosphorus concentrations increased coincident with the rising limb of the hydrograph and declined through the peak and falling limb of the hydrograph while nitrogen concentrations decreased with increased flow. Approximations based on these observed concentrations and published flow records (e.g., Díaz et al. 1994), indicate that storm flows may account for as much as 80 to 90 percent of nutrient loads to surface waters.

Surface water salinity reflected the influences of tidal exchange and differences in flushing throughout major regions of the system (Table 2). Salinity values were lowest (median = 14.6°/$_{oo}$) in Laguna San José due to the influx of freshwater from two major streams (San Anton and Juan Mendez) and limited tidal exchange through Suarez and, particularly, Martin Peña Canals. Highest salinity values in this portion of the system were in bottom waters associated with the tidal wedge near the inflow of Suarez Canal.

Salinity values for San Juan Bay and offshore ocean sites were vertically uniform and typical of coastal waters. An exception was a sheltered, near-shore site located along the western edge of the bay. Surface values here where lower due to the influx of freshwater from Quebrada Malaria and the effects of reduced mixing. Salinity values in Laguna Torrecilla were high, yet clearly influenced by freshwater inputs from Blasina Canal and Laguna San Josè (via Suarez Canal). Similarly high salinity values were observed for Piñones Lagoon. The reason for this observation is unclear, since the existence a dense mangrove forest and only a

Table 1
Summary Statistics for Selected Nutrient Concentrations for 9 Streams Draining To The San Juan Bay and Esturay. Summary Includes Sample Size (n), Median Value, and Interquartile Range (difference between 25th and 75th Quartile Value; Qrange). Samples Were Collected Under Low-Flow Conditions During The Period 5-14 July, 1995

Variable	n	Median	Qrange
Total Phosphorus ($\mu gP/\ell$)	17	527	297
Particulate Phosphorus ($\mu gP/\ell$)	17	218	279
Dissolved Inorganic Phosphorus ($\mu gP/\ell$)	17	227	153
Total Nitrogen ($\mu gN/\ell$)	17	2,010	2,670
Total Kjeldahl Nitrogen ($\mu gN/\ell$)	17	1,100	750
Ammonia Nitrogen ($\mu gN/\ell$)	17	1,200	2,030
Total Organic Carbon ($\mu gN/\ell$)	17	14,545	5,256
Particulate Organic Carbon ($\mu gN/\ell$)	17	1,991	1,957

Table 2
Summary Statistics for Selected Physicochemical Variables for Surface Waters (depth ≤ 0.5m) of Major Regions of The San Juan Bay and Estuary. Summary Includes Sample Number (n), Median Value, and Interquartile Range (difference Between 25th and 75th Quartile Value; Qrange)

Variable		San Juan Bay	Laguna San Josè	Torrecilla and Piñones	Martin Peña and Suarez	Atlantic Ocean
Temperature (C°)	n	34	25	34	20	10
	median	29.7	30.2	30.2	29.8	28.4
	Qrange	0.8	0.9	1.3	1.1	0.3
Dissolved Oxygen (mg/ℓ)	n	34	25	34	20	10
	median	5.9	5.3	4.8	2.6	5.8
	Qrange	1.1	1.8	1.9	4.8	0.8
Oxygen Saturation (%)	n	34	25	34	20	10
	median	94.4	76.7	74.7	38.5	92.2
	Qrange	17.3	25.9	29.9	68.0	12.4
Secchi Disk Transparency (m)	n	34	25	34	20	8
	median	1.5	0.8	0.5	0.6	17.9
	Qrange	1.0	0.6	0.4	0.4	4.6
Salinity (°/$_{oo}$)	n	34	25	34	20	10
	median	36.4	14.6	28.8	17.3	37.1
	Qrange	1.2	1.3	3.6	16.0	1.3

single narrow interconnecting channel would suggest a limited hydraulic exchange with Laguna Torrecilla. Freshwater inputs to Laguna Piñones were apparently minimal during the study period.

Interregional differences were observed in percent DO saturation (Table 2). Such differences are important since they reflect the balance between important biological processes (i.e., production and respiration). While percent saturation was high for San Juan Bay and Ocean sites, values were depressed in other regions of the system. This was particularly evident in Martin Peña Canal where the influx of organic material would lead to high rates of microbial respiration. Reduced percent saturation values for Laguna San Josè, and for Laguna Torrecilla and Laguna Piñones reflect the impacts of the processing of both allochthonous and autochthonous organic matter.

Nutrient concentrations displayed high spatial variability throughout the bay and estuary (Table 3). While minimal for offshore sites and throughout much of San Juan Bay, total phosphorus and total nitrogen concentrations increased dramatically at sites with marked anthropogenic influences and limited flushing. Total phosphorus concentrations ranged from 156 to 532 $\mu g/\ell$ in the eastern portion of Martin Peña Canal and were high (median = 174 $\mu g/\ell$) in surface waters throughout Laguna San José. Total phosphorus concentrations were also high in that portion of Laguna Torrecilla influenced by inputs from Blasina Canal and were unexpectedly high in Piñones Lagoon.

Table 3
Summary Statistics for Nutrient and Chlorophyll Concentration Data for Surface Strata (depth≤0.5m) of Major Regions of The San Juan Bay and Estuary. Summary Includes Sample Number (n), Median Value, and Interquartile Range (difference Between 25th and 75th Quartile Value; Qrange)

Variable		San Juan Bay	Laguna San Josè	Torrecilla and Piñones	Martin Peña and Suarez	Atlantic Ocean
Ammonia Nitrogen ($\mu g/\ell$)	n	40	32	40	30	10
	median	30	165	30	355	20
	Qrange	105	190	65	480	80
Kjeldahl Nitrogen ($\mu g/\ell$)	n	41	32	40	30	10
	median	40	440	130	390	nd
	Qrange	70	280	305	370	60
Dissolved Inorganic Phosphorus ($\mu g/\ell$)	n	40	31	39	29	10
	median	5	70	6	68	2
	Qrange	7	62	15	92	4
Total Phosphorus ($\mu g/\ell$)	n	40	32	40	30	10
	median	34	174	91	171	5
	Qrange	39	47	97	117	20
Total Organic Carbon ($\mu g/\ell$)	n	34	25	31	23	8
	median	3,768	9,700	10,026	8,433	2,731
	Qrange	2,434	2,853	9,430	3,573	4,374
Particulate Organic Carbon ($\mu g/\ell$)	n	31	25	31	23	8
	median	703	879	2,170	1,882	453
	Qrange	1,251	1,239	5,461	2,072	1,034
Chlorophyll ($\mu g/\ell$)	n	39	29	38	24	8
	median	4.0	23.6	25.1	29.3	0.3
	Qrange	5.3	36.3	36.1	33.4	0.4

Similar spatial patterns were observed for the distribution of total nitrogen concentrations. Nitrogen concentrations at off shore sites, in San Juan Bay and throughout much of Laguna Torrecilla were relatively low (Table 3), while median total Kjeldahl and ammonia concentrations in the Laguna San José and Martin Peña and Suarez Canals were as high as 440 and 335 $\mu g/\ell$. As was observed for total phosphorus, total nitrogen concentrations in Laguna Piñones and near the confluence of Blasina Canal and Laguna Torrecilla were relatively high.

In general, chlorophyll *a* concentrations were high and Secchi disk values low coincident with high nutrient concentrations (Table 3). Water clarity and chlorophyll *a* concentrations in San Juan Bay and at offshore sites were similar to those observed for other Puerto Rican coastal sites (e.g., Gilbes et al. 1996). However, unexpectedly low chlorophyll a concentrations were observed at two central sites in Laguna San José, despite relatively high nutrient concentrations. High rates of particle removal due to filtration by mussels *(Perna sp.)*, dense beds of which were observed in this area during sediment sample collection (pers. comm., T. Miller-Way, Marine Environmental Sciences Consortium, Dauphin Island, AL), may be responsible for low algal biomass. Similar impacts of mussel grazing on phytoplankton biomass have been report for other marine systems (e.g., Prins et al. 1995).

Discussion

Water and environmental conditions varied markedly throughout the San Juan Bay and Estuary system. Anthropogenic impacts (e.g., excessive nutrients and elevated chlorophyll concentrations) were pronounced, particularly for water bodies with limited flushing and significant freshwater inputs from the surrounding watershed. Characteristics of the major ecological regions of the system, as they relate to efforts to identify and evaluate management alternatives, are summarized below.

a. *San Juan Bay Region*

(1) Hydrodynamics and material transport dominated by tidal exchange.

(2) Gradients in material concentrations related to dilution, sedimentation and incomplete mixing.

(3) Low to moderate external material loads.

b. *Laguna San Josè Region*

(1) Brackish to fresh water.

(2) Extremely high external material loads, especially following runoff events.

(3) Limited tidal exchange and reduced flushing.

(4) Potential for internal material loading due to resuspension and sediment/water interactions.

(5) Hydrodynamics dominated by wind and inflow mixing.

(6) Algal production controlled by nonalgal turbidity, mixing and biological interactions.

c. *Laguna Torrecilla and Laguna Piñones Region*

(1) Moderate salinity and water quality gradients.

(2) High autochthonous detritus and carbon loads from mangrove forest.

(3) Moderate to high allochthonous nutrient and organic loads.

(4) Moderate to high tidal exchange.

d. *Coastal /Oceanic Region*

(1) Limited algal production; high water clarity.

(2) Minimal anthropogenic influences.

Conclusions

San Jose Bay and Estuary exhibits symptoms of cultural eutrophication, including elevated nutrient concentrations and excessive algal biomass. Conditions are most severe in the Laguna San José region due to limited flushing and the influx of nutrients from urban and residential landuses. Considering the apparent importance of these nutrient loads from point and nonpoint sources, particularly during storm events, efforts should be made to better define loading sources and processes. Also needed is a better definition of non-nutrient influences on plankton production, particularly as related to the potential importance of bivalve grazing in Laguna San José.

Acknowledgments

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Creating a Self-Replenishing Environment Through the Application of an Energy Equation

by
Julie Conaway[1]

Introduction

The ecology of an area can be drastically altered as a result of human interaction. Many of the environmental problems today are a result of natural ecosystems being unable to survive changes made by man. In the Northwest, watersheds support a large number of wildlife species, plant life, and aquatic life. At the same time, these basins also provide a viable economic base for urban and rural communities. Natural resources are overutilized for the most part, making it difficult for watersheds to recover naturally. The existing environmental problems need to be examined to determine where failure occurred in allowing the environment to be self-replenishing. If the current systems were to be examined and modified on a large scale to benefit the environment, future rehabilitation needs could be lowered.

Environmental Problems Presently Encountered

Salmon

On the Columbia and lower Snake rivers, many serious problems are surfacing. The once seemingly unlimited anadromous fish runs are on the brink of extinction. The Snake River sockeye, as well as Snake River spring, summer, and fall chinook have all been added to the endangered species list. More than 200 dams of various sizes have been built in the Columbia River Basin, which have limited the access of the salmon to half of their original spawning grounds. In addition, many of their original streambeds are no longer suitable for breeding due to sediment and increased water temperature.

Other Threatened Species

Many other wildlife species are also in decline. Populations of Northern spotted owl, marbled murlett, grizzly bear, and gray wolf are declining because of habitat destruction. Bald eagle numbers are decreasing due to habitat depletion and exposure to DDT, and Kootenai River white sturgeon are experiencing difficulty reproducing due to habitat modifications.

Contaminants

Many harmful contaminants, such as heavy metals, dioxins and furans, polychlorinated byphenyls (PCBs), and various pesticides, have been found in the lower Columbia River and its estuary. These contaminants have been shown to be detrimental to fish and wildlife and are a potential threat to humans. The increasing levels of PCBs can be attributed to the various industries located along the Columbia and Snake rivers and their tributaries. Evidence suggests that the pesticide DDT and its by product DDE are still entering the water from runoff 22 years after being banned, but they also reside in soils and are mobilized once again during dredging (Berwick 1995). Many heavy metals enter the environment from fuel exhausts and mining.

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[1] U.S. Army of Engineer Division, North Pacific, Portland, OR.

Habitat Destruction

Much of the decline in wildlife can be attributed to the destruction of habitat. As man moves farther from the cities, it can be predicted that the natural areas of the United States are going to grow smaller leaving less room for wildlife.

Dams and Hydropower

The Columbia River is the second largest river in the United States. Fifteen main stem, power-producing dams lay on the Columbia and lower Snake River. The hydropower supplied by these dams makes up 65 percent of the power in the Northwest. Unfortunately, a substantial amount of the natural spawning grounds for anadromous fish have been flooded by the construction of these dams. In addition, some of the dams' fish passage facilities have been inadequate in allowing for the safe migration of the fish up and down the river.

When water is spilled over the spillways of the dams, dissolved gas can reach damaging levels. If the total dissolved gas saturation percent is too high, symptoms of Gas Bubble Disease can occur in fish. This can injure or be fatal to fish. Power-producing turbines can lower chances of survival by 3 to 15 percent if fish pass downstream through them. Fish survival can also be threatened by predator fish and birds waiting below the dams to catch their disoriented prey.

Logging

The timber industry has provided a valuable economy for the Northwest for many years, but at the expense of many forms of life. Clearcutting an area increases the runoff volume and erodes the topsoil, depositing the sediment into the streams. This sediment settles on the bottom of the streams covering the gravel necessary for salmon spawning. If the riparian vegetation is removed alongside the stream, increases in water temperatures and decreases in protective cover will occur. Along with destroying important salmon habitat in the nearby streams, logging destroys habitat for the animals living in the forested areas.

Agriculture and Cattle Gazing

Another important factor in the destruction of habitat is the farming industry. Farmers divert river water from the Columbia River to irrigate more than three-million acres[1] of farmland in Oregon and Washington (Mighetto and Ebel 1995). This can lower water levels needed for fish passage. Fish can also be inadvertently diverted down irrigation channels. The farming industry uses a large amount of fertilizers and pesticides that can find their way into the streams. This raises the levels of contaminants in the system that can accumulate in the food chain. These chemicals can also change the nutrient contents of the water. If this occurs, the algae population could reach unhealthy levels and may take needed oxygen away from the fish.

Grazing animals can have a detrimental effect on the streams. Cattle can destroy streambeds if they trample through the fish spawning gravel. Riparian vegetation, which provides bank stabilization and shade, is also threatened by the presence of grazing livestock, which has a direct effect on the health of the stream. Another consideration is the high potential of accelerated erosion into streams from rangelands caused by severe summer storms and the lack of vegetation.

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[1] A table of factors for converting non-SI units of measurement to SI units is presented on page xxx.

Competing Water Uses

So much is being taken from the environment that the ecosystem is unable to repair itself. In a watershed like the Columbia Basin, many industries depend on the river. The power industry relies on the water to turn the turbines for power production. Commercial industries use the cool river water to lower temperatures of mechanical processes. They also use the river to diffuse their waste. Farmers use the land near the rivers and depend on the water for irrigation. The logging industry uses the forests as its wood supply. These trees are then manufactured into usable lumber and transported down the river. The commercial fishing industry depends totally on the harvesting of fish for its livelihood.

Along with industry, there are a large number of other water users that depend on the Columbia River. Recreational users depend on the river for boating, fishing, and windsurfing. Many like to go to the river to relax and connect with nature. The Native Americans depend on the Columbia anadromous fish for religious reasons and as a major source of food. Also, many port towns along the Columbia and Snake rivers rely on the ships moving up and down the waterways for trade.

Energy Equation for the Environment

The quality of the Northwest environment has deteriorated since the area has come under such extensive use. Many of the uses today may not be available in the future if there is a continual taking from nature. One way to prevent this extinction of valuable resources is to take a holistic view of the system. An energy equation could be developed for the environment in which the variables "removing health" are labeled as negative and those "restoring health" are positive. These variables can then be summed up to represent the energy of the ecosystem. Efforts could then be made to lower the value of the negative variables by having them take less from the environment. This would result in a higher energy (or health) of the environment. Although this equation has not been developed, it may be important to look at its concepts to see what needs to be done.

Raising the Health of the Environment

Many things can be done to lower the damage done to the environment. The hydropower dams can use various methods of operations to lower water temperatures, increase water velocity, or simulate more natural river conditions. Fish passage facilities can be modified to better accommodate both adult and juvenile fish.

Reservoirs and streams can also be improved for wildlife. Controlled floods can be simulated artificially to add nutrients and foods for fish. The flooding of banks can also temporarily create an area that is sparsely populated with fish, which naturally stimulates the reproduction and growth processes in the resident fish (Ploskey 1981). Fallen trees can provide habitat for both fish and birds, as well as reduce turbulence and erosion deposition on fish eggs. This shelter also provides protection from winds and waves for anglers.

The timber industry can employ the methods of selective cutting as opposed to clearcutting an area. In Wyoming, a study was done comparing an area that had been thinned by 60 percent to an area that had been clearcut. In the thinned area, water runoff increased by 92 percent, but after 2 years there was no significant increase in nitrate or total nitrogen outflow. In the clearcut area, water runoff increased by 227 percent. The nitrate concentrations increased by 10 to 40 times, while the total nitrogen outflow increased by 37 times. This leads to the conclusion that the surviving trees in a thinned stand are important for nutrient immobilization (Knight, Yavitt, and Joyce 1991). If an area is thinned, the logging companies may also be able to return sooner to remove some of the older, larger trees, instead of having to wait for the entire forest to grow from saplings.

Farmers could also help reduce the contaminants they put into the environment by limiting their use of fertilizers, pesticides, and herbicides. If crops were restricted along the streams, deposition of these chemicals may be reduced, and the protective riparian vegetation could be allowed to grow. Livestock could also be kept from grazing in the riparian zone of streams, and a minimum vegetative cover could be required on rangelands to prevent erosion.

In-river commercial fishing on the Columbia River was prohibited for the first time in 1994 because of the small numbers of fish returning. This allows more fish to move upriver, but a large number are still being caught in the ocean. If catch limits were lowered out in the ocean as well as in-river, the fishing industry would be able to continue, although at a decreased level, until the fish numbers were built up again. Certain methods of fishing (i.e., gill-netting) may need to be restricted so protected runs could be released if caught.

If all of these factors were modified (dams, logging, agriculture, and fishing), natural areas should be able to recover. No one item is going to raise the "environment's energy" enough to fix the problems, but a combination would help get closer to this goal. The following are some examples of how simple modifications have been implemented to raise the health of one aspect of the ecosystem. Benefits have arisen from these changes, but because the ecosystem is so vast and incorporates so many variables, these small changes are only the beginning.

Willamette Falls, Willamette River, OR

Willamette Falls is a natural 40-ft horseshoe-shaped falls that historically blocked fall chinook and summer steelhead runs on the Willamette River. These fish were only able to pass during periods of high water or floods. Stemming from a desire for more sustainable anadromous runs, crude fishways were built into the rock as early as the late 1800s and have since been modified. The present fish passage facility was completed in 1971. As a result of improved passage, the river now supports spring chinook, winter steelhead, fall chinook, coho, and summer steelhead populations. Providing new habitats and available areas for fish will only help fish populations, especially since so much of their previous habitat is no longer available. Willamette falls is also a good demonstration of how a conscious decision for more consistent fish runs and the implementation of necessary changes led to the desired result.

Colorado River

The Glen Canyon Dam was built 30 years ago on the Colorado River and has fundamentally changed the way the river runs. Downstream of this dam, sand and silt are no longer being deposited, which has put an end to the large expansive sandbars that had historically existed. Before the dam was built, 66 million tons of sediment used to move down the river each year. Now only 91 thousand tons are moving. The remainder of this sediment is being deposited behind the dam in Lake Powell. One way the U.S. Bureau of Reclamation is proposing to combat this trend is to generate a small flood in the spring, which will simulate pre-dam conditions. This change in project operations may be able to counter changes made to the environment resulting from the implementation of the Glen Canyon Dam. They hope to renew sandbars and riparian vegetation along the river, which will aid in bringing the surrounding environment closer to what had existed historically.

Elk Creek, Rogue River

On Elk Creek, minor modifications helped turn an abandoned waste area into a productive habitat. In 1989, the Corps of Engineers converted a retired rock quarry near the construction site of the unfinished Elk Creek Dam into a wetland and upland habitat. A lake had naturally formed from the excavation of rock, so a drain was installed in the small lake to maintain the water level. Dissipaters were then placed on the downside slope to prevent draining water from eroding away the soil. The wetland vegetation grew naturally over the following years, and now various waterfowl populations are beginning to thrive. This effort of restoring the environment allowed nature to reestablish itself, which resulted in a much healthier, more productive environment.

An Integrated Approach

The previous examples showed how one factor was modified to benefit one aspect of the environment (fish migration, wetland life, and riparian vegetation). The following describes the Columbia River System Operating Strategy, which is a recent attempt to evaluate the Columbia and Snake rivers as a system and determine the best way to operate the projects on these rivers together to benefit the system as a whole.

In 1991, the System Operation Review (SOR) Interagency Team was formed with members from Bonneville Power Administration, the Corps of Engineers, and the Bureau of Reclamation. The purpose of this group was to evaluate several operation strategies for the Columbia River and Snake River system to determine the most beneficial to the system, with anadromous fish as one of the driving forces. A 5 year study was completed, analyzing several alternatives for the system. A Preferred Alternative was established and the Final Environmental Impact Statement was released in December 1995. This study was a major undertaking, focusing on the Columbia River system as a whole.

The intent of the Preferred Alternative is to store water in the reservoirs in the fall and winter to meet spring and summer flow targets during the anadromous juvenile downstream migration. Maximum summer draft limits are to be used to try to minimize detrimental effects on other wildlife. Flood protection will still be provided, along with a reasonable amount of power generation. Instead of only focusing on the project operations, the Preferred Alternative proposes other measures to assist in fish recovery. A combination of in-river passage and the transporting of juveniles is one way the Interagency Team attempts to balance water needs, gas levels, and fish survival. Another way is the coordination of spill, drawdown, and flow augmentation.

This change in river operations will affect the wildlife in both positive and negative ways. It is predicted that the Preferred Alternative will aid Methow summer chinook, Hanford fall chinook, and Snake River fall chinook. Implementing the plan should also provide significant benefits to Kootenai River white sturgeon and improve conditions for resident fish at many, but not all, of the reservoirs. There will be a decrease in habitat, however, especially at Lake Umatilla above John Day Dam, and there will be a decrease in waterfowl, bald eagle, western painted turtle, beaver, and otter populations.

The Preferred Alternative was compared with six other alternatives. Some of the other alternatives had higher anadromous fish survival rates or higher benefits to other wildlife. The Preferred Alternative was chosen as the most beneficial to the system as a whole, including economically. One item the Preferred Alternative does not require is that the other contributing factors (industry, agriculture, etc.) modify their uses of the system. The SOR study conducted on the Columbia and lower Snake rivers tried to determine the most beneficial operating plan for the river system as a whole, but did not look past the dams and projects. The Preferred Alternative cited by the Interagency Team has obvious benefits and drawbacks. This exemplifies much of the future struggle that will come with attempting to balance benefits and drawbacks in an effort to raise the health of the environment.

Summary

These specific projects are all examples of ways conscious efforts can be implemented to try and improve conditions. From project inception, all potential environmental problems should be considered in order to balance ecosystem needs with the proposed changes to the system. A conscious decision must be made initially regarding the type and quality of ecosystem desired after the implementation of the new project. The surrounding ecosystem and the project should be constructed together in hopes of creating a self-replenishing environment. This should lower the need for future rehabilitation efforts because plans would already be made for the problem, and the ecosystem would have a better chance of recovery. As a whole, man needs to take less from the environment to allow for the recovery of what has already been done. The many groups of people harming the environment need to take responsibility for raising the energy again. If the health is raised enough to allow the environment to be self-sustaining, natural resources can be preserved for future use.

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Evaluation of Sluicing Versus Station Service Unit Releases for Aquatic Habitat Improvement Below Buford Dam, Gwinnett County, Georgia

by
Jerry D. Jones[1] *and John E. Zediak*[1]

Buford Dam and Powerhouse forms Lake Sidney Lanier and is located on the Chattahoochee River in Gwinnett and Forsyth counties, Georgia, approximately 35 miles northeast of Atlanta and 4.5 miles northwest of Buford, GA. The 1,040 square-mile drainage area above the dam lies on the southern slope of the Blue Ridge Mountains and is characterized by steep slopes and mountain streams. These physical attributes and other special features make Lake Sidney Lanier one of the most visited lakes in the United States. The Chattahoochee River below Buford Dam can best be characterized as having broadly rolling hills and occasional low mountains. Over this section, the Chattahoochee River is contained within high steep banks that create a rectangular river channel (Figure 1). Channel widths range from 150 to 300 ft while containing the dam's peak hydropower pulse of 9,600 cfs. Exposed rock outcroppings form an

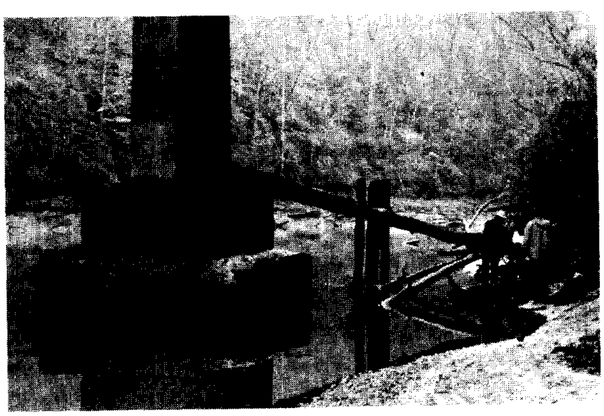


Figure 1. View of Chattahoochee River below Buford Dam

irregular riverbed in the river section. A thin layer of sand and gravel ranging from approximately 2 to 5 ft in depth overlays other portions of the top of rock. The river flows downstream 30 miles to Morgan Falls Dam dropping approximately 50 ft and creating a hydraulic gradient of 1.6 ft per mile. The project is the headwater dam for the series of dams that make up the Apalachicola-Chattahoochee-Flint Waterway System. Buford Dam is a multiple purpose project with primary authorization to provide flood control storage and generate hydropower and low-flow augmentation on the lower Chattahoochee River. Secondary benefits include water supply, public recreation, and fish/wildlife conservation.

Buford Dam is operated as a peaking power plant with power normally being generated Monday through Friday during the afternoon hours. The water management plan provides a minimum of 2 hr of generation 5 days a week under peak capacity as long as the lake level is in the conservation pool. Flows during peaking power generation average about 9,600 cfs through two 40 and one 6 MW generators. When peaking power is not being produced, a minimum release of approximately 600 cfs is made through the station service unit (6 MW) for downstream water supply, fish and wildlife, and water quality and to supplement navigational needs.

Lake Sidney Lanier is a fairly deep lake (over 150 ft at the dam) that maintains cold water at its depths year round. A benefit of

[1] U.S. Army Engineer District, Mobile; Mobile, AL.

these cold waters has been that the releases made from Buford are cold year round; as a result, the tailwater of Buford has been able to support one of the southernmost trout fisheries in the country. Like most deep reservoirs in the southeastern United States, the lake stratifies during the warmer months of the year, resulting in lower dissolved oxygen (DO) levels and elevated metal concentrations in the deeper cold waters of the reservoir. Ironically, without the cold water coming from Lake Lanier, there would be no trout fishery in the river downstream of Buford Dam. On the other hand, the same cold waters that breed aquatic life in the tailwater early in the year diminish aquatic life later in the year when DO levels are low and metal concentrations are high. During the severe stratification period (August-December), the release waters from Lake Lanier routinely do not meet State standards for DO, iron, and manganese during low-flow generation (station service unit operating only). Associated with the decrease in water quality is the elimination of a vibrant cold water fishery in the tailwaters during this period. By matter of chance, it was observed during a sluicing operation (maintenance of minimum releases while the house service unit is being repaired) that sluice releases (equal to the station service unit releases) made during the stratification period exhibit significantly higher DO concentrations while elevated metal concentrations decreased.

In an attempt to document the observed phenomena, Georgia Environmental Protection Division and Mobile District personnel performed the following test during December 4-6, 1995 at Buford Dam. Continuous reading water quality equipment was strategically placed at four locations (750 ft, 3 , 8, and 15 miles) below Buford Dam to measure DO and temperature. Additionally, several pull samples were taken at each site on different occasions for the purpose of measuring metal concentrations. The following 3-day test was performed at the dam, and the previously mentioned water quality parameters were recorded at each site.

- **Monday, December 4, 1995.** Normal operation. Station service unit operated alone from 0001-1545. From 1545-1745, peak generation occurred (station service unit + two main units). The remainder of the day the station service unit operated alone (this operation constituted normal operation).

- **Tuesday, December 5, 1995.** Normal operation + sluice. Station service unit operated alone from 0001-1450. At 1450, the station service unit releases were replaced with releases from the sluice. From 1545-1745, peak generation occurred along with the sluice. At 1745, peak generation ceased, and only the sluice continued to operate for the remainder of the day.

- **Wednesday, December 6, 1995.** Returning to normal operation. Sluice operated alone from 0001-1450. At 1450, the sluice unit releases were replaced with the station service unit releases. From 1545-1745, peak generation occurred. At 1745, peak generation ceased, and only the station service unit continued to operate for the remainder of the day.

Operating under the assumption (based on past observation) that peaking generation produced releases with adequate water quality values, the real effort of this test focused on the comparison of sluicing water versus releasing water solely through the station service unit. The monitoring station at the dam showed a strong increase in DO with sluicing as indicated in Figure 2. Due to the natural terrain of the river, as the release waters flow over rock ledges and other oxygen-forming mechanisms, one would expect that natural reaeration would occur fairly rapidly within this portion of the river. Therefore, the measurable effects of sluicing would dampen out within a few miles (assuming no other oxygen-depleting sources existed below the dam).

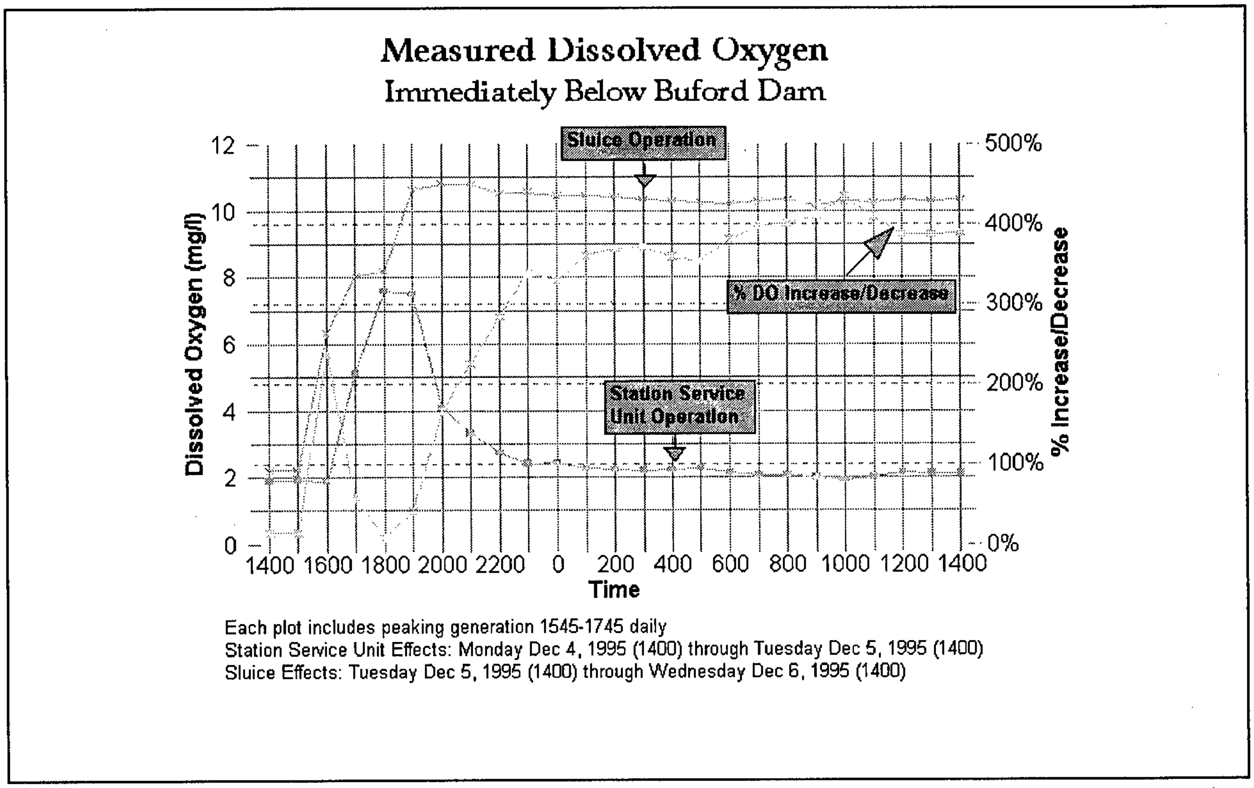


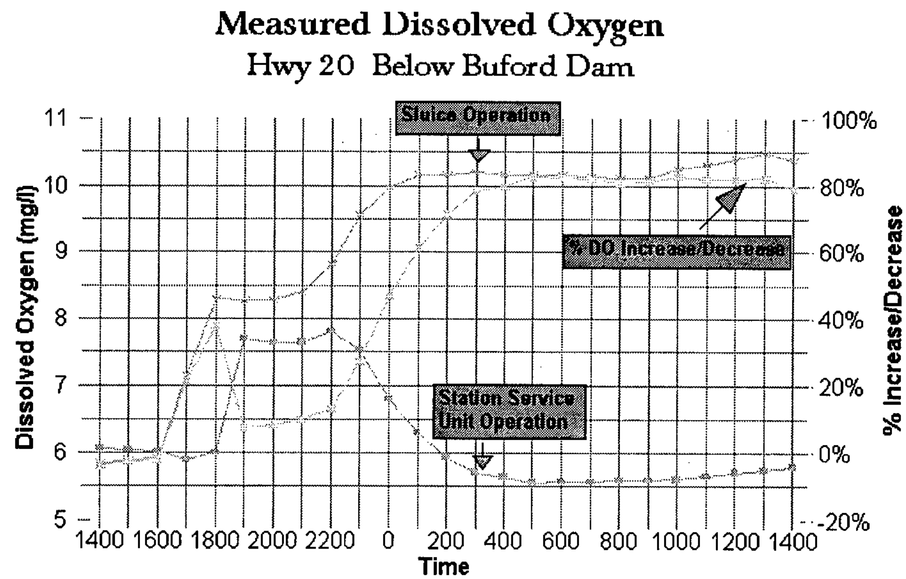
Figure 2. Measured DO immediately below Buford Dam

However, water quality samples taken in the past at the Buford Trout Hatchery located between the dam and Georgia Highway 20 have shown that DO levels during the severe stratification period are often still extremely poor by the time the water reaches the hatchery (under low-flow conditions). The test results observed at Georgia State Highway 20, which is approximately 3 miles below the dam and 1 mile below the Buford Trout Hatchery, strongly suggest that sluicing continues to significantly increase the DO at this point in the river (Figure 3). However, the other sites sampled (Figures 4 and 5) during this test reflected insignificant increases in DO. Additionally, iron and manganese measured adjacent to the Buford Trout Hatchery (not during this study, but during the same period of the year) show a strong inverse relationship between DO and these two parameters (Figures 6 and 7). It is expected that measured iron and manganese concentrations taken during this test would have yielded similar results.

Results of the test strongly suggest that the sluicing versus station service unit operation significantly increases DO and reduces iron and manganese concentrations downstream to Georgia Highway 20 (3 miles away). The effects at the other stations are not so evident.

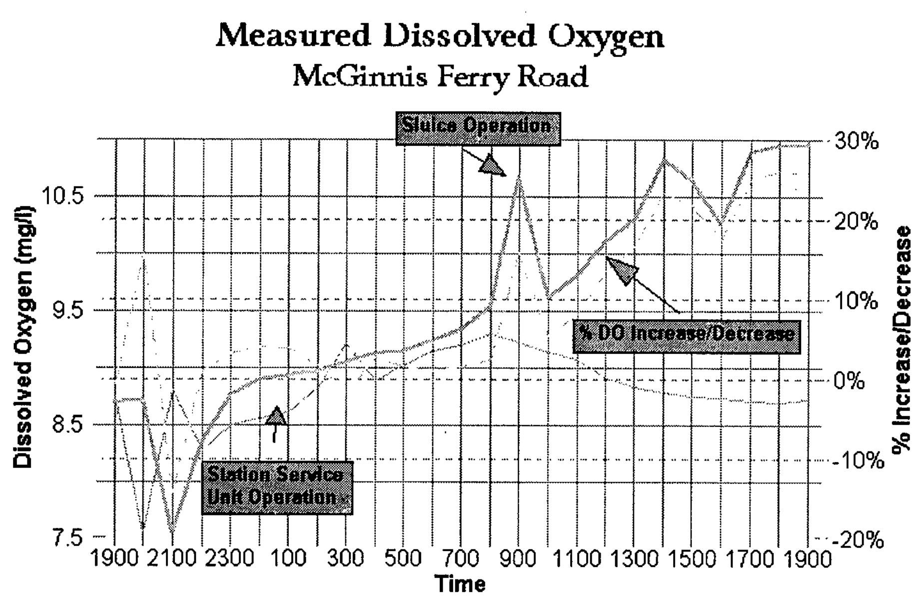
One of the down sides of sluicing is related to the hydropower ideology that water released to the downstream riverine environment should be done so only through the generating units. This idealogy places no monetary value on maintaining aquatic habitat for human use. As a result, in addition to the sluicing alternative for correction of the water quality predicament below the dam, the following alternatives are being examined as a basis for correcting the DO problem downstream of Buford Dam.

- **Self-Aspirating Turbine.** Some research and deployment of runners of



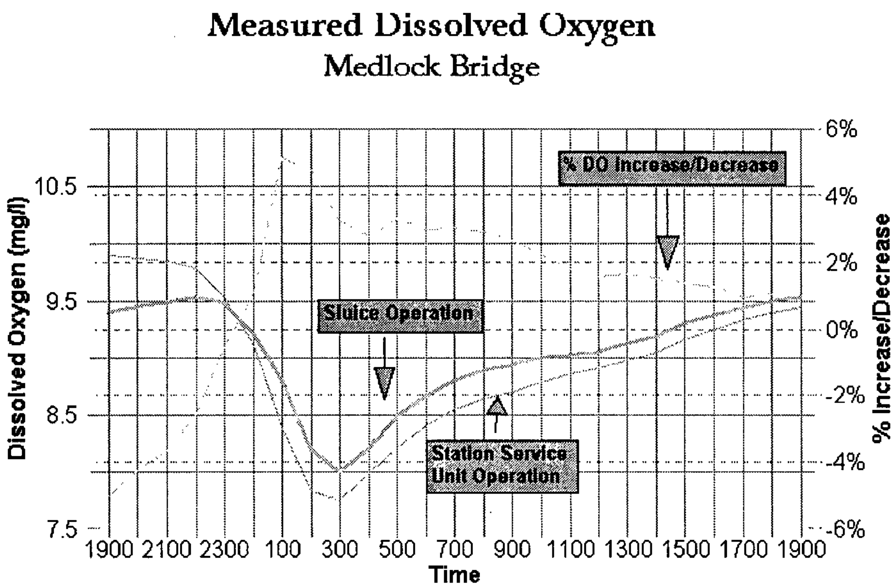
Each plot includes peaking generation 15:45-17:45 daily
Station Service Unit Effects: Monay Dec 4, 1995 (1400 hrs) through Tuesday Dec 5, 1995 (1400 hrs)
Sluice Effects: Tuesday Dec 5, 1995 (1400 hrs) through Wednesday Dec 6, 1995 (1400 hrs)

Figure 3. Measured DO at Georgia Highway 20



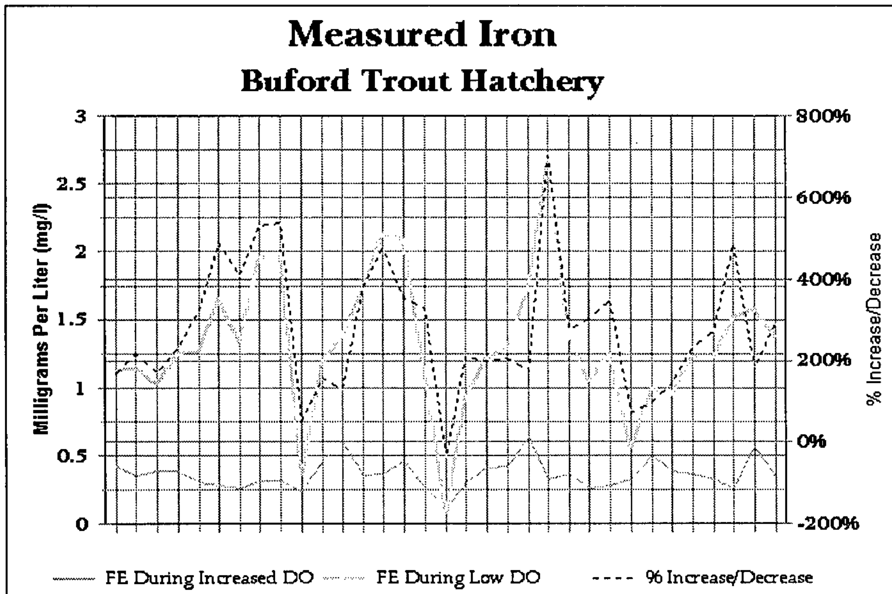
Each plot includes peaking generation 15:45-17:45 daily
Station Service Unit Effects: Monday Dec 4, 1995 (1400 hrs) through Tuesday Dec 5, 1995 (1400 hrs)
Sluice Effects: Tuesday Dec 5, 1995 (1400 hrs) through Wednesday Dec 6, 1995 (1400 hrs)

Figure 4. Measured DO at McGinnis Ferry Road



Each plot includes peaking generation 15:45-17:45 daily
Station Service Unit Effects: Monday Dec 4, 1995 (1400 hrs) through Tuesday Dec 5, 1995 (1400 hrs)
Sluice Effects: Tuesday Dec 5, 1995 (1400 hrs) through Wednesday Dec 6, 1995 (1400 hrs)

Figure 5. Measured DO at Medlock Bridge



Source: Buford Trout Hatchery Personnel: Data collected during critical lake stratification period
Note: Increased Do implies high flow conditions, Decreased DO implies low flow conditions

Figure 6. Measured iron adjacent to Buford Trout Hatchery

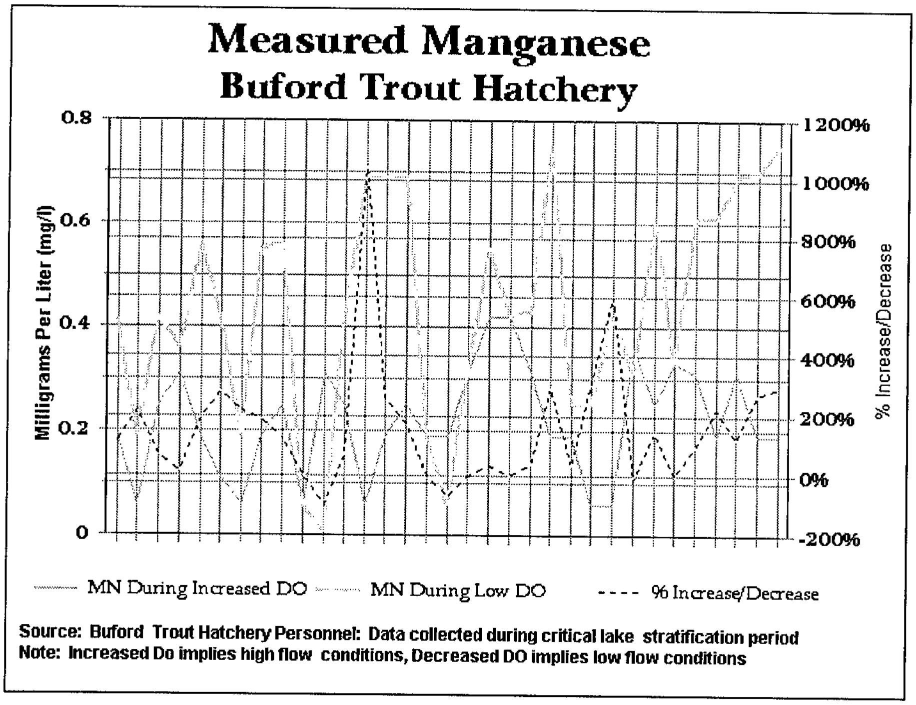


Figure 7. Measured manganese adjacent to Buford Trout Hatchery

this type have taken place in recent years. However, at present, the technology to accurately predict the results to be obtained in service from this type of runner does not exist outside of empirical testing. In addition, to accurately determine the performance in the areas of output, efficiency, and DO improvement of a selfaspirating runner, either a model or a prototype or both must be built and tested. With a self- aspirating runner, it is known that there is always some amount of efficiency lost over that of a standard runner. The amount of DO level improvement in the discharge can only be determined after careful experimentation with laboratory model(s) and/or the prototype runner. For replacement of the existing station service unit turbine runner with a modern state-of-the-art runner, it is not considered feasible to require a laboratory model test. For small turbines such as the station service unit turbine at Buford, it is usually sufficient to rely on the present-day ability of manufacturers to closely predict the replacement runner performance targets using computer models. The cost of laboratory model testing could quite possibly exceed the cost of the new runner itself. Should a self-aspirating replacement runner become the preferred alternative, model testing would certainly be anticipated and recommended in order to ensure that the runner could achieve the target DO levels in the plant discharge. Laboratory testing would also be necessary in order to tailor prototype details to achieve the best combination of output and efficiency over the operating range in order to meet performance guarantees.

- **Reconfiguration of Station Service Unit Discharge.** As noted previously, the DO levels in power plant discharges under circumstances when both the main and the station service units are operating have been found to be satisfactory the majority of the time even though DO levels originating through the intake structure are extremely low during the stratification period. Apparently, the large amount of turbulence and boiling of the discharge in the power plant tailrace that occurs during main unit operation is sufficient to aerate the discharge. As the power plant discharges from the main units still contain a considerable amount of energy, the dissipation of this energy results in the turbulence and surface disturbance that ultimately aerates the discharge to acceptable DO levels. During periods when only the station service unit is operating, the tailrace is relatively calm with only a minimal amount of surface disturbance. Discharges from the station service unit (only) do not produce enough surface action to sufficiently aerate the tailrace to acceptable levels. Reconfiguration of the station service unit discharge using baffling or other directional control methods that would create turbulence on the tailrace surface provides a possible means of creating the mechanism for increased aeration of the tailrace. However, as the surface turbulence produced by turbine discharge creates a positive effect from the DO perspective, it is not desirable with respect to unit performance as it represents energy that was not captured by the turbine.

- **Construction of a Weir Downstream.** Under this option, construction of a weir downstream would provide a cascading effect on the releases during low-flow periods, which should significantly increase DO during the severe stratification period. During high flow, the weir would have no effect on the releases since the depth of the water during high flow would be well above the crest of the weir.

- **Reconfiguration of Sluice Gate.** The sluice gate was originally designed to pass high-water flows when the generating units were physically unable to pass them. Additionally, it was not designed to operate continuously or at small-gate openings. However, over the years, the sluice has typically been used to replace the station service unit flows when either the station service unit was down or when special requests for sluice releases were made by the State of Georgia. Replacing the approximately 600 cfs that the station service unit puts out requires only a small raising of the sluice gate, which results in a tremendous amount of nozzle pressure being applied to the lower portion of the gate. Presently, the gate is raised/lowered by a motor hoist located atop the dam that lifts the gate by way of chains connected to the top of the gate. The gate rolls up and down on rollers, which are situated inside of metal tracks that were built into the concrete structure of the dam. Discussions with Buford powerhouse personnel indicate that the installation of a valve-type opening/closing mechanism and a minor redesign of the sluice gate (to withstand the nozzle pressure when the gate is less than fully opened) would probably allow for continued safe operation of the sluice during periods of need. This option if developed correctly would be a one-time cost and provide years of trouble-free service. However, water released through the sluice provides no hydropower benefits but, on the otherhand, provides important environmental benefits.

- **Injection of Compressed Air Into the Station Service Unit's Vacuum Breaker.** Under this potential alternative, compressed air generated by an industrial compressor would be injected into the turbine by way of the vacuum breaker. Ideally, as water rotates through the turbine, compressed air would be forced into the water. As a result, the chemical process that occurs would introduce dissolved oxygen that otherwise would not be present. Obviously, there will be a slight loss of efficiency in the turbine. However, turbine efficiency would not be sacrificed continuously for an environmental problem that is not continuous. Additionally, if the potential alternative is successful, minor modifications to the turbine and the purchase of an adequate compressor would be necessary to implement the plan. Through a series of tests scheduled for the summer of 1996, the viability of this alternative will be evaluated to determine its effectiveness.

In conclusion, the sluicing alternative showed promising results. Sluicing could possibly be incorporated into the operational scheme of Buford Dam, thereby significantly maintaining/raising water quality in Buford's tailrace year round. The other alternatives discussed in this report will be studied in more detail during the summer of 1996 with the assistance of the Corps Technical Consultants at the U.S. Army Engineer Waterways Experiment Station. If either alternative provides similar results as those yielded during sluicing, a decision will be made by the Mobile District to incorporate the most feasible alternative into its ongoing major rehabilitation project for the Buford Power-house.

Ecosystem Management—Evolving Policy

by
Lynn R. Martin[1]

Introduction

The Corps of Engineers policies concerning the restoration of ecological resources have evolved considerably in the last few years. These policies, founded in broad authorities related to the protection and management of natural resources, and the development of water resources, have evolved in response to Administration and Congressional priorities and emphasis. The Administration's emphasis on an ecosystem approach to natural resource management, as well as specific study and implementation authorities supported by the Administration and Congress have influenced the emerging ecosystem management policies for the Corps Civil Works program. Corps policy is evolving to accommodate these positions and priorities. Most recently, the Administration has undertaken a number of initiatives that strive to define an ecosystem approach to managing natural resources, as well as to foster this approach among the Federal agencies. Guidance was issued last year that incorporates an ecosystem approach to planning and implementing Corps projects for the restoration and protection of ecological resources. Policy will no doubt continue to evolve to accommodate new positions and priorities. This paper presents a summary of the Administration initiatives that have influenced the evolution of Corps policies regarding ecosystem restoration and management. A summary of the important changes to the policies is also presented.

Administration Initiatives

Included as part of the Federal programs and activities examined by the National Performance Review (NPR) was the management of natural resources by Federal agencies. A supporting report to the NPR, "Reinventing Environmental Management," points out that maintaining healthy ecosystems and sustaining their productivity is vital to ensure high quality of life for future generations. The report states, however, that "piece-meal" restoration of ecosystems is ineffective, and there are numerous managing agencies that have "inconsistent statutory missions." Among the problems that the report identifies are incompatible data, distinct agency cultures, inconsistent planning and budgeting cycles, differing agency organizational structures, and demands of special interests. The report asserts that "regional boundaries differ between agencies, ...[and] can further fragment Federal management within a given area. ...Federal agencies, even those with similar mandates, are managing the same ecosystem differently, often at cross-purposes. ...[There has been] an absence of the necessary political will to address decisions needed to ensure the long-term health of the environment and a sustainable economy." Better integration of Federal, State, local, tribal, and private agencies' efforts concerning natural resource management is recommended.

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The NPR report recommends that "[a]n improved Federal approach to ecosystem management would be based on ecological, not political, boundaries. It would then seek and consider input from all stakeholders affected by Federal responsibilities in the area. Within such a framework, Federal agencies, state, local, and tribal governments, businesses, public interest groups, citizens, and Congress could work in collaboration to develop specific strategies, refocus current programs and resources, and better ensure the long-term ecological and economic health of the country."

The Government Accounting Office (GAO) released a report in August 1994 that also examines Federal ecosystem management initiatives. The GAO report focuses on the management of Federal lands by four Federal land management agencies (National Park Service, Bureau of Land Management, Fish and Wildlife Service, and the Forest Service). The report states that even though many laws have been enacted to protect individual natural resources, ecological conditions on many Federal lands have declined. It also acknowledges that many of the historic levels of commodity production and other resource uses cannot be sustained indefinitely. Thus, management practices, including commodity production/harvesting and recreational use, have had to be reduced or modified. In some instances, these changes have disrupted local economies and communities, contributing to intractable conflicts between ecological and economic values and concerns.

In 1993, the White House Office on Environmental Policy established an Interagency Ecosystem Management Task Force to pursue the recommendations of the NPR report. The task force is chaired by the White House Office of Environmental Policy and is made up of Assistant Secretaries from 12 departments and agencies, as well as representatives from Office of Management and Budget and the Office of Science and Technology Policy. Both the Department of Defense and the OASA(CW) participate on the working group for this task force. The task force was charged with establishing overarching goals for all Federal agencies; removing the barriers that frustrate more effective, efficient interagency cooperation; and learning from large-scale ecosystem-based management efforts.

The interagency effort is using the following definitions as part of their efforts: *An ecosystem is an interconnected community of living things, including humans, and the physical environment within which they interact. The ecosystem approach is a method for sustaining or restoring natural systems and their functions and values. It is goal driven, and is based on a collaborative developed vision of desired future conditions that integrate ecological, economic and social factors. It is applied within a geographic framework defined primarily by ecological boundaries.*

A number of case studies were conducted to gain information that would support development of guidelines for implementing an ecosystem approach. These case studies include the Anacostia River Watershed, South Florida, Great Lakes Basin, Coastal Louisiana, the Pacific Northwest Forests, Prince William Sound, and Southern Appalachians. The task force issued the first volume of their report: *The Ecosystem Approach: Healthy Ecosystems and Sustainable Economies* in June 1995 (Interagency Ecosystem Management Task Force 1995a). This report describes the ecosystem approach as defined by the task force and identifies principles to guide agencies in implementing and participating in ecosystem efforts. These principles are summarized in Table 1. The report also identifies a number of recommendations to improve agency implementation of the ecosystem approach. The second volume of the Task Force report (Interagency Ecosystem Management Task Force 1995b), released in the fall of 1995, examines issues that are impediments to effective application of the approach by the Federal agencies. The issues addressed in the report are grouped into the following areas: Budget process issues, institutional issues, public participation, science and information management, and legal authorities.

Table 1
Principles for Federal Agency Implementation of the Ecosystem Approach Recommended by the Interagency Ecosystem Management Task Force (1995a)

- Develop a shared vision of the desired ecosystem condition.
- Develop coordinated approaches among Federal agencies to accomplish the ecosystem objective and collaborate with local State and tribal parties.
- Use ecological approaches that restore and sustain biodiversity, health, and productivity of ecosystems.
- Support actions that incorporate sustained economic, sociocultural, and community goals.
- Respect private property rights and work cooperatively with private landowners.
- Recognize that the ecosystems and institutions are complex, dynamic, and heterogeneous over space and time.
- Use adaptive management to achieve both desired goals and new understanding of ecosystems.
- Integrate the best science and knowledge available to the decision-making process while continuing research to improve the knowledge base.
- Establish baseline conditions for ecosystem functioning and sustainablity against which change can be measured. Monitor and evaluate actions and their outcomes to determine if goals and objectives are being achieved.

An interagency Memorandum of Understanding (MOU) on Ecosystem Management was signed in December of 1995. This MOU draws upon findings of the interagency task force and contains recommendations regarding Federal implementation of the ecosystem approach. The Acting Assistant Secretary of the Army for Civil Works signed the MOU, but no implementation guidance has been issued to date.

Evolving Corps Policy

Significant changes in Corps Civil Works (CW) policies concerning ecological resources have occurred in recent years. The evolution of this policy can be traced through the CW budget guidance documents over the last several years. Starting from a historically narrow emphasis on the evaluation and mitigation of environmental impacts, from water resources development actions, the policy later evolved such that projects could be formulated specifically for fish and wildlife habitat restoration objectives. The CW program and budget guidance for fiscal year 1993 (issued in 1991 via EC 11-8-2) identified the restoration of environmental resources, including fish and wildlife habitat, among the priorities for new start candidates. More recently, "restoring and protecting the environment, *including ecosystem functions and values* [emphasis added]" was identified as a CW program goal (EC 11-2-166, March 1995).

The Corps recently issued policy and implementation guidance pertaining to ecosystem restoration in the form of Engineer Circular (EC) 1105-2-210 (U.S. Army Corps of Engineers 1995c). The policy and guidance contained in this EC attempts to convey the philosophy of moving from a focus on fish and wildlife habitat, with a emphasis on individual species, to a broadened focus on restoring ecosystem structure and function. While this EC does not formally acknowledge the principles of an ecosystem approach outlined by the Interagency Ecosystem Task Force, many of the principles are incorporated implicitly.

The implementation of ecosystem restoration projects as part of the Civil Works program is supported by the authorities provided by Section 1135 of the Water Resources Development Act (WRDA) 1986, and Section 204 of WRDA 92, as well as through individual study authorities derived from Congressional resolutions. Ecosystem restoration projects can also result from favorable reconnaissance studies initiated under Section 216 of the River and Harbor and Flood Control Act of 1970. Policy and implementation guidance regarding Section 1135 and Section 204 are provided in EC 1105-2-206, Project Modifications for Improvement of the Environment (U.S. Army Corps of Engineers 1995a), and EC 1105-2-209, Implementing Ecosystem Restoration Projects in Connection with Dredging (U.S. Army Corps of Engineers 1995b).

Significant Policy Changes and Points of Clarification

The Nature of Project Outputs

Ecosystem restoration initiatives consist of restoring or protecting the structure and function of an ecosystem, or parts thereof. The focus is not on restoring habitat for individual species, or resource commodities, but the physical, chemical, and biological attributes of ecosystems and the roles of plants and animals in the context of community and ecosystem frameworks. (U.S. Army Corps of Engineers 1991 and 1995c).

Analytical Framework Clarified

The analytical framework of the Principles and Guidelines is to be followed, and NED benefits are to be considered, but the projects are not justified based on NED benefits (U.S. Army Corps of Engineers 1990; U.S. Water Resources Council 1983). Instead, both monetary and nonmonetary benefits are used to justify ecosystem restoration projects. The relative values of the outputs must be described but they do not have to be monetized. These values are considered in relationship to costs, but the traditional cost-benefit analysis is not applied. Both the with- and without-project conditions are considered (described, quantified, and displayed). Tradeoffs between environmental outputs gained and losses in NED output can be considered. Changes in the levels of existing project outputs (i.e., NED benefits) can be considered. Increases in NED benefits are to be portrayed as part of the project justification, but as incidental benefits, and not the primary basis for project justification.

Measuring Outputs—The Use of HEP Is Not A Requirement

As mentioned above, environmental restoration project outputs or benefits are no longer limited to "fish & wildlife habitat," nor is it mandatory to express the outputs in habitat units. The Habitat Evaluation Procedure (HEP) is just one of many methods for measuring habitat related outputs. Other kinds of outputs can be measured via other methods. However, because of either the misunderstanding regarding requirements for measuring outputs, or because of the widespread familiarity with HEP, HEP and HEP-like methods have been the most frequently used in the formulation and evaluation of Corps environmental projects. One of the primary constraints on applying these and other procedures to measure benefits of wetland projects is the difficulty in measuring causal linkages between the measures proposed and their effects on plant and animal populations or other aspects of ecosystem structure and function. Uncertainty in the information related to ecological relationships and project results limits the ability to forecast ecological changes and to quantify the anticipated benefits associated with wetland projects.

Cost Effectiveness and Incremental Cost Analysis

The difficulty in weighing the nonmonetary benefits against the costs of wetland projects presents a challenge to decision makers interested in ensuring wise investments in wetland projects. Cost-effectiveness analysis and incremental cost-analysis procedures provide a structured, yet flexible framework to assist in comparison and selection of alternative plans (Robinson et al. 1995). These analyses use output information from one of the nonmonetary (e.g., ecological assessment) evaluation techniques, comparing the costs of a range of alternatives to outputs from each alternative in order to provide more information for decision making. Cost-effectiveness analysis can be used to screen alternatives that are not cost effective from further consideration. Incremental cost analysis reveals changes in costs as levels of environmental outputs increase. These procedures can assist in ensuring that economically rational plans and projects are identified and considered for selection. They provide a means to progressively compare alternative levels of outputs and ask if each additional level of output is worth the expenditure required. While these procedures have frequently been applied to habitat functions, where outputs are typically measured in terms of area, habitat units, or numbers of individuals, they can also be applied to outputs related to other types of ecosystem functions and outputs. The only constraint is that the outputs be quantifiable. A technique such as the Wetland Evaluation Technique (WET), for example, is not useful with these procedures because it does not provide quantitative measures of outputs.

Cooperation With Other Agencies

Successful ecosystem restoration requires considerations of features and activities in the landscape, and cooperation with other agencies is essential. Also, tight budgets necessitate cooperative application and use of resources where feasible. The recent guidance states that Corps ecosystem restoration initiatives should complement authorities and programs of other agencies, and that the Corps role will vary. Sometimes the Corps may have the lead in these efforts and other times play more of a supporting role (e.g., planning or technical assistance).

New Non-Federal Partnerships

Environmental restoration and protection initiatives generally require 25-percent local cost sharing in implementation and full non-Federal operation and maintenance of the completed work. The development of new partnerships and involvement of what are perhaps nontraditional sponsors is encouraged. Some grants and other funds provided by other Federal agencies to State and local governments may be available for the non-Federal sponsor as their cost-share portion; however, this is not the case in all instances, and such opportunities must be carefully evaluated with the other Federal agency.

Streamlining Small-Project Management and Execution

The guidance issued last year concerning management and implementation of the Section 1135 program (U.S. Army Corps of Engineers 1995a) reflects changes in policies concerning funding nonproject specific coordination activities, delegation of project approval authority, and clarifies procedural concerns. The new policy reflects a substantial empowerment of project approval of the Major Subordinate Commands. These and other changes were made to streamline the program and to facilitate implementation of the program.

Multifunction and Multidisciplinary Teams Are Essential

The approach to ecosystem management is not limited to the assessment of environmental impacts. A variety of professions can contribute to ecosystem restoration initiatives, including planners, designers, ecologists, economists, natural resource managers, and others. The formulation of ecosystem restoration projects requires the talents of a number of disciplines. Multiple function and multidisciplinary teams can lend to improved identification of problems and opportunities, refinement of objectives, formulation of more efficient and effective alternatives to address the specified objectives, better description of the outputs desired and anticipated, and improved evaluation of both nonmonetary and monetary outputs in relation to costs.

Adaptive Management

The relative newness of the ecorestoration science and the uncertainty of the ecorestoration planning theories and tools points to the need for contingencies to address restoration problems that occur either during or after construction. Adaptive management provides a means to both address these problems and improve the knowledge base and tools associated with ecorestoration (National Research Council 1992). It involves monitoring to determine whether restoration measures are working as designed and is especially helpful when new, unproven techniques are being applied, or when significant levels of uncertainty exist. EC 1105-2-210 addresses implementation of adaptive management as part of ecosystem restoration initiatives in the CW program.

What's Next?

Corps headquarters, with assistance from the Institute for Water Resources, is currently examining the findings and products of the Interagency Ecosystem Task Force in order to determine how to incorporate the principles and concepts contained in these products into the Civil Works traditional water resources missions and the newer ecosystem restoration initiatives. Some limitations in existing authorities have been recognized, and pursuit of additional authorities that would allow broader opportunities to participate in ecosystem restoration is being considered.

The Corps ecosystem restoration policies and guidance contained in EC 1105-2-210 is primarily focused on the formulation and evaluation of ecosystem restoration initiatives. While the philosophy is implicit to the entire Civil Works program, it does not address in detail, programs and activities associated with natural resources management, project operations and maintenance, or dredging. In some instances, it may be appropriate to mesh regulatory considerations with mitigation as well as broad ecosystem restoration initiatives. It is possible that there will be policy development work in these areas in the future.

The basic premise of an ecosystem approach to environmental restoration and natural resource management requires that a shared vision be developed by and among interested stakeholders. The Shared Vision Model and the planning and evaluation approach developed as part of the National Drought Study managed by IWR may be readily applicable to ecosystem management. A user-friendly, graphical simulation tool (STELLA II) is used to capture the expertise and experience of people in a region to develop a shared vision upon which to base negotiation. Such an approach as well as other decision support methodologies and software may be available and applicable to ecosystem restoration and management initiatives.

Ecosystem Management and Watershed Studies

EC 1105-2-210 discusses the use of watersheds as an organizing tool to help address ecosystem restoration opportunities. Watershed-based thinking (planning) and decision making has a long history in Federal water resources programs. Recently, the watershed focus has been reemphasized in both the debate over reauthorization of the Clean Water Act and the National Performance Review, which recommended "cross-agency ecosystem planning and management" throughout the Federal government. The U.S. Environmental Protection Agency has subsequently developed a "watershed protection approach," and other Federal agencies have adopted similar "ecosystem" and "watershed" approaches.[1] The Corps will be initiating a number of "watershed" reconnaissance studies this year. While the Corps has extensive experience in using a watershed approach, the renewed watershed emphasis has highlighted many issues related to what a watershed study is and how the Corps should go about doing watershed studies.

Among the issues associated with watershed studies are their relationship to ecosystem restoration and management objectives and opportunities. One option is for the Corps to provide leadership in "true multi-objective planning" through an ecosystem approach to water resources management and development at the watershed level. Most of the agencies active in the Administration's ecosystem management efforts are focusing on missions associated with natural resource stewardship. A major void seems to be the integration of ecosystem management principles with resource development initiatives into a closer approximation of "sustainable development." Efforts are currently underway to define the Corps role in watershed planning and management initiatives. Ecosystem perspectives and restoration opportunities will no doubt be incorporated, but "how" has yet to be determined.

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In-Reservoir Effects of Water-Level Drawdowns, Dworshak Reservoir, Idaho

by
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Introduction

The depletion of salmon populations in the Northwest, and specifically the Snake and Clearwater river runs, have necessitated drastic changes in the management of the Northwest's water resources. Dworshak reservoir has been drawn down over the past 3 years to increase flows in the lower Clearwater and Snake rivers to assist outmigrating juvenile salmon and steelhead in their journey to the ocean. Reservoir levels dipped to as low as 60 ft below full pool (1,600 ft above datum) in the summer of 1993, and 110 ft and 80 ft in the summers of 1994 and 1995, respectively. The major reductions in reservoir level and induced limnological effects have been a recent focus of study by reservoir managers.

In the spring of 1993, the Nez Perce Fishery Department (NPFD) and the State of Washington Water Research Center (WRC) initiated a study to evaluate the effects of these system operations on the biota and water quality of the reservoir. The study included two main channel sites located in the lower 20 miles of the reservoir, River Miles 3 (NFC-3)and 19 (NFC-19), and a major tributary site, Elk Creek (EC-4) (Figure 1). Primary productivity determinations at NFC-3 (light-dark bottle ^{14}C method) accompanied by nutrient and algal analyses were conducted throughout the 1993 season. The study was continued in 1994 and expanded with additional funding from the U.S. Army Corps of Engineers (COE) during the last 3 months of the season to include three additional upper reservoir sites: River Miles 35 (NFC-35), 45 (NFC-45), and the Little North Fork (LNF-1) tributary (Figure 1). Primary productivity, nutrient, chlorophyll *a*, and algal analyses were completed for these sites. In 1995, the limnological investigation continued at all six sites from March through October, during which time chemical, physical, and biological

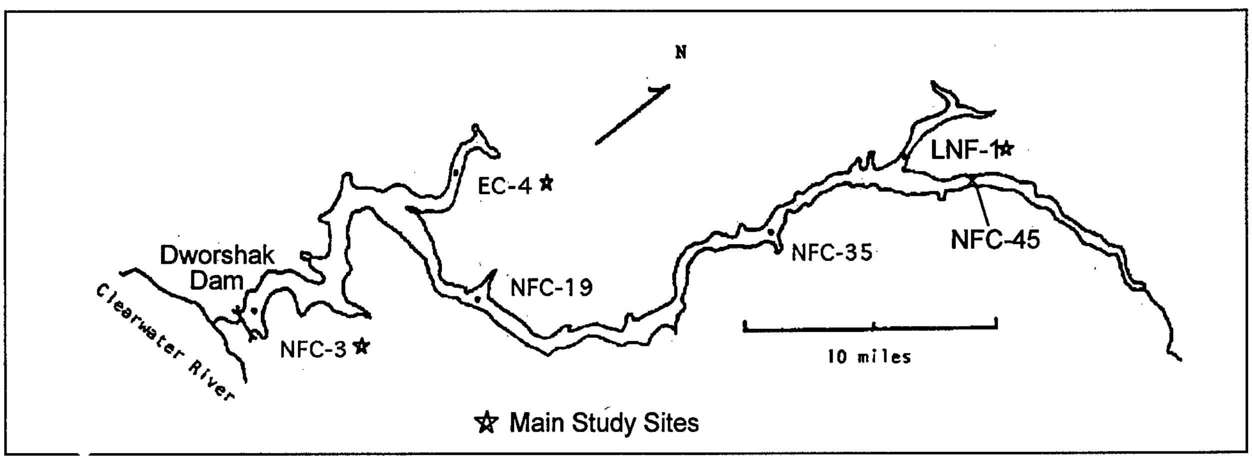

Figure 1. Map of Dworshak Reservoir, Idaho, including study sites

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parameters were evaluated. In addition, several of the reservoir's inlets and the outlet were monitored in 1995 to help evaluate allochthonous impacts.

This paper will address the seasonal and spatial trends in primary productivity and associated parameters between 1993 and 1995 and examine correlations between these trends and drawdown levels. The main focus will be on the growing season, from June to September. Three study areas will be used to assess the trends: NFC-3 as a main channel reference, and EC-4 and LNF-1 as tributary references.

Dworshak dam, constructed in 1971, retains 53 miles of the North Fork Clearwater River, creating a deep, coldwater reservoir. Dworshak has a surface area of 17,600 acres, a maximum volume of 3.47 million acre-ft at full pool, a maximum depth of 630 ft in the forebay, and receives water from 2,440 miles^2 of a mainly forested, mountainous watershed (Falter et al. 1977). The morphology of the reservoir is characterized by steep banks and dendritic arms extending into the surrounding mountains. The lower 10 miles of reservoir is monomictic and deeper than the dimictic, upper portion. Early in its existence, Dworshak was considered mesotrophic (Falter et al., 1977). However, over the past 20 years, the abundant nutrients from the flooded forest land have been depleted, and productivity in the reservoir has become more nutrient limited.

Primary Productivity

Primary productivity in reservoirs exhibits great variation both seasonally and spatially due to differences in morphology and chemical/ physical attributes associated with different sites. Reservoirs with numerous arms and bays, such as Dworshak, have distinct trends in productivity. The productivity in these arms can be much higher than the main basin due to increased loading of nutrients and higher temperatures (Wetzel 1983). However, productivity can also be suppressed in these bays due to higher loads of suspended particles that attenuate light and decrease photic zone depths.

Several trends in primary productivity were noted in 1993 and 1994 at Dworshak. First, productivity was generally higher in bays and upper reaches of the reservoir (EC-4 and LNF-1) throughout the season (Figure 2). During the growing season of 1993, productivity at EC-4 was higher than the main channel

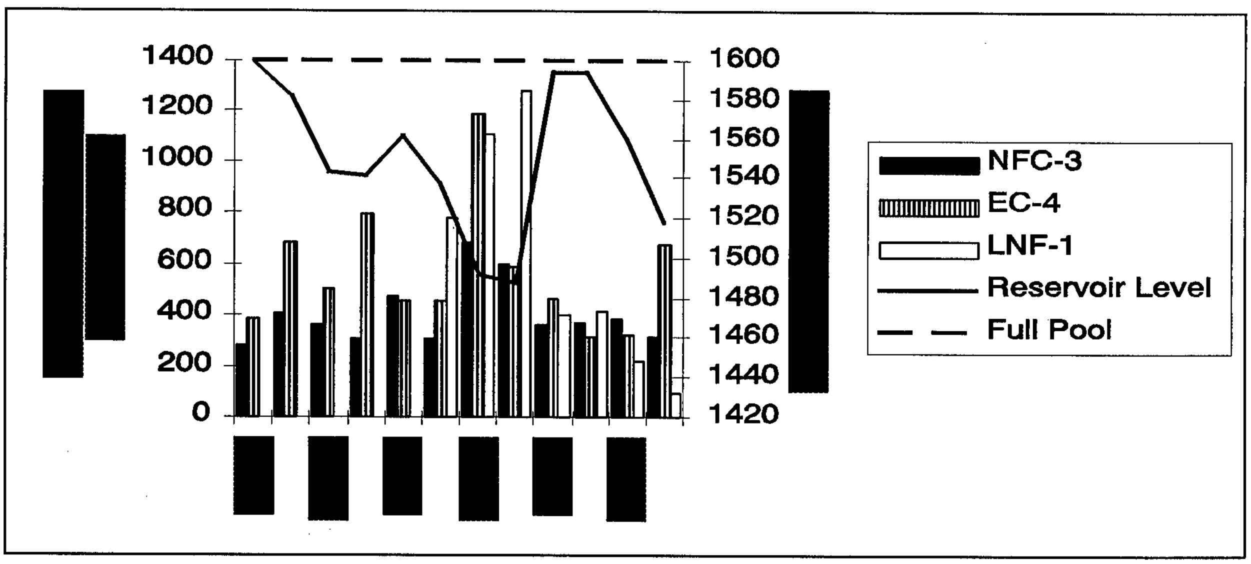

Figure 2. Primary productivity trends in relation to reservoir level at Dworshak Reservoir

station at NFC-3. However, during 1994, NFC-3 often exhibited higher rates of growth than EC-4. For the few months that productivity at LNF-1 was monitored in 1994, it appeared to be higher than other areas, especially in September. Secondly, productivity at all stations showed increased growth in August, after summer drawdowns had taken place. Finally, fluctuations in primary productivity between sampling dates in 1994 were much more pronounced than the previous year, varying by as much as 55 percent at NFC-3.

In 1995, notable trends in productivity were present, but there was little similarity to previous years (Figure 2). Fluctuations between months at all sites appeared to be dampened. Productivity at NFC-3 varied by as little as 15 percent throughout the growing season, peaking slightly in August. At EC-4, changes in productivity were insignificant until September, when a near doubling occurred, and summer averages indicated higher productivity than at NFC-3. Primary productivity at LNF-1 declined steadily throughout the season to reach a 3-year low in September; in contrast, the peak at that site occurred in September of 1994. Average primary productivity was lower in 1995 than in previous years throughout the reservoir.

Spatial Variability and Initial Conditions

The spatial and temporal variability between sites can account for differences in productivity among regions of the reservoir. Proximity to inlets, overall depth, orientation, shoreline morphology, and current/circulation patterns all influence rates of productivity. The geology of the watershed at Dworshak is not continuous, with distinct changes in rock and soil type occurring down the reservoir (Falter et al. 1977). This variety of geologic inputs affects the levels of nutrients and ions essential for growth, as well as the amount of inorganic particulates that remain suspended in the water column. Water temperature and volume of spring runoff can affect seasonal patterns in productivity (Wetzel 1983). In June of 1994, surface water in most of the reservoir was up to 3 °C colder than in 1993; however, by August of 1994, water temperatures were about the same as the previous year. The exception was LNF-1 where water temperatures were nearly 2 °C warmer than recorded at other stations in August of 1994.

At the beginning of the growing season in 1995, all stations were warmer than the preceding year. However, by August of 1995, temperatures within the photic zone were much colder than in 1994, particularly at NFC-3 where it was nearly 7 °C colder. This could possibly be due to a reduction in the depth of the epilimnion. In August 1994, the photic zone and thermocline at NFC-3 were located at 5.9 m and 7.0 m below the water's surface, respectively. In 1995, however, the metalimnion began at about 6.0 m, and included part of the 6.5-m photic zone.

The inlets monitored in 1995 were colder than the reservoir stations throughout the growing season. Mixing of the inlet and reservoir water appeared to take place within the photic zone. There was a noticeable temperature decrease in August, especially at North Fork Clearwater and Elk Creek stations.

Nutrients

Tables 1-3 present data for parameters related to productivity from 1993 to 1995. Nutrient trends are important because there are frequently good correlations between concentrations of available nutrients and productivity. In 1993, soluble inorganic nitrogen concentrations decreased steadily from June to September, ranging from 30 μg/L to below 10 μg/L at NFC-3. In 1994, inorganic nitrogen concentrations increased in the later months of the growing season, along with an increase in productivity. All stations appeared to have elevated concentrations of inorganic nitrogen in September of 1995, with values at EC-4 and LNF-1 as high as 80 μg/L and 70 μg/L, respectively. While productivity increased at EC-4, the relationship does not hold at LNF-1 in September, where productivity was unusually low.

Table 1
Chemical/Physical Parameters From NFC-3 at Dworshak Reservoir Between 1993-1995

Month	Reservoir Level, ft	TN:TP Ratio	Inorg-N mg/L	Inorg-P mg/L	Suspended Solids, mg/L	Photic Zone, m	Temperature °C	Secchi Disk, m
1993								
June	1,599	12	0.03	0.004	*	8.6	16.1	*
July	1,582	11	0.02	<0.001	*	8.5	16.7	*
August	1,543	10	0.02	<0.001	*	7.3	20.2	*
September	1,541	12	0.00	<0.001	*	10.2	18.5	4.7
1994								
June	1,562	23	0.02	<0.001	*	8.4	13.5	3.4
July	1,537	34	0.01	<0.001	*	7.2	16.9	2.7
August	1,492	21	0.03	0.001	3.1	5.9	23.4	1.8
September	1,488	26	0.03	0.001	2.3	10.1	21.1	3.7
1995								
June	1,594	18	0.05	<0.001	2.6	8.3	17.7	2.5
July	1,594	17	0.02	<0.001	2.5	7.5	19.4	2.3
August	1,560	17	0.01	<0.001	3.0	6.3	16.4	2.1
September	1,518	15	0.04	<0.001	2.0	8.1	17.8	2.4

Table 2
Chemical/Physical Parameters From EC-4 at Dworshak Reservoir Between 1993-1995

Month	Reservoir Level, ft	TN:TP Ratio	Inorg-N mg/L	Inorg-P mg/L	Suspended Solids, mg/L	Photic Zone, m	Temperature °C	Secchi Disk, m
1993								
June	1,599	11	*	*	*	5.1	15.8	*
July	1,582	22	*	*	*	7.6	16.8	2.4
August	1,543	21	*	*	*	5.5	19.2	2.9
September	1,541	20	*	*	*	6.5	18.4	3.5
1994								
June	1,562	16	0.01	<0.001	*	6.6	15.1	1.7
July	1,537	18	0.01	<0.001	*	4.1	15.3	1.2
August	1,492	25	0.03	0.001	6.0	5.3	19.7	1.8
September	1,488	21	0.03	0.001	2.0	6.7	20.6	2.1
1995								
June	1,594	17	0.03	60.001	3.2	5.9	17.4	1.7
July	1,594	14	0.02	<0.001	3.8	6.0	20.2	1.8
August	1,560	13	0.02	<0.001	3.2	5.1	17.2	1.5
September	1,518	13	0.08	<0.001	2.5	5.6	17.4	2.9

Table 3
Chemical/Physical Parameters From LNF-1 at Dworshak Reservoir Between 1993-1995

Month	Reservoir Level, ft	TN:TP Ratio	Inorg-N mg/L	Inorg-P mg/L	Suspended Solids, mg/L	Photic Zone, m	Temperature °C	Secchi Disk, m
1993								
June	1,599	*	*	*	*	*	16.5	*
July	1,582	*	*	*	*	*	18.0	*
August	1,543	*	*	*	*	6.7	19.2	2.1
September	1,541	*	*	*	*	7.5	18.4	3.8
1994								
June	1,562	*	*	*	*	7.6	15.1	2.1
July	1,537	14	0.01	<0.001	*	8.3	15.3	1.8
August	1,492	14	0.05	<0.001	*	5.6	19.7	2.1
September	1,488	30	0.03	<0.001	1.6	8.4	18.4	3.4
1995								
June	1,594	15	0.02	<0.001	1.4	9.1	15.1	2.9
July	1,594	10	0.03	<0.001	1.4	11.7	19.4	4.6
August	1,560	20	0.01	0.001	1.3	7.9	17.5	2.9
September	1,518	11	0.07	<0.001	3.1	4.6	18.3	1.9

Concentrations of soluble phosphorus were usually exhausted during the growing season, with values rarely exceeding 1 μg/L. Therefore, TN:TP ratios were used to determine trends and evaluate nutrient limitation. The reservoir was close to nitrogen limitation in 1993, as ratios were near or below a value of 10 (Organization for Economic Cooperation and Development (OECD) 1982). Falter et al. (1977) observed nitrogen limitation as well, due to an abundance of phosphorus during the reservoir's early years. In 1994, however, ratios were more indicative of phosphorus limitation due to apparent increase in nitrogen. NFC-3 appeared to be phosphorus limited throughout the summer, whereas EC-4 and LNF-1 were P-limited only in August and September. During the rest of the 1994 and 1995 growing seasons, TN:TP ratios were within the range of nitrogen and/or phosphorus limitation (OECD 1982). Preliminary bioassay results confirm colimitation in 1995.

Variations in nutrient concentrations and subsequent effects on productivity are due to a variety of sources. First, inlets vary in concentrations of nutrients and stations that are influenced by these streams should exhibit greater variation. The composition of in-flowing water is a function of land-use activities, such as timber harvest, stream substrate and gradient, and volume of runoff. Concentrations of inorganic nitrogen and phosphorus in streams ranged from 10 to 90 μg/L and 1 to 10 μg/L, respectively. Elk Creek and Long Meadows Creek, the main inlets at EC-4, had higher levels of orthophosphorus and lower TN:TP ratios. Timber harvest in the drainages of these streams may have contributed nutrients and suspended matter. Breakfast Creek, which flows to LNF-1, appeared to have higher levels of inorganic nitrogen compared with neighboring streams.

Second, nutrient concentrations increased with depth in the reservoir, particularly the inorganic constituents. Nitrate concentrations, for example, typically ranged from 100 to 150 μg/L below a depth of 30 m. It was noted earlier in the study that destratification did occur in upper reaches of the reservoir in late July and early August; in fact, areas that had been characteristic of impounded water were now free-flowing. We do not have definitive hydrologic information on the reservoir, but it is not unreasonable to assume that some of the upstream hypolimnetic nutrients were mixed into the euphotic zone and transported to other parts of the reservoir as destratification progressed.

Turbidity and Solids

A third source of nutrients is the resuspension of deposited material and bank erosion during the drawdown period. As the reservoir level declined and water receded from upper reaches and tributaries, previously deposited sediment and nutrients were dislodged and resuspended. In addition, bank erosion was evident in the exposed sediment, which led to an increase in levels of suspended particles.

It is difficult to detect drastic increases in turbidity in data collected in 1994 and 1995; however, vertical extinction coefficients generally increased from 1993 to 1994, and this trend carries through 1995. Photic zone and Secchi disk depths both decreased in August of 1995 immediately after drawdown periods (Tables 1-3). In all 3 years of study, light penetration increased in September, except for LNF-1 in 1995, where the photic zone was unusually shallow for that time of year, as was the primary productivity. Although, no phytoplankton biomass data were available, the reduction in photic zone depth appears to be due primarily to suspended inorganic material rather than algal cells. Productivity at LNF-1 was low, and the effect of self-shading was unlikely (Figure 3).

Parameter Relationships

In order to identify sources of fluctuation in primary productivity, it is important to identify relationships between productivity and the parameters that influence it. Pearson correlation coefficients were determined between several parameters and primary productivity. The data are presented in Table 4.

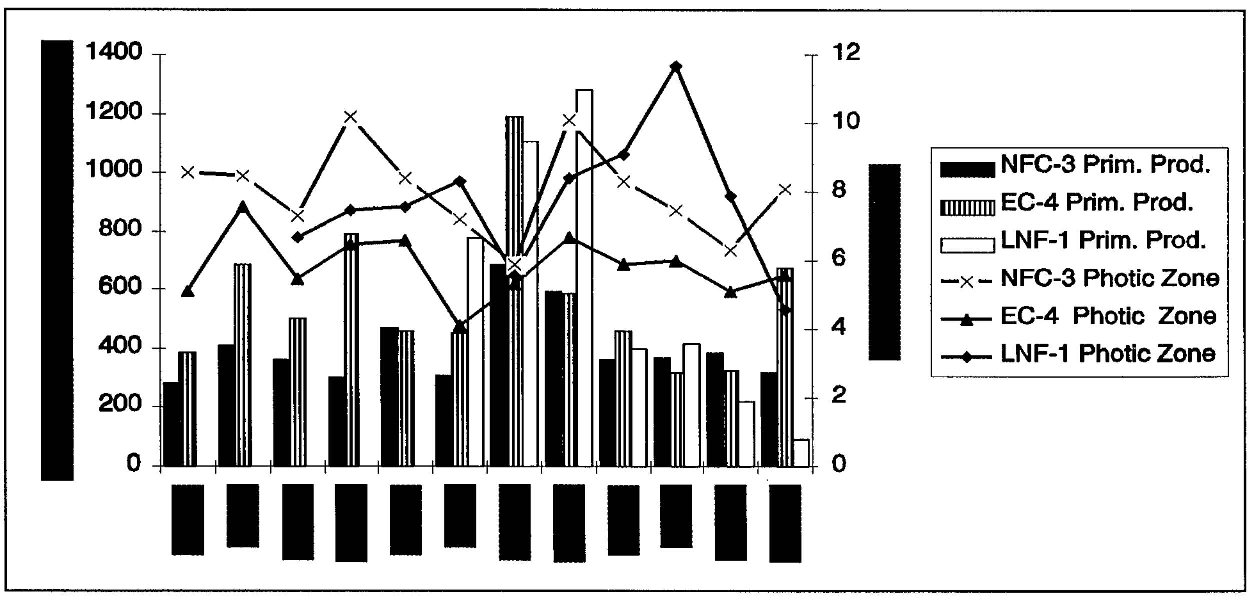

Figure 3. Primary productivity and photic zone trends at Dworshak Reservoir

Table 4
Pearson Correlation Coefficients for Selected Parameters, Dworshak, 1993-1995

Parameters	NFC-3	EC-4	LNF-1
Primary Productivity to Reservoir Level	-0.63	-0.72	-0.90
Primary Productivity to Total-N	0.76	0.72	0.33
Primary Productivity to Inorganic-N	0.09	0.30	0.60
Primary Productivity to Temperature	0.56	0.33	0.32
Primary Productivity to Suspended Solids	0.43	0.64	0.92
Primary Productivity to Photic Zone	-0.17	0.17	-0.49
Reservoir Level to Total-N	-0.45	-0.14	-0.48
Reservoir Level to TN:TP ratio	-0.33	-0.66	-0.45
Reservoir Level to Photic Zone	-0.10	0.04	0.67
Reservoir Level to Temperature	-0.56	-0.31	-0.46
Reservoir Level to Suspended Solids	0.07	-0.08	-0.48
Suspended Solids to Photic Zone	-0.76	-0.54	-0.66
TN:TP ratio to TP	-0.70	-0.59	-0.82

As shown in Table 4, relationships exist at certain sites and not at others, but a few hold for all sites. The study areas show a negative relationship between primary productivity and reservoir level. As the water level declined, primary productivity increased, although this relationship was not absolute. Increased productivity was probably a result of resuspension of settled particulates, including nutrients.

Primary productivity and suspended solids had a positive relationship. This was primarily due to the fact that the resuspension of particles, as stated earlier, included nutrients, enabling higher growth rates. As shown in Table 4, suspended solids and photic zones had a negative relationship. For the drawdowns studied thus far, the interaction between resuspended nutrients and productivity appeared to be stronger. However, there may be a point where the relationship between inorganic suspended matter and photic zones becomes overwhelming, and reduced light penetration will limit productivity.

As the reservoir level declined, average photic zone temperatures increased. Productivity increased as well; however, this may not be related to temperature.

Relationships between nutrients, reservoir level, and primary productivity were difficult to define. Primary productivity at LNF-1 showed some correlation with inorganic nitrogen, but this trend did not hold for the late season in 1995 when nutrient levels increased but productivity continued to decline. It was likely that changes in nutrient concentrations during a drawdown led to an increase in productivity; however, when combined with other factors, this relationship was masked.

Summary

Conclusions regarding the effect of summer drawdowns on the biota and overall limnology of Dworshak Reservoir are indeterminate. However, it can be confidently stated that there is a relationship between primary productivity and reservoir level and that large fluctuations during the growing season will affect the growth of algae and organisms in trophic levels dependent on them. There are a myriad of relation ships and factors associated with reservoir level that may work to either increase or decrease productivity. In 1994, there appeared to be an abundance of nutrients during times of low water, leading to an increase in productivity. Yet increases in productivity after the drawdown in 1995 were much less dramatic; in fact, productivity decreased at LNF-1. Decreasing photic depths, or increased attenuation of light, may have had a greater impact in 1995, leading to the reductions in productivity.

Data concerning trophic relationships between zooplankton and algae were collected in 1995, but are not available at this time. These data will undoubtedly help explain the differences in productivity. Falter et al. (1977) studied the loss of phytoplankton to zooplankton grazing and found that these losses were minimal. However, the situation may have changed over the past 20 years, or during a drawdown, and an understanding of this trophic relationship will be helpful. Correlating productivity with changing reservoir level is essential in predicting the effects of various system operations on phytoplankton biomass. Furthermore, it has been shown in a study on Hungry Horse and Libby reservoirs, Montana, that zooplankton production was significantly reduced by fluctuations in phytoplankton production, in turn affecting the survival of fish in the reservoirs (Marotz, Gustafson, and Althen 1994). We are confident that there is a loss of phytoplankton and zooplankton from the reservoir in the spill water during the summer drawdown. This loss does not seem to substantially affect primary productivity at the lower station. However, no definitive statements can be made regarding whether other regions, particularly tributary arms, are affected by this loss.

In 1996, we hope to continue the study at Dworshak, initiating further investigation into the variations in primary productivity. This work will include identification of growth-limiting parameters and a closer look at complex changes in chemical/physical parameters associated with the morphology of the reservoir. A narrowing of factors has already taken place, but to correctly predict the effects of system operations, more research is necessary. Accurate forecasts of changes in trophic relationships is essential in managing the fishery at Dworshak and will be a primary focus in 1996.

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Temporal Trends in the Limnology of Lower Snake River Reservoirs, Washington

by
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Introduction

A limnological investigation of the lower Snake River was conducted between April 1994 and October 1995 by the State of Washington Water Research Center. The study was commissioned by the U.S. Army Corps of Engineers and was designed to determine current physical, chemical, and biologic characteristics of the lower Snake system. Few limnological investigations of lower Snake River reservoirs have been undertaken, with the study by Funk, Falter, and Lingg (1985) being a notable exception.

The lower Snake River consists of a series of reservoirs (Lower Granite, Little Goose, Lower Monumental, and Ice Harbor) that extend from Lewiston, ID, at River Mile (RM) 139 to the confluence with the Columbia near Pasco, WA (Figure 1). The lower Snake reservoirs are operated mainly for power production, navigation, and recreation. The reservoirs vary from 47 to 79 km in length, with maximum depths of approximately 40 m. The hydraulic characteristics are typical of run-of-the-river type reservoirs, well-mixed, with an average hydraulic residence time of 5 days per reservoir.

Three stations on the lower Snake (RM 108 at Lower Granite Dam, RM 129 near Silcott Island, and RM 140 near Clarkston, WA) and one station on the Clearwater River (CLW-1 near Lewiston) were sampled every 2 weeks throughout most of the study. The remaining three stations (RM 18 at Fishhook Park, RM 83 at Central Ferry, and RM 118 near Centennial Island) were sampled monthly. The study included analyses of nutrients, cations and anions, suspended solids, phytoplankton, zooplankton, chlorophyll *a* and ^{14}C primary

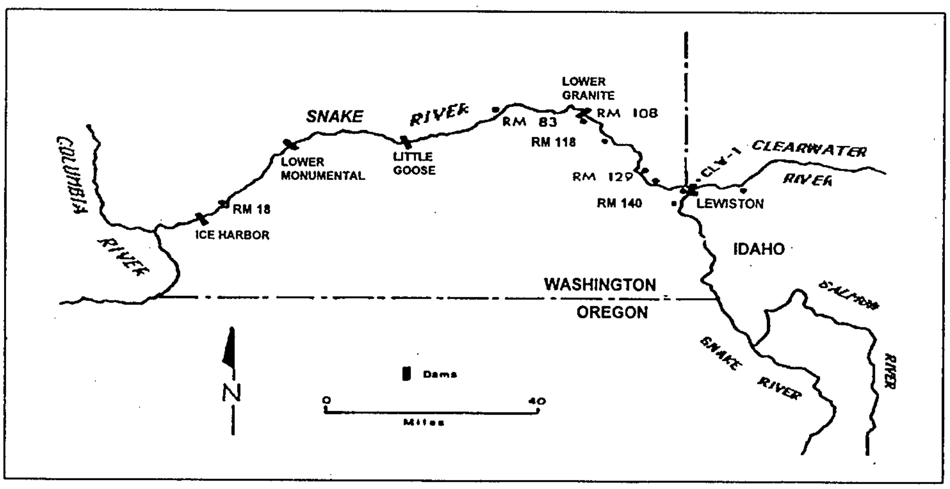

Figure 1. Map of Snake River study area

[1] State of Washington Water Research Center, Washington State University, Pullman, WA.

productivity, as well as standard field measurements of physical parameters.

General limnological characteristics of the lower Snake river as a whole, and RM 108 specifically, will be the focus of this paper. RM 108 was isolated for a more detailed analysis of the hydrologic and physicochemical processes that largely govern the chemical and biological characteristics of the lower Snake sampling stations because it is typical of the entire system and would likely be selected for study in the event that Lower Granite reservoir is utilized for flow augmentation in the future.

All Stations

Nutrients

Nutrient concentrations, averaged over the growing season, are very similar at all Snake River sampling sites. Table 1 summarizes nitrogen and phosphorus values for each station during the 1994 and 1995 growing seasons. Generally, nutrient concentrations are highest during low-flow periods (fall and winter) and lowest during high-flow periods (spring).

Several trends are shown in Table 1. Nutrient concentrations are drastically lower in the Clearwater River than in the Snake River above the confluence (RM 140). Generally, the Clearwater River chemistry reflects the character of the watershed, with clear, cold, nutrient-poor waters emerging from a largely forested watershed. Snake River chemistry is also largely determined by watershed characteristics, with warmer, more turbid, nutrient-rich waters resulting from a largely agricultural watershed. Differences in the geology and soils of the watersheds also influence their chemistries.

Moving downstream, mixing is complete by RM 118, and mean nutrient levels remain relatively constant down to RM 18. The general spatial trend (between sites) observed during the 1994 growing season was a decline in nutrient concentrations downstream. Total nitrogen (TN) decreased from 0.46 mg/L at RM 118 to 0.32 mg/L at RM 18. Biological uptake and settling probably account for this trend. Nutrient concentrations during the 1995 growing season did not decrease downstream. Total flow into the lower Snake was markedly higher in 1995, keeping nutrients in suspension, while primary productivity was lower, reducing biological nutrient uptake.

While there is some spatial and temporal variation in mean nutrient concentrations, statistical analysis using boxplots to characterize the data shows that there is not significant difference in nutrient concentrations between sites at the 95-percent confidence interval. Figures 2 and 3 show that, collectively, all reservoir sites had comparable nutrient concentrations during the 1994 growing season.

Primary Productivity

Measurements were made for ^{14}C primary productivity bi-weekly at reservoir stations

Table 1
Mean Nutrient Concentrations at Snake River Sampling Sites

	Total Phosphorus mg/L		Orthophosphorus mg/L		Total Nitrogen mg/L		Inorganic Nitrogen mg/L	
Station	1994	1995	1994	1995	1994	1995	1994	1995
RM 18	0.03	0.04	0.01	0.01	0.32	0.37	0.08	0.20
RM 83	0.04	0.04	0.01	0.01	0.38	0.42	0.11	0.20
RM 108	0.04	0.04	0.01	0.01	0.43	0.40	0.10	0.17
RM 118	0.04	0.04	0.02	0.02	0.46	0.42	0.13	0.22
RM 129	0.05	0.04	0.02	0.01	0.50	0.45	0.20	0.24
RM 140	0.06	0.05	0.03	0.02	0.62	0.54	0.34	0.25
CLW-1	0.02	0.01	0.00	0.00	0.21	0.14	0.02	0.02

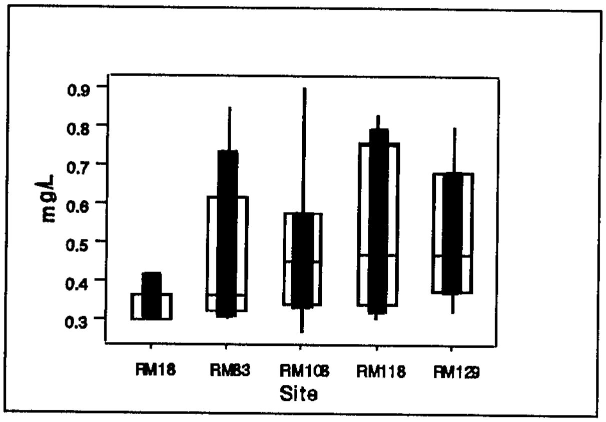

Figure 2. Total nitrogen - 1994

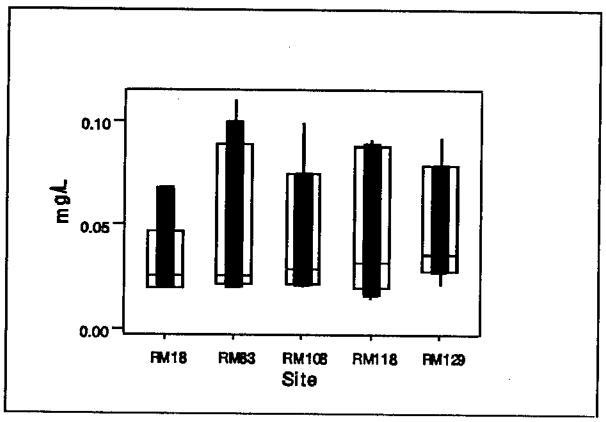

Figure 3. Total phosphorus - 1994

RM 108 and RM 129, and monthly at stations RM 18, RM 83, and RM 118. No measurements were made from November 1994 to February 1995. Statistical analysis ofproductivity showed that all stations along the lower Snake showed comparable productivity rates in 1994 (Figure 4). There were slight variations in the median productivity values, but the large ranges around each of those medians suggest that there were no statistical differences between the sites. While productivity data for the late summer and early fall of 1995 have yet to be analyzed for stations other than RM 108; a similar trend is expected for 1995.

Productivity peaked in mid-July in 1994 and 1995, and, as expected, waned in early fall. Algal bioassays were conducted at all reservoir sampling stations through the 1994 growing

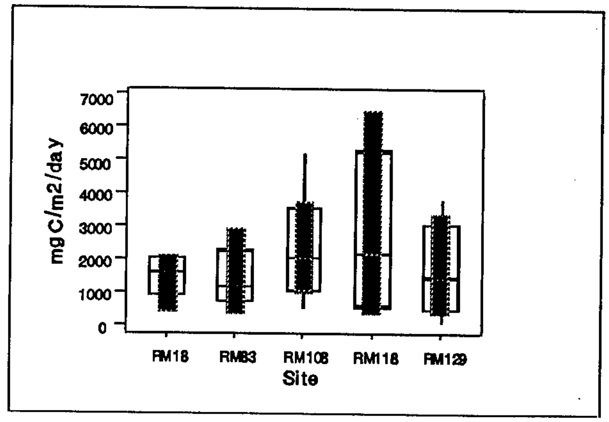

Figure 4. Primary productivity of the lower Snake - 1994

season and into the fall. Results indicate that algal growth potentials were often very high (Greene 1995). Additionally, detectable levels of orthophosphate (OP) as well as ammonia (NH_3-N) and nitrate (NO_3-N) (<0.001, 0.01, and 0.01 mg/L, respectively) were observed during peak production periods at all stations in 1994 and at RM 108 in 1995. Assuming that productivity will not be nutrient limited as long as OP, NH_3 and NO_3 are present, peak productivity in the lower Snake system is not usually limited by nutrient availability. The rates of productivity were, in almost all cases, indicative of eutrophic systems.

Algae

The algal community also demonstrated some temporal variability. During the early part of the 1994 growing season, Chrysophyta were the major component of the phytoplankton community, consisting mainly of *Cyclotella*, *Fragilaria*, *Stephanodiscus*, and less frequently *Melosira*. Later in the summer, Cryptophyta, primarily in the form of *Cryptomonas*, were more prevalent. Finally, towards the end of the summer, the blue-greens, principally, *Aphanizomenon* and, to a less extent, *Microcystis*, were the representatives. It should also be noted that blue-green algae blooms were not evident during the summer or fall of 1995, and it is suspected that this was due to the greater total flow through the reservoirs.

RM 108 Sampling Station

The sampling station at RM 108, just upstream from lower Granite Dam, showed temporal trends that are representative of the entire lower Snake system. Effects of flow, temperature, and suspended solids on primary productivity illustrate the impact of hydrologic conditions on the general limnology of the lower Snake.

Flow

Seasonal trends and year-to-year fluctuations of limnological conditions in the lower Snake system are strongly correlated with flow. Total flow into the lower Snake was estimated using U.S. Geological Survey flow data from the Snake River at Anatone, WA (RM 163), and the Clearwater River at Spalding, ID (RM 13). Figure 5 shows the strong relationship between total flow and primary productivity at RM 108. The correlation coefficient (r^2) for the 1995 growing season was -0.78.

Total flow, due to increased spring freshet, was substantially higher in 1995 than in 1994. Consequently, primary productivity was much lower for 1995 than 1994.

Temperature

The water temperature at RM 108 ranged from 2.9 °C in February 1995 to 23.5 °C in August 1994. The mean water temperature was 19.7 °C for the 1994 growing season and 17.6 °C for the 1995 growing season. Figure 6 shows temperature trends at RM 108.

Cooler temperatures often inhibit phytoplankton growth and may affect seasonal succession in the Snake River. In addition to increased total flow, cooler summer water temperatures may partially explain why no blue-green blooms occurred in 1995, as they have in years past.

Water from Dworshak Reservoir was released into the Clearwater River during mid-summer 1994 and 1995 as part of the Snake River flow augmentation program. Under normal conditions, the Clearwater River is several degrees cooler than the Snake River and accounts for approximately 30 percent of the total flow into lower Granite Reservoir. During flow augmentation periods, however, the Clearwater contributed approximately 55 and 45 percent in 1994 and 1995, respectively. During both spill periods, a slight disruption of normal summer warming was noted. While the Dworshak summer spills had a small effect on temperature at RM 108, both the Clearwater River and the Snake River were colder in 1995 than 1994.

Suspended Solids

Suspended solids measurements were made between March 1994 and October 1995. Values

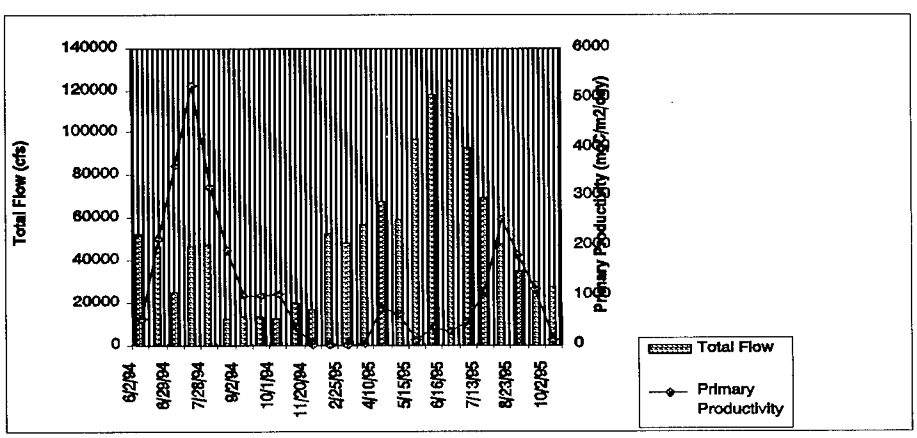

Figure 5. Total flow and primary productivity at River Mile 108

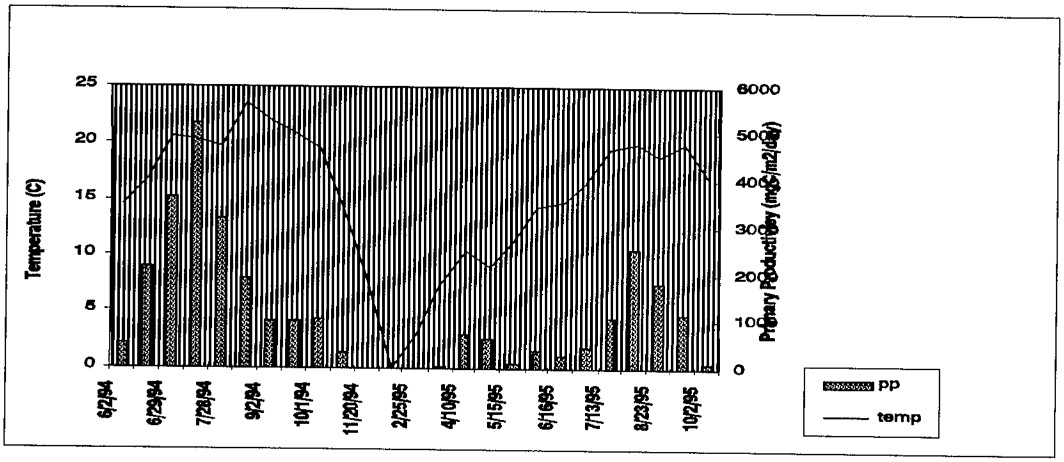

Figure 6. Temperature and primary productivity at River Mile 108

ranged from 1.7 to 11.6 mg/L and were highest in the Spring.

High concentrations of suspended solids reduce water clarity, which reduces the depth of the photic zone (a function of light penetration) and lowers the rate of primary productivity. At RM 108, suspended solids concentrations are correlated with total flow (r^2 = 0.76) and depth of the photic zone (r^2 = -0.69). Vertical profiles of rates of primary productivity through the photic zone illustrate the relationship between suspended solids and primary productivity. Profiles for the 1994 growing season are typical of a water column with low turbidity (suspended solids in this case), with photoinhibition occurring at the surface and maximum growth at about 1 m. The 1995 profiles are indicative of turbid conditions, with maximum growth being highest at the surface and quickly diminishing with depth.

Conclusion

The limnological characteristics of the lower Snake system vary according to prevailing hydrologic and physicochemical conditions. More than any other single factor, total flow into the system had the largest impact on the limnology of the reservoirs in 1995. High flow dominates other relationships that would normally govern primary productivity and general limnology. When flows are lower, as in 1994, relationships between light, temperature, phyto-zooplankton interaction, and nutrient chemistry become more apparent. Nutrient concentrations and primary productivity values, for 1994 and 1995, are indicative of a eutrophic system. Primary productivity was lower, and blue-green algae blooms failed to develop in 1995; but water clarity was compromised due to increased suspended solids.

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Twentymile Creek Habitat Restoration Project

by
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Authority and Project Linkage

The Congress of the United States has delegated continuing authority through Section 1135(b) of the Water Resources Development Act of 1986, as amended, to the Secretary of the Army to make modifications in the structures and operations of existing water resources projects that are feasible and consistent with the authorized project purpose in order to improve the quality of the environment. The water resources project on Twentymile Creek was authorized in Section 203 of the Flood Control Act of 1958. Construction of the authorized project was completed, and the project was turned over to the non-Federal sponsor for operation and maintenance. The authorized project limits extended from the mouth to Mile 11.7. The area of interest for assistance in the restoration of fish and wildlife habitat under the Section 1135 Program is the 2.6 miles of Twentymile Creek between Mile 9.1 and Mile 11.7.

Location and Basin Description

Twentymile Creek rises in the northwest corner of Prentiss County, Mississippi, near Booneville in northeast Mississippi. It flows southerly for 30.3 miles through Prentiss, Lee, and Itawamba counties to enter the East Fork of the Tombigbee River through the right bank about 6 miles north of Fulton. Twentymile Creek drains a watershed with an area of almost 169 square miles and has a length of about 31 miles and a maximum width of about 11 miles.

Project History

Twentymile Creek has an extensive history of modifications. In the early 1900s, 12 miles of the creek were initially channelized by non-Federal drainage districts. The U.S. Army Corps of Engineers (Corps) completed work in May 1940 to remove accumulated snags and other debris from Twentymile Creek under the authority of Section 2 of the Flood Control Act of 1937, as amended. The flood problems along Twentymile Creek continued to be critical, and Twentymile Creek was 1 of 22 streams entering the Tombigbee River that were authorized for modification by the Corps in the interest of flood control in Section 203 of the Flood Control Act approved 3 July 1958. The modifications on Twentymile Creek in Itawamba and Lee counties, Mississippi, were completed, and in January 1967, the completed project was turned over to the non-Federal sponsor for operation and maintenance in accordance with the authorizing act.

Problems in the Twentymile Creek project area were realized shortly after construction was completed. In April 1967, after only 4 months of project operation, the U.S. Army Engineer District, Mobile, received a letter from the non-Federal sponsor that cited two complaints about caving and erosion of channel banks along the project. Over the next 10 to 12 years, channel degradation developed and worked upstream to Mile 22.0. As the bed of Twentymile Creek degraded, waves of degradation propagated up the tributaries also. Degradations

[1] U.S. Army Engineer District, Mobile; Mobile, AL.

and subsequent bank failures doubled the channel cross-sectional area of the channel in the reach between Mile 12.0 and Mile 22.0. Bridges crossing Twentymile Creek were damaged or destroyed by the channel degradation, and farm acreage was lost due to bank sloughing. The reach between Mile 5.5 and Mile 12.0 had also experienced bank failures, although not as severe as those associated with the degradation further upstream. Sediments derived from upstream bed and bank erosion and tributary erosion caused the channel below Mile 5.5 to aggrade, reducing channel capacity and perhaps aggravating flood problems. The lack of instream cover (e.g., woody debris, reduced stream bank vegetative cover) increased solar radiation on the water surface that caused significant increases in water temperature that stressed aquatic organisms. The aquatic system also suffered from a loss of organic material entering the stream from the leaf fall. The decomposition of leaf litter into particulate organic matter was a vital link in the aquatic food chain from macroinvertebrates to fish. These instream cover impacts collectively acted to diminish the aquatic diversity and abundance of the stream channel. The deficiency (poorly developed or not developed) of the riffle-pool habitat eliminated the variability in depths and hydraulic conditions that served to provide resting and feeding areas for both forage and predaceous fish. Benthic macroinvertebrate populations that also benefit from this variability in habitat were impacted. The amount of oxygen added to the water because of the increased air/water interface at the riffle-pool habitat was decreased. These combined impacts on the riffle-pool habitat have effectively reduced the diversity of the aquatic organisms.

Congress provided authority to rectify some of these problems in the Supplemental Appropriations and Recession Act of 1980. Under this authority, the Mobile District constructed grade control structures at five locations, constructed rock bank protection at approximately 60 sites, and planted willows between Mile 11.7 and Mile 22.0. Additional rock bank protection and grade control structures were completed in the 1980s at eight bridge locations in the upper portion of the basin under the authority of Section 14 of the Flood Control Act of 1946, as amended.

Study Scope

Twentymile Creek was the focus of this habitat restoration study because of significant physical habitat degradation that has occurred due to hydrologic modifications, changes in sediment loadings, and channelizations that moved the stream reaches away from geomorphic equilibrium. The study concentrated on additional modifications of Twentymile Creek between Mile 9.1 and Mile 11.7 to restore and/or improve aquatic and stream corridor riparian habitat. Modifications investigated for incorporation into the habitat restoration project and, when possible, incorporated into the project were as follows: stable scour holes, stable habitat, diversity of habitat, solid substrate for invertebrates, canopy cover, and woody debris. These modifications, when incorporated, would serve as an offset to compensate for the aquatic and stream corridor riparian habitat altered by construction and operation of the "flood control" projects since the early 1900s.

Proposed Modifications

Selection of the desired amount of environmental restoration was influenced by several factors together with the cost-effective and incremental costs. The most important of these factors were the available budget, significance of resource, consideration of historical conditions, and the ability to properly maintain the project in the future. The proposed modifications include constructing bendway weirs with appurtenant structures plus planting willows and bottomland hardwoods at 12 reaches along Twentymile Creek between Mile 9.1 and Mile 11.7.

Bendway Weirs and Appurtenant Structures

The bendway weirs would be keyed into the bank to prevent flanking of the structures occurring and allowing flow to go around the structures. The top elevation of this hardpoint would be set at the 2-year flood elevation at each reach being protected. The bendway weir would then slope down on a 1 vertical to 4 horizontal slope to the channel bottom. At an elevation of 3 ft above channel bottom, the bendway weir would be angled upstream and extended out into the channel a distance of 37.5 percent of the channel top width. This weir section would extend out into the stream with a level top elevation and would have a top width of 3 ft. The wide, level crests would promote the formation and maintenance of larger, deeper scour holes. The bendway weirs would be placed on a filter fabric blanket to prevent the rock from settling into the sand and shortening the dikes. Stone would be a quarry run gradation with a stone size range of 650 to 1,000 lb. Stone toe protection would also be used at several locations to provide bank protection to reaches that have a stable bottom and would not respond as well to the bendway weirs and to provide additional protection downstream of the last bendway weir. Overbank drainage structures would be provided in two reaches to control the runoff from the agricultural fields that is entering the channel at these points. Care would be taken during construction to protect existing natural features, such as pools, hard clay substrate, and woody debris, so as not to cause any degradation to the habitat that these natural features have produced.

Willows

Native black willow (*Salix nigra*), common along Twentymile Creek, would be the mainstay for the willow plantings; however, use of dwarf willow species would be considered for supplemental plantings among the black willows. Based on habitat restoration work on Harland Creek, located in northern Mississippi, the following guidelines would be incorporated into the planting contract: areas currently supporting native woody vegetation would not be included in the planting area; planting sites infested with kudzu would be controlled prior to planting; willow posts must be planted when they are dormant; willows would be kept wet after cutting; the elapsed time between cutting and planting of the native willow material would not exceed 48 hr; the tops of the posts would be marked to ensure that the posts would be planted upright; the spacing would be a 3-ft grid; minimum post diameter would be 3 in. at the butt end—however, smaller diameter post within a bunch could be used; minimum willow post length would be of 10 ft; posts would be planted at least 8 ft deep using an 8-in.-diam auger with no more than 4 ft of the post showing above the ground; the first row would start at the water's edge (based upon low-water elevation), and each row would extend for the entire length of the bend; however, willow post would not be planted in permanently flooded soils or those soils too impermeable to permit significant groundwater movement, which would not allow rooting; no willows would be planted above top bank; the planting contractor would be required to excavate material from the channel and dump on top of planted posts to ensure holes are filled and to provide a near-surface medium for root development. If dwarf willow species are utilized to supplement the native black willow planting, the smaller dwarf willow plant materials would be refrigerated for a period of time before planting to form calluses and aid in rooting

Hardwoods

Nursery stock, bare root trees would be planted in the easement area along the top of bank in the restoration reaches along Twentymile Creek. Species to be planted would include those that are native to the bottomland area in this region of northeast Mississippi. Planting of these woody species would be accomplished during the winter plant dormancy period. Measures such as plastic guards for tree trunks, netting, mesh, etc., would be incorporated into the plantings to minimize herbivory from animals such as beaver and deer. Planting sites infested with kudzu would be controlled prior to planting.

Inspection of Plantings

The willow and hardwood plantings would be inspected at the end of the first growing season to ensure adequate survival. Replanting during the first 3 years following construction might be necessary in areas experiencing significant plant mortality. This mortality could be from such factors as poor soil conditions; inadequate moisture; herbivory from insects, beaver, and deer; inadequate soil aeration; or plant competition from kudzu. Attempts would be made to control kudzu in planting areas infested with this exotic plant species.

Costs

The total project first cost for the proposed modification has been estimated at $970,000 with an average annual cost of $96,000 (includes $7,000 annual operation and maintenance cost) for a 25-year project life.

Benefits

The stream corridor measures would serve to reestablish a series of riffles-pools on the degraded shallow stream with unstable substrate. Sections of channelized streams that afford substantial cover, coarse or cohesive substrates, and increased depth could therefore harbor more complex fish faunas due to broader food bases and increased habitat availability. In the case of Twentymile Creek, the excellent opportunity currently exists to install modifications to create depth, velocity, and substrate diversities to form food-producing areas, spawning and rearing areas, and instream and overbank cover to maintain reproductive populations. These modifications would also slow or halt the channel widening, thereby allowing recovery of natural vegetation, which also would offer riparian and riverine habitat benefits. As the stream corridor stabilizes, riparian vegetation would reestablish (accelerated by willow and hardwood plantings), thus improving the riparian wildlife habitat and providing shade for the stream, which would improve the aquatic habitat. These types of environmental restoration benefits have been demonstrated on portions of Twentymile Creek located upstream of the study area and as part of the Yazoo River Basin Demonstration Erosion Control Project to produce substantial habitat restoration benefits. The rock size and quantities for the bendway weirs (380 to 1,000 cu yd per reach, yielding about 8,000 total cu yd) would improve the stability of the aquatic habitat, enhance scour hole formation, provide hard substrate (currently a sparse substrate type in the project area), and provide biologically valuable interstitial spaces between the rocks. The hardwood plantings on the top bank areas at the 12 reaches would provide long-term benefits to the wildlife community.

Fishery benefits from restoration of channelized streams in northern Mississippi have been substantiated in a number of studies (Shields et al. 1990; Shields and Hoover 1991; Cooper and Knight 1987; Knight and Cooper 1991; Shields, Cooper, and Knight 1992, 1993, 1995; Shields, Knight, and Cooper 1993, 1994, 1995; and Dardeau, Killgore, and Miller, In Preparation). Studies on stabilized portions of Twentymile Creek compared with unstabilized reference channelized streams nearby (Shields et al. 1990; Shields and Hoover 1991) demonstrated a substantial increase in fish species diversity: 40 species on Twentymile Creek compared with 22 species collected on Mubby-Chiwapa Creeks. Work by Hoover et al. (1995) indicate broad distributions of most fish species throughout the 12 study reaches and correlations between fishes and hydraulic variables which are indicative that the fish community could benefit from the proposed habitat restoration.

In more directly related work on restoration of northern Mississippi degraded stream channels, Shields, Knight, and Cooper (1993) evaluated environmental benefits by placing rock spur dikes in the low-flow channel and by planting woody vegetation on the berms. This work shows that following installation of environmental restoration features to a stabilized channelized stream, there has been a significant increase in mean water depth and mean maximum scour hole depth (44 and 82 percent, respectively). They found that the mean length of fish and the number of fish species approximately doubled and the total weight of fish captured by a unit of sampling effort increased by an order of magnitude. Mean scour hole width increased by 130 percent. Woody cover along the treated bank increased from 38 to 66 percent of bank line after one growing season, although survival of young plantings was reduced from an estimated 60 percent to an observed 34 percent by competition from the exotic kudzu vine.

Follow-up research by Shields, Cooper, and Knight (1995) and Shields, Knight and Cooper (1994, 1995) continues to validate the environmental restoration benefits associated with stone spur dikes, stone toe protection, and planting of willows. Major changes in channel characteristics did not occur following restoration. Channel geometry, channel sinuosity, bed material size, and hydraulic roughness remained fairly stable. However, the fraction of sandbar bank line supporting vegetation more than doubled, and pool habitat availability increased fivefold, principally due to the enlargement of scour hole adjacent to spur dike extensions. Fish populations have favorably responded by growing larger, more diverse, and by retaining larger individuals. These researchers note that the stone spur dikes and willow posts created more pool habitat per unit length than standard stone toe protection. The willow posts are contributing significant amounts of woody debris, thus improving the quality of aquatic habitat within a warmwater stream that had been damaged by erosion and channelization.

The effects of various types of stone bank protection on stream fishes have been evaluated by Knight and Cooper (1991). The 3-year study evaluated natural bank areas of an unstable channelized stream, old (>5 years old) and new (≤ 5 years old) lateral dike (stone bank armoring), and transverse dike (spur dike) habitat. No significant difference in species diversity was observed; however, catch per unit of effort was higher along transverse dikes than lateral dikes. Total catch in both numbers and weight was greater around the transverse dikes. Scour holes associated with transverse dikes provide increased deeper water refuges that support larger numbers of fish and larger fish than the other habitats sampled.

Quantification of the fishery benefits of the proposed modifications (compared with status quo and another structural alternative—bank armoring) was performed through use of regression models developed by Hoover et al. (1995) for the orangefin shiner and brook silverside. These models defined relationships between populations of these two native fish species with physical habitat parameters. This habitat-based analysis showed that the proposed modifications (bendway weirs) would produce higher fishery benefits than the status quo or the bank-armoring alternative.

Schedule for Accomplishment

The construction of the proposed modifications should begin in the fall of 1996 and should take about 9 months to complete.

Findings and Conclusions

The primary aquatic and stream corridor riparian habitat degradation problems along the 2.6-mile reach of Twentymile Creek between Mile 9.1 and Mile 11.7 are the result of previous construction and operation of the "flood control" projects since the early 1900s. The degradation has occurred due to hydrologic modifications, changes in sediment loadings, and channelizations that have moved the stream reaches away from geomorphic equilibrium. Studies were undertaken to identify measures that could be undertaken to restore the aquatic and stream corridor riparian habitat within this 2.6-mile reach. A recommended habitat restoration plan was identified with a total project first cost estimated to be $970,000. The recommended plan includes constructing bendway weirs with appurtenant structures and planting willows and bottomland hardwoods at 12 reaches along Twentymile Creek between Mile 9.1 and Mile 11.7. Given the findings of the technical analyses and the stated goal to restore and/or improve aquatic and stream corridor riparian habitat, there is a Federal interest in the implementation of a project to improve the quality of the environment.

Acknowledgment

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Late-Summer Temperature and Dissolved Oxygen Dynamics in the Tailrace of a Southeastern Hydropower Reservoir

by
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Introduction

Tailraces below hydropower projects are an important yet frequently overlooked component of aquatic systems. These areas often possess characteristics of both lakes and rivers and contribute to the water quality of the entire downstream system. Tailraces provide recreation, navigation, and water supply for downstream users. Furthermore, they provide valuable biological habitat.

One type of tailwater is that in which a downstream reservoir acts as a receiving pool for hydropower releases. During extended periods of non-release, these tailwaters are typically lacustrine in nature and often stratify during the summer period. These areas are influenced by solar radiation, algal and macropyhte bioactivity, and existing downstream pool conditions. Tailrace water quality conditions during release typically emulate conditions found in the upstream pool at depths corresponding to the withdrawal zone. This is due to the displacement of ambient tailrace water by hydropower generation release. These flows often impact temperature and dissolved oxygen (DO) conditions throughout the downsteam pool. Typically, a "plug flow" response is observed from the immediate tailrace to some distance downstream, at which time the water plunges beneath the warmer surface layer, creating a "plunge point," and mixes with lake water of similar density.

Study Site

Hartwell (HW) Dam is a Corps of Engineers hydropower project located on the Savannah River on the Georgia-South Carolina border. HW Dam is a peaking power project, with generation designed to coincide with maximum power demand within its service area. During the summer months, releases typically occur Monday through Friday, commencing in midmorning and ceasing operation in late afternoon or at night. The HW Dam releases flow directly into the Richard B. Russell (RBR) Lake headwaters. These releases directly influence RBR Lake water quality. The immediate area downstream of HW Dam is constricted (0.25 km in width) and bordered by rocky substrate. Tailrace waters are extremely clear in comparison to upstream and downstream lake clarity conditions. The tailrace supports a "put and take" trout fishery. Tailrace water quality conditions often become critical during July through September, as temperatures increase and DO concentrations decrease, reflective of hypolimnetic conditions in the upstream HW Lake forebay.

Objectives

A study to better understand the dynamics of temperature and DO in the HW Lake tailrace under generation and non-generation conditions was undertaken by the Trotters Shoals

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Limnological Research Facility during August through mid-September 1995. The mid- to late summer months have historically produced the maximum water temperatures and the minimum DO concentrations in the HW Lake tailrace. Objectives were to (1) investigate lateral, longitudinal, and temporal variability in the HW Lake tailwater, (2) monitor HW Dam releases into RBR Lake, and (3) contrast habitat dynamics and suitability during generation/non-generation periods.

Methods and Materials

Study design incorporated manual and automated sampling techniques to measure lateral, longitudinal, and temporal dynamics. Manual sampling of in situ parameters was accomplished both via boat and from shore using Hydrolab DataSonde3 and H20 multi-parameter sondes (Hydrolab Corporation, Austin, TX). Remote temperature sampling was conducted using Onset Stowaway self-contained loggers (Onset Corporation, Pocasset, MA). Four Datasonde3s were deployed for remote temperature and DO data collection. All instruments were programmed to collect data at 30 minute intervals.

Station Location

Sampling sites were selected at distances ranging from 0.1 to 15 km downstream of HW Dam to obtain near- and far-field data within the tailrace and tailwater. Primary emphasis was placed on locating the compact temperature loggers in the immediate vicinity of HW Dam to detect short term temporal and lateral near-field variation (Figure 1). DO logging instruments were placed in the Savannah River thalweg at distances of 0.2 (Station 3C), 1.5 (Station 198), 5 (Station 190) and 15 (Station 180) km downsteam of HW Dam.

Deployment Procedure

Sixteen Stowaway loggers were deployed in the HW Dam tailrace attached directly to anchors in 2-3 meters of water. At station 180, located 15 km downstream, two additional temperature loggers were attached to a tethered buoy in the center of flow at depths of 0.5 and 8 meters.

Four DataSonde3s were deployed approximately 1-2 meters above bottom at depths of 2-3 meters at stations 3C, 198, and 190 and at the 14 m depth at station 180. These instruments logged temperature and DO data from 1 August through 19 September and were calibrated and downloaded weekly.

Results

Pre-Test Profiling

In preparation for the remote logger deployment, in situ profiles were taken via boat along the HW tailwater from 4 to 15 km downstream prior to generation and one hour following start up on Monday, 31 July 1995. These data depict stratified conditions following a weekend period of non-release, as well as the rapid change in water quality conditions that occurred within one hour of hydropower discharge.

Temperature

Temperatures 4 km downstream of HW Dam ranged from 25 °C to 16 °C, surface to bottom, respectively during non-release (Figure 2). These temperatures reflected those observed downstream in the deeper transitional zone. Within one hour of hydropower start-up, the upstream 5 km reach had been completely filled with 14 °C release waters. Warmer waters had been displaced downstream, effectively "stacking up" at the tailwater plunge point.

Dissolved Oxygen

DO concentration 4 km downstream also exhibited distinct gradients during an extended period of non-release, as concentrations ranged from nearly 8 to 6.5 mg/ℓ, surface to bottom, respectively (Figure 3). As with temperature, these gradients in the upper tailwater were erased by HW Dam release flows shortly after

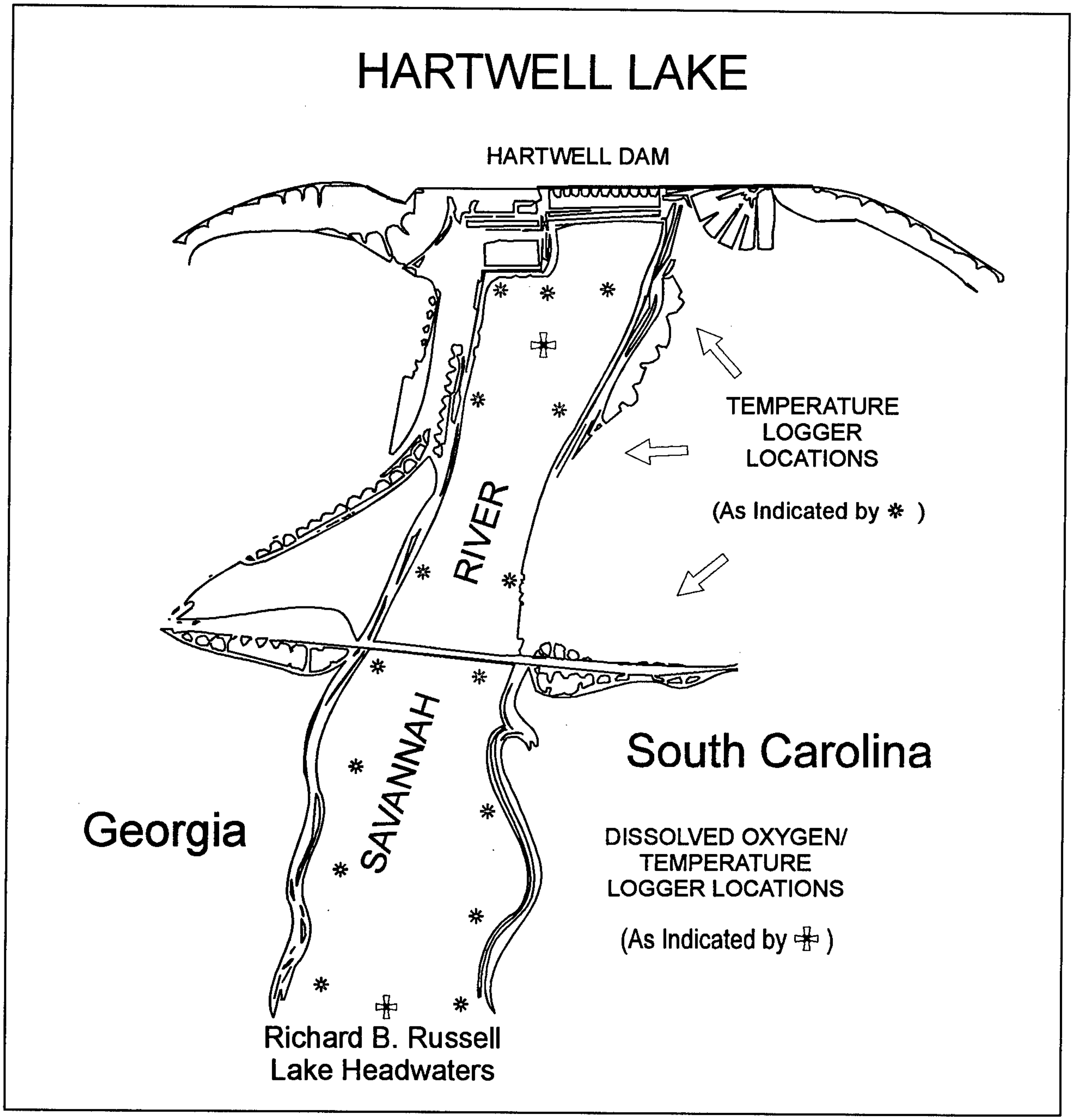

Figure 1. Near-field sampling stations in the Hartwell Lake tailrace

start-up, as 7.5 mg/ℓ waters filled the area. The downstream hypolimnion was supplemented by the increased DO concentration of the release, increasing from 6.0 to 6.5 mg/ℓ downsteam of the plunge point.

Hydrograph Dynamics

A one week hydrograph of temperature and DO concentration during 19-26 August 1995 is depicted in Figures 4 and 5. Both temperature

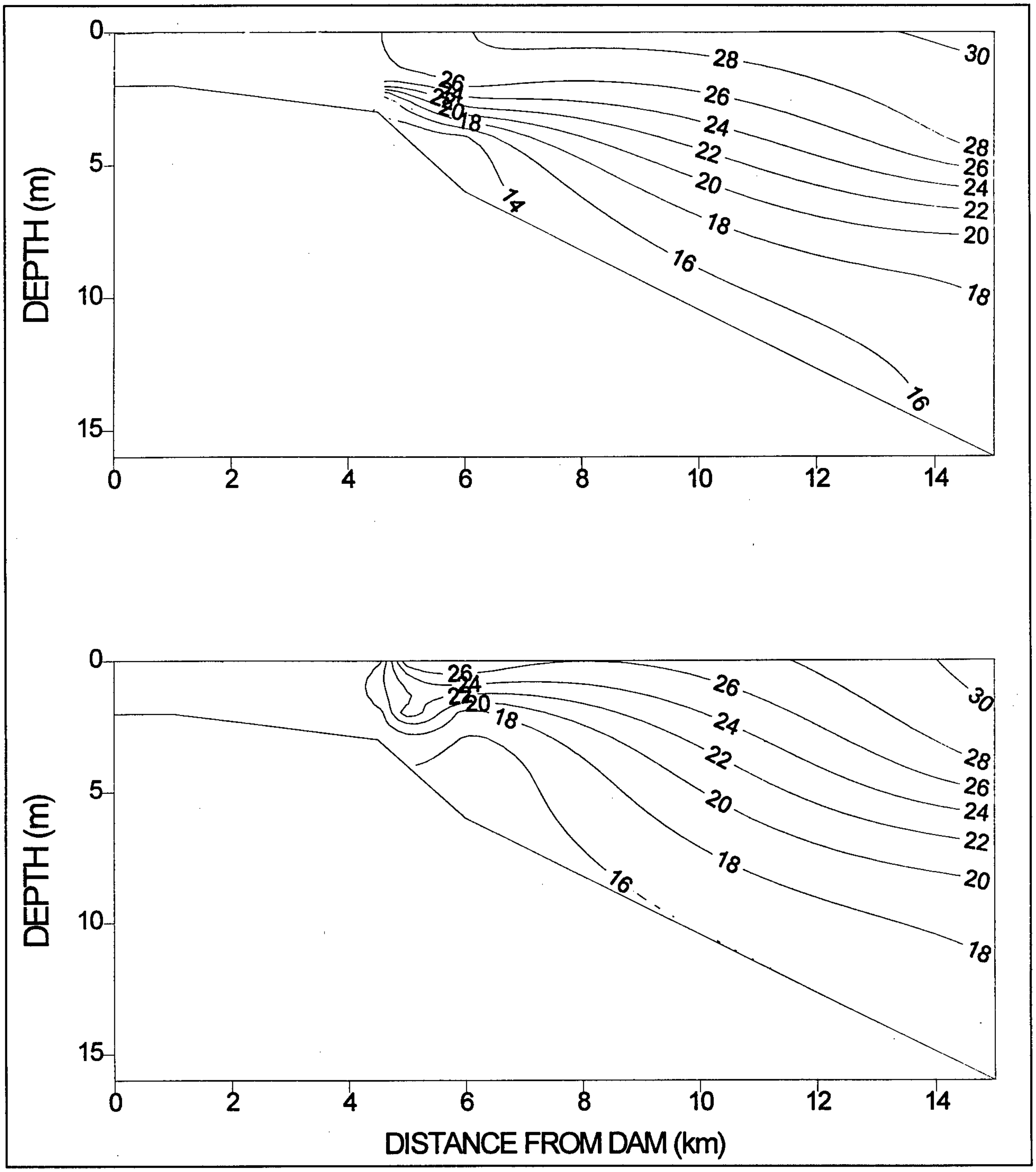

Figure 2. Distribution of temperature (°C) in the Hartwell Lake tailwater prior to generation (upper) and one hour following start-up (lower) on Monday, 31 July 1995

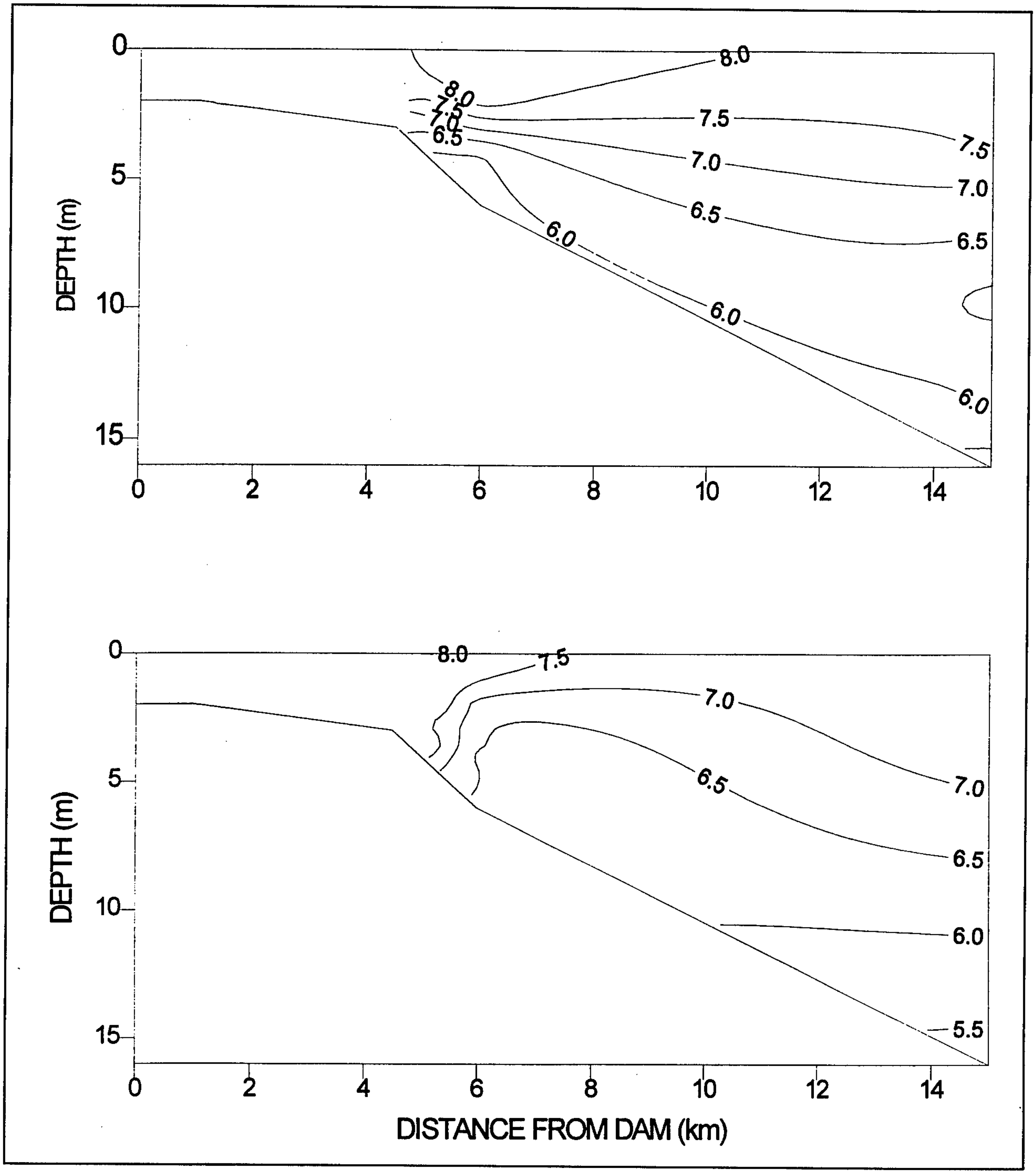

Figure 3. Distribution of dissolved oxygen concentration (mg/ℓ) in the Hartwell Lake tailwater prior to generation (upper) and one hour after start-up (lower) on Monday, 31 July 1995

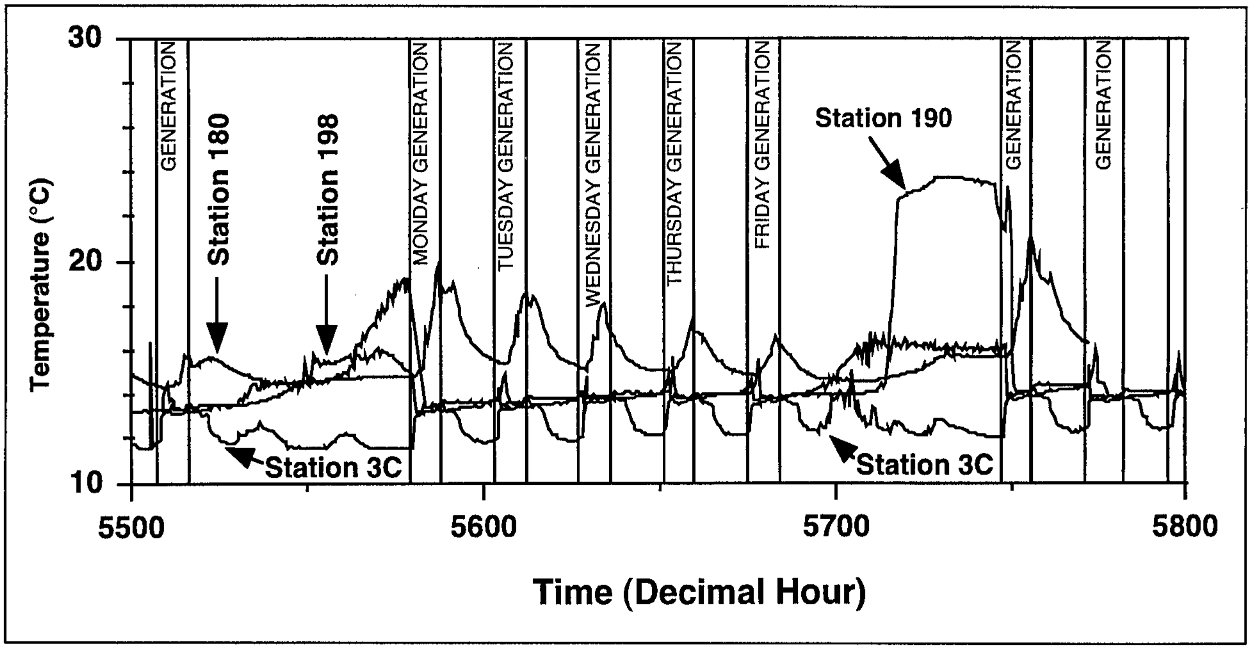

Figure 4. Temperature (°C) patterns at stations 3C (0.2 km), 198 (1.5 km), 190 (5 km), and 180 (15 km) in the Hartwell Dam tailrace and tailwaters. Two weekends of non-generation preceed and follow five weekday generation cycles

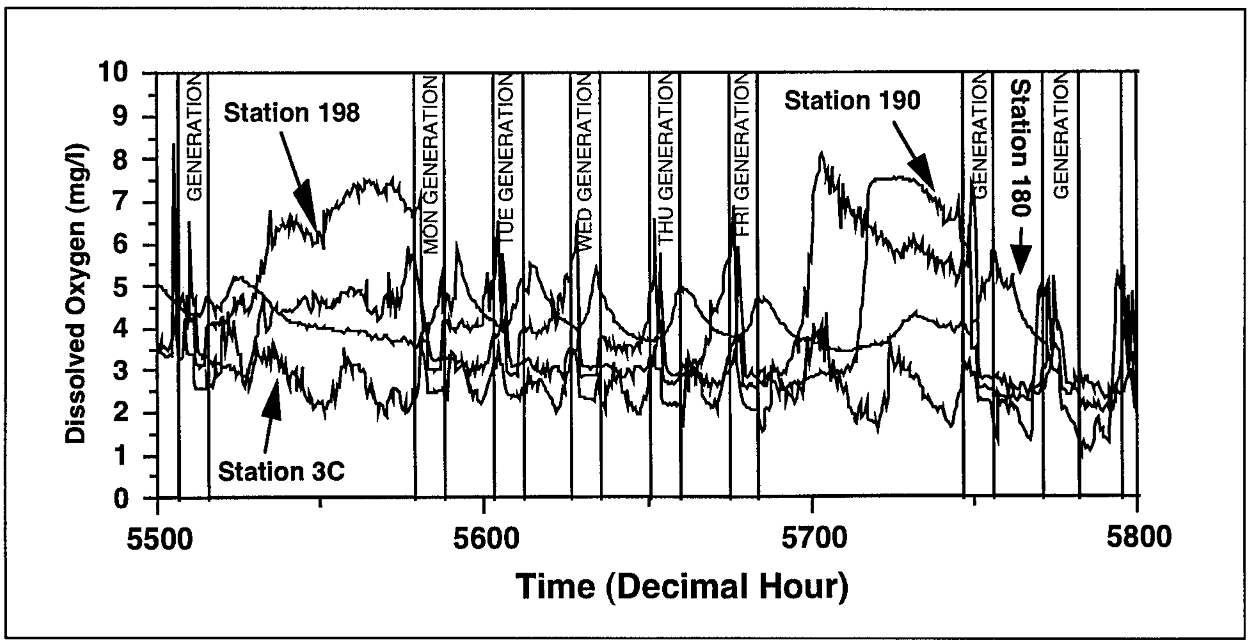

Figure 5. Dissolved oxygen (mg/ℓ) patterns at stations 3C (0.2 km), 198 (1.5 km), 190 (5 km), and 180 (15 km) in the Hartwell Dam tailrace and tailwaters. Two weekends of non-generation preceed and follow five weekday generation cycles

and DO concentration in the HW Dam tailwater followed distinct cyclic patterns. These patterns were in response to operational and, to a lesser degree, diel conditions.

Station 3C

Temperatures at station 3C, located at the buoy line 0.2 km downsteam, indicated that non-generation releases through HW Dam maintained cool temperatures that averaged 12 °C during the weekend period of non-operation. Diel effects were observed during the period. Commencement of hydropower release on Monday through Friday increased station 3C temperatures. Cessation of generation resulted in a tailing off of temperature, as the generation waters were in turn displaced by non-generation releases. DO concentration also exhibited diel and generation effects. Weekend concentrations depicted diel cycling, as DO ranged from 2 to 3.5 mg/ℓ. Weekday generation typically resulted in a spike in the DO concentration to approximately 4 mg/ℓ with a decrease occurring following shut-down.

Station 198

Station 198, 1.5 km downsteam, exhibited a gradual increase in temperature from 13 to 16 °C due to solar input and RBR Lake contribution during the weekends. Hydropower generation decreased temperatures and resulted in 13 to 14 °C temperatures throughout the week. DO concentration at station 198 exhibited the greatest degree of variability during the study, increasing from 4 to 8 mg/ℓ during the weekend. Following generation startup, DO concentrations experienced a rapid spike, attaining 7 mg/ℓ, followed by a decrease of approximately 3 mg/ℓ during the remainder of the discharge cycle.

Station 190

Station 190, located 5 km downsteam, was sufficiently distant from HW Dam that no non-generational release effects were observed. Temperatures increased steadily during the weekend periods from 13° to greater than 20 C. Temperatures during the week of generation closely followed the patterns observed at station 198 and averaged 14 °C with minimal variability. DO concentration did not show apparent diel effects at station 198. Concentrations averaged 5 mg/ℓ during non-release and 3 mg/ℓ during generation. Onset of daily generation resulted in a DO concentration spike to 6 mg/ℓ.

Station 180

A considerable distance (10 km) separated station 180 from the other sampling locations. The temp/DO logger, located 2 m above bottom at a depth of approximately 14 m, recorded hypolimnetic water quality data. Consequently, variability due to diel cycles was much less pronounced. Temperatures were 15 °C throughout the weekend periods. Due to the time lag corresponding to the 15 km distance from HW Dam, weekday generation effects were not observed until the halfway point of the release cycle. The observed effect was a fairly rapid warming of the hypolimnion from 15 to nearly 20 °C on Monday, gradually decreasing in magnitude during the week. On Friday, an increase from 15 to 16.5 °C was recorded. DO concentrations followed the dynamic pattern that were observed with temperatures. Relatively stable concentrations averaging 4 mg/ℓ during non-release increased to greater than 6 mg/ℓ following Monday generation. The magnitude of the change decreased throughout the week. With respect to both temperature and DO concentration, station 180 reflected the effect of the downsteam movement of warmer, more highly oxygenated surface waters. The plunging effect distributed these displaced waters throughout the water column.

Lateral Effects

During non-release periods, solar activity resulted in surface warming and, it may be assumed, increased productivity in the numerous pools adjacent to the thalweg. Within the thalweg itself, temperatures in the immediate vicinity of the dam remained cool (averaging 14 °C) due to non-generational releases through HW Dam. Little to no variability in temperatures was observed along a 1 km reach. During generation, near immediate displacement of stratified conditions by isothermal release was evident within the pools. Reaeration of the HW Lake release occurred rapidly due to turbulence over rocky strata and atmospheric contribution in the immediate tailrace.

Summary

A number of factors influenced temperature and DO dynamics in the HW Lake tailwater. Temperature and DO concentrations exhibited distinct patterns, indicating cause and effect relationships in response to operational and diel cycles, in addition to late summer solar warming. Near-field lateral differences were significant only during periods of non-operation in areas outside the main channel. The near-dam thalweg was most influenced by non-generation contribution through HW Dam.

The immediate tailrace area (approx. 1 km) became isothermal within 15 minutes following generation start-up. The greatest differences in water quality conditions, both near and far-field, were those seen on Sunday afternoons following a weekend period of non-generation as contrasted to the first full day of operation on the following Monday. Downlake effects, while muted, were evident at a distance of 15 km.

Quantifying Sedimentation Using a Three-Dimensional Numerical Sedimentation Model

by
Brad R. Hall[1]

Introduction

The term sedimentation embodies the processes of erosion, entrainment, transportation, deposition, and compaction of sediment (Vanoni 1975). For modeling sedimentation, a variety of criteria are used to describe thresholds and rates of these processes. Particle erosion occurs upon sufficient shear and lift forces on the bed, typically characterized by evaluating Shields stress. Entrainment of sediment particles is established by sufficient near-bed turbulence, typically characterized by a Rouse number criteria. Sediment particle transportation occurs through a combination of particle transposition, advective transport, or diffusive transport. Particle deposition occurs through gravitational settling. Bed compaction is generally characterized as a time dependent function of particle size and overbearing pressure. One- and two-dimensional numerical simulation models of sedimentation have been developed and applied for several years; however, successful application of three-dimensional numerical simulation models is a relatively new development.

Theoretical Basis

The basis for the mobile bed CH3D model is a solution of conservation equations. The numerical model consists of two primary components: the hydrodynamic model and the sedimentation model. The model can be executed in a stand-alone hydrodynamic model simulation and has been used to investigate a variety of coastal and estuarine circulation studies (Johnson et al. 1991). Mobile bed capability was added to the hydrodynamic model under a joint effort by the Hydraulics Laboratory of the U.S. Army Engineer Waterways Experiment Station and the Iowa Institute of Hydraulic Research (Spasojevic and Holly 1994). A boundary-fitted nonorthogonal finite difference approximation in the horizontal plane and a sigma-stretched approximation in the vertical direction of the governing equations are used for the approximations of the governing equations.

The hydrodynamic model solves the depth-averaged Reynolds approximation of the momentum equation for velocity and the depth-averaged conservation of mass equation for water surface elevation. The deviation from the depth-averaged velocity is computed for each cell by solving the conservation of mass equation in conjunction with an algebraic closure for vertical momentum diffusion. Sedimentation computations are based on a two-dimensional solution of the conservation of mass for the channel bed (i.e., the Exner equation) and a three-dimensional advection-diffusion equation for suspended sediment transport.

Auxiliary Relationships

Technological and theoretical constraints on solving the governing equations require incorporation of empirical relationships for developing a useable simulation model. Typically, relationships that define frictional loss, bed material movement, turbulence dissipation, and near-bed reference concentrations are used to supply the

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necessary link between computed physical parameters such as depth, velocity, suspended sediment concentration, and sediment exchange on the bed. The relationships presently employed in the CH3D model are described below:

- **Bed Material Load.** The bed load predictor developed by Van Rijn (1984a) is presently used. Although the original development was for a uniform grain-size of bed material, the relationship was generalized to account for multiple grain size availability and transport. Efforts are presently underway to implement the Laursen, Yang, and Acker-White bed material load relationships for use in the CH3D model (Engel, Hotchkiss, and Hall 1995).

- **Reference Concentration.** The relationship developed by Van Rijn (1984b) is used to compute a reference concentration for the near-bed source term for suspended sediment load. Concentration gradients in the near-bed zone are estimated from the computed concentration profile to obtain the vertical flux of bed sediment entrained in the flow field.

- **Vertical Diffusion Coefficient.** The vertical diffusion coefficient for suspended sediment is adjusted to account for the influence of concentration and near-bed turbulence intensity on momentum and mass diffusion. Methods presented by Van Rijn (1984b) are presently used in the model.

- **Bed Material Exchange.** Any sedimentation model that accounts for multiple grain-size characteristics of the channel bed requires a specification of the active, or mixing layer, of the channel bed. The active layer description and hiding effects due to nonuniform bed material gradation are described in Spasojevic and Holly (1994).

- **Transport Mode Allocation.** A unique feature of the mobile bed CH3D model is the allocation of bed material load for transport as either bed load or suspended load. The two modes of transport are independently accounted for in the two-dimensional solution of the Exner equation. The criteria is based on a relationship developed from data presented by Van Rijn (1984b).

Application

The Horseshoe Bend Reach of the Atchafalaya River lies approximately 13 miles downstream of Morgan City, LA. The river takes a relatively sharp left hand bend with a mid-channel bar, splitting the thalweg into the outer Horseshoe Bend channel and the inner Crewboat Cut channel (Figure 1). The navigation alignment follows the Horseshoe Bend channel. Shallow-draft vessels often follow the shorter inner channel. Dredging to maintain project dimensions of the Horseshoe Bend navigation channel has increased in recent years. The modeling effort focused on quantifying relative amounts of erosion and deposition in the two channels (Figure 2) under a variety of hydrologic scenarios. This information was used to determine if depositional processes would favor maintaining the navigation channel in either the Horseshoe Bend or Crewboat Cut navigation alignments.

Synopsis of Results

Average annual deposition volumes for both alignments were estimated from model results, using flow-duration characteristics of the 1986 through 1994 recorded daily discharge at the Atchafalaya River at the Morgan City gauge. These volumes of sediment deposition are described for the two conditions below:

Deposition volumes were computed for the quantity deposited in either of the navigation channels. This entailed identifying the computation cells that overlap with the navigation channel and exporting the deposition quantities for each of these cells into a spreadsheet for analysis. The total deposition volume was computed by integrating the computed deposition quantity over an average annual hydrograph. The computed deposition volumes were 790,000 cu yd per year for the Horseshoe Bend alignment and 1,130,000 cu yd per year for the Crewboat Cut alignment. Computed deposition volumes in the Crewboat Cut alternative were approximately 50 percent higher than the computed volume in the Horseshoe Bend alternative.

Interpretation of Numerical Model Results

This analysis indicates that average annual sediment deposition along a navigation channel maintained to a 20-ft draft by 400-ft width dimension along the Crewboat Cut alignment would be higher than the average annual sediment deposition that would occur on a 20-ft draft by 400-ft width dimension channel maintained through Horseshoe Bend. The morphology of the study reach indicates that the sediment deposition through the study reach is very similar to the deposition pattern that would be expected on a point bar deposit. The effects of curvature in the flow field (Figures 3 and 4) at this location result in higher near-bed suspended sediment concentrations and sediment deposition along the inside of the channel bend (Figures 5 and 6). Dredging a channel through the Crewboat Cut increases the discharge through the proposed alignment, which increases the quantity of sediment transported and eventually deposited in this proposed alignment.

Acknowledgments

The tests described and the resulting data presented herein, unless otherwise noted, were obtained from research conducted under the Civil Works Program of the United States Army Corps of Engineers by the U.S. Army Engineer Waterways Experiment Station. Permission was granted by the Chief of Engineers to publish this information.

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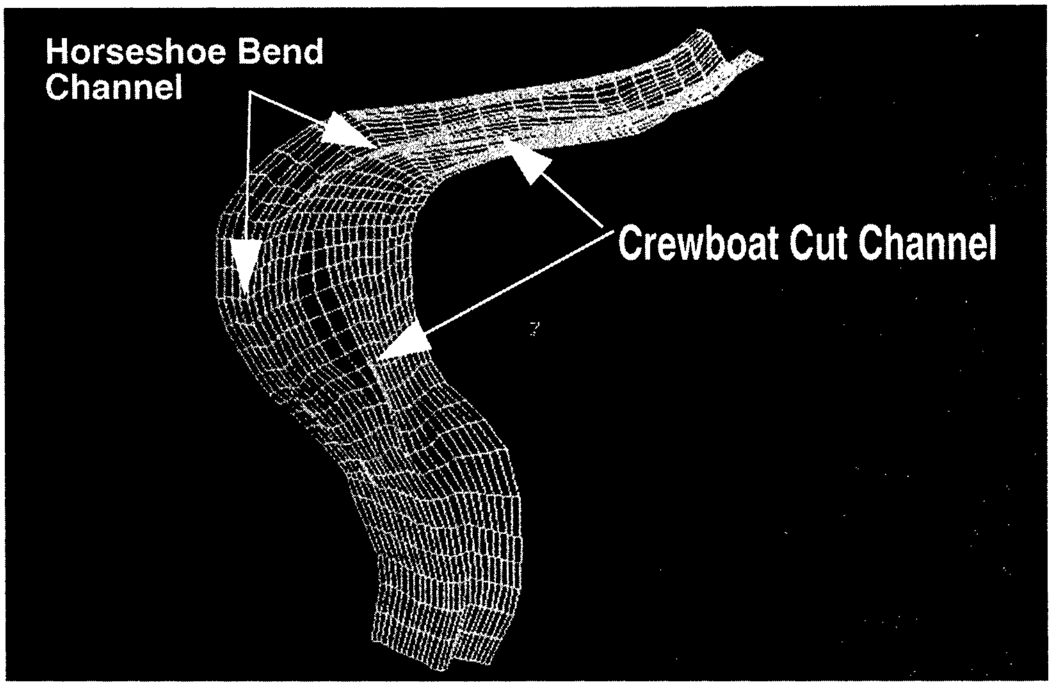

Figure 1. Computational grid, showing existing Horseshoe Bend and Crewboat Cut channels

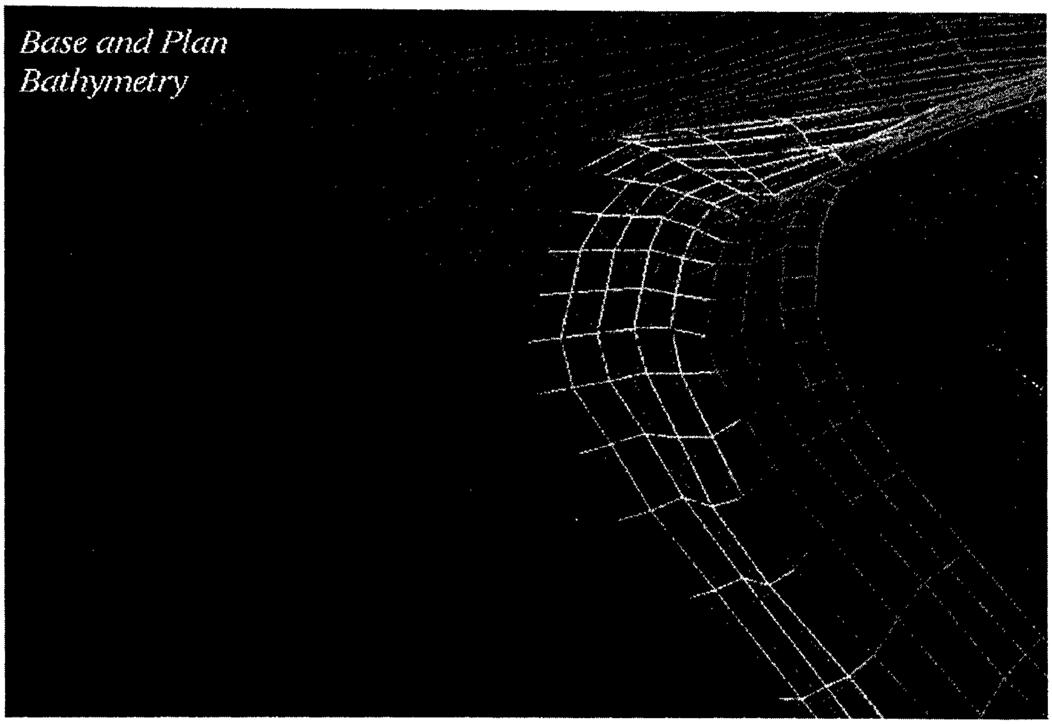

Figure 2. Detail of computational grid highlighting dredge template for Crewboat Cut channel

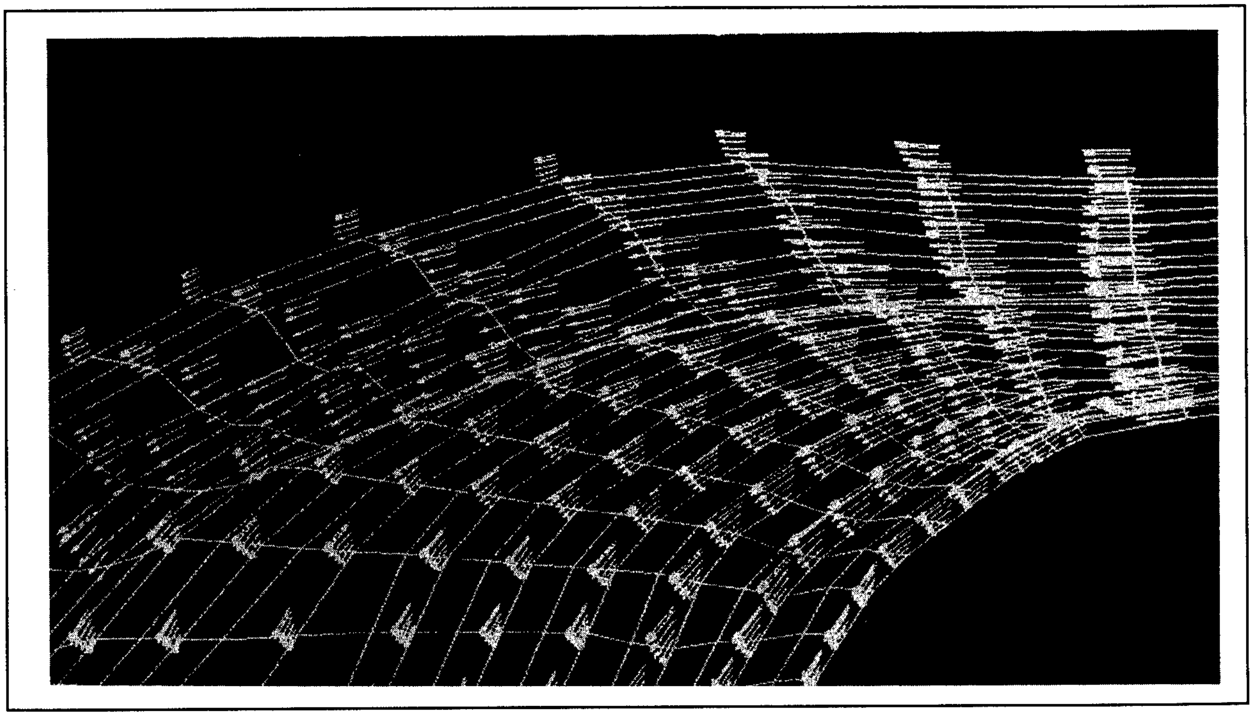

Figure 3. Perspective view of computed velocity in study area

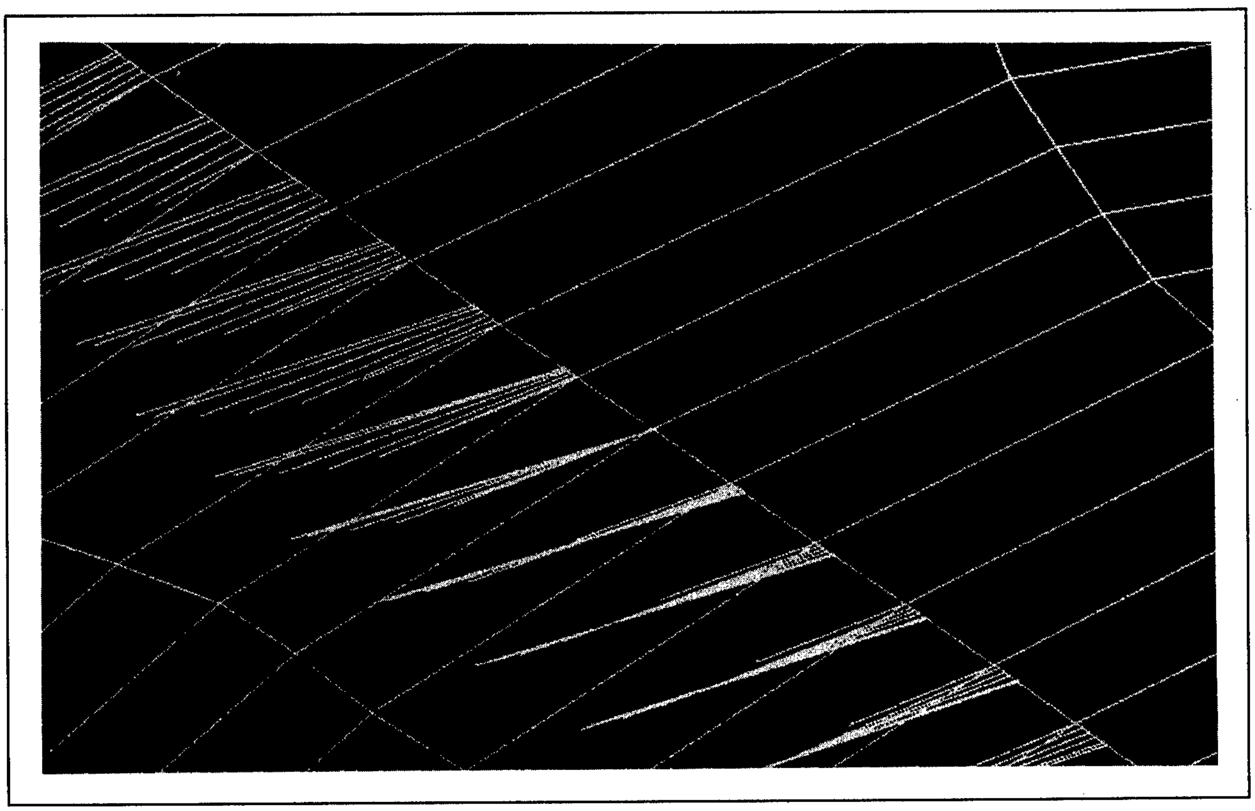

Figure 4. Plan view of computed velocity in study area

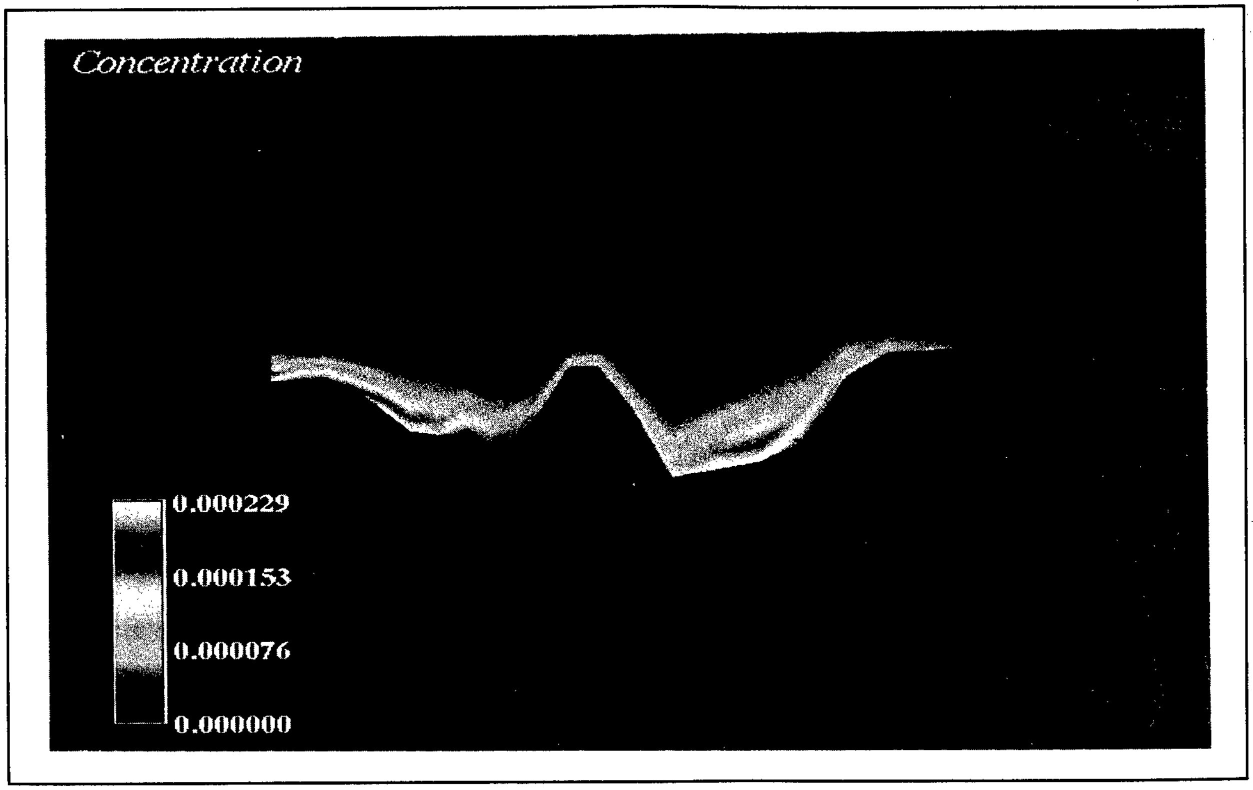

Figure 5. Computed suspended bed material concentration (Concentration is given in dimensionless units, mass basis (grams sediment per gram water))

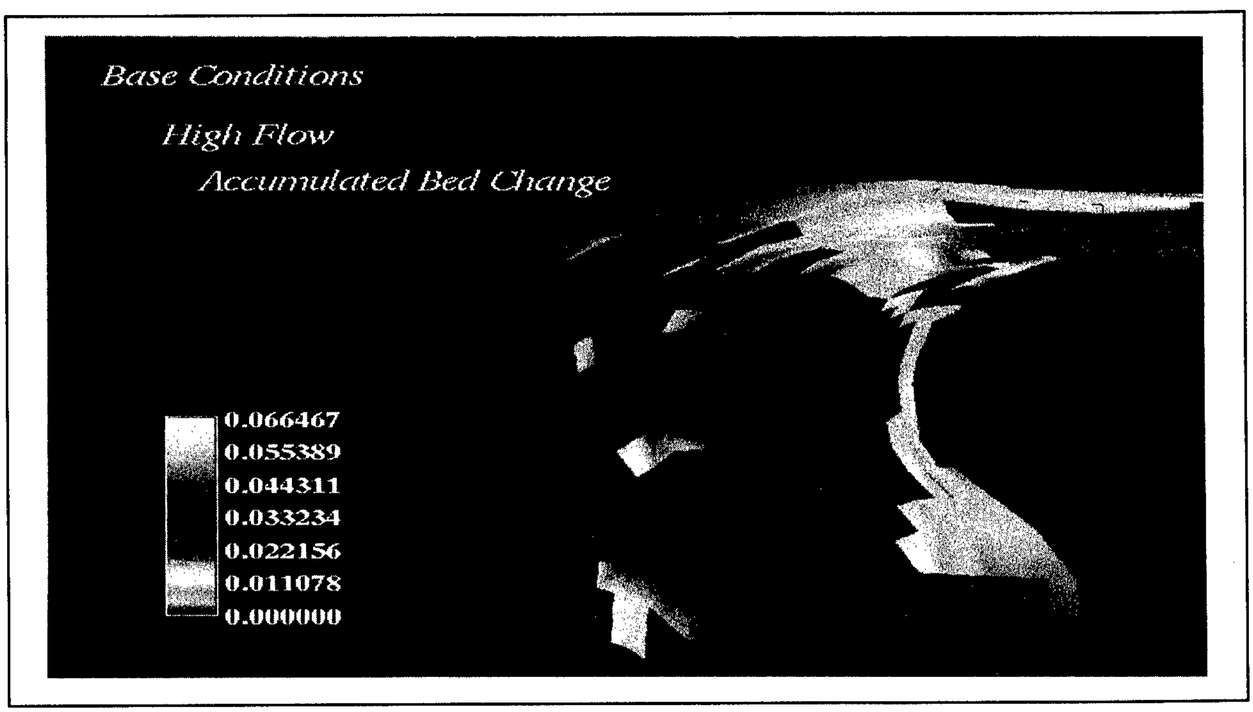

Figure 6. Base condition daily bed change (Units are in meters per day)

Effect of Wave-Induced Resuspension of Fine Sediment on Water Quality in Nearshore Zone

by
T. M. Parchure,[1] *Keu Kim,*[1] *and W. H. McAnally, Jr.*[1]

Introduction

Wave-induced resuspension of fine sediment in littoral zones has ecological significance due to its effect on submersed aquatic vegetation (SAV). Resuspension of a very thin layer, on the order of a couple of millimeters thick, releases millions of primary particles in the water column. These particles with a large specific surface area significantly reduce light penetration to the bed. It has been reported that changes in water quality have caused light reduction, which in turn has resulted in a very serious decline of SAV in the tributaries of Chesapeake Bay (Kemp et al. 1983; Orth and Moor 1983). Response of fine sediment bed to wave action has been studied only very recently, and response of different types of SAV systems to (a) the changing nutrient loading scenarios and (b) the changing suspended sediment concentration has still not been adequately understood. Based on the available information, a module for predicting the resuspension concentration of fine sediment in the littoral zone is being added to an existing water quality model. Details of development of this module have been described in this paper.

Fine Sediment Processes

Grain-size analysis of sediment along the length of a river typically shows that the upper reaches of rivers have coarse sediment in the range of gravel and sand, whereas the estuarine reach at the river mouth has fine, cohesive sediments in the range of clays and silt. In an estuarine environment, the processes of erosion, deposition, and consolidation of this fine sediment take place in a cyclic order as shown in Figure 1 (Mehta et al. 1982). Due to time-varying magnitude and direction of flow under tidal regime, the time available for the sediment consolidation process is relatively very small. Hence, a very thin and easily erodible layer (on the order of a few centimeters thick) is formed at the surface that participates in these processes. This thin layer between the water column and firm bed is in the form of a fluid mud. The initial removal of sediment from a consolidated sediment bed is termed as erosion, whereas removal of the sediment that has been recently deposited from the water column is termed as resuspension. Although erosion or resuspension occurs mainly due to tidal flow in an estuary, action of locally generated waves can also be significant in the processes, particularly in the littoral zone. A field study conducted by Halka, Sanford, and Ortt (1994) has shown that (a) shoreline erosion makes a significant contribution to suspended sediment concentration and (b) resuspension activity under normal tidal conditions is greater in deep water where the tidal currents are strong. In a study of Chesapeake Bay, Ward (1985) noted that in water depths less than 6 m, resuspension of bottom sediment by wind provides a major source of suspended sediment.

The combined effect of both waves and current is more complex than their individual effect. Research of this phenomenon in physical models has shown that when action of each

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

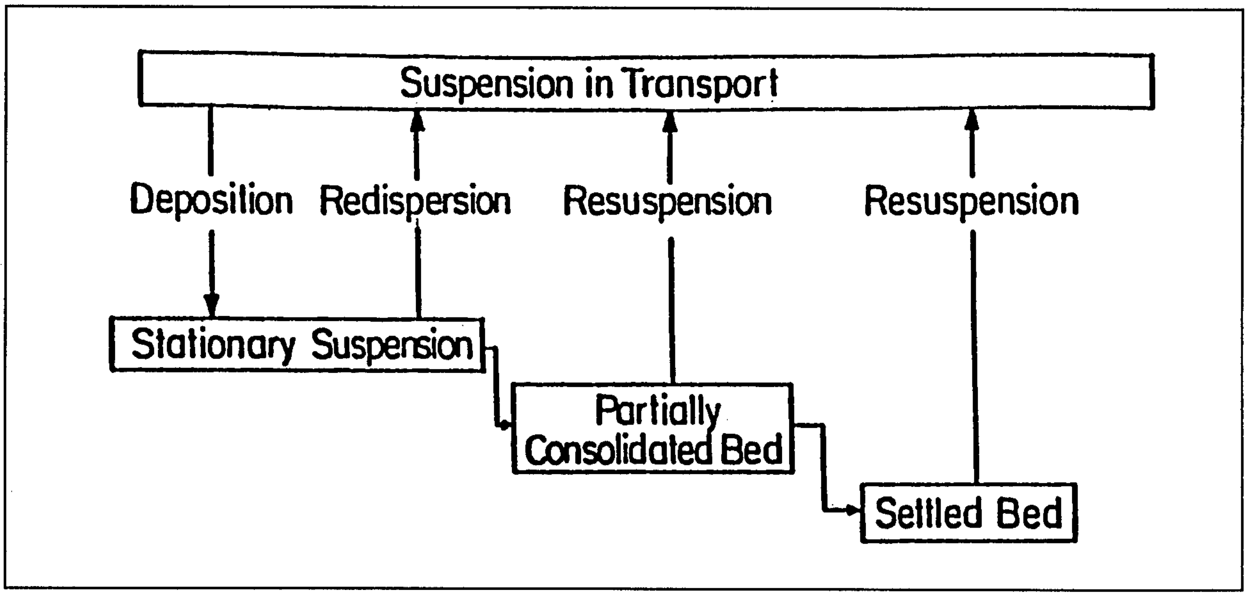

Figure 1. Cyclic processes of fine sediment dynamics

both waves and current, physical modeling factor is so important as to require simulation of becomes more complex, and the similitude laws related to each action are generally not compatible (Latteux 1986). In view of these difficulties, it is necessary to develop numerical models to study the aspects of sediment resuspension under combined action of waves and currents.

A three-dimensional numerical hydrodynamic, salinity, and temperature model of Chesapeake Bay was developed at the U.S. Army Engineer Waterways Experiment Station (WES) by Johnson et al. (1991). The objective of this model was to provide flow field to the three-dimensional water quality model of the Bay, which is also being developed at WES (Cerco and Cole 1993). The water quality model does not include a sediment module at present. Hence the objective of the study described here was to incorporate a code for estimating suspended sediment concentration caused by locally generated wind-induced waves and to also include interaction of SAV and suspended sediment that is significant in water quality of estuarine littoral zones.

Significance of SAV

The depletion of SAV can be a reason for considerable concerns on the ecological balance of a water body. Hence the related parameters have been examined, one of which is the suspended sediment. Interaction between SAV and suspended sediment has been established through field investigations. Ward et al. (1984) have made the following observations: (a) suspended particulate matter (SPM) concentrations were significantly lower inside grass beds than adjacent unvegetated areas, because wave energy was attenuated by vegetation that enhanced sediment deposition, (b) when water levels were higher during spring tides or storm surges, plants were less effective in attenuating wave energy, and the SPM concentration inside the seagrass beds was higher, (c) in tidally averaged water depth, surface waves and seagrass structure interact to influence spatial and temporal patterns of SPM in shallow estuarine embayments, and (d) in water depth less than 2 m, resuspension of bottom sediment increased 10 times in unvegetated regions when wind velocity was greater than 25 km/hr. Although qualitative observations are available, quantification has been attempted only by very few researchers. Fonseca et al. (1982) correlated Froude Number (F) with the mean bending angle of the seagrass canopy as a whole. They have noted that maximum bending occurred at F = 1.0, but most bending had taken place at F = 0.4 at a flow velocity of 40 to 50 cm/s. They have also correlated shear velocity and roughness height with seagrass surface area.

Sediment Characterization

Field observations of Sanford (1994) have shown that the estimated shear stress that resulted in the largest resuspension was about four times smaller than the estimated shear stress that produced a lower resuspension at the same site. The reason for this apparently contradicting result has been explained by the fact that the sediments were less erodible during the period of higher stress than they were during lower stress. Hence, characterization of fine, cohesive sediments is more important than that of coarse, noncohesive sediments because of the large number of parameters that affect the sediment properties of cohesive sediments and hence their erosional and depositional behavior. As many as 32 parameters have been identified that influence the properties and behavior of fine sediments (Cohesive Sediment Research Newsletter, 1992). These parameters may be grouped under (a) physico-chemical properties of fluid, (b) physico-chemical properties of the mud, and (c) water-mud exchange processes. While dealing with erosion of fluid mud, rheological properties are very significant, and the model presented in this paper requires their values as input.

Wave-Mud Interaction

Fine sediment beds are classified as (a) uniform bed, which has a constant shear strength over depth, (b) stratified bed, which has layers of varying shear strength, and (c) fluid mud bed. The erosional properties of these are quite different from each other. The entrainment of sediment due to hydrodynamic instability at the interface resulting in the generation and breakup of sediment billows has been described in terms of the balance between production of turbulent kinetic energy, buoyancy work in entraining sediment, and viscous energy dissipation (Kranenburg 1994). Thus, this process is distinct from surface erosion of sediment flocs that occurs over a consolidated cohesive sediment bed. It is therefore essential to determine the type of bed and select an appropriate mathematical formulation.

It is noted from recent research on fluid mud formation that the action of waves on cohesive sediment bed can cause fluidization of bed, often without any significant change in its bulk density (Ross and Mehta 1991). This is achieved only by breaking the interparticle bonds, and hence a very weak and easily erodible layer of fluid mud is created by the wave action. It has been assumed in the present model that fluid mud already exists at the site under consideration.

The vertical suspended sediment transport equation is given as

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial z}\left(w_s C + K_z \frac{\partial C}{\partial z}\right) \tag{1}$$

where

w_s = sediment-settling velocity

C = wave-averaged suspension concentration

z and t = water depth and time, respectively

K_z = sediment diffusion coefficient, which is a function of diffusivity K_0 and density stratification factor ϕ.

Two components make up K_0, one due to waves and the other due to current. The wave diffusion coefficient is a function of angular wave frequency, wave number, and scaling coefficient. The current-induced diffusion coefficient is related to the Karman constant, Manning's bed resistance coefficient, and mean current velocity. The stratification factor ϕ is a function of gradient Richardson Number. The net quantity of sediment remaining in suspension is determined from the difference between erosion flux and deposition flux averaged over a preselected duration of time. The following equation is used to determine sediment settling velocity w_s:

$$W_s = \begin{cases} W_{sf} & (C < C_1) \\ \dfrac{aC^n}{(C^2 + b^2)^m} & (C > C_1) \end{cases} \qquad (2)$$

In this equation, W_{sf} is the free settling velocity, and a, b, n, and m are coefficients. The concentration C_1 defines the limit below which w_s remains constant, equal to W_{sf}.

Considering all the factors mentioned above, the following main equation is used to determine the net entrainment rate of fluid mud, taking into account the flux of sediment from fluid mud to the water column and its removal through settling:

$$F_n = \begin{cases} \rho_m u_b \beta (R_{ic}^2 R_{ig}^{-1} - R_{ig}) - W_s C|_{z=0} & (R_{ig} < R_{ic}) \\ - W_s C|_{z=0} & (R_{ig} \geq R_{ic}) \end{cases} \qquad (3)$$

where

F_n = net rate of sediment entrainment under combined action of waves and a weak current u_b

ρ_m = sediment density

R_{ig} and R_{ic} = global and critical Richardson Numbers, respectively

β = entrainment coefficient, which depends on flow field.

Numerical Model

The numerical model has been developed as a "box-type" model, which does not take into account advection between adjacent cells. Sediment concentrations as a function of depth and time are given by the model. Option for plotting the profiles is included. The model has the following composition:

a. Analysis of wind data and estimation of wave heights and wave periods: Wind data can be obtained from the weather station nearest to the site of interest. Sustained wind for a sufficiently long duration together with a sufficiently high wind speed is the essential requirement for generation of waves. It is therefore necessary to filter out data consisting of short wind durations and very low wind speed. Only certain wind directions that approach the site over the water surface (and not from the land) would be significant for wave generation. Hence rapidly varying wind from significant directions are reassigned into groups of adjacent directions. A time series of average values of all the three parameters (wind direction, wind speed, and wind duration) is generated.

b. Determination of fetch: Fetch is defined as the area in which waves are generated by a wind having a fairly constant direction and speed. Waves locally generated by wind in estuaries are fetch limited because of a relatively small width between the banks and often a very irregular geometry. Computer programs exist for working out fetch length; however, for a given site, fetch length for any significant wind direction can be easily determined using area maps.

c. Estimation of wind-induced wave height and wave period: Wind-wave generation on restricted fetches has been described by Smith (1991). Codes with necessary modifications have been written for estimating wave heights and wave periods for the given simultaneous combinations of wind speed, wind duration, and fetch length.

d. Sediment characterization: The type of bed present at the site is selected from field data, and values of different parameters used for its characterization are determined from laboratory measurements. The parameters include rheological properties, bed density, diffusion coefficient, and empirical constants used in various equations related to erosion and deposition.

e. Calculation of suspension concentration: The code calculates the global Richardson Number, and using the specified value of critical Richardson Number calculates the sediment flux. The sediment settling velocity is calculated for the given input. Using these two time-varying quantities, suspension concentration as a function of water depth and time is determined.

Model Verification

Field data on total suspended solids (TSS) collected at Chesapeake Bay during August 1993 were used for preliminary verification of the model. A comparison of computed and observed suspension concentration is given in Figure 2. Although the comparison is quite satisfactory for preliminary modeling purposes, following are the possible reasons for deviation between the two data sets: (a) the amount of sediment in the form of background concentration in the field data, (b) presence of suspended organic particulates in the field that have a source other than bed sediment, and (c) sediment advected from other areas not simulated in the model.

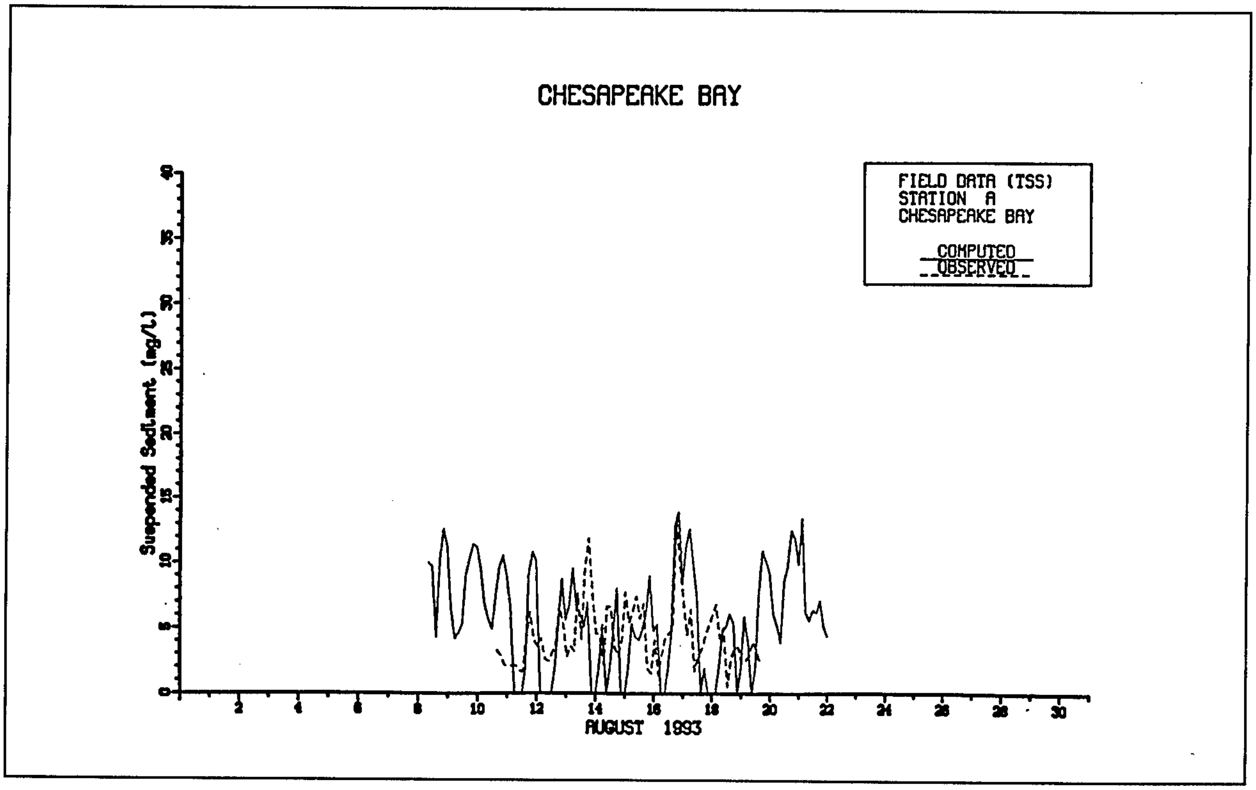

Figure 2. Verification of computed suspension concentration with field data

Conclusion

The numerical model presented in this paper can be used for a reasonable prediction of wave-induced resuspension of sediment in an estuarine littoral zone with cohesive sediments.

Acknowledgment

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Shoreline Erosion—Management Techniques and Practices

by
John L. Andersen[1]

Introduction

Shoreline erosion is a major problem occurring on nearly all reservoirs located in areas of erodible soils such as the Midwest. The Omaha District alone has over 6,000 miles of reservoir shoreline of which between 70 and 90 percent is eroding. Common sense dictates that taking sediment from the shoreline and allowing it to enter the reservoir will result in major ecological damage and, in turn, add to massive future reservoir renovation costs.

It is unfortunate that shoreline erosion problems are often ignored by agencies that manage reservoirs unless something considered important is being threatened such as recreation areas, cultural resources, or structures (McCartney 1976). Few will dispute the importance of reservoir functions such as flood control, hydropower, recreation, etc., yet, the reservoir and its problems such as shoreline erosion are commonly ignored as though thought of as unimportant or something that can be delayed indefinitely. Data have shown that reservoir usefulness declines with age (Kimmel and Groeger 1986). Shoreline erosion is a part of the aging process and plays a significant role in the loss of spawning and rearing habitat utilized by species such as the largemouth bass. The decline or loss of sport fisheries is but one example of impaired reservoir function considered valuable by recreators. Data have shown that reservoirs decline in biological productivity with age and that useful functions are impaired well before the basins are filled with sediment (Kimmel and Groeger 1986).

The instances where shoreline erosion prevention efforts have been undertaken are often piecemeal, ineffective, and too small to be of significant benefit to the reservoir system. In addition, monitoring to determine the success or failure of a particular effort is rarely done. Further, since monitoring is essentially nonexistent, communication between individuals or offices regarding the success or failure of various protection efforts is not communicated. The combination of a lack of monitoring and lack of communication has resulted in a pattern of repeating failures and not repeating successes.

Shoreline erosion problems can be divided into two types: (a) reservoirs with relatively stable pools and (b) reservoirs with large annual pool fluctuations due to hydropower, flood control, navigation, etc. Methods to resolve shoreline erosion problems on reservoirs with relatively stable pools are numerous and readily available and consist of using vegetation, rock, geotextiles, or some combination thereof. These methods are often manpower intensive; however, labor problems can often be resolved by utilizing civic groups such as Boy Scouts, fishing clubs, environmental groups, etc. Shoreline erosion prevention on reservoirs with large annual fluctuations is difficult, and standardized methods other than the expensive placement of rock do not exist. It is imperative that additional methods be sought and that a

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systematic program of resolving the problem be initiated. The practice of purchasing more land when shoreline erosion exceeds Federal boundaries is not the answer and constitutes a short-sighted and economically unwise solution to a long-term problem.

Causes of shoreline erosion such as wind-wave action, boat wakes, ice, and freeze-thaw cycles may act alone or in concert to damage the reservoir system. The movement of ice during the spring can loosen bankline soils, yet the sediment is removed during the open-water season by the action of waves. As a result, potential solutions to the problem must prevent erosion due to wave action as well a being able to withstand the actions of ice or other causative factors.

Shoreline erosion has many impacts upon reservoir systems that will ultimately result in the loss of one or more reservoir functions. Continued shoreline erosion, for instance coupled with sedimentation from the watershed, will result in the loss of fisheries such as the bass-bluegill populations common in so many reservoirs. This effect has been documented in the Salt Valley Reservoirs located near Lincoln, NE. In addition, the continued loss of shoreline habitat and resultant sediment focusing will also result in the loss of significant parts of the food web. This effect has been documented in the benthic community of the Salt Valley Reservoirs, a system of 10 reservoirs constructed for flood control near Lincoln, NE. In these reservoirs, it was found that benthic diversity decreased significantly and that benthic biomass decreased approximately 90 percent between the period 1968-1970 and 1991-1992 (Popp and Hoagland 1995). Grundy (1993) states that continued shoreline erosion not only damages the fishery, but reduces water clarity and diminishes water quality in addition to reducing reservoir depths.

Methods

Several methods of stopping shoreline erosion have been attempted in the Omaha District. These methods, excluding the standard of massive riprapping, consist of placing hay bales or fiber rolls offshore, using geotextiles, vegetative plantings, creative use of rock, or some combination of the above. These methods are described in the following paragraphs.

Bales or Fiber Rolls

Using hay bales or fiber rolls for shoreline protection involves placing them in the reservoir at a selected distance from the eroding bank. Such efforts may or may not be combined with aquatic planting between the eroding shoreline and the bales (See Figure 1). To date, shoreline erosion prevention efforts using bales or fiber rolls have exhibited a failure rate in excess of 90 percent on reservoirs within the Omaha District. These efforts commonly last 1 to 3 years and, at best, simply delay shoreline erosion for a short time.

The purpose of the bales is to provide quiet water, which in turn provides an area suitable for planting and maturation of the aquatic vegetation. The plants, once established, protect the eroding bank from wind-wave action after the bales or fiber rolls have decayed. A number of problems associated with the use of bales or fiber rolls have become evident, these problems are as follows:

a. Not planting the quiet area between the eroding bank and the bales with selected species may result in colonization of the area by terrestrial plants. Once the bales or fiber roll has decayed away, constant wetting by wind-wave action or increased pool elevations will destroy the terrestrial

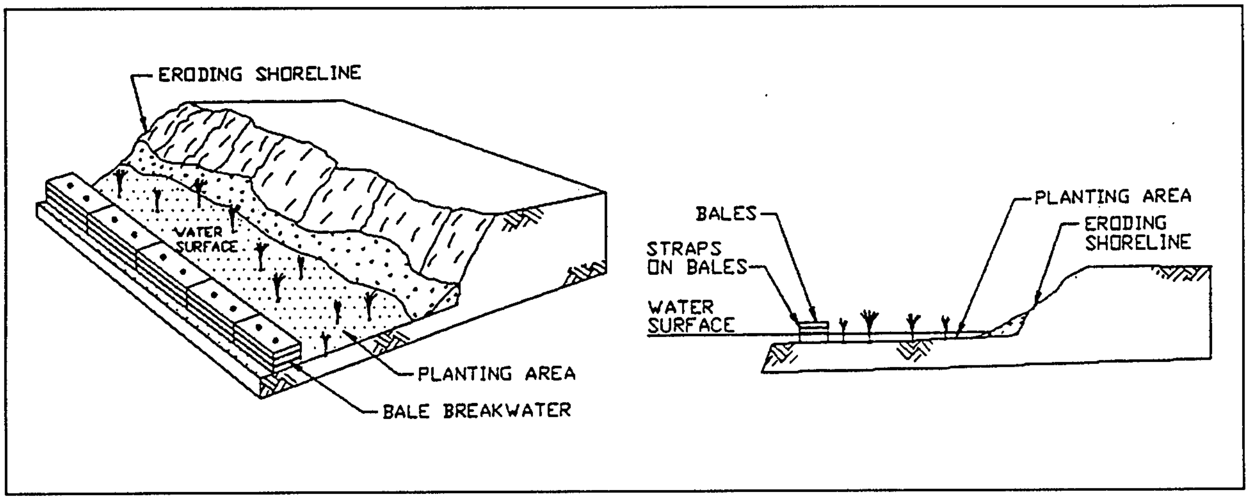

Figure 1. Diagrammatic representation of hay bale usage to halt shoreline erosion (Large round bales now common in farming may also be used but require heavy equipment for placement)

plants; the shoreline erosion will proceed where it left off prior to the erosion prevention effort.

b. The bales or fiber roll should not be tied into the bank or shoreline. Littoral drift should be allowed to freely enter and settle out in the area protected by the bales or fiber roll. The sediments moving along the shoreline will deposit and heal an eroding bank, thus providing suitable habitat for the growth of aquatic vegetation. Sediment deposition will form beaches being built naturally, which in turn can result in a wider band of established vegetation.

c. In most efforts with the Omaha District, the bales or fiber rolls were placed 3 to 4 ft from the eroding bank. This placement is too close to the bank and can result in two problems: (1) The eroding bank can collapse and push the bales out of the way, thus causing a breach in protection, or (2) The band of vegetation established will be too narrow to withstand the impact of wind-wave action and will be too small to allow regrowth to heal the impacted area.

Geotextiles of Fabric Blankets

The use of geotextiles has been attempted only once as a reservoir shoreline erosion preventative in the Omaha District. The effort appeared to be initially successful; however, drought caused the reservoir pool to drop below the lower edge of the fabric blanket that had been under water. The exposed edge of the blanket began to unravel, and shoreline erosion proceeded as a vertical scarp beneath the fabric. Ice may also have played a roll in damaging the geotextile.

Vegetative Plantings

Numerous attempts at halting shoreline erosion using vegetative plantings, both with and without the use of bales or fiber rolls, have been attempted. Successes have been few. The problems with vegetative plantings are numerous. Examples include the following:

a. Newly placed plants must be able to withstand wind-wave action and other erosive forces where terrestrial and aquatic habitats are coupled. This area can be a difficult environment to establish plants without the use of protective measures such as bales or breakwaters.

b. Waterfowl, especially goose populations established at many reservoirs have become a problem in that they will pull up and eat all the newly placed plants. The geese are more efficient at removal than humans are at planting. The use of fencing, chemicals, or other protective measures are vital in attempting to create a viable stand of vegetation in areas of resident goose populations.

c. Drought can cause the loss of established vegetative plantings. In one instance, an Omaha District effort was totally eliminated by drought. The drought caused the reservoir pools to drop, leaving newly established plants high and dry, well away from the reservoir. The aquatic plants such as cattails, river bulrush, and soft-stemmed bulrush withstood this condition for a period of months. However, as the reservoir receded, terrestrial plants invaded the area. The terrestrial vegetation, often well in excess of 8 ft in height, simply outcompeted the aquatic vegetation, even though the area was periodically watered.

d. Another type of shoreline protection has been attempted using reed canary grass, Chinese silver grass, and other species. This method involves establishing vegetation on flat or gently sloping surfaces and allowing the plants to prevent the formation of an eroding vertical scarp. Observations on this type of effort indicate that the established vegetation provides excellent surface cover but will not prevent shoreline erosion. A vertical scarp will become established and proceed up the slope despite the established vegetation. The root systems may slow the rate of erosion, but will not stop shoreline erosion.

Innovative Use of Rock

Nearly all reservoir shoreline erosion prevention efforts involving the use of rock have consisted of using standard methods of riprap. Unfortunately, cost prohibits the rip-rapping of thousands of miles of eroding shoreline. Cheaper and more creative uses of rock should be sought. Figure 2 illustrates one such method currently being attempted in the Omaha District. The placement of rock in separate piles or hard points can decrease the amount of rock used and eliminate the effort and cost of spreading the rock over the entire bankline. In addition, the areas between the rock hard points can be vegetated with aquatic plants thus creating habitat diversity.

Figure 3 illustrates a conceptual method of utilizing rock along eroding shorelines of reservoirs with large annual pool fluctuation. Large annual fluctuations preclude the use of vegetation for obvious reasons. This method requires the use of a piece of farm machinery, a rock picker, and using rock left on the shoreline as a result of bank recession. The rock is picked up and placed in a preselected configuration and location. This type of shoreline protection eliminates the purchase of quarried rock and the expense of trucking the rock to the appropriate location. Obviously, enough rock must exist on the beached area of eroding bank to make this method feasible.

Successes

Despite all the previously mentioned problems and failures, some successes have been achieved. One test area was planted four times before the vegetation became established. It is unfortunate that no data were obtained so that what was right about the fourth attempt could be documented. Despite the lack of data, a dense viable stand of vegetation protects an area of shoreline for unknown reasons.

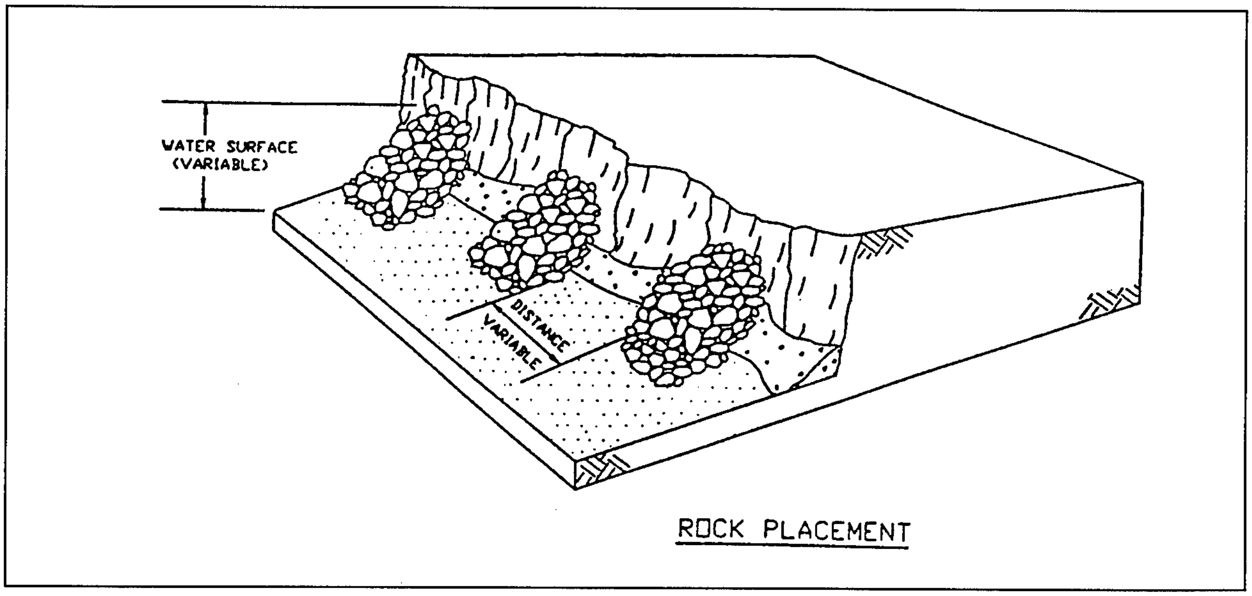

Figure 2. Diagrammatic representation of using rock placed as hard points

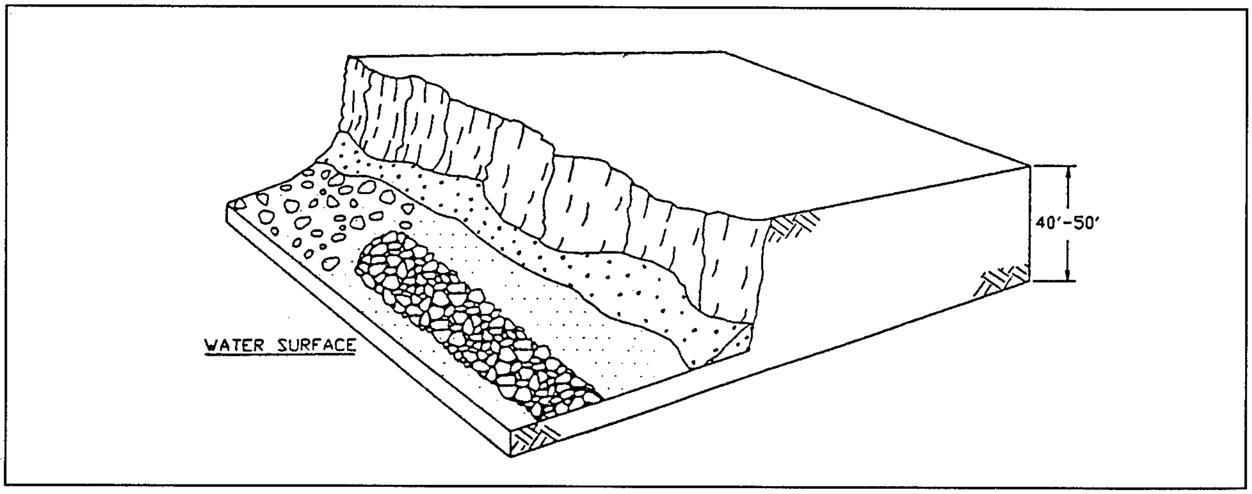

Figure 3. Rock placement on eroding shorelines utilizing readily available materials

A second success consists of planting an area of eroding shoreline with brush mats, wattling bundles, and other measures. This area was eroding so rapidly that the vegetation would have had no chance to become established. However, personnel of the Nebraska Game and Parks Commission noticed and hand placed a row of rocks at the eroding scarp near the base of the newly install vegetation. The combination of a small quantity of rock and vegetation has survived many years, even though the bank on either side of the test area has receded approximately 3 feet.[1]

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1 Personal Observation, 1994, John L. Anderson, U.S. Army Engineer District, Omaha, Omaha, NE.

Conclusions

As a result of the efforts to control shoreline erosion that have been attempted, several lessons have become evident.

a. Bales or fiber rolls should be placed in the reservoir as far as possible from the eroding bank. Efforts to date indicate that a narrow band of vegetation established along the shoreline will not withstand significant wind-wave action or storm surges after the bales or fiber rolls have decayed away. The vegetative band should be as wide as possible to allow for self-healing after damages due to wind-wave action and to prevent the total elimination of the vegetative band due to storms.

b. A job or task should be established within the district office. This job could come under the heading, "other duties as assigned." The work would entail tracking, communicating, and reviewing shoreline erosion efforts. This task is not viewed as dictating who does what, rather, it would simply keep track of where and what methods were attempted at which reservoir projects. New and innovative techniques of managing shoreline erosion could also be communicated to the appropriate agencies or agency segments.

c. All shoreline erosion prevention efforts should be reviewed over a period of years to determine the success or failure of a particular method or effort. A project that does not last more that 5 years should not be considered a success. The success or failure should be communicated to all appropriate District elements so the successful methods can be duplicated by others and so that unsuccessful methods will not be duplicated by others.

d. A systematic method of managing shoreline erosion at each reservoir project should be initiated.

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Sediment Resuspension and Export in a Shallow Minnesota Reservoir

by
William F. James[1] *and John W. Barko*[1]

Introduction

The water quality of shallow lakes and reservoirs are often dominated by wind-induced sediment resuspension leading to low-water clarity, high turbidity, enhanced nutrient recycling, and stimulated algal growth (Dillon, Evans, and Molot 1990; Maceina and Soballe 1990; Hellström 1991; Søndergaard, Kristensen, and Jeppesen 1992). By reducing sediment resuspension, wetland and submersed aquatic vegetation can have beneficial effects on the water quality of shallow impoundments and lakes. For example, Dieter (1990) showed that sediment resuspension was reduced in regions colonized by emergent vegetation in Sand Lake, South Dakota. Similarly, James and Barko (1990) demonstrated that submersed vegetation reduced sediment erosion in the littoral zone of Eau Galle Reservoir, Wisconsin. Thus, aquatic vegetation may play an important role in mitigating water quality problems in shallow lakes subjected to frequent winds.

In this study, we examined sediment resuspension in Marsh Lake, Minnesota, and the export of sediment from the system to downstream Lac Qui Parle Reservoir, Minnesota. Sediment resuspension was examined under a variety of wind conditions in Marsh Lake to determine critical thresholds of wind velocity that resulted in high concentrations of suspended sediment (i.e., seston) in the water column. Sediment resuspension was examined during a year when submersed aquatic vegetation in Marsh Lake was densely established (1991) and during a year when it was almost completely absent (1992), thus providing information on the influence of aquatic vegetation on sediment dynamics in a shallow reservoir.

Methods

Marsh Lake, a flood control impoundment located near the headwaters of the Minnesota River in western Minnesota, is large (surface area = 1,862 ha) and very shallow (mean depth = 0.9 m, and maximum depth = 1.5 m; Figure 1). Tributaries to Marsh Lake include the Pomme De Terre River, which enters the lake near the dam and outlet structure, and the Minnesota River (Figure 1). The tributaries drain a mostly agricultural watershed. Discharges from Marsh Lake occur over an uncontrolled fixed-crest overflow structure located at an elevation of 285.78 m National Geodetic Vertical Datum (NGVD). Discharge also occurs through a sluice gate located at 284.26 m NGVD. Discharges from Marsh Lake travel approximately 20 km downstream before entering the headwaters of Lac Qui Parle Reservoir.

Sediment loading and resuspension were examined in Marsh Lake during June through September 1991 and during May through September 1992. During the summer and autumn of 1991, the entire reservoir was densely populated by *Potamogen* sp., with surface coverages > 90 percent (D. Hatfield and K. Bonema, Personal Observation). Submersed macrophyte densities diminished greatly in 1992, to the point where plants could not be observed visually in

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

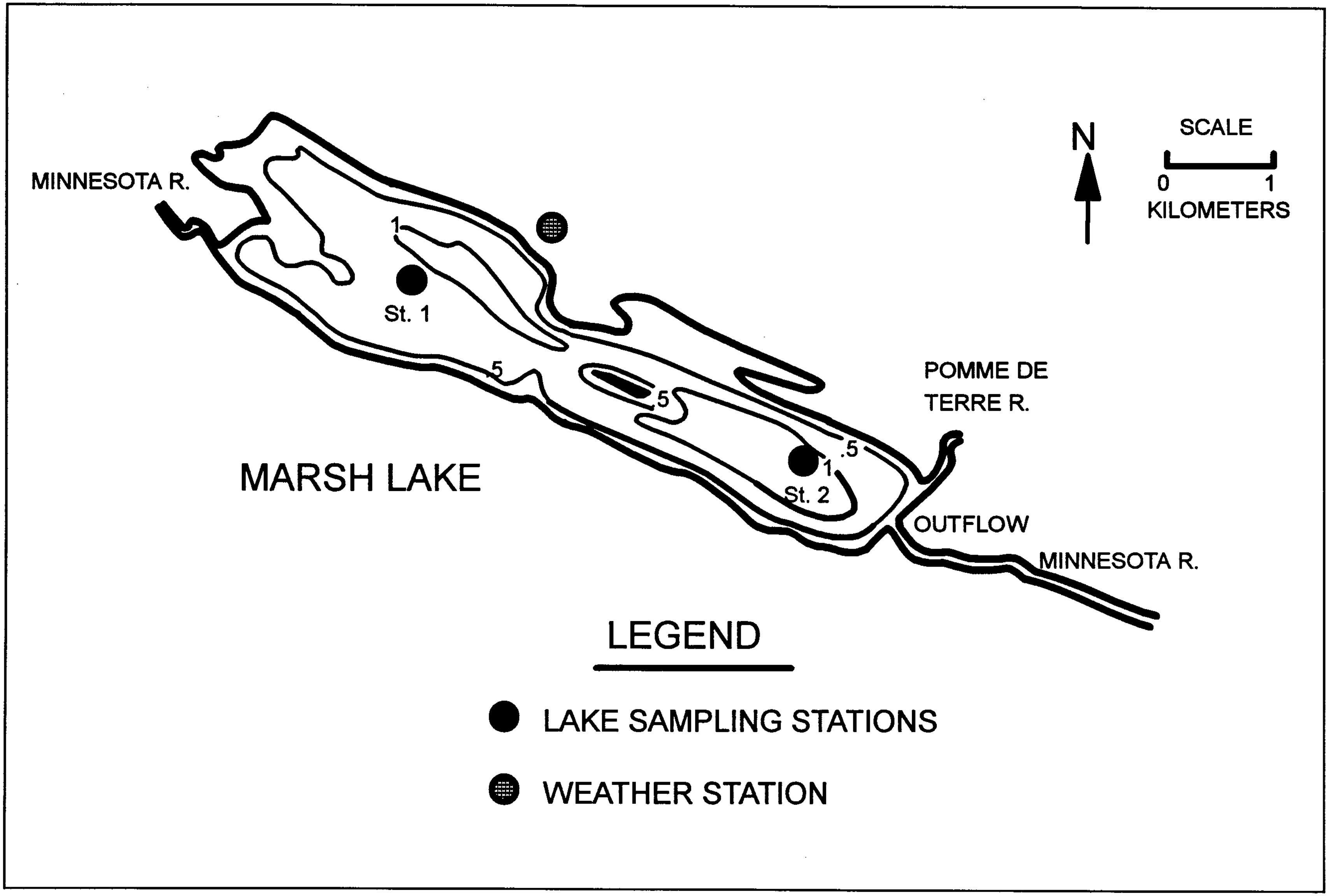

Figure 1. Morphometric map and station locations in Marsh Lake (Depth contours are in meters)

the water column throughout most of the reservoir. Differences in macrophyte coverage were not quantified, since we did not anticipate such a marked contrast in abundance between the 2 study years. The cause of this macrophyte decline has not been determined.

In 1991, two midchannel stations (0.7- to 0.8-m deep) were established in the lake, with water sampling (integrated water column) attempted by boat every Monday, Wednesday, and Friday. However, strong winds and high waves prevented manual collection of water samples on many dates. In 1992, automated water samplers (ISCO) were placed on permanently anchored platforms at these same lake stations (Figure 1). The samplers were programmed to composite three water samples, collected at 8-hr intervals at a depth of 0.3 m below the lake surface on a daily basis. Thus, water samples could be collected even under severe wind conditions.

River water samples were collected daily (automated water samplers) or weekly (grab samples) at fixed stations located on the Minnesota River upstream of Marsh Lake, on the Pomme De Terre River, and at the outflow of Marsh Lake during both years. Flows were gauged continuously using continuous recording gauges or stage height recorders. The water level of Marsh Lake was measured daily with a recording gauge.

All lake and river samples were analyzed for seston (i.e., total suspended solids), particulate organic matter (POM), and total phosphorus (total P) concentrations. Samples for seston and POM were filtered onto precombusted glass fiber filters. The filters were dried at 105 °C for seston analysis and then combusted at 550 °C for 2 hr for determination of POM (i.e., loss on ignition; American Public Health Association (APHA) 1989). Total P of unfiltered samples was determined colorimetrically on a Technicon Autoanalyzer II following persulfate oxidation (APHA 1989).

Daily external loadings and discharges of seston, POM, and total P in Marsh Lake were calculated by multiplying the daily volumetric flow rate by the measured concentration. Linear interpolation was used to estimate concentrations on dates when water samples were not collected. Retention (kilograms/day) of these constituents in Marsh Lake was estimated according to the following equation: Retention = External Load - Outflow. The external load represented the sum of daily loading estimates from the Minnesota and Pomme De Terre rivers, while the outflow represented the daily discharge load from Marsh Lake.

A weather station was established on the northern shoreline of Marsh Lake for wind monitoring (Figure 1). The average of data collected at 5-min intervals was recorded every 15 min to one-half hour for wind velocity and direction using an automated data logger (Omnidata International). Mean daily wind velocities were computed from these data for comparison with trends in water quality variables, because mean daily velocity reflects sustained winds. Daily wind vectors were calculated as the wind direction weighted with respect to wind velocity and averaged over a 24-hr period. Statistical analysis of the wind data were performed using Statistical Analysis System (1988).

The ability of waves, generated by wind activity, to disturb and resuspend the sediment surface in Marsh Lake was estimated for a variety of wind directions using the wave model developed by Carper and Bachmann (1984). In general, the model calculates wavelengths (L) of surface waves using mathematical wave theory as applied by the U.S. Army Coastal Engineering Research Center (1977) and compares 1/2L to the water column depth. If, 1/2L is greater than the water column depth, the wave is assumed to touch the lake bottom and resuspend sediment. The wave model assumes that there are no obstructions in the water column such as submersed or emergent aquatic macrophytes that might create drag or redirect currents and wave activity.

Results

During both years, winds blew most frequently out of the southeast (~40 percent) and southwest (~39 percent), followed by winds blowing out of the northwest (18 percent). Relatively low mean daily wind velocities of 5-10 km hr^{-1} were most frequent (47 percent) during both years, followed by greater wind velocities of 10-15 km hr^{-1} and 15-20 km hr^{-1}, occurring 28 and 11 percent of the time, respectively. Mean daily wind velocities of < 5 km hr^{-1} occurred on only 9 percent of the days during both years. In general, mean daily wind velocities for all directions were ≥ 10 km hr^{-1} on 44 percent of the days and ≥ 20 km hr^{-1} on 5 percent of the days during both years.

Based on wave theory, only 17 to 22 percent of the lakebed was potentially affected by waves at low-wind velocities (i.e., ≤ 5 km/hr; Table 1). However, at wind velocities of 10 km hr^{-1}, the percentage of the lakebed affected by waves increased markedly (Table 1), particularly for winds blowing in the direction of maximum effective fetch (i.e, SE and NW winds). Virtually 100 percent of the lakebed was potentially affected by waves at wind velocities of 15 km hr^{-1} blowing from any direction. Water sampling stations in Marsh Lake were affected by wave disturbances at critical (theoretical) wind velocities ranging between 10.5 and 15 km hr^{-1}, depending on wind direction (Table 2).

In 1991, mean daily wind velocities fluctuated between about 5 and 19 km hr^{-1} from June through mid-September (Figure 2). Peaks in mean daily wind velocity of >19 km hr^{-1} occurred in late September (Figure 2). Wind direction during these periods of high wind velocity was predominantly from the northwest. Seston concentrations were relatively low (i.e., < 50 mg/L) in Marsh Lake from June through mid-September 1991 (Figure 2); however, concentrations of seston increased markedly in late September (i.e., >100 mg L^{-1}) in association with high mean daily wind velocities of ≥ 19 km hr^{-1}. A similar seasonal pattern was observed for seston concentrations in the outflow of Marsh Lake during 1991 (Figure 2), suggesting the discharge of resuspended sediment.

Table 1
Percent of Lakebed Disturbed by Waves at Various Wind Speeds and Directions

Wind Speed km/hr	Winds Blowing From: NE	SE	SW	NW
5	22	22	17	17
10	49	67	37	75
15	86	95	81	100
20	100	100	100	100

Table 2
Comparison of Effective Fetches and Critical Wind Velocities Required to Resuspend Sediment at Stations 1 and 2 for Various Wind Directions at Nominal Water Column Depths (i.e., ~ 0.8 m)

	Site 1		Site 2	
Wind Direction	Fetch, m	Critical Velocity km/hr	Fetch, m	Critical Velocity km/hr
NE	1,100	15.0	1,150	15.5
SE	1,200	14.5	2,500	11.0
SW	1,100	15.0	1,200	14.5
NW	2,900	10.5	1,900	12.0

During 1992, mean daily wind velocities were generally greatest in May and late August through September and much lower during late June through mid-August (Figure 3). However, the highest mean daily wind velocity of

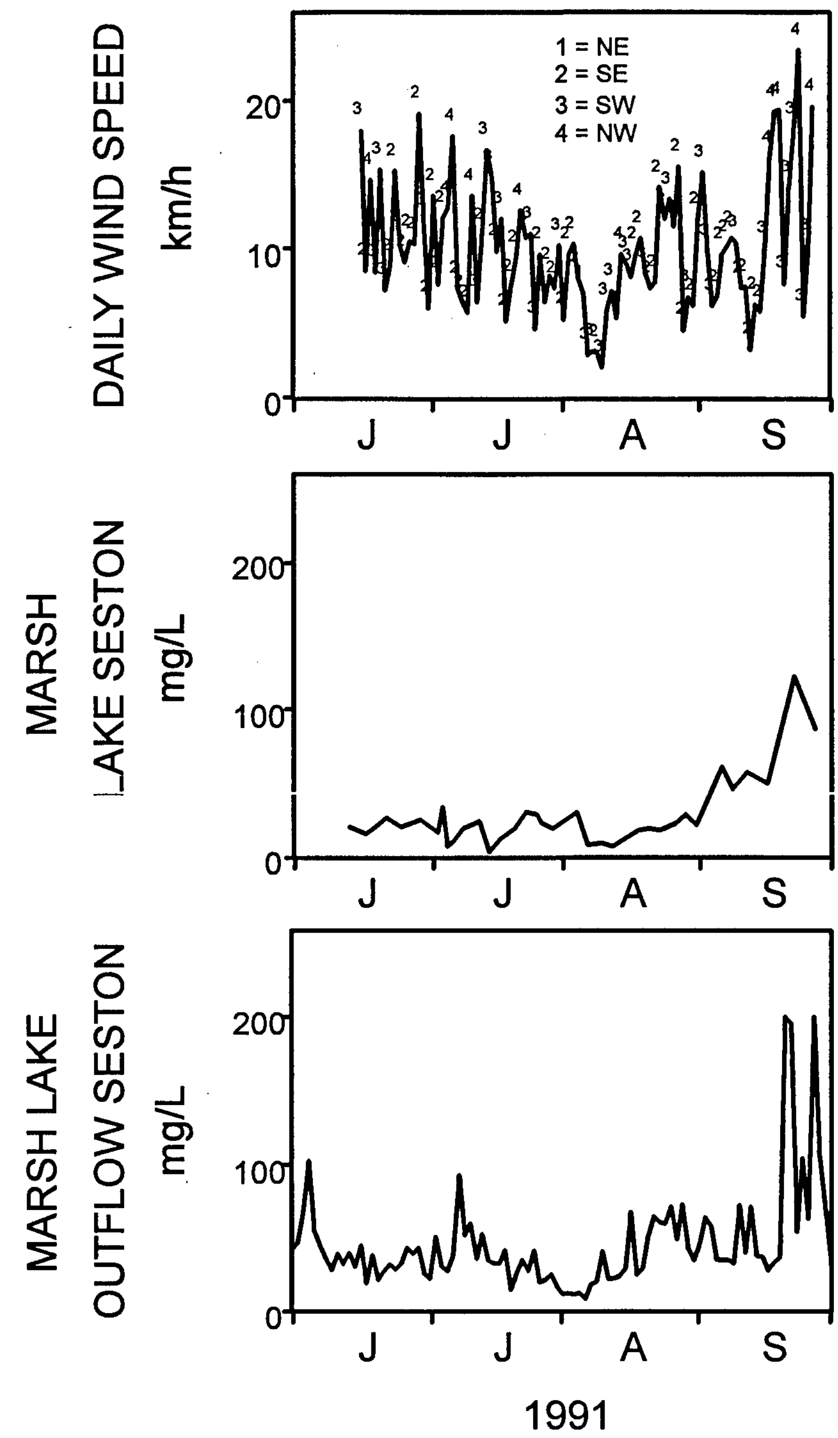

Figure 2. Variations in mean daily wind velocity (upper) and seston concentrations in Marsh lake (middle) and the outflow (lower) during 1991 (Seston concentrations in Marsh Lake represent the average in concentration from stations 1 and 2)

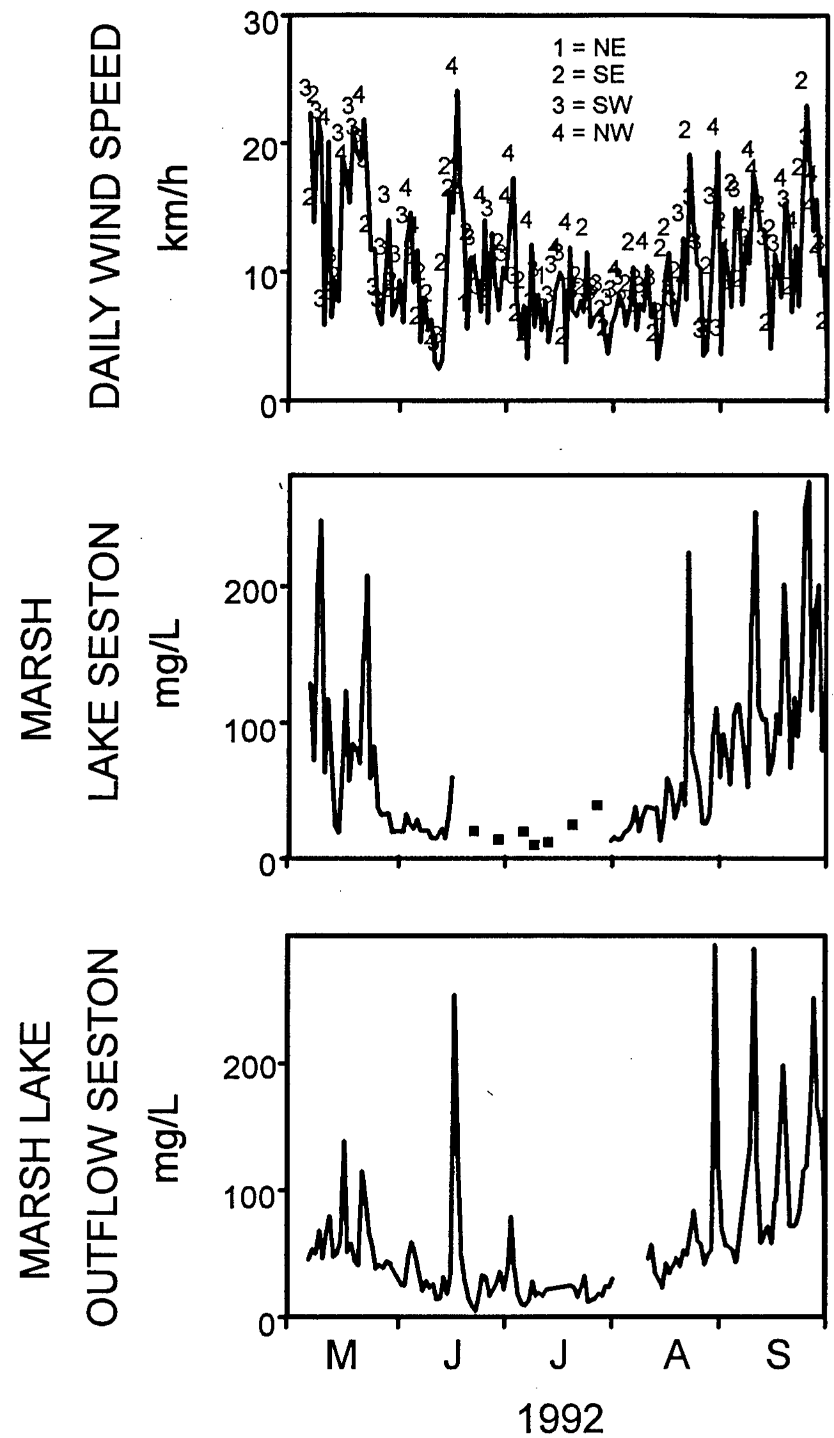

Figure 3. Variations in mean daily wind velocity (upper) and seston concentrations in Marsh lake (middle) and the outflow (lower) during 1992 (Seston concentrations in Marsh Lake represent the average in concentration from Stations 1 and 2)

24 km hr^{-1} occurred in mid-June of that year. Seston concentrations increased substantially in both Marsh Lake and the outflow (i.e, 100-300 mg L^{-1}) during these windy periods in 1992 (Figure 3). In contrast, seston concentrations (<50 mg L^{-1}) were much lower during the calm er summer months.

Although we predicted from the wave model (Carper and Bachmann 1984) that sediment resuspension would occur at wind velocities of only 10.5 to 15 km hr^{-1}, depending on wind direction (Table 2), the actual critical wind velocity for sediment resuspension was about 20 km hr^{-1} in 1991, based on relationships be tween mean daily wind speed and Marsh Lake seston concentrations (Figure 4). At these wind velocities, which blew primarily from the north-west direction, seston concentrations were also elevated in the outflow in 1991 (Figure 4), and the seston retention rate was high and extremely negative (i.e., < -100 × 103 kg/day; Figure 4), indicating a net loss of sediment from the system during sediment resuspension events. The discrepancy between predicted and observed critical wind velocities may be attributed to the presence of high densities of submersed aquatic macrophytes, which potentially dampened wave activity and sediment resuspension in 1991.

In contrast, relationships between mean daily wind speed and seston concentrations in Marsh Lake suggested that a critical wind velocity of about 12 km hr^{-1} was required for sediment resuspension in 1992 (Figure 5). The decrease in the critical wind velocity in 1992 was within the range of predicted values determined from the wave model (Table 2) and coincided with an almost complete absence of submersed aquatic macrophytes in Marsh Lake during that year. Above the critical wind velocity of 12 km hr^{-1}, seston concentrations increased in a linear fashion with increasing wind velocity ($p < 0.05$; $r^2 = 0.58$) regardless of wind direction. Below this critical wind velocity, seston concentrations were usually < 50 mg L^{-1}. Anomalously high seston concentrations (i.e., between 50 and 100 mg L^{-1}) below this critical wind velocity were representative of seston samples collected 1 day after the occurrence of a sediment resuspension event (wind velocities > 15 km hr^{-1}). This pattern in 1992 suggested a time lag of at least 1 day for the recovery of seston concentrations to nominal values following wind-generated sediment resuspension.

Discussion

The presence or absence of aquatic vegetation appears to play an important role in sediment resuspension dynamics in Marsh Lake. As an apparent result of the presence of aquatic vegetation in 1991, the critical wind velocity required to cause sediment resuspension was 5 to nearly 10 km hr^{-1} higher than the values predicted from the wave model. In contrast, the critical wind velocity was much lower in 1992, because of the absence of aquatic macrophytes. This change in the critical wind velocity from about 20 km hr^{-1} in 1991 to 12 km hr^{-1} in 1992 resulted in a >25-percent increase in the frequency of occurrence of sediment resuspension in 1992 (i.e., from 5-percent occurrence in 1991 to 32-percent occurrence in 1992). The wave model predicted a 30-percent frequency of occurrence in sediment resuspension during both years, based on the wind velocity record, which was similar to the observed frequency of occurrence in 1992 when submersed macrophytes were virtually absent in the lake. These results strongly suggest that aquatic vegetation was beneficial in reducing the occurrence of sediment resuspension in Marsh Lake by dampening wave activity.

Several studies have shown that emergent and submersed macrophytes can reduce and/or redirect currents and wave activity that promote sediment resuspension (Fonseca et al. 1982; Gregg and Rose 1982; Madson and Warncke 1983; Eckman, Duggins, and Sewell 1989). Others have shown that sediment resuspension and erosion are reduced, while net sedimentation is enhanced, in shallow regions of lakes occupied by aquatic vegetation (Moeller and Wetzel 1988; Deiter 1990; James and Barko 1990; Anderson 1990; Petticrew and Kalff 1991, 1992). In particular, although nearly 80 percent

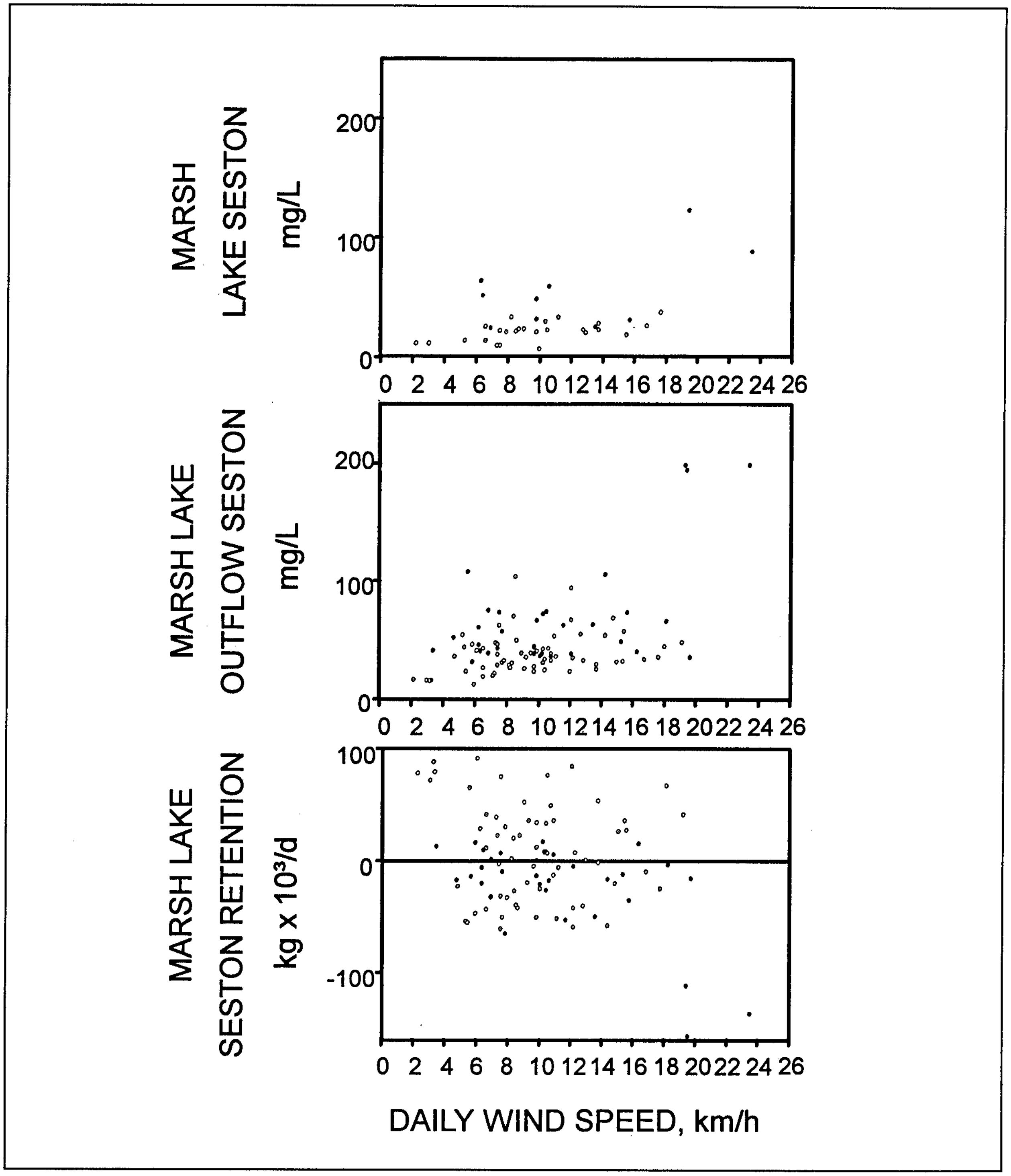

Figure 4. Relationships between mean daily wind velocity and seston concentrations in Marsh Lake (upper) and the outflow (middle) and daily retention of seston in the lake (lower) during 1991 (Solid dots represent data collected at normal pool elevation. Circles represent data collected at higher pool elevations. Higher pool elevations occurred during the summer (June-August) in association with storm inflows)

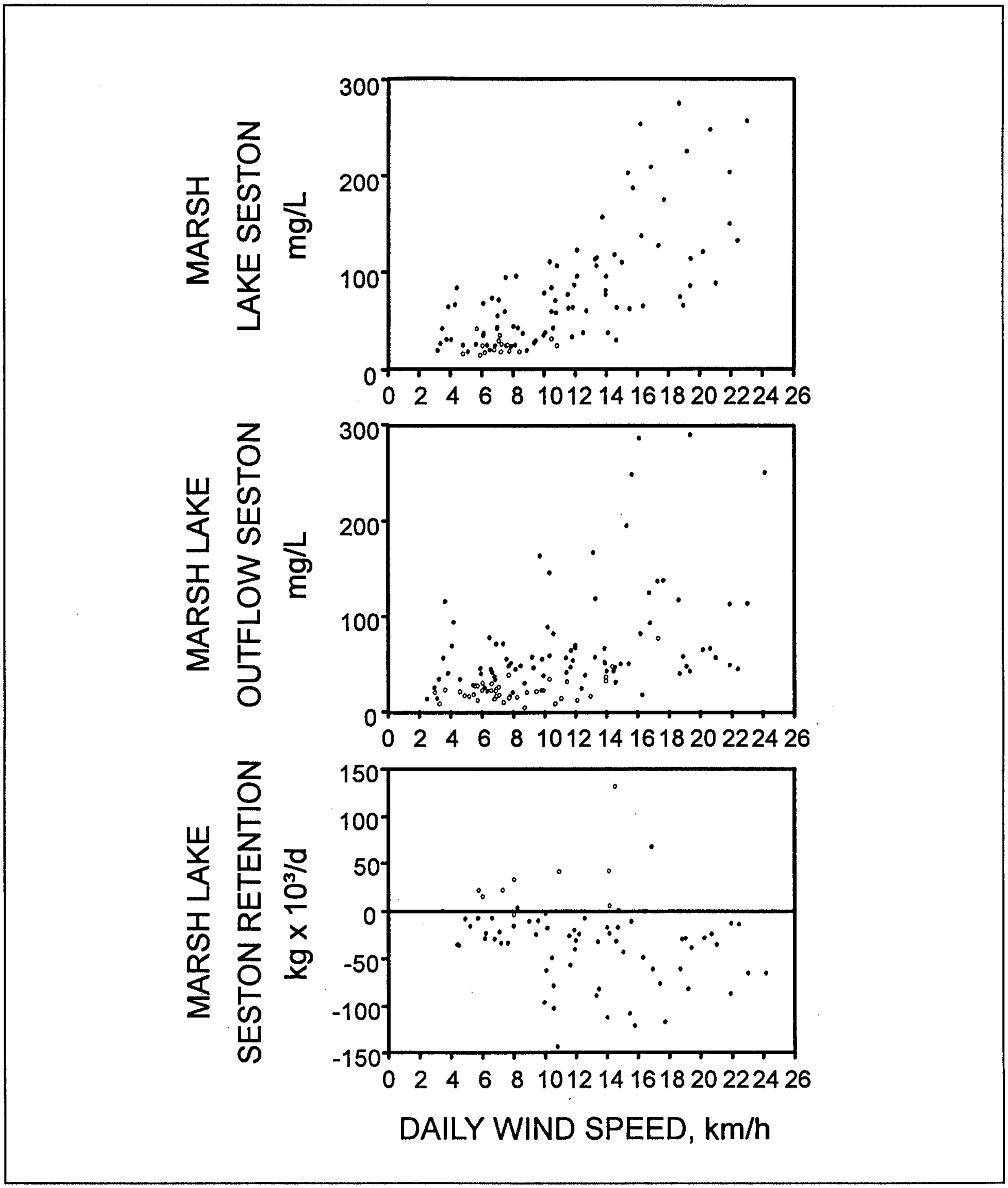

Figure 5. Relationships between mean daily wind velocity and seston concentrations in Marsh Lake (upper) and the outflow (middle) and daily retention of seston in the lake (lower) during 1992 (Solid dots represent data collected at normal pool elevation. Circles represent data collected at higher pool elevations. Higher pool elevations occurred during the summer (June-August) in association with storm inflows)

of the sediment surface area of shallow Lake Tämnaren (mean depth = 1.5 m) is potentially resuspended as a result of wind and wave activity, all vegetated regions along the shoreline represent zones where sediments accumulate (Hellström 1991).

Sediment transport from Marsh Lake during periods of resuspension could have an important impact on Lac Qui Parle Reservoir, located immediately downstream, by both exacerbating accretion rates and accelerating the reduction in water storage capacity. In addition, accumulation of POM associated with resuspended sediment from Marsh Lake could have an impact on dissolved oxygen conditions in Lac Qui Parle Reservoir during ice formation. For instance, Gunnison and Barko (1988) and James et al. (1992) suggested that the resuspension and downstream transport of sediment-associated chemical oxygen-demanding materials were responsible for the rapid development of anoxia during winter drawdown in Big Eau Pleine Reservoir, Wisconsin. Discharge of resuspended phosphorus and other nutrients could also accelerate eutrophication and algal productivity in impoundments located downstream of Marsh Lake.

The results of these studies have implications for management alternatives to improve habitat and water quality in both Marsh Lake and Lac Qui Parle Reservoir. Techniques need to be developed to maintain the abundance of aquatic macrophytes in Marsh Lake to reduce sediment resuspension. These techniques might include periodic drawdown for sediment consolidation and seed germination, macrophyte transplanting, regulating pool elevation at least initially for the establishment of both submersed and emergent vegetation, and construction of artificial islands in strategic locations to reduce wind fetch and turbidity associated with sediment resuspension. Application of these techniques would encourage the development and maintenance of extensive stands of aquatic vegetation, thereby creating valuable habitat for waterfowl and limiting wind-driven sediment resuspension and sediment discharge to Lac Qui Parle Reservoir.

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A Comparison of Water Quality in Channelized and Unchannelized Streams in the Upper Coldwater River Watershed

by
David R. Johnson[1]

Introduction

This study was performed as part of the water quality evaluation for the Demonstration Erosion Control Project, Coldwater River Watershed, Environmental Impact Statement. The Upper Coldwater River watershed includes the Coldwater River and its tributaries from just below its confluence with the Camp's Creek. The following channelized tributaries are included: Camps Creek, Pigeon Roost Creek, and the Pigeon Roost tributaries, Red Banks Creek and Cuffawa Creek. This study was to describe the impacts due to construction of 14 floodwater retarding structures in the Upper Coldwater River Watershed.

The State of Mississippi, Department of Environmental Quality, Bureau of Pollution Control (MSDEQ) rates the Coldwater River above Arkabutla Reservoir as partially supporting its Fish and Wildlife classification. It rates that classification because of high nutrients and organic enrichment due to point and non-point pollution from industry and agriculture.

This study used water quality data on 14 parameters collected at 12 stations in the upper Coldwater River Watershed. Eight of the stations in that study were on channelized streams and four on natural streams. Initially, there were eight stations in the study area and water quality data was collected weekly. In 1991, four stations in the upper Pigeon Roost Watershed were added, and the sampling interval was reduced to once every other week. Additional data on the occurrence of heavy metals, pesticides, and herbicides at the 12 sites were also available. Arsenic and mercury were the only metals detected. Several pesticides were detected including: trifluralin, metribuzin, DDE, Atrazine, Alachlor, chloropyrifos and pendimethalin.

All of the data used in the study came from the U.S.D.A., ARS, National Sedimentation Laboratory. The data cover the period from January 1989 to March 1995. The 12 stations used are shown in Figure 1. The water quality data in this study have received extensive statistical analysis by USACE. Statistical Analysis System (SAS) release 6.03 for personal computers (SAS Institute Inc., Cary, NC 27513) was used to make the analyses. The means, medians, and percentile ranges were calculated with Proc Univariate. The Analysis of Variance (ANOVA) were performed with Proc GLM (General Linear Models), and the correlations were calculated with Proc Corr.

It has often been observed that the means of the water quality data from channelized streams appeared different (generally higher) than the means from the unchannelized streams. An

[1] U.S. Army Engineer District, Vicksburg; Vicksburg, MS.

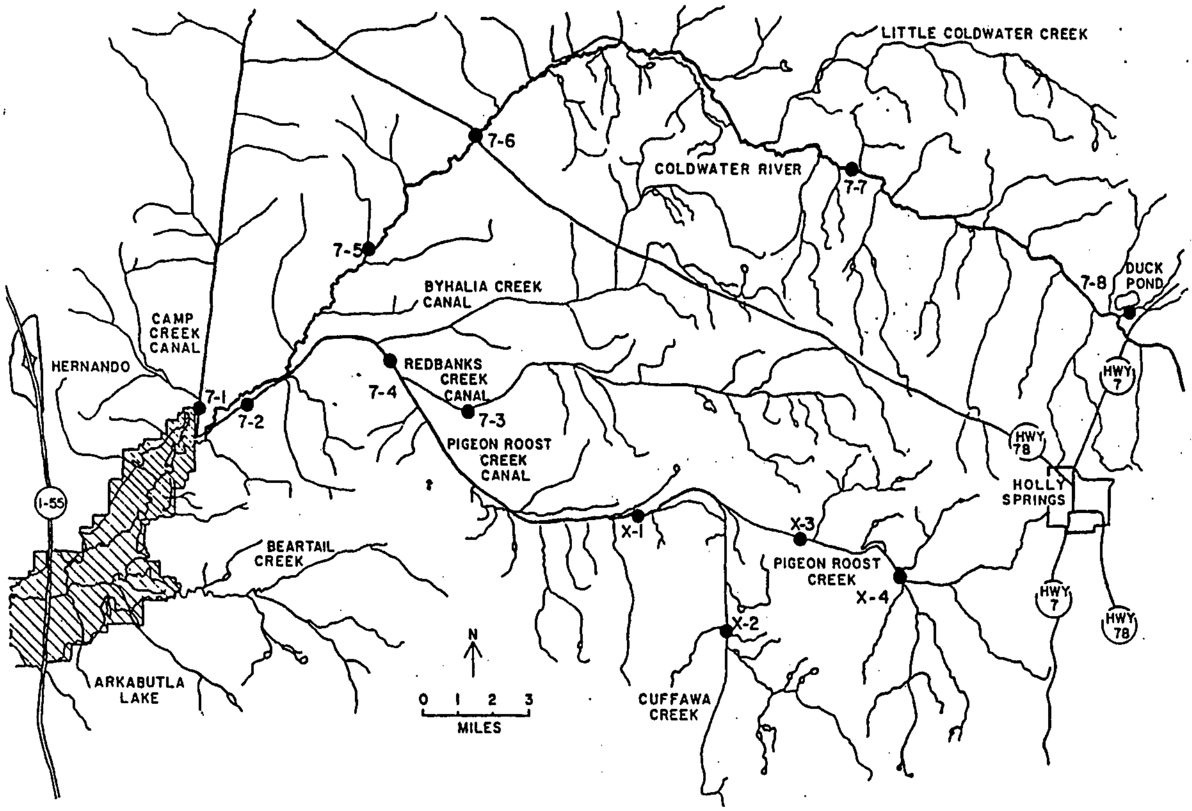

Figure 1. Water quality sampling sites for the Coldwater River Drainage Systems

ANOVA and Unplanned Comparison Tests (UPCT) were performed on the individual stations and various groupings of the stations to determine if there were statistically significant differences between the station means for the 14 water quality parameters analyzed. The first ANOVA compared the means of all of the stations individually to see if any stations were significantly different from the others. The ANOVA on the individual stations showed that there were significant differences between some stations for most parameters. Based on the results of the first ANOVA, a second ANOVA with 8 groups of stations was performed. Only stations which were not significantly different for most of the parameters were grouped together. The following groupings were used: Stations 7-2 and 7-4; Stations 7-5, 7-6, and 7-7; and Stations X-1 and X-3. Stations 7-1, 7-3, 7-8, X-2, and X-4 did not consistently group with any other station and were left as independent stations. The Lower Pigeon Roost stations (7-2 and 7-4) and the Red Banks station (7-3) were frequently associated in the same group in the second ANOVA. These stations were associated together for 5 of the 11 parameters which had significant differences. Differences are considered significant, when the $Pr > F$ statistic is less than 0.05. For most parameters, the $Pr > F$ statistic was 0.0001, which indicates that the observed differences were highly significant. The results of this ANOVA are presented in Table 1.

The third ANOVA compared the means of the data which had been separated into two periods (early and late). The ANOVA was performed to test if the amount of data or the period of the data collection, was affecting the results. The results indicated the same basic trends for data collected before 1991 as after. However, when pre and post 1991 data were treated as different stations, significant differences were noted for data collected at the same locations between the two time periods. In particular, the mean suspended solids of all four groups of pre 1991 data were higher than the same stations post 1991. The means for three of four pairings were significantly higher.

Significant differences were noted between the means of the pre and post 1991 data collected at the same station for 10 of the 12 parameters tested. These results are presented in Table 2. 1991 is significant not only because the new stations were added, but because DEC construction for bank stabilization and riser pipes had been initiated in the Coldwater River Watershed.

The water quality data were then sorted by year and station, and an ANOVA performed comparing the annual means of each station for 10 of the 14 parameters. Significant differences were noted at all stations for some parameters, with the means frequently sorted chronologically by years. This trend implies a correlation of the means by station for each parameter with the sampling year. The annual means of 10 parameters were tested for correlation by station. Strong correlations were present for dissolved oxygen, conductivity, total solids, suspended solids, total phosphorus, ammonia, nitrate and chlorophyll. Dissolved oxygen, conductivity, ammonia, nitrate and chlorophyll were positively correlated, while total solids, suspended solids and total phosphorus were negatively correlated. Although differences in the annual precipitation may be affecting the means, it is likely that the changes observed result from the completion of bank stabilization and riser pipe structures within the watershed.

Comparison of Existing Water Quality Between Natural and Channelized Streams

It is generally accepted that channelization has negative impacts on stream water quality. The ARS in a review of the water quality of the Coldwater River observed that the monthly means of most water quality parameters were higher for channelized versus natural streams. The most frequently cited impacts to water quality due to channelization are increased temperatures, decreased dissolved oxygen, increased suspended solids, increased nutrients, and decreased algae primary productivity. The

Table 1
ANOVA by Grouped Water Quality by Grouped Sampling Sites

Temperature (0.096)

Duncan Group	Mean	N	Station
A	18.2	231	Coldwtr1
A	17.5	436	PigeonR1
A	17.1	93	PigeonR3
A	17.0	61	Cuffawa
A	16.7	436	Coldwtr1
A	16.7	169	Red Bank
A	16.6	217	Camp Cr
A	16.6	440	Coldwtr2

pH (0.0031)

Duncan Group	Mean	N	Station
A	6.57	217	Camp Cr
A	6.55	436	Pigeon R1
B A	6.50	436	Coldwtr1
B A	6.48	169	Red Bank
B A	6.45	231	PigeonR2
B	6.41	440	Coldwtr2
B	6.41	61	Cuffawa
B	6.40	93	PigeonR3

Dissolved Oxygen (0.0001)

Duncan Group	Mean	N	Station
A	9.4	61	Cuffawa
B	8.9	229	PigeonR2
B	8.7	167	Red Bank
B	8.6	432	PigeonR1
B	8.5	92	PigeonR3
C	7.9	430	Coldwtr1
C	7.9	216	Camp Cr
C	7.5	434	Coldwtr2

Conductivity (0.0001)

Duncan Group	Mean	N	Station
A	118	214	Camp Cr
B	92	92	PigeonR3
C	60	166	Red Bank
D	48	59	Cuffawa
D	48	431	Coldwtr2
E D	46	227	PigeonR2
E F	41	429	PigeonR3
F	38	430	Coldwtr1

Total Solids (0.0001)

Duncan Group	Mean	N	Station
A	270	209	Camp Cr
B	208	164	Red Bank
C B	182	56	Cuffawa
C B D	175	423	PigeonR1
C E D	144	84	PigeonR3
C E D	130	213	PigeonR2
E D	120	422	Coldwtr1
E	101	428	Coldwtr2

Dissolved Solids (0.0001)

Duncan Group	Mean	N	Station
A	95.9	209	Camp Cr
B	85.8	83	PigeonR3
C	70.6	56	Cuffawa
D	62.7	163	Red Bank
E D	57.6	209	PigeonR2
E	53.3	424	Coldwtr2
E	52.8	424	PigeonR1
E	52.0	420	Coldwtr1

Suspended Solids (0.0001)

Duncan Group	Mean	N	Station
A	172.0	208	Camp Cr
A	146.7	163	Red Bank
B A	123.5	420	PigeonR1
B A C	113.4	55	Cuffawa
B D C	73.6	208	PigeonR2
B D C	63.0	419	Coldwtr1
D C	59.6	82	PigeonR3
D	44.6	213	Coldwtr2

Chlorophyll (0.001)

Duncan Group	Mean	N	Station
A	58.3	55	PigeonR3
B	35.4	32	Cuffawa
B	35.1	132	PigeonR2
C B	22.2	165	Camp Cr
C	17.4	167	Coldwtr2
C	14.7	333	PigeonR1
C	14.1	498	Coldwtr1
C	11.3	135	Red Bank

Total Phosphorus (0.0001)

Duncan Group	Mean	N	Station
A	269.6	209	Camp Cr
B	208.4	164	Red Bank
B	181.9	56	Cuffawa
C B	174.6	423	PigeonR1
C D	144.1	84	PigeonR3
C D	130.3	213	PigeonR2
D	113.4	636	Coldwtr1
D	102.0	214	Coldwtr2

Orthophosphorous (0.0001)

Duncan Group	Mean	N	Station
A	.083	221	Camp Cr
B	.054	94	PigeonR3
C	.033	61	Cuffawa
D C	.027	235	PigeonR2
D E	.023	447	PigeonR1
D E	.021	173	Red Bank
E	.018	670	Coldwtr2
E	.010	225	Coldwtr3

(Continued)

Table 1 (Concluded)

Ammonia (0.0001)				Nitrate (0.0001)			
Duncan Group	Mean	N	Station	Duncan Group	Mean	N	Station
A	.483	92	PigeonR3	A	.261	59	Cuffawa
B	.174	213	Camp Cr	B	.060	94	PigeonR3
C B	.142	231	PigeonR2	C	.660	233	PigeonR2
C D	.124	217	Coldwtr3	D	.502	220	Camp Cr
D	.091	642	Coldwtr2	E	.412	173	Red Bank
D	.085	431	PigeonR1	F	.320	445	PigeonR1
D	.081	166	Red Bank	G	.220	224	Coldwtr3
D	.080	60	Cuffawa	G	.162	667	Coldwtr2

following paragraphs will examine those impacts with regard to the Coldwater River system.

Temperature increases due to channelization are often attributed to loss of vegetative shading. This impact would be anticipated on low order streams with closed canopies prior to channelization. None of the ANOVA performed on the data from this system found any significant differences in temperature between any stations. Stream channelization was originally done in the 1920's and channel cleanout was last performed 20 to 30 years previously. In spite of the length of time since channelization, most channelized reaches do not have closed canopies. All of the channelized streams have long reaches with little or no canopy cover, and tilled fields often extend to top bank. The annual mean temperatures at the eight water quality stations are plotted in Figure 2. With the exception of Red Banks Creek, the annual means appear very similar (1995 represents only partial data), with little variation between years. Previous studies have reported temperature increases as much as 9 °C, averaging 5 to 6 °C. No such increases are observed in the channelized streams in this basin.

The second major water quality change generally attibuted to channelization is lower dissolved oxygen levels. This generally is attributed to the lower solubility of oxygen at higher temperature and increased organic loading resulting from the loss of vegetative cover. All the ANOVA indicate that the opposite is generally true in the Coldwater River basin. In the ANOVA where the 12 sites were grouped into 8 stations (Table 2), the two natural streams are ranked lowest and third lowest, and they both have significantly lower DO levels than all other stations except Camp Creek. The high DO levels in the channelized streams are likely due to higher primary productivity and better reaeration from steeper stream gradients.

The third water quality parameter where channelized streams are normally thought to suffer is suspended solids. The results of the ANOVA for the 8 stations (Table 1) have the natural streams lower than most channelized streams. Two channelized streams are grouped with them (Group D) and their means are not significantly different. Those two stations are the upper and middle Pigeon Roost Creek stations. The other channelized stations have means 2 to 4 times higher than the lower groups, but it is clear that increased suspended solids are not found in all channelized streams in this study. The results of the ANOVA separating the data into 2 periods, pre and post (Tables 3 and 4), are somewhat different. In the pre project data (Table 3), Duncan's group A contains all of the channelized stations, indicating they are significantly higher than the two natural stations. However, the other two Duncan's groupings (B&C) include one or both natural stations with three or more channelized stations. As all groupings have the same significance, the results are somewhat ambiguous. The station means do indicate large differences in suspended solids between channelized and natural stream.

Table 2
ANOVA Comparing Pre and Post 1991 Data

Dissolved Oxygen (0.0001)

Duncan Group	Mean	N	Station
A	9.4	61	Cuffawa p
B A	9.2	80	Red Bank p
B	8.9	232	PigeonR1 p
B C	8.9	229	PigeonR2 p
D C	8.5	92	PigeonR3 p
D	8.3	87	Red Bank
D	8.3	116	Camp Cr p
D	8.2	200	PigeonR1
D	8.1	466	Coldwtr p
D	7.4	100	Camp CR
D	7.2	398	Coldwtr

Conductivity (0.0001)

Duncan Group	Mean	N	Station
A	122.1	100	Camp Cr p
B	113.7	114	Camp Cr
C	91.6	92	PigeonR3
D	62.8	79	Red Bank
D	57.3	87	Red Bank p
E	48.2	59	Cuffawa
F E	45.6	227	PigeonR2
F E	44.8	402	Coldwtr p
F E	41.9	202	PigeonR1 p
F E	41.4	459	Coldwtr
F	40.2	227	PigeonR1

pH (0.0001)

Duncan Group	Mean	N	Station
A	6.7	116	Camp Cr p
B A	6.7	232	PigeonR1 p
B A C	6.6	81	Red Bank p
B D C	6.6	470	Coldwtr p
D C	6.6	231	PigeonR2 p
D C	6.4	101	Camp Cr
D C	6.4	204	PigeonR1
D	6.4	61	Cuffawa p
D	6.4	93	PigeonR3 p
D	6.4	88	Red Bank
D	6.4	406	Coldwtr

Chlorophyll (0.0001)

Duncan Group	Mean	N	Station
A	58.3	55	PigeonR3 p
B	35.4	32	Cuffawa p
B	35.1	132	PigeonR2 p
B	31.9	83	Camp Cr p
C B	25.5	335	Coldwtr p
C B	25.0	167	PigeonR1 p
C B D	21.3	52	Red Bank p
C D	12.4	82	Camp Cr
D	5.1	83	Red Bank
D	4.4	166	PigeonR1
D	4.2	330	Coldwtr

Total Solids (0.0001)

Duncan Group	Mean	N	Station
A	337.1	101	Camp Cr
B	242.0	87	Red Bank
C B	206.5	108	Camp Cr p
C B	200.0	204	PigeonR1
C B D	181.9	56	Cuffawa p
C E D	170.5	77	Red Bank p
C E D	151.0	219	PigeonR1 p
C E D	144.1	84	PigeonR3 p
E D	130.3	213	PigeonR2 p
E D	117.5	406	Coldwtr
E	104.2	444	Coldwtr p

Suspended Solids (0.0001)

Duncan Group	Mean	N	Station
A	246.9	101	Camp Cr
B	184.0	87	Red Bank
C B	153.2	204	PigeonR1
C D	113.4	55	Cuffawa p
C D	103.9	76	Red Bank p
C D	101.2	107	Camp Cr p
C D	95.4	216	PigeonR1 p
D	73.6	208	PigeonR2 p
D	67.9	406	Coldwtr
D	59.6	82	PigeonR3 p
D	49.5	437	Coldwtr p

Total Phosphorus (0.0001)

Duncan Group	Mean	N	Station
A	.390	101	Camp Cr
A	.341	120	Camp Cr p
B	.260	94	PigeonR3 p
C B	.230	61	Cuffawa p
C D	.200	88	Red Bank
C D	.200	204	PigeonR1
C D E	.180	243	PigeonR1 p
C D E	.180	84	Red Bank p
D E	.153	488	Coldwtr p
D E	.150	406	Coldwtr
E	.140	234	81

Orthophosphorous (0.0001)

Duncan Group	Mean	N	Station
A	.083	101	Camp Cr
A	.083	120	Camp Cr p
B	.053	94	PigeonR3 p
C	.032	61	Cuffawa p
C	.030	85	Red Bank p
C	.030	243	PigeonR1 p
D C	.030	235	PigeonR2 p
D E	.020	489	Coldwtr p
E	.020	204	PigeonR1
F E	.014	88	Red Bank
F	.009	406	Coldwtr

(Continued)

Table 2 (Concluded)

Ammonia (0.0001)				Nitrate (0.0001)			
Duncan Group	Mean	N	Station	Duncan Group	Mean	N	Station
A	.482	92	PigeonR3 p	A	1.26	59	Cuffawa p
B	.213	118	Camp Cr p	B	1.06	94	PigeonR3 p
C	.141	231	PigeonR2 p	C	0.66	119	Camp Cr p
D C	.130	95	Camp Cr	C	0.66	233	PigeonR2 p
D C E	.120	477	Coldwtr p	D	0.45	85	Red Bank p
D C E	.100	239	PigeonR1 p	E	0.38	241	PigeonR1 p
D C E	.092	84	Red Bank p	E	0.37	88	Red Bank
D E	.080	60	Cuffawa p	E	0.32	101	Camp Cr
D E	.080	382	Coldwtr	F	0.24	204	PigeonR1
D E	.071	192	PigeonR1	F	0.21	485	Coldwtr p
E	.070	82	Red Bank	G	0.13	406	Coldwtr

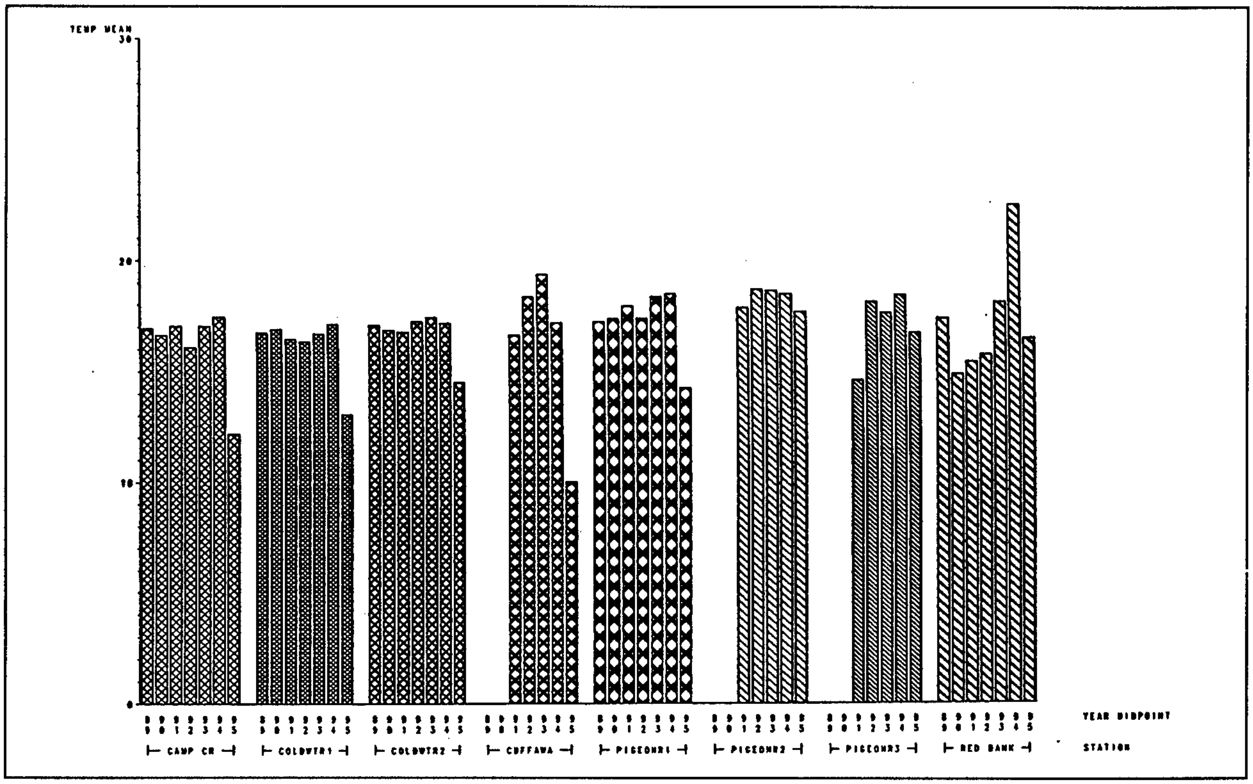

Figure 2. Annual mean temperature by station

The fourth area where differences in water quality are noted is nutrients. Data for four nutrients have been collected. The results for total phosphorus and dissolved orthophosphorus are similar. In both cases the two channelized stations have the lowest means (Table 2), but in neither case are the means significantly lower than all of the channelized stations. Total phosphorus has just 2 Duncan's groupings with the natural stream stations grouped with 3 of the channelized stations. For orthophosphorus, the "B" grouping of station means groups the natural channels alone, but the other groupings associate one or more channelized stations with the natural stream stations. With regard to the two forms of nitrogen, nitrate and ammonia, the

Table 3
ANOVA Comparing Pre DEC Construction Water Quality

Dissolved Oxygen (0.0001)				Conductivity (0.0001)			
Duncan Group	Mean	N	Station	Duncan Group	Mean	N	Station
A	8.8	24	Cuffawa	A	118.7	136	Camp Cr
B A	8.6	74	PigeonR2	B	72.7	19	PigeonR3
B A	8.5	115	Red Bank	C	58.2	137	Coldwtr3
B A	8.3	276	PigeonR1	C	53.8	114	Red Bank
B C	8.2	19	PigeonR3	D	39.6	274	PigeonR1
D C	7.6	412	Coldwtr2	D	38.3	72	PigeonR2
D	7.5	138	Camp Cr	D	36.5	411	Coldwtr2
E	6.6	138	Coldwtr3	D	33.5	22	Cuffawa

pH (0.0001)				Total Solids (0.0001)			
Duncan Group	Mean	N	Station	Duncan Group	Mean	N	Station
A	6.3	139	Camp Cr	A	312.0	135	Camp Cr
B A	6.3	280	PigeonR1	A	292.4	21	Cuffawa
B A	6.2	117	Red Bank	B A	248.9	114	Red Bank
B A	6.2	421	Coldwtr2	B A C	234.0	19	PigeonR3
B	6.1	141	Coldwtr3	B A C	212.3	275	PigeonR1
C	5.8	20	PigeonR3	B A C	211.2	68	PigeonR2
C	5.8	76	PigeonR2	B C	123.0	412	Coldwtr2
C	5.8	24	Cuffawa	C	111.9	138	Coldwtr3

Suspended Solids (0.0001)				Total Phosphorus (0.0001)			
Duncan Group	Mean	N	Station	Duncan Group	Mean	N	Station
A	219.1	135	Camp Cr	A	.380	139	Camp Cr
A	216.2	21	Cuffawa	A	.372	20	PigeonR3
B A	188.6	114	Red Bank	A	.350	24	Cuffawa
B A C	161.2	275	PigeonR1	B	.213	281	PigeonR1
B A C	148.2	68	PigeonR2	B	.213	116	Red Bank
B A C	123.8	19	PigeonR3	B	.200	76	PigeonR2
B C	71.8	412	Coldwtr2	B	.170	421	Codlwtr2
C	53.4	138	Coldwtr3	B	.140	141	Coldwtr3

Nitrate (0.0001)				Chlorophyll (0.0001)			
Duncan Group	Mean	N	Station	Duncan Group	Mean	N	Station
A	.90	20	PigeonR3	A	26.4	6	PigeonR3
B	.74	24	Cuffawa	B A	20.7	2	Cuffawa
C	.57	74	PigeonR2	B C	13.1	95	Camp Cr
D	.41	117	Red Bank	B C	10.9	24	PigeonR2
D	.36	138	Camp Cr	C	7.3	96	Coldwtr3
E	.27	279	PigeonR1	C	5.8	192	PigeonR1
F	.17	140	Coldwtr3	C	5.8	89	Red Bank
F	.14	418	Coldwtr2	C	3.8	286	Coldwtr2

ANOVA groupings are quite different. The pre project nitrate levels are significantly lower than all other stations (Table 3), but the ammonia means are intermediate. Thus this study shows that natural streams have generally lower nutrient levels than channelized streams, but those levels are significantly lower only for nitrate.

The final water quality parameter that is generally affected by channelization is chlorophyll, which is an indicator of primary productivity. Canopy removal results in more available light, which would increase primary production, but increased suspended solids levels decrease light penetration which reduces primary production. Increased nutrients also

Table 4
ANOVA Comparing Post DEC Construction Water Quality

Dissolved Oxygen (0.0001)

Duncan Group	Mean	N	Station
A	9.9	37	Cuffawa
B	9.1	52	Red Bank
C B	9.1	156	PigeonR1
C B	9.0	155	PigeonR2
C B	8.5	73	PigeonR3
C	8.5	78	Camp Cr
C	8.5	235	Coldwtr2
D	7.7	79	Coldwtr3

Conductivity (0.0001)

Duncan Group	Mean	N	Station
A	115.6	78	Camp Cr
B	96.5	73	PigeonR3
C	73.3	52	Red Bank
D	57.7	79	Coldwtr3
D	57.0	37	Cuffawa
E	49.0	155	PigeonR2
F E	43.5	155	PigeonR1
F	40.4	234	Coldwtr2

pH (0.0001)

Duncan Group	Mean	N	Station
A	7.0	156	PigeonR1
A	7.0	78	Camp Cr
A	7.0	52	Red Bank
B A	7.0	235	Coldwtr2
B C	7.0	37	Cuffawa
B C	7.0	155	PigeonR2
D C	7.0	79	Coldwtr3
D	7.0	73	PigeonR3

Total Solids (0.0001)

Duncan Group	Mean	N	Station
A	192	74	Camp Cr
B	118	65	PigeonR3
B	116	50	Red Bank
B	115	35	Cuffawa
C B	105	148	PigeonR1
C B	96	224	Coldwtr2
C B	92	145	PigeonR2
C	84	76	Coldwtr3

Suspended Solids (0.0001)

Duncan Group	Mean	N	Station
A	84.9	73	Camp Cr
B	51.9	145	PigeonR1
B	49.9	34	Cuffawa
B	49.1	49	Red Bank
B	46.2	218	Coldwtr2
B	40.2	63	PigeonR3
B	37.3	140	PigeonR2
B	28.6	75	Coldwtr3

Total Phosphorus (0.0001)

Duncan Group	Mean	N	Station
A	0.344	82	Camp Cr
B	0.229	74	PigeonR3
C	0.152	37	Cuffawa
D C	0.149	249	Coldwtr2
D C	0.146	166	PigeonR1
D C	0.138	56	Red Bank
D C	0.111	158	PigeonR2
D	0.101	83	Coldwtr2

Nitrate (0.0001)

Duncan Group	Mean	N	Station
A	1.62	35	Cuffawa
B	1.10	74	PigeonR3
C	0.73	82	Camp Cr
C	0.70	159	PigeonR2
D	0.42	56	Red Bank
D	0.40	166	PigeonR1
E	0.29	84	Coldwtr3
E	0.20	249	Coldwtr2

Chlorophyll (0.1336)

Duncan Group	Mean	N	Station
A	62.2	49	PigeonR3
B A	40.5	108	PigeonR2
B A	36.4	30	Cuffawa
B A	34.6	70	Camp Cr
B	30.9	71	Coldwtr3
B	28.0	212	Coldwtr2
B	26.9	141	PigeonR1
B	22.0	46	Red Bank

stimulate primary production. Thus channelization could either increase or decrease chlorophyll concentrations. In this system both have occurred. All three ANOVA show significant differences between stations for chlorophyll. The Duncan's test groups the natural streams with some of the channelized streams. The natural streams have low chlorophyll levels, as do some of the channelized streams. Chlorophyll concentrations in this system seem to be stimulated more by nutrients than inhibited by high suspended solids.

To this point, all differences between the water quality in natural and channelized streams have been attributed to the differences in the channels. This assumption may or may not be valid. Another factor which can play a major role in water quality is land use. If the land use is essentially the same in all the basins, the assumptions may be valid. However, if the channelized reaches are more intensely farmed, the differences may be due to land use.

Effects of Previous DEC Construction on Water Quality

The water quality data base for the upper Coldwater River basin covers the period 1989 through the present. DEC construction was initiated in 1991, but the first items were not completed until October and November 1991. Thus the 1991 water quality data are representing pre DEC project conditions. As mentioned in the previous section the water quality data was sorted into two groups labeled pre and post. An ANOVA was performed on both groups. The results of the analysis of variance comparing the means of the post data are summarized in Table 4. The data used in the ANOVA covers the period 1992 to the present. It is pre project with regard to the proposed work covered by this EIS, but it provides a means to analyze the impacts of other DEC construction and to predict the likely impacts of this proposed construction. In order to evaluate the impacts of previous DEC construction any trends reflecting changes in the basin's water quality need to be discerned. Figures 2 through 11 plot the annual means of 10 water quality parameters by station. These figures will allow the changes in water quality over time to be examined. They also aid in the comparison of water quality between stations. DEC construction in this period includes bank stabilization, grade control structures, and riser pipes.

Figure 2 which plots the annual temperatures does not reveal any trends in temperature among the stations. Dissolved oxygen does appear to be steadily increasing over time at nearly every station. Camp Creek, Upper Coldwater, and Red Bank Creek all have slight dips in the mean annual DO during 1994 (Figure 3). Hydrogen Ion concentrations (Figure 4) go up and down at every station. All stations have a major decline in mean annual pH during 1991. The extreme amounts of rainfall during 1991 are likely responsible for those lower pH values. The annual mean conductivity is plotted in Figure 5 and shows a pattern similar to pH. Like pH, conductivity is generally lower at all stations during 1991. The dilution of the surface water by the large amount of rain received in 1991 is again the likely cause for the observed change. Figures 6 and 7, which plot the annual means for total and suspended solids, show very definite trends toward lower values with time. The most significant changes are seen at the channelized stream stations. The annual means for total solids at Red Bank Creek fall from around 300 mg/ℓ pre project to around 100 mg/ℓ in the post period, while the suspended solids means at that station fall from approximately 200 mg/ℓ to 60 mg/ℓ. The ANOVA for the post period (Table 4) places all stations except Camp Creek into two overlapping groups for total solids and one group for suspended solids. Even the natural stream stations show reductions in total and suspended solids. Although there are no grade control or bank stabilization items on the Coldwater River, over 20 riser pipe structures were constructed on creeks which drain directly into the upper and middle Coldwater River. The post period DEC construction affects these streams in descending order, Byhalia, Cuffawa, Red Bank, Camp, Coldwater, Pigeon Roost. Byhalia, Camp, and Red Bank Creeks construction item were completed first. There is no water quality sampling site on Byhalia Creek, but site 7-4 (Pigeon R1) on Pigeon Roost Creek is immediately downstream of the confluence. The annual mean total phosphorus falls at most stations in a manner similar to total and suspended solids. Cuffawa and Red Banks Creeks have had many riser pipes constructed and they show greater changes in total phosphorus than Camp Creek which has received several bank stabilization items.

In contrast to the changes in total solids, suspended solids, and total phosphorus; orthophosphorus, nitrate and ammonia levels appear to be increasing with time. This observed trend

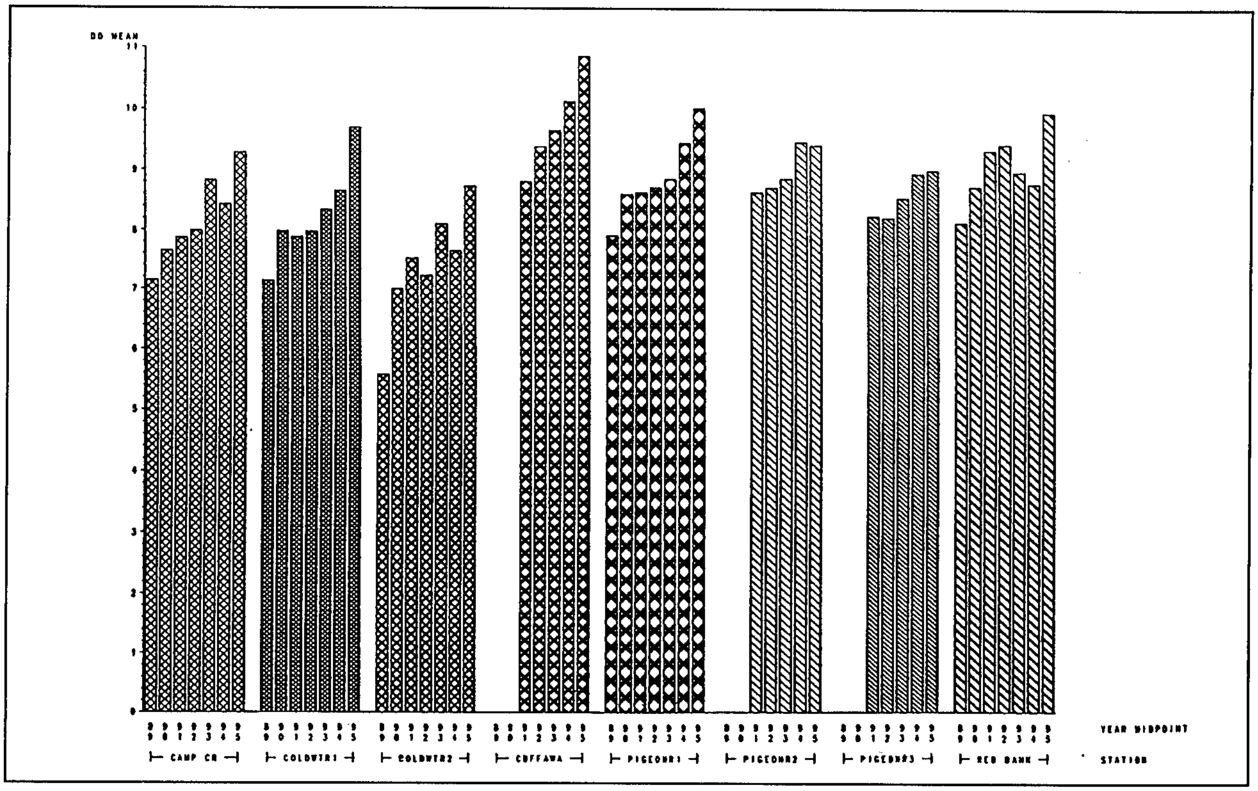

Figure 3. Annual mean dissolved oxygen by station

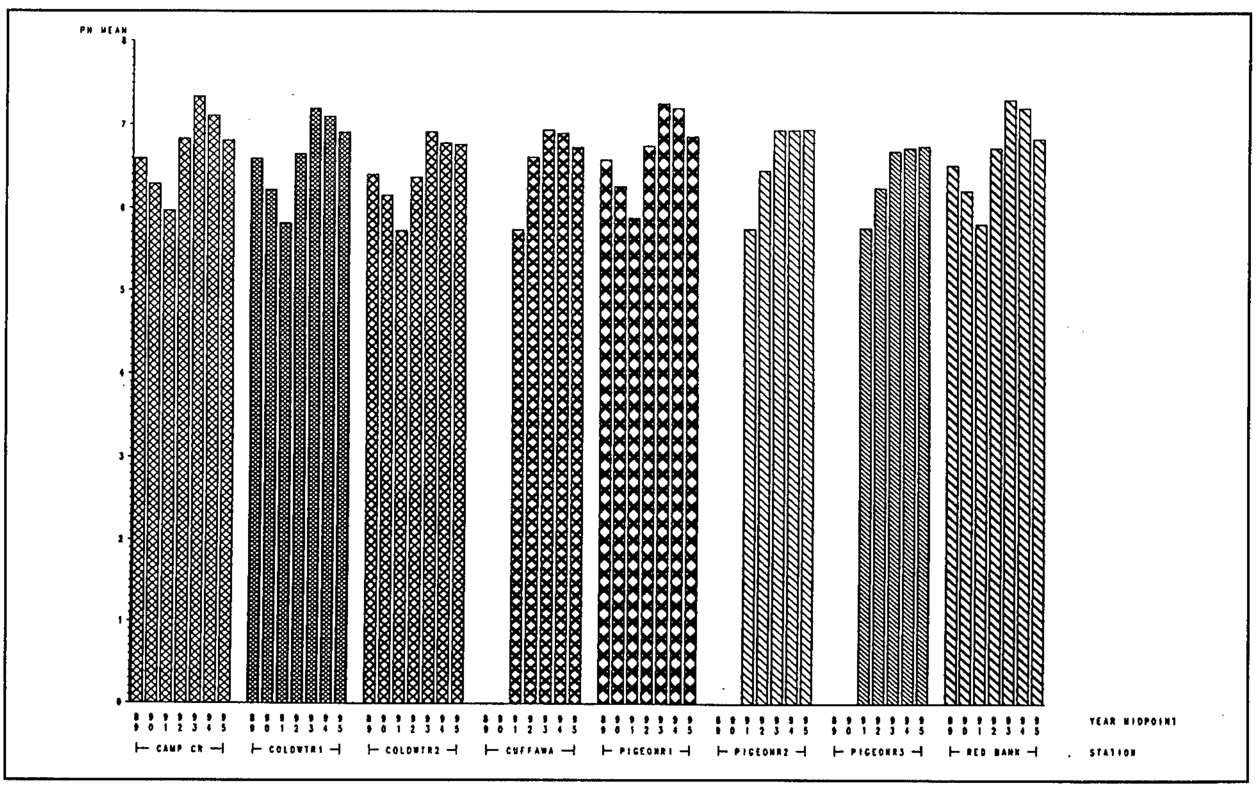

Figure 4. Annual mean pH by station

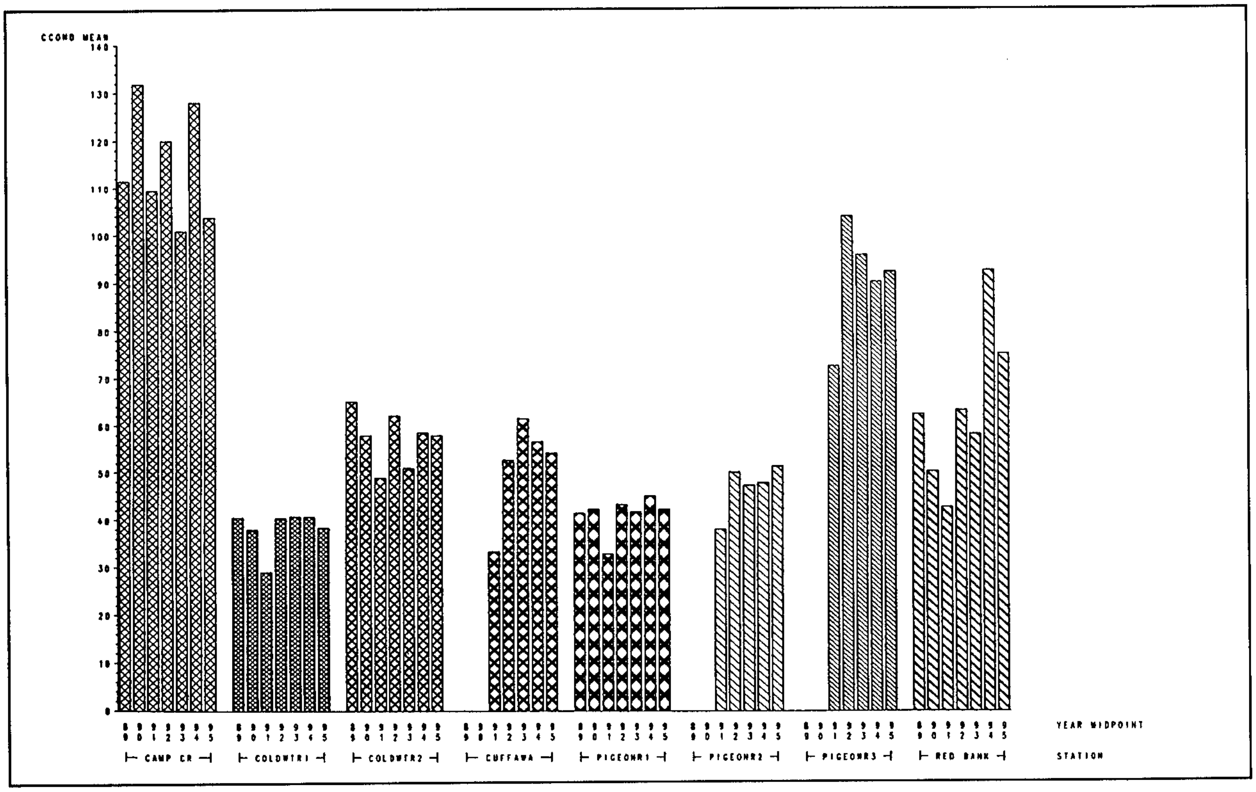

Figure 5. Annual mean conductivity by station

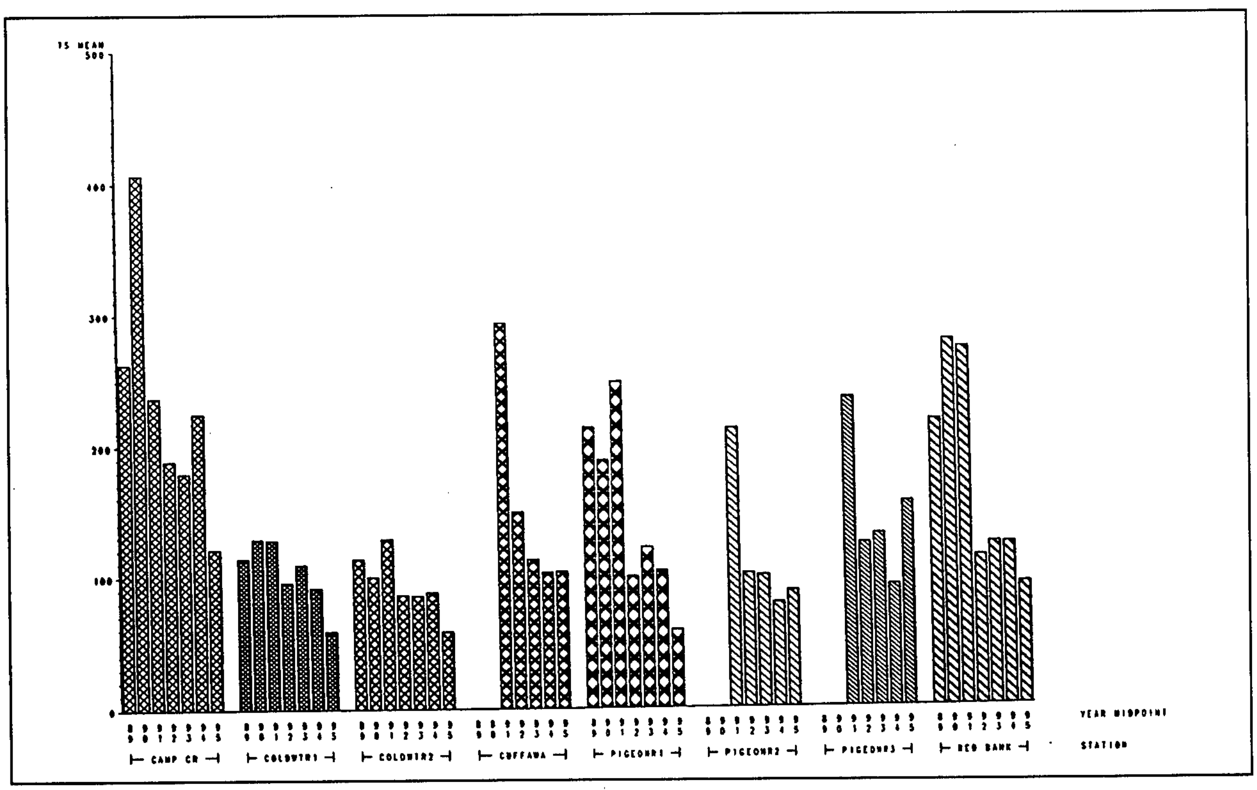

Figure 6. Annual mean total solids by station

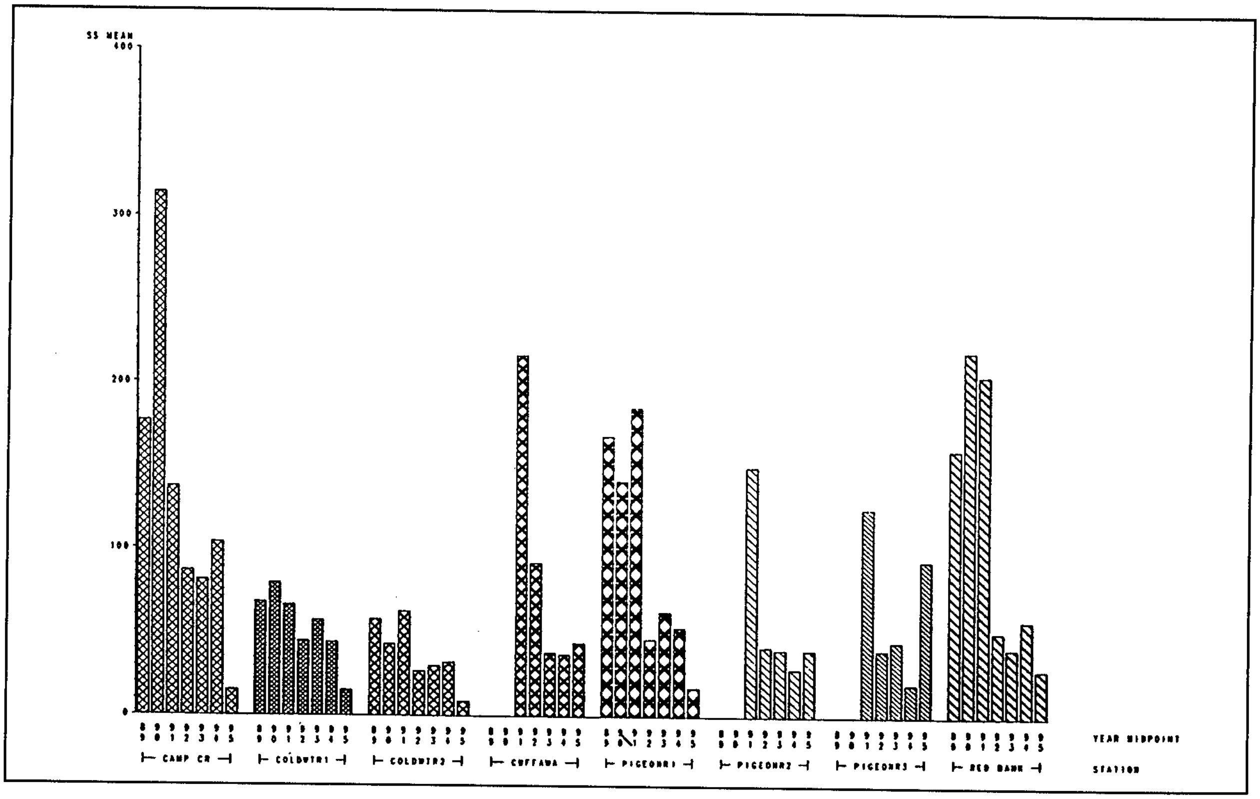

Figure 7. Annual mean suspended solids by station

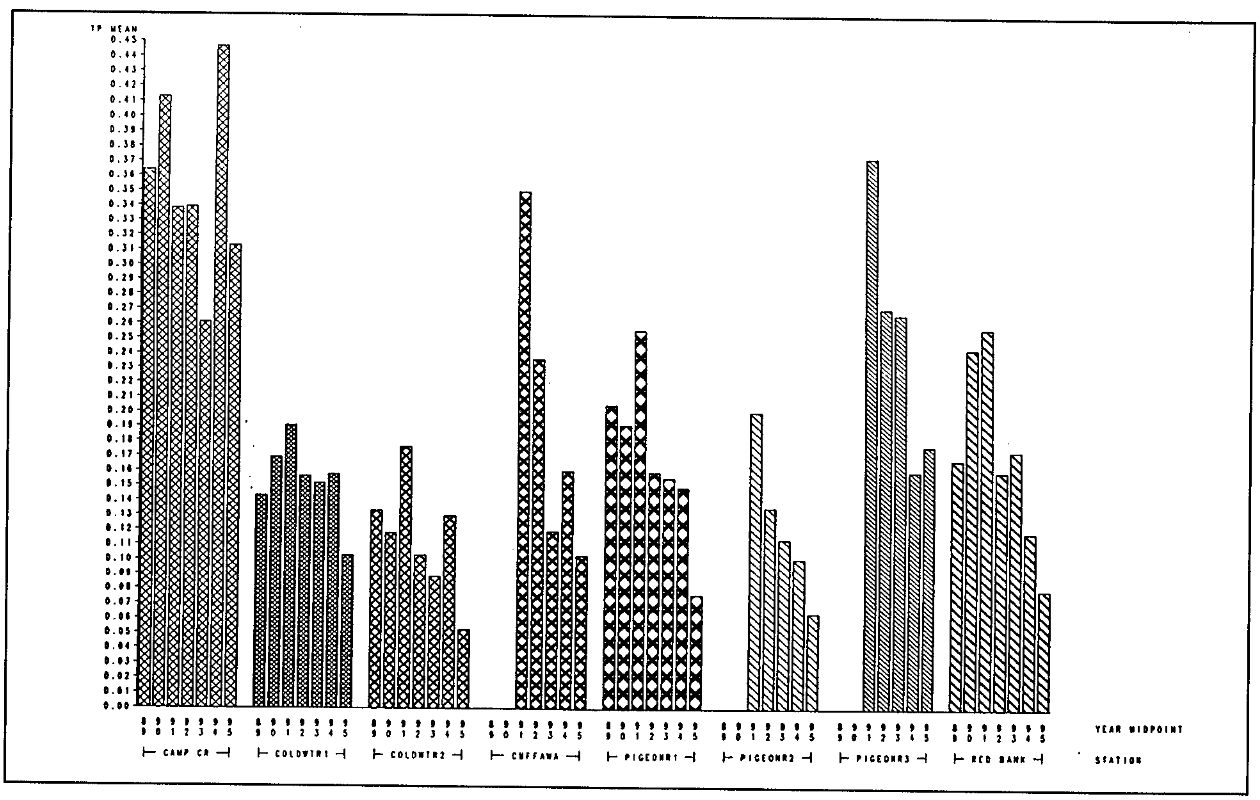

Figure 8. Annual mean total phosphorus by station

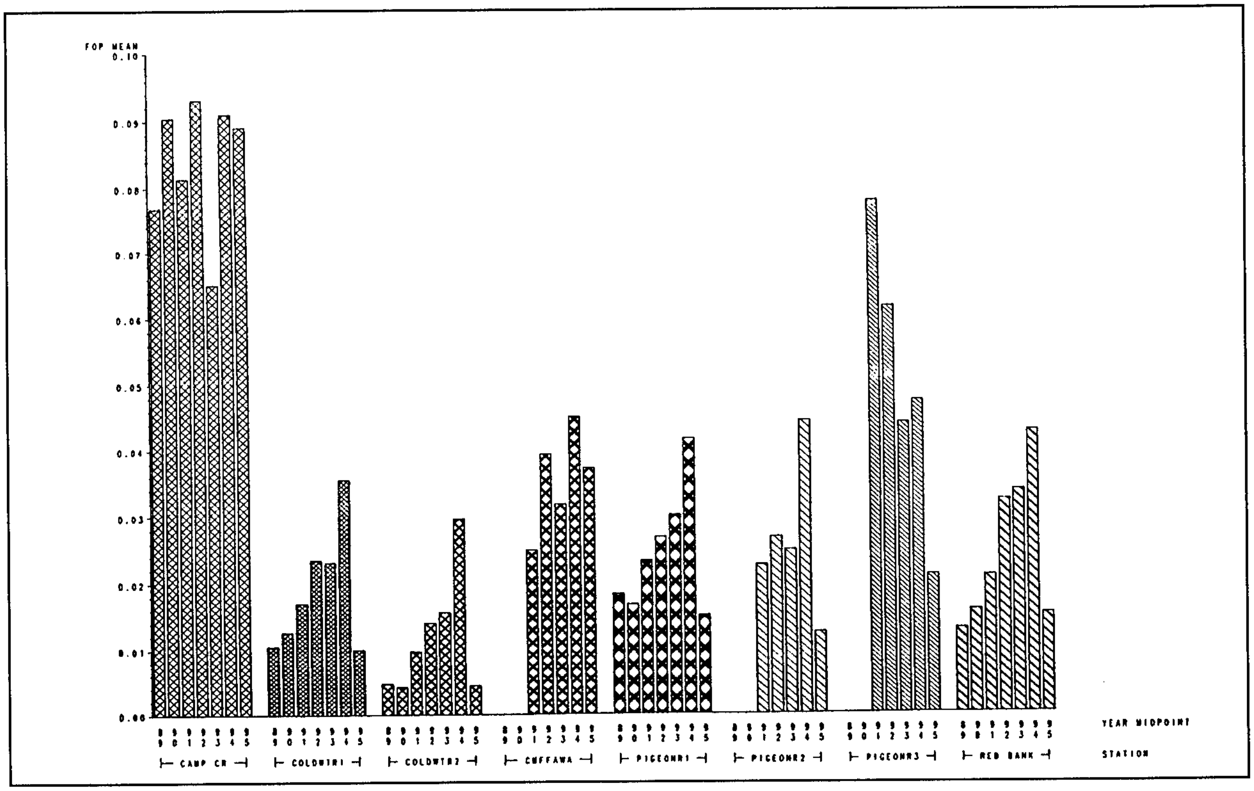

Figure 9. Annual mean orthophosphorus by station

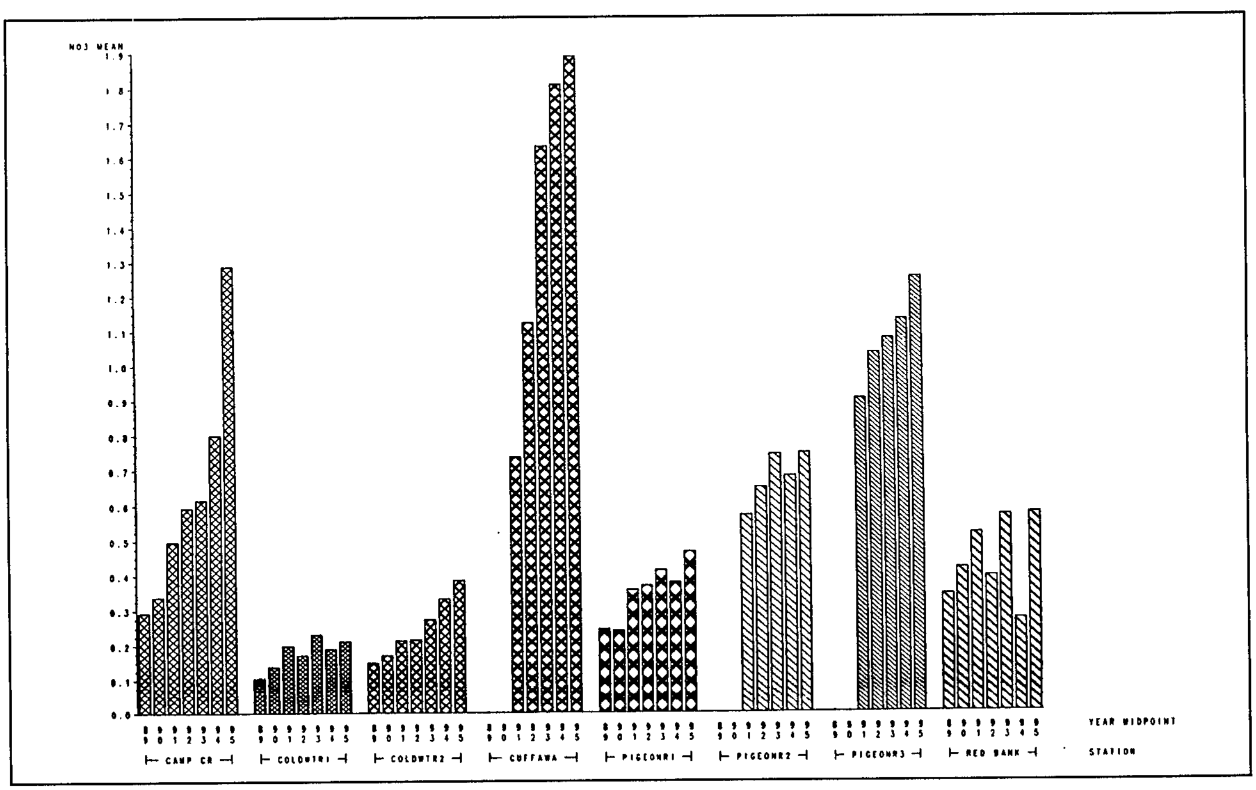

Figure 10. Annual mean nitrate by station

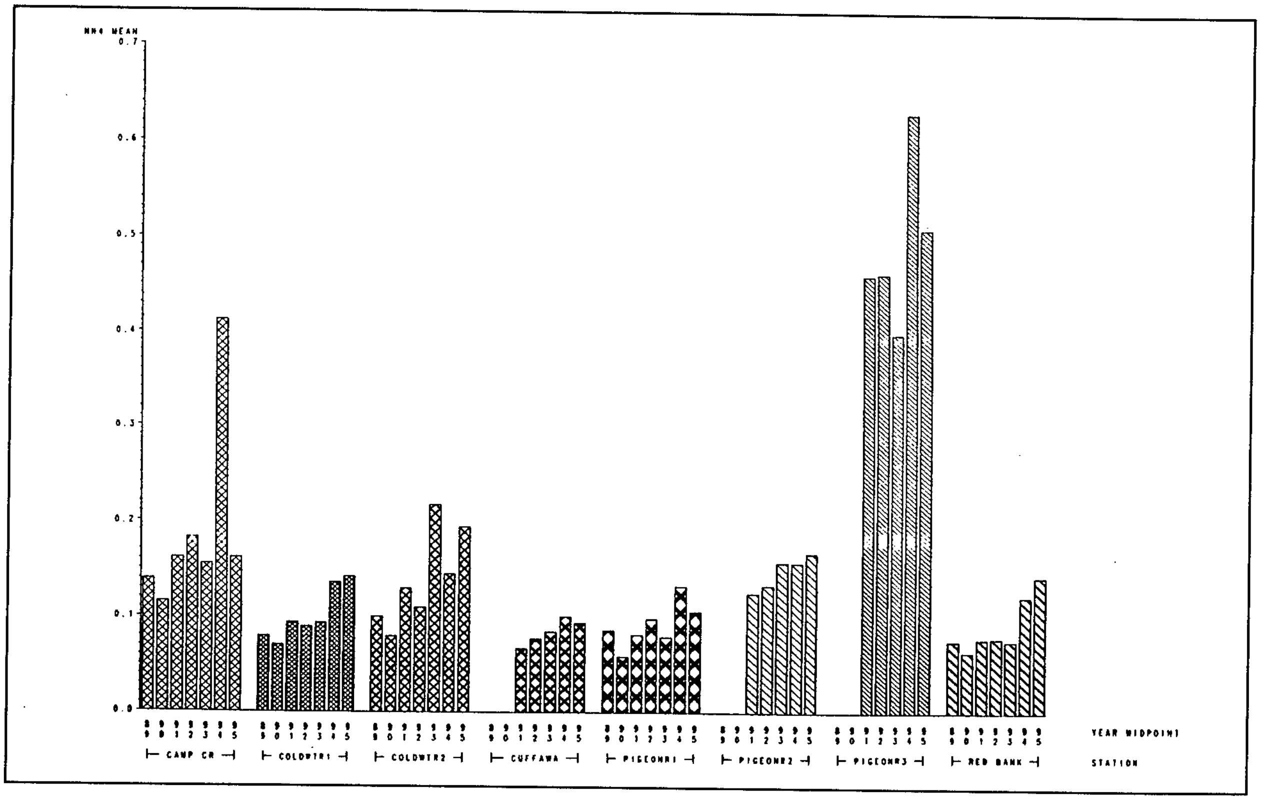

Figure 11. Annual mean ammonia by station

is somewhat puzzling, because nutrients are frequently bound to suspended solids. However, these are measurements of dissolved nutrients. These nutrient levels have increased at all stations even the natural stream stations and thus the changes may reflect something other than DEC construction. Finally Figure 12 plots the annual means for chlorophyll. Chlorophyll levels go up sharply at all stations. High suspended solids levels may have been inhibiting primary production, but the loss of suspended solids and the increase in nutrient levels has significantly increased primary productivity.

Overall, the construction of previous DEC work items in the upper Coldwater River Basin has significantly improved water quality. The area of greatest impact from channelization (suspended solids) identified in the previous section has been improved the most. The post period conditions find no significant differences between most of the channelized streams and the natural streams for total solids, suspended solids, and total phosphorus. The levels of all three are significantly reduced at most stations.

Conclusion

The results of this study have shown that the effects of channelization vary greatly between streams. In general, the channelized streams in the Coldwater River system do not show increased temperatures or reduced dissolved oxygen levels. Most channelized streams did have higher suspended solids and total solids levels. This study also showed that the channel stabilization techniques used in the DEC Program can dramatically improve water quality in a short period of time.

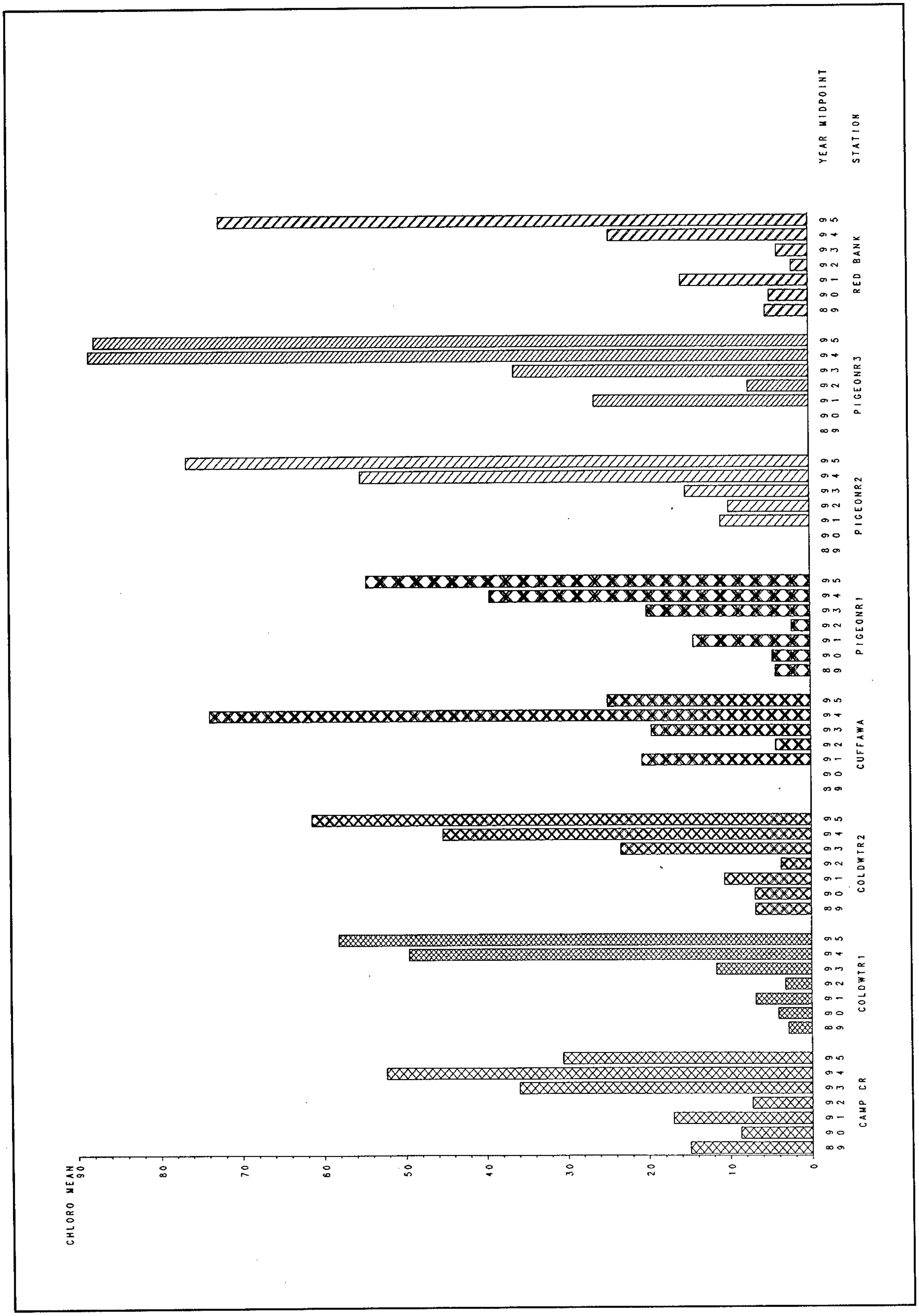

Figure 12. Annual mean chlorophyll by station

Water Quality Studies in America's Greatest Wetland

by
Richard Punnett[1]

Abstract

The original Everglades, covering about 18,000 square miles of south Florida, has undergone tremendous landscape and water quality changes since 1900. Increases in nutrients, pesticides, and soluble mercury—coupled with a staggering influx of hominids—have dramatically transmogrified the Everglades. This paper will characterize the water quality problems, research, modeling activities, and potential solutions within the remaining Everglades.

Many state and federal agencies, private businesses, tribes, and coalitions are working toward the ultimate goal: the sustainability of the south Florida ecosystem including the hominids. To date, the losses have exceeded the gains; however, the hominids are learning the right moves and making the right sounds.

(This report was submitted late. The complete report can be found at the end of this proceedings.)

1 U.S. Army Engineer District, Jacksonville; Jacksonville, FL.

Overview of the Groundwater Modeling System

by
Jeffery P. Holland[1]

Introduction

Activities at Department of Defense (DoD) installations have produced contamination of groundwater resources that may pose problems for human health and the aquatic environments adjacent to or on these posts. The U.S. Army Engineer Waterways Experiment Station (WES), in concert with the U.S. Army Environmental Center, is leading a consortium of Army, Air Force, Department of Energy (DOE), Environmental Protection Agency (USEPA), and academic researchers in the development of a new state of the art in subsurface modeling: the Groundwater Modeling System (GMS). The GMS provides a consistent computational basis for site characterization and for simulating groundwater and unsaturated zone flows, the transport/fate of subsurface contaminants, and the efficacy of remedial actions associated with contaminated groundwater resources at DoD installations and other Federal sites. The system is also being applied for a variety of Army Civil Works concerns associated with salinity intrusion, channel deepening, and surface water/groundwater interactions. The system is modular in design for operation across a variety of computing platforms. The system has been fielded and is in use for a number of site-specific project applications.

The development of the GMS focuses on key aspects of the remediation of contaminated groundwater resources and upon the development of engineering aids for site characterization and contaminant assessment. The system also addresses multiple civil works concerns such as: salinity intrusion resulting from navigation channel deepening, cleanup of contaminated Corps of Engineers properties (e.g., new real estate procured for water resources projects), flow and transport of contaminants from dredged material placement areas, and surface water/groundwater interactions (e.g., South Florida water management).

A two-pronged technical development approach is being followed: (a) conduct development to improve DoD's immediate use of the better of existing models and tools through existing model enhancement and guidance, thereby bringing DoD's state of practice up to the state of the art by the middle of 1996, and (b) concurrent with (a) (and extending beyond), develop the tools DoD will need in the future to model its remediation activities. The technological centerpiece of the effort is the development of the GMS. A schematic of the five development levels for the GMS is presented in Figure 1. The current versions of the GMS concentrate on improved use of existing models through coupling them with a comprehensive user interface, visualization tools, parameter estimation techniques, and subsurface conceptualization methods. As such, many of the features of Levels 1-3 as shown in Figure 1 are complete. Future versions of the GMS are incorporating Levels 2-5 improvements.

An implicit part of the demonstration of the developed technology is thorough testing and verification of the modeling components of the GMS. This will be done through use of controlled laboratory data and well-characterized field data sets from DoD and DOE sites under active cleanup and from a number of Army civil works projects.

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[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

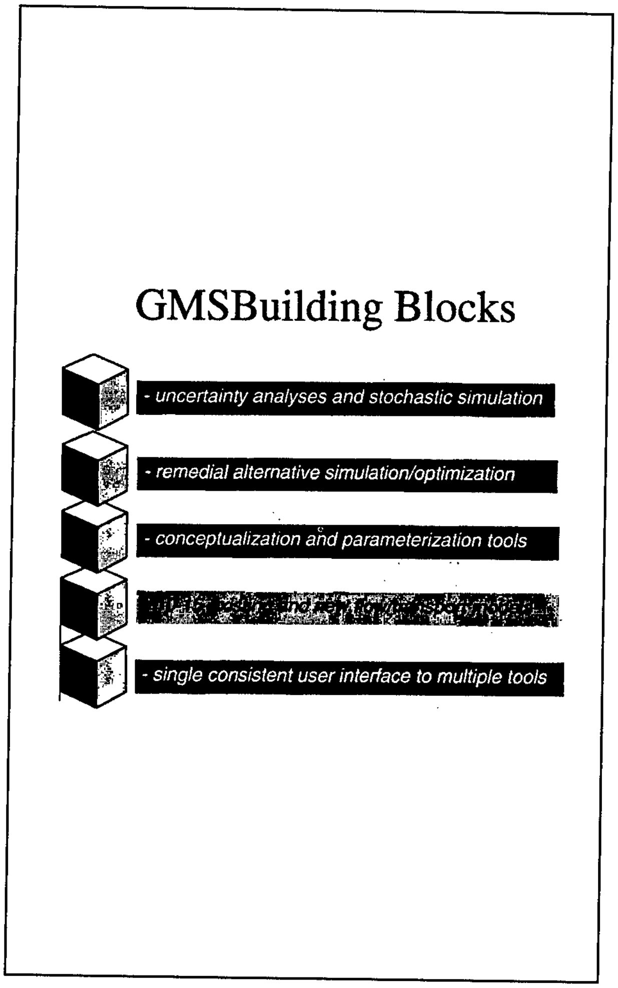

Figure 1. GMS major research and development components

Current GMS Capabilities

Flexibility and portability are prime design considerations for the GMS. The DoD ground-water modeling user community is known to compute on personal computers (PCs), UNIX workstations, and supercomputers. Additionally, due to regulatory requirements, DoD model users may require access to up to 10 existing subsurface flow and/or transport models as part of remedial evaluation and design. Version 1.2 of the GMS has been developed for application of multiple subsurface models (FEMWATER, LEWASTE, MT3D, and MODFLOW) on multiple UNIX workstations running primitive X-windows and for personal computers running Windows 3.1. The GMS's user environment, which is a joint development of Brigham Young University (BYU) and WES, has been developed in C. This basic user environment will front all models integrated into the GMS. The same environmental look and feel is being employed for each GMS-supported model. In this manner, the users will have to become acquainted with only a single computational environment in their use of multiple modeling and analysis tools. An additional four models will be incorporated within the GMS in 1996.

The GMS user environment is divided logically according to task. These tasks include those required for conceptualization of a given site cleanup, subsurface flow and transport modeling for the site, and visualization and animation of model results. Commands are divided into nine consistently designed modules. Selecting a new module entails choosing an appropriate icon from an array of nine icons in a tool palette. The interface changes to reflect a new set of tools and menu selections that are applicable for the new module. The modules are briefly defined in Table 1. By dividing the program up into task-oriented modules, the user is effectively shielded from unnecessary complexity of other tasks in the system while focusing on specific procedures at hand.

The GMS provides a series of tools that have been designed specifically for defining surfaces and solids of geologic origin. The system will accept field data in the form of borehole logs and allow the user to process the data to directly define a solid model. The approach taken is to first define surfaces separating the various stratigraphic units, followed by a series of extrusions, and set operations to define a solid model of the site. GMS uses triangulated irregular networks to model terrain and the interfaces between stratigraphic units.

Semiautomated and fully automated procedures are also provided within the GMS for generating two- and three-dimensional (2-D and

Table 1
Description of Modules Used in GMS

Module Name	Description
TIN	Tools for building and editing TINS
Borehole	Tools for viewing and manipulating borehole data
Solid	Tools for constructing solid models of geologic units
2-D Finite Element Mesh	Tools for building and displaying a 2-D finite element mesh
2-D Finite Difference Grid	Tools for building and displaying a 2-D finite difference grid
2-D Scattered Data	2-D interpolation and geostatistical tools
3-D Finite Element Mesh	Tools for building and displaying a 3-D finite element mesh. Preprocessing/postprocessing for 3-D finite difference code
3-D Finite Difference Grid	Tools for building and displaying a 3-D finite difference grid. Preprocessing/postprocessing for 3-D finite difference code
3-D Scattered Data	3-D interpolation and geostatistical tools

3-D) finite element and finite difference meshes and grids. A series of dialogs and graphical tools can then be used to define specific boundary conditions relevant to the analysis code to be used. Geometry and boundary condition files can then be saved and the solution computed using the analysis code.

In keeping with the objective of providing a fully integrated environment for groundwater modeling, significant effort has been made to provide state-of-the-art tools for scientific visualization. Once the analysis engine has been run, GMS will directly read the analysis results and display them in a variety of formats. Methods for visualizing 2-D and 3-D data include contours and fringes, vector arrows, isosurfaces, and cutting planes, as well as several animation techniques.

Near-Term GMS Developments

The GMS is undergoing a number of enhancements in fiscal years 1996 and 1997. These include the following:

a. Integration of four new codes into the GMS. These are RAND3D (the 3-D version of the RANDOMWALK model), NUFT3D (a 3-D multiphase model developed by Lawrence Livermore National Lab (LLNL) that simulates steam injection, vapor extraction, and electrical heating), ADH (an adaptive subsurface 3-D model under development by WES), and SWGW (an adaptive surface water-groundwater interaction model under development by WES). Additional models under development within the DOE may also be added as appropriate.

b. Linkages to Arc/Info, ArcView, and GRASS geographic information systems (GISs). The GIS packages are deemed those most in demand by subsurface cleanup specialists at present. These linkages will include import/export features. Analogous features will be provided for linkage to computer-aided design and drafting packages.

c. Linkages to installation restoration project databases in use by the Army and Air Force (e.g., IRPIMS, IRDIMIS, and

and their successor) for easy downloading of site data.

d. Initial remedial design modules.

e. Incorporation of optimization modules for design of optimal borehole and well placement and for design of remediation measures.

f. Conceptualization methods for integration of "hard" and "soft" data.

Remedial Alternative Simulators

The prime driving force behind the development of the GMS is to vastly improve the cost-effective design and operation of remediation methods for site-specific military cleanups. Presently, simulation (and optimization) of remedial alternative operations is based on one or more of the following: (a) inferences derived from subsurface flow and/or transport simulations; (b) specialized, remediation-specific simulation capabilities developed as enhancements to initial flow and transport models; and (c) coupled simulation/optimization methods built around either (a) or (b). However, there is a need for an efficient computational means for comparing multiple remedial alternatives (both economically and from an exposure risk perspective) for a given site cleanup. Such development is the goal of this work area. Remedial module development will proceed in coordination with associated efforts to develop optimization algorithms and methods to incorporate parameter uncertainty into model predictions. WES has developed a multiyear, phased approach that will provide the ability to simulate and evaluate differing remedial alternatives within the GMS. These technologies were chosen because of their interest to DoD. Space prohibits detailed discussion of these developments herein; the reader is referred to Holland (1996) for complete details.

Map Module Development

In order to facilitate a new approach to conceptual model development, a new module called the Map Module is being added to the GMS. Four types of objects are supported in the module: DXF objects, image objects, drawing objects, and feature objects. The first three objects are primarily used as graphical tools to enhance the development and presentation of conceptual models. DXF objects consist of drawings imported from standard CAD packages such as AutoCAD or MicroStation. Externally produced site drawings can often provide a useful backdrop or supplement to the graphical desktop during model construction. Drawing objects are essentially a simple set of tools that allow the user to draw text, lines, polylines, arrows, rectangles, etc., to add annotation to the graphical representation of the conceptual model. Image objects are digital images representing aerial photos or scanned maps in the form of TIFF files. TIFF files can be imported and registered to real-world coordinates. Construction of the conceptual model can then be accomplished using a high-resolution image in the background of the graphical desktop.

Feature objects are used to used to construct the actual conceptual model. Feature objects are patterned after the data models used by GIS such as Arc/Info. The GISs data model utilize points, arcs (polylines), and polygons to represent spatially distributed information. For example, points represent data such as wells or point sources for contaminants; arcs represent rivers or model boundaries; and polygons represent areal data such as lakes or zones defining different recharge zones or hydraulic conductivities. Sets of points, arcs, and polygons can be grouped into layers or coverages. Since a GIS approach is used, the conceptual model can be exported to or imported from another GIS. However, a separate GIS is not required because the entire model can be constructed within

GMS. Details of this development are available in Richards and Jones (1996). The use of the previous two paragraphs from these authors is gratefully acknowledged.

Summary of Current Applications

There are a large number of GMS applications ongoing. These fall into two categories: military installation cleanups and Army civil works projects. The installations within the first category include: Fort Dix, New Jersey; Schofield Barracks, Hawaii; Memphis Depot, Tennessee; Louisiana Army Ammunition Plant; Columbus AFB, Mississippi; and, Aberdeen Proving Grounds, Maryland. These investigations center around development of site conceptual models, on contaminant transport simulations, and on evaluation of the efficacy of varying remediation scenarios. In addition, WES has conducted several groundwater modeling studies for the USEPA such as the Walnut Creek Watershed, Iowa, and the Savannah River Nuclear Site, Georgia.

Example Army civil works applications include: surface water/groundwater interactions in the Central and South Florida Project and the C-111 Project (for the Corps of Engineers Jacksonville District in conjunction with the South Florida Water Management District); salinity intrusion into coastal aquifers possibly resulting from proposed channel modifications to Wilmington Harbor, North Carolina (for the Wilmington District); and conceptualization of hydrocarbon plumes in the vicinity of a residential area (for the Philadelphia District). As pointed out above, the GMS is designed to provide support to both military and Army civil works applications (including Superfund sites). Additional civil works applications include evaluation of capped dredged material placements; riverine and reservoir interactions with groundwater resources (particularly for regional water supply investigations); leachate releases from landfills; seepage through earth embankments and dams; and several analogous types of water resource concerns.

Summary and Acknowledgment

The DoD is continuing its development of a comprehensive groundwater modeling system for use in the cleanup of contaminated groundwater resources at military sites and to address a variety of Army civil works concerns. The multiyear research effort strongly leverages ongoing and near-future research through partnering with several Federal agencies and universities. Substantial improvements in the effectiveness and defensibility of remedial actions have already been achieved through implementation of this system, thereby resulting in significant savings in remediation costs.

This paper was prepared from research and development conducted under the Groundwater Modeling Program of the WES, Vicksburg, MS. Permission was granted by the Chief of Engineers to publish this information.

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Predicting the Removal of Pollutants by Wetlands

by
Mark S. Dortch[1]

Introduction

Constructed and natural wetlands can function as effective treatment systems for removing water pollutants. Various processes, such as settling, nitrification, denitrification, biodegradation, hydrolysis, volatilization, and photolysis, can transform and remove pollutants. Mathematical models can be used to estimate the amount of pollutant material removed for evaluating wetland functions or when designing constructed wetlands for water quality improvement.

The amount of pollutant material removed from receiving water can be quantitatively expressed in terms of the removal efficiency (*RE*), where *RE* is the percentage of influent trapped or removed by the wetland and is computed from

$$RE = \frac{Mass_{in} - Mass_{out}}{Mass_{in}} \times 100 \qquad (1)$$

where mass is the time integral of pollutant mass flow rate (kilograms/day) entering or leaving the wetland over a specified period. Pollutant mass flow rate, or loading, is the product of water flow rate (Q) and pollutant concentration (C). *RE* of 100 percent denotes total removal of a pollutant.

The objective of this work was to develop a relatively simple, easy-to-use method for predicting RE given basic wetland characteristics. An analytical, screening-level, mathematical model was formulated for this purpose. The model is referred to as PREWet, Pollutant Removal Estimates for Wetlands.

Model Description

The primary assumption made with the analytical model to achieve simplicity is that the wetland is at steady state (i.e., flow and concentrations are constant in time). Although wetlands may not be at steady state, steady-state analyses are useful for evaluating long-term, average conditions. Mean annual input conditions (e.g., flows and depth) are consistent with this assumption.

Either of two conditions is assumed for spatial gradients in concentration: a) fully mixed (no gradients); and b) longitudinal gradients exist along the main flow axis (well mixed laterally and vertically, or plug flow). *RE* for these two conditions can be calculated from conservation of mass, considering longitudinal mass advection, neglecting dispersion, and assuming steady-state conditions. For fully mixed conditions, the relationship for *RE* is

$$RE = \left[\frac{K\tau}{1 + K\tau} \right] \times 100 \qquad (2)$$

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

where

K = bulk removal rate, day^{-1}

τ = water detention time, days

For fully mixed conditions, τ equals the hydraulic residence time (HRT), which is the wetland water volume (V) divided by the mean flow rate (Q). When longitudinal gradients exist, the relationship is

$$RE = \left(1 - e^{-K\tau}\right) \times 100 \tag{3}$$

For this case, τ can be less than HRT due to dominant flow paths that result in dead zones and short-circuiting of flow. However, for truly well mixed and plug flow conditions, τ = HRT.

The bulk removal rate depends on pollutant-specific processes and wetland ambient conditions, such as water temperature, depth, and flow velocity. Thus, obtaining a representative K value can be problematic. The approach here was to focus on the dominant long-term removal mechanisms, making use of literature values or mathematical formulations for those mechanisms when possible. The model contains methods for estimating or specifying K values for the following:

- Total suspended solids (TSS) or inorganic suspended solids (ISS).
- Total coliform bacteria (TCB).
- Biochemical oxygen demand (BOD).
- Total nitrogen (TN).
- Total phosphorus (TP).
- Contaminants (e.g., organic chemicals and trace metals).

The removal rates for TSS, TP, and contaminants, which have sedimentary interactions, are based on mechanistic concepts taken from the literature, whereas more empiricism is employed for BOD, TCB, and TN, which are predominantly biologically mediated. Consistent with the steady-state assumption, the model utilizes dominant, long-term, net removal processes, such as denitrification for TN and sediment burial for TP and trace metals. Seasonal and annual processes such as plant growth and death are not included. Thus, this model should not be used to evaluate such short-term effects. Nutrients are cycled through plants, but at steady state, plants do not have a net effect on nutrient removal except through detrital deposition to the sediments, thus contributing to burial. Details of the removal rate formulations for estimating K values for each water quality constituent are too lengthy to be presented here but are described by Dortch and Gerald (1995).

Model Confirmation

For confirmation purposes, the analytical model was tested against available field observations from the literature and from the Cache River wetland (CRW), a major study site of the Corps of Engineers Wetlands Research Program. Flows and concentrations observed at wetland upstream and downstream boundaries were used to estimate "observed" RE values. With estimates of detention time and other wetland characteristics, such as τ and mean depth, it was possible to calculate RE for comparison with observed RE values.

Upstream and downstream flows and concentrations observed during 1987-1990 on the Cache River (i.e., Patterson and Cotton Plant gauges, respectively) were used to integrate mass fluxes for estimating long-term RE values for TSS, TN, and TP. Observed and predicted RE values for the CRW are summarized as follows:

Constituent	Observed *RE*, %	Predicted *RE*, %
TSS	29.5	25.9
TN	21.4	18.1
TP	3.0	3.1

These results tend to support confirmation of the model for TSS, TN, and TP. The details of the application of PREWet to the CRW are discussed by Dortch and Gerald (1995).

The CRW effectively removes TSS and TN relative to TP. Higher RE values for TN are reasonable considering that denitrification in benthic sediments provides a significant removal pathway indefinitely. TP is removed by sorption to solids that settle and are eventually buried. The low TP removal efficiency is related to the saturated conditions of the benthic sediments, i.e., the sediments have a low capacity to accept and retain additional phosphorus. Net phosphorus removal for an established bottomland hardwood wetland, such as the Cache River, is controlled more by sediment burial rather than through net settling from the water column as with a new constructed wetland. Removal through burial is much smaller than through settling. PREWet can estimate whether a wetland has established equilibrium or whether it can be classified as a new wetland. The *RE* for TP is then computed with the appropriate equations (i.e., established or new wetland).

As mentioned above, the algorithms for estimating the TP bulk removal rate coefficient are based on the assumption that the growth and death of aquatic plants have no net effect on the long-term fate of TP. The justification for this is that plants take up P when they grow, but recycle it when they die; therefore, for steady-state conditions, there is no effect. TP is assumed to partition only between water and solids, and its fate is affected by settling of suspended solids, resuspension of sediment, diffusion of sediment pore water, and sediment burial. A steady-state mass balance for the water column and sediment bed is used to derive the relationship of TP removal rate to these processes. This approach is probably adequate for most forested wetlands, such as the CRW, where peat buildup is minimal. Over 80 percent of the solids entering and leaving the Cache is inorganic. However, the use of this approach is questionable for marshes where peat building is a dominant process. Some experts have revised their thinking and now consider peat building to be an important removal mechanism for phosphorus. The idea is that plants die and fall to the bottom to form peat. Part of the peat organic matter is refractory, or decays very slowly. This refractory organic matter, which holds some phosphorus, is eventually buried, thus removing phosphorus from the system so not all the phosphorus taken up by plants is recycled. Therefore, the algorithms for estimating the TP removal rate coefficient may not be applicable for marshes since peat building is not included.

Model Implementation

PREWet has been coded into a user-friendly, interactive computer program operational on PCs. The equations and logic are programmed in C. A commercially available graphical user interface library, ZINC, is implemented in C++ to provide windows, buttons, and menus for data input and selection of variables and parameters. The model is designed to be as self-explanatory as possible with on-line help features available.

The user may select the features shown in Figure 1 from the main screen. Selection of system properties invokes a series of screens that allows the user to describe the physical characteristics of the wetland under study, i.e., the size (including volume, area, width, length, and depth), water temperature, and flow rate. Other system properties, HRT, τ, and mean velocity, are calculated from these and displayed. Within the system properties section, the user also specifies the assumption for spatial gradients, i.e., fully mixed or longitudinal gradients.

The constituent selection button displays a window that contains the selection list of constituents for which RE can be calculated. The user may select any or all of the six constituents, TSS, TCB, BOD, TN, TP, and contaminants. If the user selects TP and/or contaminants, then TSS must also be selected since TSS is required

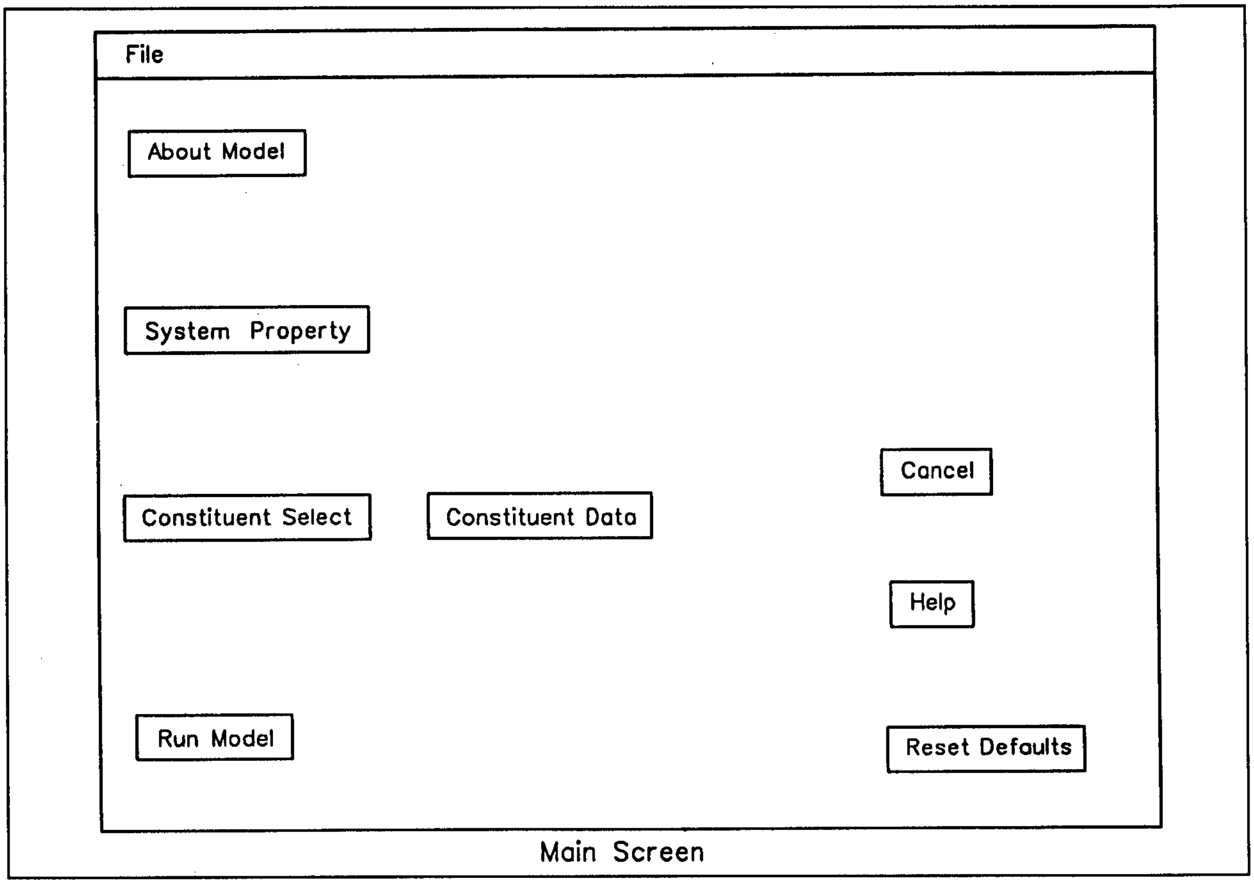

Figure 1. Main screen features

for TP and contaminant *RE* calculations. The constituent data button should be pressed next to begin input data specification. This button brings up input windows for each of the constituents selected.

Each interactive session can be saved as a file for subsequent use or modification. Default values of parameters are provided. Whenever possible, parameters are calculated to reduce burden on the user, but the user has the option to override default values and computed parameters. After all input data has been specified, the user can select the run model button. Accepting this button displays the calculated removal efficiencies for all variables selected.

Summary

The steady-state, analytical model PREWet provides a relatively simple method for estimating pollutant removal efficiency for free surface wetlands. The model is applicable to TSS (or ISS), BOD, TCB, TN, TP, and contaminants and to either fully mixed conditions or one-dimensional (longitudinal) spatial gradients. A first-order removal rate law is used. The model contains methods for estimating removal rates. PREWet has been developed for interactive, user friendly, PC application.

PREWet was tested against TP, TN, and ISS data obtained from the CRW. The model performed well for this application where *RE* predictions were within 15 percent of *RE* estimates obtained for a 3-year period of observations. Similar long-term data sets were not available from the CRW for testing the other model variables (i.e., BOD, TCB, and contaminants). However, information from the literature was used to make recommendations for estimating rate coefficients needed to compute *RE* for each of the constituents. As with any new model, additional applications are needed to gain a better understanding of the model's strengths and limitations.

Acknowledgments

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Passaic River Flood Diversion Tunnel Water Quality Model Study

by
Barry W. Bunch[1]

Introduction

The Passaic River drains 935 square miles of northern New Jersey and southern New York and empties into Newark Bay, which is part of the New York/New Jersey harbor system, Figure 1. The Passaic and Hackensack rivers are the only freshwater flows into northern Newark Bay. The Passaic River basin is heavily developed and is susceptible to destructive flooding during storm events. A diversion tunnel has been proposed as a means of alleviating the flooding in the lower portion of the Passaic basin by diverting floodwaters from the upper basin directly to Newark Bay. Floodwaters would enter the tunnel at two locations in the upper basin and be discharged at an outlet structure located between the mouths of the Passaic and Hackensack rivers. Total tunnel length would be 20.1 miles, and the tunnel would have a diameter of 42 ft. Due to construction requirements, the proposed tunnel will have a profile of an inverted siphon with a volume of 2.65×10^6 m^3 below sea level. Under one set of operating conditions, this portion of the tunnel would remain filled with floodwaters after a flood event.

The objective of this study was to determine what impact operation of the Passaic River Flood Diversion Tunnel would have upon living resources in Newark Bay, specifically finfish and shellfish. With the aid of the Marine Fisheries Service, the parameters impacting these living resources most were identified as temperature, dissolved oxygen, and salinity. Information on these parameters and the impact of tunnel operation upon them would be used by others to determine the effect upon living resources.

Models

Two models were required to conduct this study. One for hydrodynamics, Computational Hydrodynamics in 3 Dimensions (CH3D), and one for water quality, CE-QUAL-ICM (ICM). CH3D is a finite difference hydrodynamic model capable of simulating temperature, salinity, and density at cell centers. CH3D uses a curvilinear grid that captures the physical features of a system. CE-QUAL-ICM was developed for the Chesapeake Bay Eutrophication study (Cerco and Cole 1994) and has subsequently been applied to numerous other studies. ICM stands for "Integrated Compartment Model." CE-QUAL-ICM is a multi-dimensional, finite volume water quality model with a suite of 22 independently activated state variables. CE-QUAL-ICM interfaces with a fully predictive sediment submodel. ICM is a very flexible model that can be configured for each study. A description of ICM state variables and kinetics can be found in Cerco and Cole (1994). Specific information of ICM kinetics for the Passaic River study can be found in Cerco and Bunch (In Preparation).

Hydrodynamic information required by ICM consists of cell surface areas, cell lengths, cell volumes, flow face areas, horizontal flows, and vertical flows. This information is generated with CH3D and stored in a file. Because of this arrangement, hydrodynamic information for

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

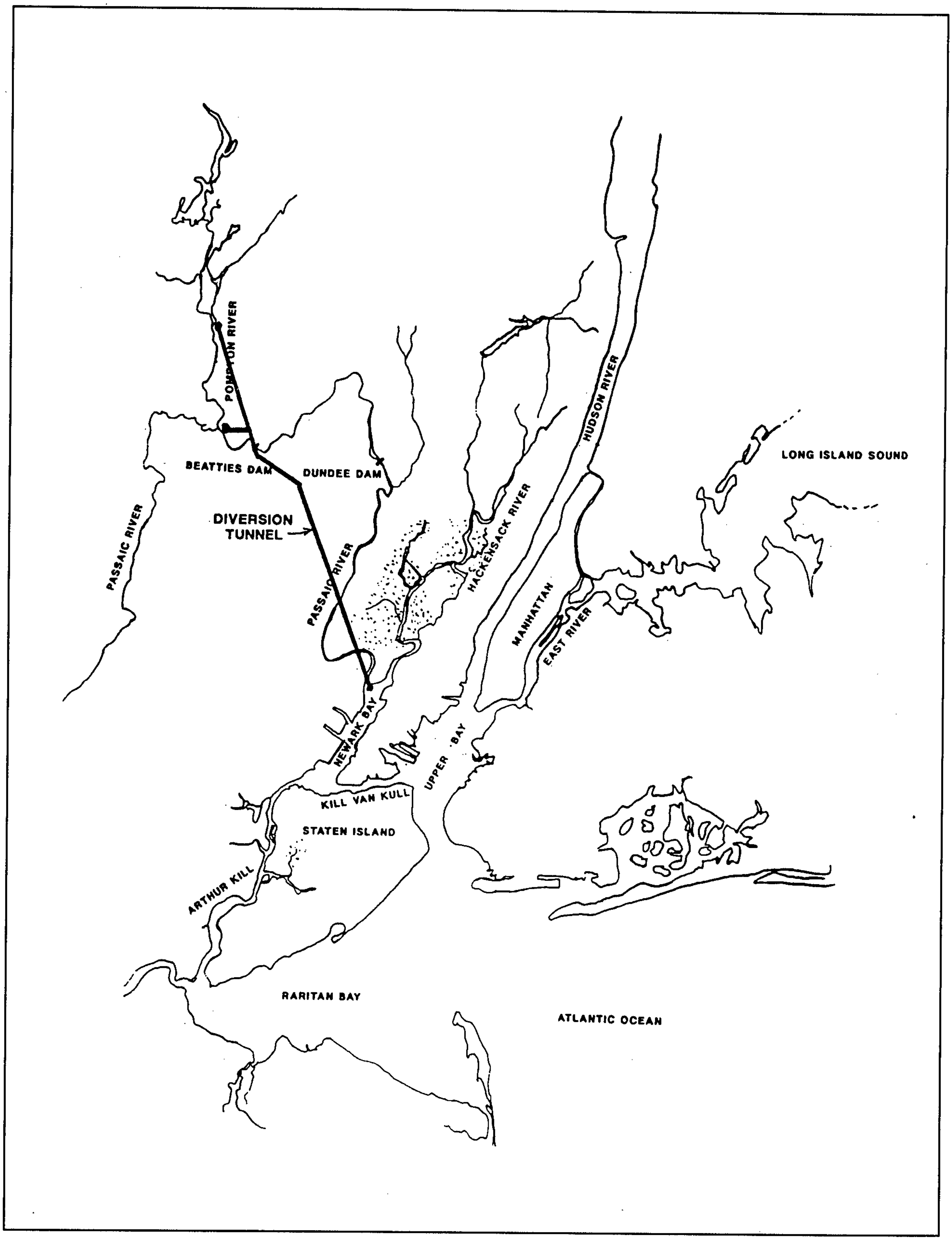

Figure 1. Passaic River study area showing diversion tunnel route

ICM need only be generated once. Multiple ICM runs can be made using "stored" hydrodynamic information that decreases computational requirements and run time.

Water Quality Model

Grid

The water quality grid for this study was a one-to-one overlay of a portion of the hydrodynamic grid. The region covered by the hydrodynamic grid extended from the Hudson River Fall line to the New York Bight. The size of this grid was necessitated by the availability of information for hydrodynamic boundary conditions and was far beyond the farthest extent of any impacts arising from tunnel operation. For computational efficiency and in order to decrease the amount of input data required, a smaller portion of this grid, Figure 2, was used for water quality simulations. Though smaller than the CH3D grid, the ICM grid was still extensive enough that the Hudson River boundary and the open ocean boundaries had negligible impact upon the tunnel outlet site and would therefore not bias the results. This was documented in numerous simulations run prior to locating the boundaries at these locations. The ICM grid, after moving the boundaries, had a total of 6,815 cells in five layers of 1,363 cells each. The total number of cells in the ICM grid was approximately 40 percent of the total in the CH3D grid.

State Variables

In this study, the question to be answered was what effect the tunnel operation would have upon water quality. In order to answer this question the following state variables were modeled: temperature, salinity, dissolved oxygen (DO), chemical oxygen demand (COD), and ultimate biochemical oxygen demand (BODu).

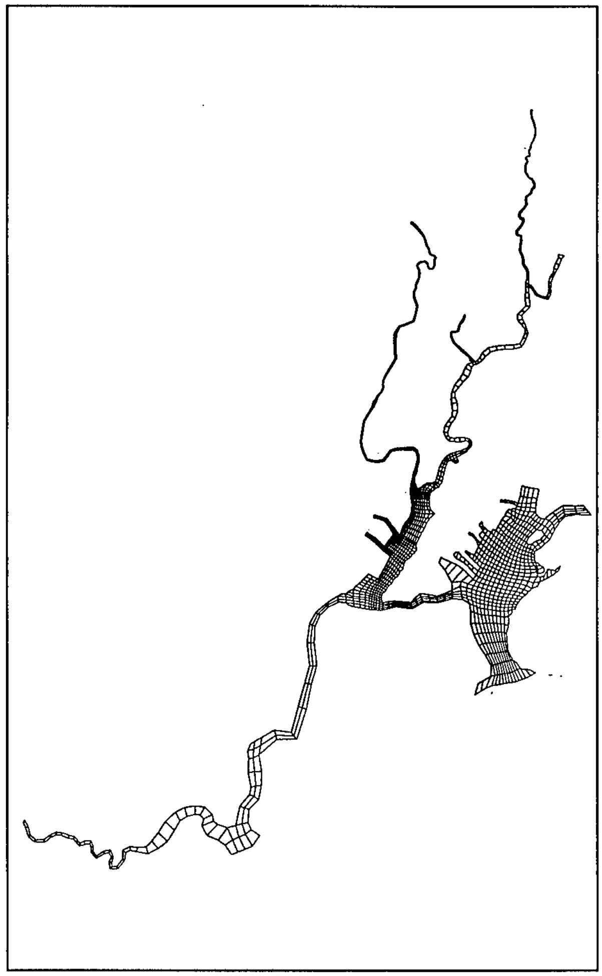

Figure 2. Water quality grid

Temperature and salinity were modeled because of the short-term changes in both arising from tunnel operation that could be detrimental to living resources. Both parameters are already calculated in CH3D as part of its hydrodynamics computations. Output from CH3D is adequate for determination of tunnel operation upon these two parameters. These variables were included in the ICM suite of state variables because of the effect that they have upon other variables. Both temperature and salinity affect dissolved oxygen solubility. Temperature is also modeled because of the important effect it has upon biochemical reactions. Computation of salinity in ICM allows for verification that ICM and CH3D are linked properly and transport material in a like manner.

Dissolved oxygen was modeled because it is essential for higher order life forms. Transient effects upon DO could have serious impacts upon all aquatic life in the area. The extent of any impact upon dissolved oxygen arising from tunnel operations could be ascertained. DO also plays an important role in many chemical reactions and processes that can affect water quality. BODu and COD were included because of the effect that both have upon DO. BODu is a measure of the oxygen requirement to biologically decay all organics in water. In this study, COD was used to represent the oxygen demand required to oxidize reduced substances that might be generated by tunnel operation. COD can also be generated when DO levels in the water column are insufficient to satisfy benthic oxygen demands.

Model Applications

Application details of CH3D are covered in the hydrodynamic report and are beyond the scope of this paper. Application of ICM can be separated into three distinct operations: data collection and assembly, ICM calibration, tunnel scenario testing. Each of these is covered below.

Data Collection and Assembly

WES contracted the field sampling program for this study to Stevens Institute of Technology. Stevens personnel made in situ measurements of temperature, salinity, and dissolved oxygen at 30 stations throughout the study area at monthly intervals during the period July to September 1994. During each sampling event, grab samples were collected at 25 of those stations and analyzed for 5-day BOD (BOD5) and BODu. Data collected by Stevens was augmented with dissolved oxygen and temperature measurements made by the National Marine Fisheries Service during the period May 1993 to May 1994 and with data obtained from the New York Department of Environmental Protection.

Stevens also compiled a database of point source dischargers for the entire New York-New Jersey Harbor area. From this list, the most substantial dischargers in the study area were identified and their discharge records for the year preceding the calibration period obtained from the New Jersey Department of the Environment. Information on point source dischargers obtained by Stevens was augmented with data contained in reports generated for the U.S. Environmental Protection Agency (EPA) on waste loads to the New York-New Jersey harbor system (HydroQual 1991).

Combined Sewer Overflows (CSOs) generate a significant nonpoint source loading to this system. Loads due to CSOs were developed for each watershed discharging into the study area using daily precipitation records and information contained in the New York/New Jersey Section 208 report (Hazen and Sawyer 1978). Nonpoint source loads were applied as distributed loads to all edges of stream cells below the fall lines. Freshwater inflows representing point source and nonpoint source flows were included in CH3D because of the magnitude of the flows and the effect that they have upon circulation in certain areas of the system.

Ocean boundary conditions were set using data collected by Stevens Institute and the New York City Department of Environmental Protection. Fall line BODu boundary conditions were set using a relationship developed for each stream based upon flow and observed BOD. Fall line loadings were calculated as the product of the calculated fall line BOD and flow.

ICM Calibration

ICM was calibrated for the period July 1, 1994, through September 11, 1994. Loadings to the system and boundary conditions were generated as indicated above. Artificial initial conditions corresponding to July 1 were used for calibration. They were generated by looping the model with constant inputs until equilibrium was reached. ICM output was compared with data collected during the July to September field study using time series plots, longitudinal transect plots, and predicted/observed scatter plots. Time series plots of model output and observed data for mid-Newark Bay are shown in Figure 3. The solid line represents ICM daily depth-averaged concentrations, and the shaded region indicates maximum and minimum daily ICM values. Circles represent mean observed concentrations, and the vertical bar indicates the range of observations.

ICM predictions for the overall system were good. ICM performs well in Newark Bay below the proposed tunnel's outlet. It captures the minimum DO levels, which is important for living resources impact determination. ICM does an excellent job on temperature predictions, and the salinity predictions are within the range of daily observations.

ICM Scenarios

A matrix of scenarios were developed to assess impacts of tunnel operation. Scenarios consisted of three tunnel configurations: no tunnel, dry tunnel, wet tunnel. The no tunnel configuration simulated future conditions as if the tunnel were not constructed and served as a point of comparison for the other two configurations. In the dry tunnel configuration, the below-sea-level portion of the tunnel is pumped dry after each flood. In the wet tunnel configuration, the tunnel contains floodwater from the preceding flood, which is forced out by the next flood. Since the tunnel was designed to operate on floods with periods of 2 years or greater, the water in the tunnel would have adequate time for extensive water quality degradation. Floodwater BODu levels in excess of DO levels could result in anoxic conditions leading to production of reduced substances and methane. It was feared that this situation would severely degrade DO levels in the receiving waters of Newark Bay when this water was discharged from the tunnel by the next flood.

The effects of prolonged storage on floodwaters was determined by incubating samples of Passaic River water for 6 months under conditions similar to those expected in the tunnel. DO levels were measured daily, and samples were withdrawn periodically for determination of BOD. Gas production was monitored to observe if hydrogen sulfide or methane were generated by to anaerobic decay. Experimental results indicated that biological decay continued until DO reached 1.5 mg/L, then ceased. BODu levels were slightly higher than DO levels at the end of the experiments, and no gas production or immediate oxygen demand was observed. Based upon these results, it is reasonable to expect that DO and BOD in the tunnel eventually would, under worst case conditions, reach 0 mg/L.

The incubation results corroborated conclusions reached after comparing observed DO and BODu levels of the upper Passaic River. Historical observations indicated that mean fall-line DO and BODu levels are approximately equal.

Flood conditions resulting from three storm events were simulated for each tunnel configuration: 2-year storm, 25-year storm, 100-year storm. The 2-year storm flood is the minimum flood that will cause tunnel operation. All scenarios were run for February and July mean meteorological conditions. Newark Bay temperatures are coldest (1 °C) in February and warmest (28 °C) in July. These months provide worst case conditions for temperature shock to living resources resulting from a discharge of 12.7 °C tunnel water. July also provides the lowest receiving water DO and less capability to handle anoxic tunnel discharges.

Scenarios had durations of 27 days. The first 13 days were model "spinup," which allowed the model to equilibrate with scenario loads and boundary conditions. Spinup was followed by a 14-day storm simulation.

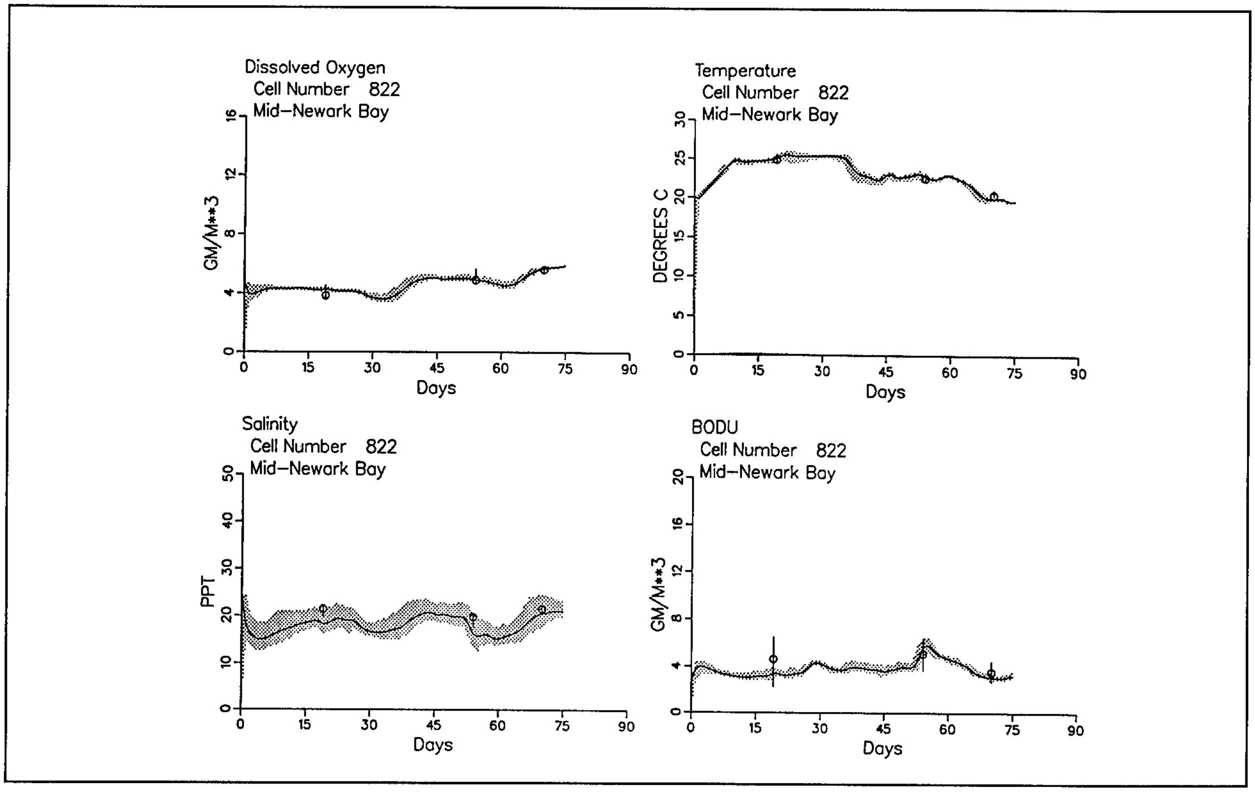

Figure 3. ICM calibration results

Scenario Results

A total of 18 scenarios were run. Results from wet and dry tunnel simulations were compared with base (no tunnel) simulations for the same conditions. Any differences in the simulations with the tunnel and the base simulation were attributed to tunnel impact. DO results for four scenarios are shown in Figures 4-7. Solid lines indicate base (no tunnel) conditions, and dashed lines indicate wet or dry tunnel conditions.

Results from the dry tunnel simulations indicated DO concentrations increased in upper Newark Bay with the tunnel in place. This occurred because the tunnel was placing oxygen saturated floodwaters directly into upper Newark Bay. Without the tunnel, the DO of these floodwaters would be decreased by CBOD and SOD as it traveled down the Passaic River.

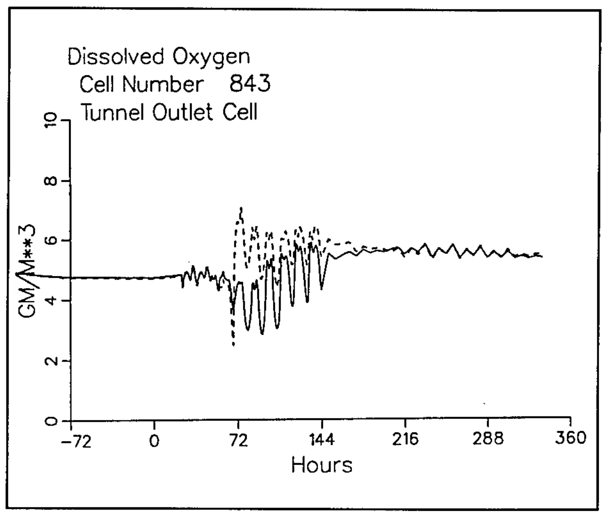

Figure 4. 2 year July wet tunnel scenario

In the wet tunnel scenarios, a short-term decrease in DO occurs near the tunnel outlet. For the 2-year, 25-year, and 100-year storms,

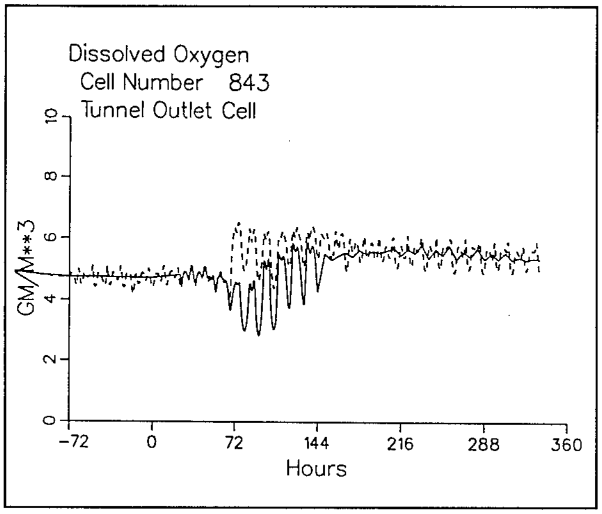

Figure 5. 2 year July dry tunnel scenario

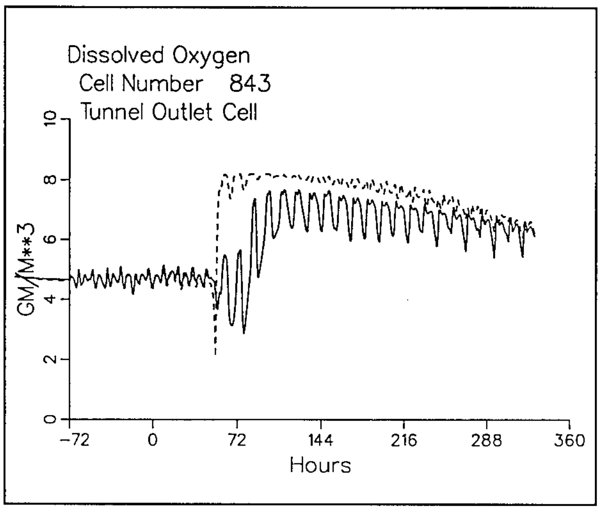

Figure 6. 100 year July wet tunnel scenario

the minimum DO next to the tunnel outlet was approximately 2 mg/L. The minimum DO predicted at the tunnel outlet for the 2-year storm was only slightly lower than the minimum DO predicted without the tunnel. Results indicated that the duration of the DO decrease near the tunnel outlet for the 2-year flood was longer than that of the 25-year and 100-year floods. This is because a longer period is required to purge the tunnel with the 2-year flood. Once

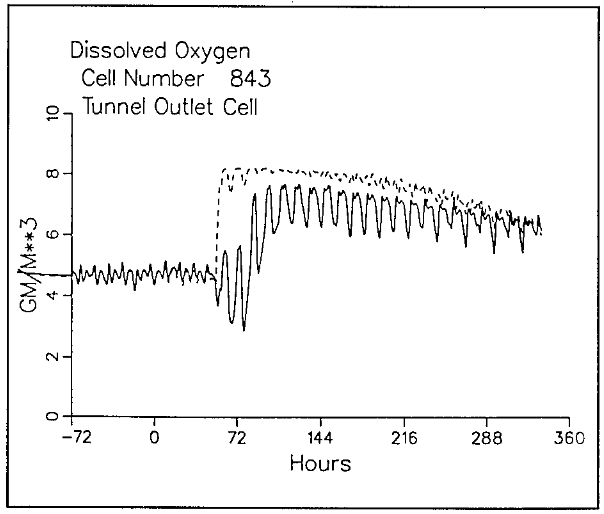

Figure 7. 100 year July wet tunnel scenario

the tunnel has been purged of the 0 mg/L DO water, the tunnel conveyed DO-saturated upper Passaic River water, which resulted in an increase in DO concentrations near the outlet.

Conclusions

Scenario results indicated that DO impacts resulting from discharge of DO-depleted water were limited in duration and extent. Dilution and reaeration combined to limit tunnel impacts. DO depleted tunnel water mixed quickly with the receiving waters of Newark Bay. Mixing was enhanced by DO-saturated water from the tunnel and the flood flows on the Passaic and Hackensack rivers. Reaeration, which is proportional to DO depletion, rapidly replenishes the deficit caused by the DO-depleted tunnel waters.

Acknowledgments

This project was funded by the Passaic River Division, U.S. Army Corps of Engineers (USACE). Permission to publish this work has been granted by Headquarters, USACE.

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Simulation of Richard B. Russell and J. Strom Thurmond Reservoirs for Pump-Storage Using CE-QUAL-W2

by

Dorothy H. Tillman[1] *and Thomas M. Cole*[1]

Introduction

Richard B. Russell (RBR) is the most recent of a series of projects built on the Savannah River which are managed by the Savannah District (SAS). Construction of the project was begun in 1985. It is located between Hartwell Lake and J. Strom Thurmond (JST) Lake on the Georgia, South Carolina border (Figure 1). RBR is a pumped-storage project and operates in tandem with JST. During the pump-storage operation, four variable speed pump-turbines remove cold water from the hypolimnetic waters of RBR and return a mixture of recently generated and warmer surface water from the RBR tailwater. Stored water in RBR is later used to generate power during peak demands (e.g., week days).

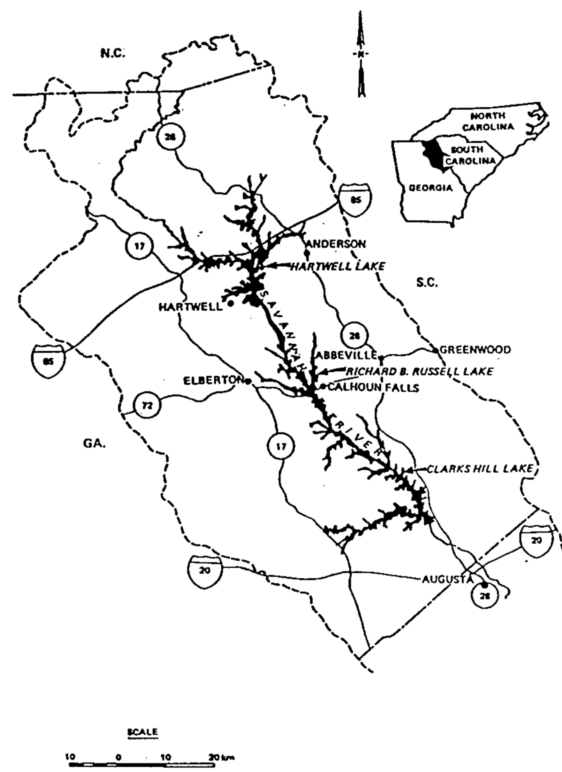

Figure 1. Project location and vicinity map

As the pump-storage operation of Richard B. Russell (RBR) starts gearing up for commercial operations, concerns over environmental impacts to the thermal and dissolved oxygen regimes in the RBR lake and tailwater remain for the Savannah District. Both RBR and JST reservoirs will be significantly affected by the commercial operations. In the early 1970's and 1980's, pre-impoundment studies were conducted to provide the best estimates of the future water quality conditions in the RBR forebay at the time. Since these studies, improved numerical modeling methodologies became available. In an effort to refine estimates of the thermal and dissolved oxygen predictions in the RBR forebay and tailwater from previous studies, CE-QUAL-W2, a two-dimensional, laterally averaged hydrodynamic and water quality model was applied to both systems. Results for the thermal calibrations for both reservoirs will be presented in this paper.

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Study Objective

The Environmental and Hydraulics Laboratories at the Waterways Experiment Station (WES) are assisting the Savannah District in Studying the tandem operation of RBR and JST. The study will refine previous estimates from earlier studies to determine the effects of pump-storage operation on the water movement, temperature, and dissolved oxygen (DO) in the forebay and tailwater of RBR for varying hydrologic conditions and with oxygen injection.

General Modeling Approach

CE-QUAL-W2, the two-dimensional (laterally averaged) hydrodynamic and water quality model, is being applied to both RBR and JST for temperature, DO, algae, soluble reactive phosphorus, ammonia, nitrate-nitrate, dissolved organic matter (DOC), and particulate organic matter (POC). Many modifications to CE-QUAL-W2 will be necessary to represent the system and to meet study needs. A major modification to the code which has been completed was to dynamically link RBR and JST for pumpback. This entailed modifying the code to be able to model two reservoirs at once in order to simulate the tandem operation (bi-directional flow of water). Modifications to include the effect of entrainment/mixing of the pumpback jet have not been completed as yet, but will be in the final version of the code. The equation representing the pumpback jet will be a function of the discharge rate, outlet geometry, and stratification pattern and will be used to determine boundary conditions at the dam during pumpback operations. Finally the code will be modified to include the oxygen exchange of the diffuser systems. Empirical relationships developed from field observations will be added to the code to represent this process.

The model will be calibrated for two extreme water years, a dry and wet year. Scenario testing on the proposed pump-storage operation for a dry, average, and wet year will be simulated with an improved tailwater channel to estimate impacts on RBR and JST water movement, temperature, and DO.

Status of Study

Due to the delay of Phase 3, monitoring of commercial operations at RBR, the study has been suspended until start up of Phase 3 begins. Presently, the status of the study is:

- CE-QUAL-W2 has been modified to link RBR and JST.
- RBR and JST have been thermally calibrated for 1988 and 1994.
- RBR and JST are being initially calibrated for water quality for both years.
- Modifications are being made to the code to include entrainment/mixing of the pumpback jet and the oxygen diffuser systems.

Thermal Calibration/Verification

CE-QUAL-W2 was applied to both reservoirs for two extreme water years, a dry year (1988) and a wet year (1994) which included pump-storage operation. Flow and water quality data for the study years were provided by the Trotter Shoals Laboratory personnel. Bathymetry data were provided by the Savannah District.

Bathymetry

CE-QUAL-W2 conceptualizes the reservoirs as branches which are discretized into longitudinal segments and vertical layers that can vary in length and size. The RBR/JST tandem system was segmented into ten branches with a total of 166 segments and 59 layers. Branch one represented the mainstem of RBR and branch six represented the mainstem of JST. The remaining branches represented tributaries of both reservoirs. A comparison of computed volume-elevation curves and U.S. Army Corps of Engineers (USACE) data is given in Figure 2 for both reservoirs.

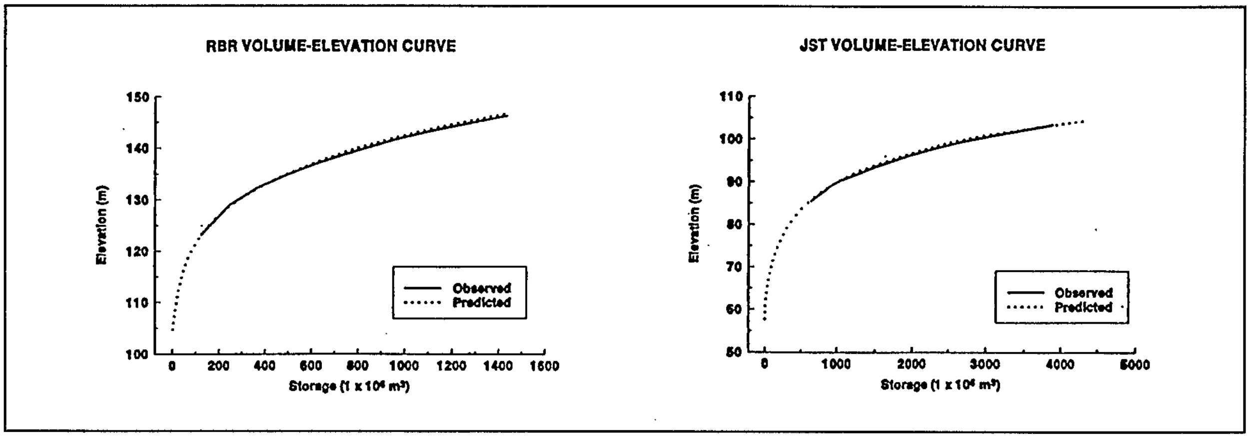

Figure 2. Volume-elevation curves for RBR and JST

Water Surface Elevation

Adjustments to bathymetry data and elevation of the bottom datum were made to correct for water imbalances to the system. In addition, a distributed tributary was modeled for the mainstem branches to account for flow from nonpoint sources. Figure 3 shows the comparison of computed versus the observed water surface elevations for 1988 and 1994 for both reservoirs. Differences in computed and observed stages were less than 0.5 meters (m) for the simulation periods.

1988 Temperature Calibration

Once satisfactory results for the water balance in both reservoirs were obtained, temperature calibration was begun. During temperature calibration, the boundary inflow temperatures for RBR were set to Hartwell Reservoir release temperatures and for JST to computed RBR release temperatures. Tributary boundary inflow temperatures were estimated using the response temperature calculator (RTC) developed by J. E. Edinger Associates (1984). The RTC program uses meteorological data to estimate the inflow temperatures. Meteorological conditions measured at Athens, Georgia were used to represent weather conditions at RBR, and meteorological data measured at Augusta, Georgia represented weather conditions at JST.

Table 1 shows the values of all coefficients that affect temperature computations and their final value. These coefficients were initially set to the recommended values from the CE-QUAL-W2 user manual (Cole and Buchak 1995). Temperature predictions appeared to be most sensitive to changes in the wind sheltering coefficient for both reservoirs.

Observed and computed temperature profiles are shown for stations in RBR and JST closest to the dams (Figures 4 and 5, respectively). Close agreement can be seen for both reservoirs indicating that the bathymetry and boundary conditions are representing the system well. Discrepancies between observed and computed for most dates are less than 1 °C and are probably due to using meteorological data that were not measured at the projects. For the stations closest to the dam, meteorological data that were not measured at the projects. For the stations closest to the dam, meteorological data have more influence on computed water temperatures than inflow temperatures because these stations are usually the greatest distance from the boundary.

1994 Temperature Verification

During 1994 temperature verification, the boundary inflow temperatures to the mainstem branches were set as discussed above. Tributary

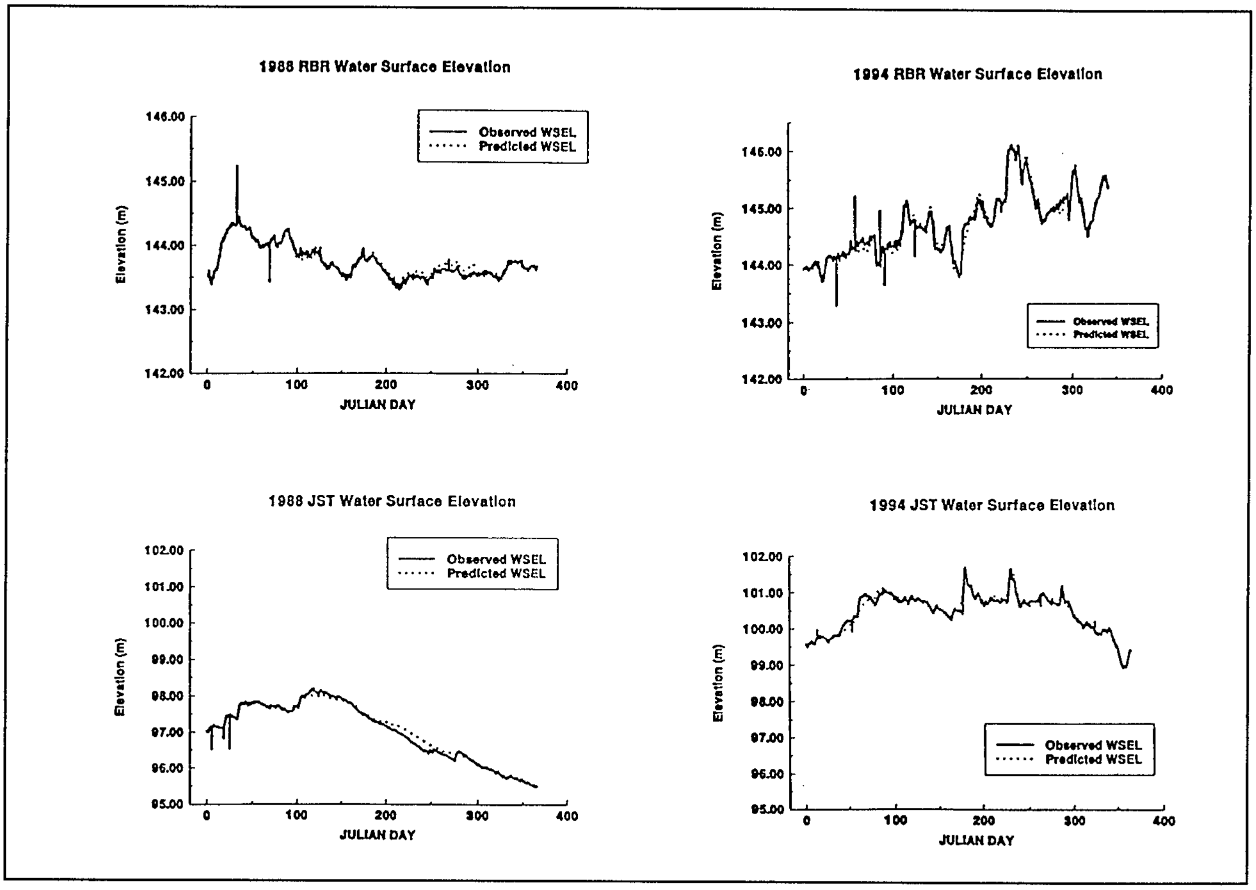

Figure 3. Observed and computed water surface elevations of RBR and JST for 1988 and 1994

Table 1
Final Values for Coefficients Adjusted During Temperature Calibration

Coefficient	Value
Horizontal eddy viscosity	$1.0\ m^2\ s^{-1}$
Minimum vertical eddy viscosity	$1.4 \times 10^{-7}\ m^2\ s^{-1}$
Horizontal eddy diffusivity	$1.0\ m^2\ s^{-1}$
Minimum vertical eddy diffusivity	$1.4 \times 10^{-6}\ m^2\ s^{-1}$
Bottom frictional resistance	$70.0\ m^{0.5}\ s^{-1}$
Fraction of solar radiation absorbed at the water surface	0.65
Light Extinction - water	$0.65\ m^{-1}$
Wind sheltering coefficient - RBR	0.90
Wind sheltering coefficient - JST	1.00

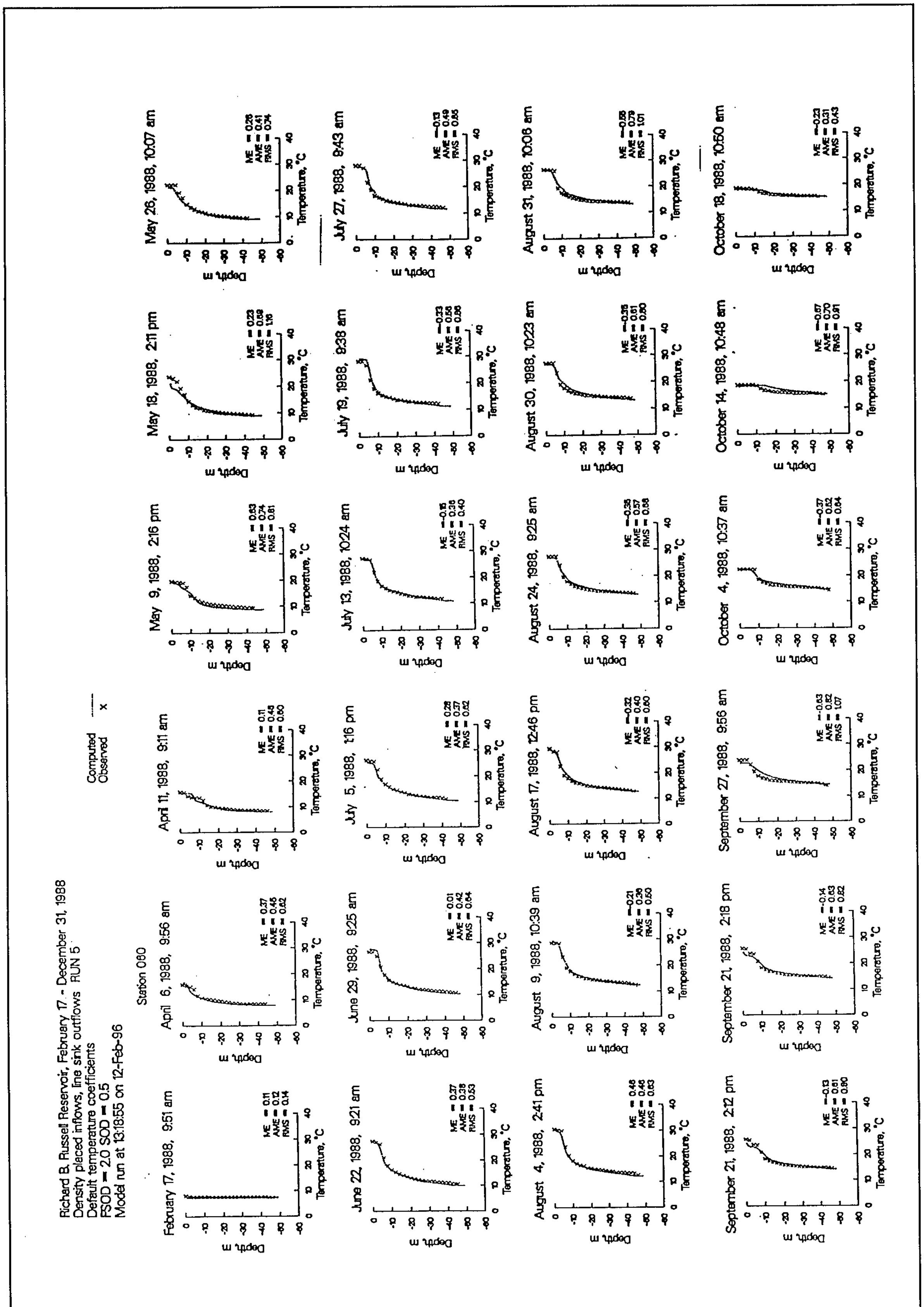

Figure 4. RBR 1988 temperature results

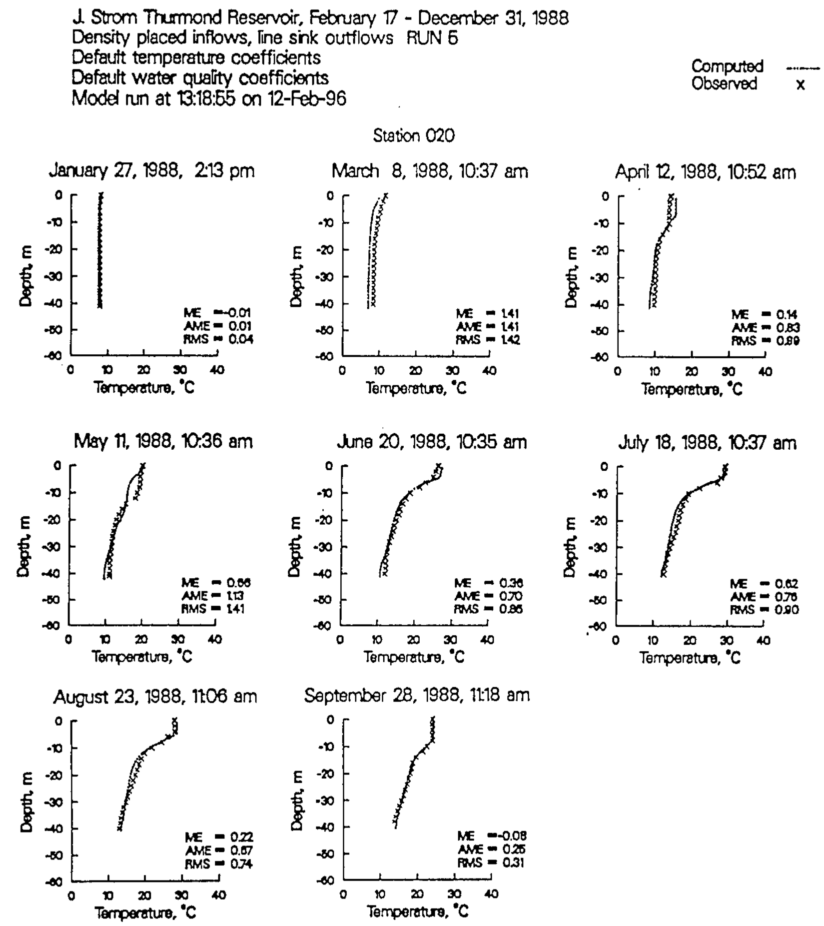

Figure 5. JST 1988 temperature results

boundary inflow temperatures were again estimated using the RTC. Coefficients were set to the values used in 1988.

Observed and computed temperature profiles for RBR and JST are shown for the same stations discussed above in Figures 6 and 7, respectively. Close agreement can be seen for both reservoirs, and discrepancies again are less than 1 °C for most dates.

Conclusions

At this point in the study, conclusions and recommendations cannot be made to SAS until all modifications have been made to CE-QUAL-W2, and all simulations have been completed. However, temperature calibration is a very important milestone to complete in the water quality calibration process since most water quality constituents are temperature dependent. Even without the inclusion of the entrainment routine, temperature results were in very good agreement for both reservoirs, especially in 1994 results where pumpback is being simulated. This may indicate that there is not as much entrainment as was thought, or entrainment may be more important during the water quality calibrations.

Acknowledgments

The authors greatly thank the Trotter Shoals personnel for providing the observed data necessary for the study. Savannah District provided the digitized bathymetry data which Mr. Mike Schneider of the Hydraulics Laboratory (WES) used in generating the grid for both reservoirs. Permission was granted by the Chief of Engineers to publish this information.

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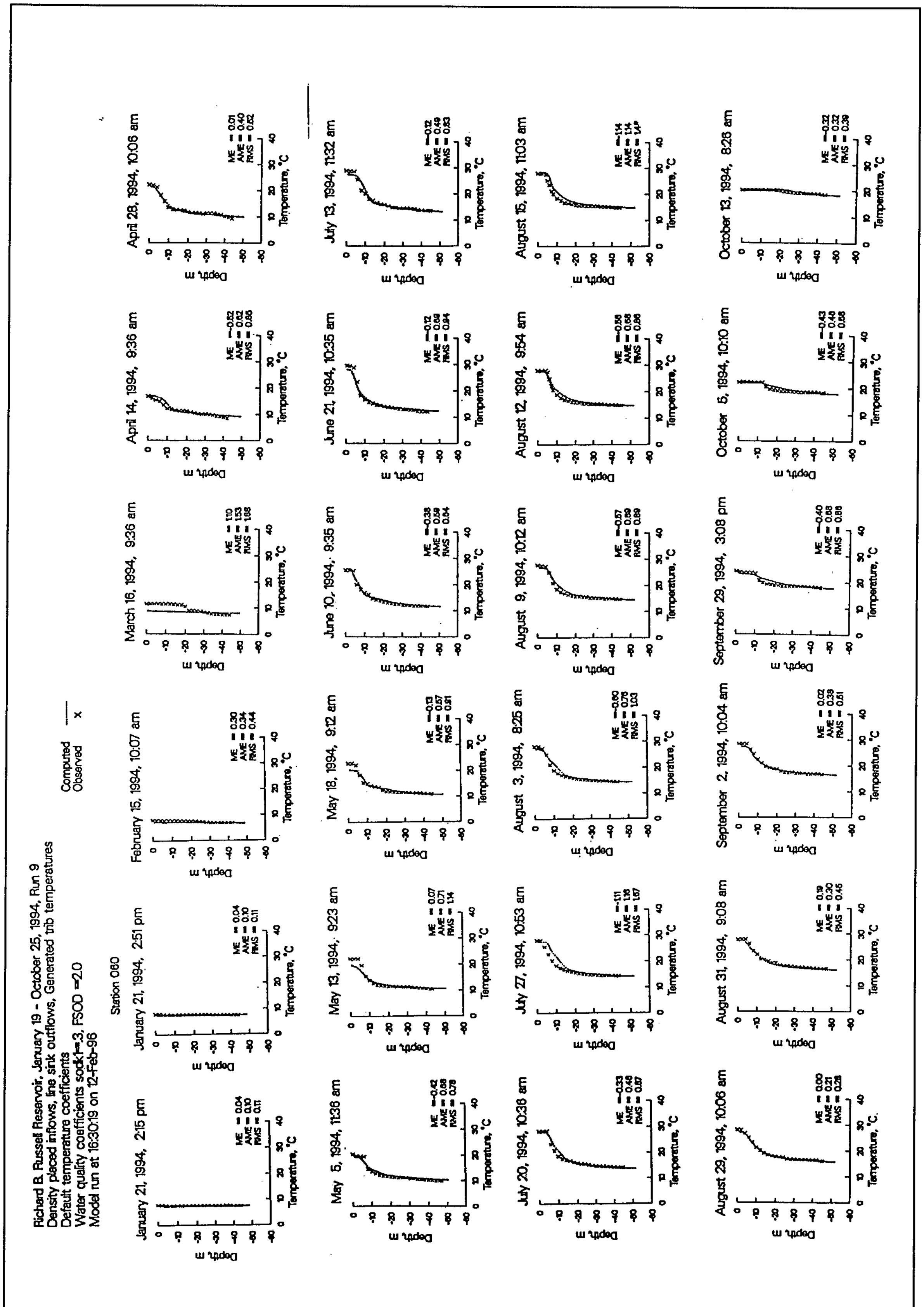

Figure 6. RBR 1994 temperature results

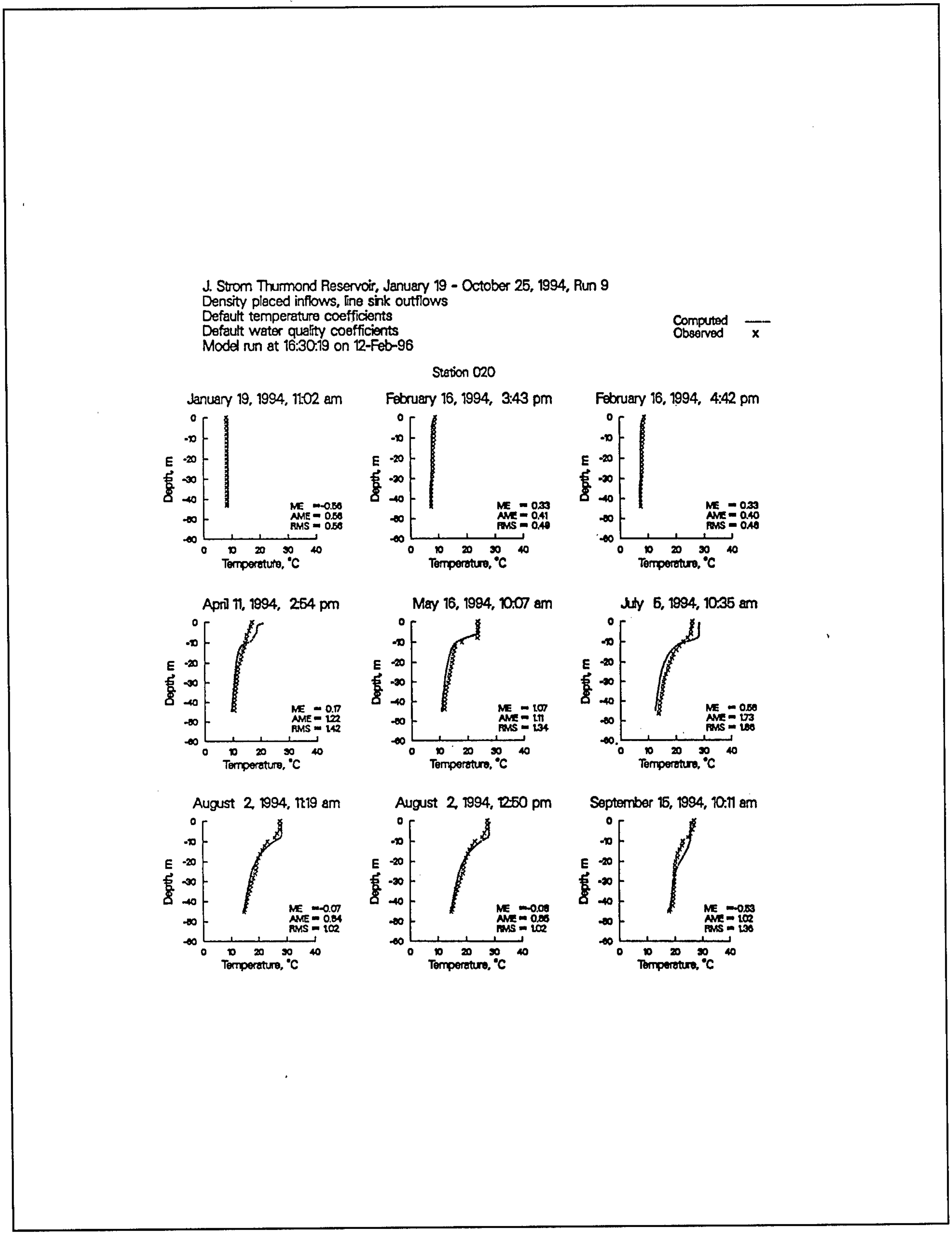

Figure 7.   JST 1994 temperature results

Submersed Aquatic Macrophytes on Sediment and Water Quality

by
John W. Barko[1] *and William F. James*[1]

Summary

Submersed aquatic macrophytes rely primarily on sediment as a direct source of nitrogen (N) phosphorus (P) and micronutrients for their nutrition. The availability of these elements in sediments is affected markedly by sediment type, but can also be influenced by macrophyte growth. Results of a variety of studies have indicated that macrophyte species, even with relatively diminutive root systems, can significantly deplete sediment N and P pools. From fertilization experiments involving sediments from a variety of locations, macrophyte growth on nutritionally-depleted sediments has been shown to be limited by the availability of sediment N, but not P.

Through uptake from the sediment, aquatic plants transport nutrients directly to the overlying water column. Elevated pH, associated with plant photosynthesis, further enhances nutrient (phosphorus) flux from sediments. Water circulation induced by diel heating and cooling of surface water in aquatic plant beds facilitates nutrient exchanges with the adjacent open water of aquatic systems. These processes can result in enhanced phytoplankton (chlorophyll) production and deteriorated water quality conditions. However, in shallow-high energy environments, these potential negative effects on water quality may be overshadowed by the ability of aquatic macrophytes to moderate current and wave energies, thereby reducing sediment resuspension, turbidity, and concentrations of suspended particulate materials.

The vigor of submersed macrophyte beds is likely maintained by nominal inputs of sediment providing a nutritional subsidy. However, excessive inputs of sediment can result in macrophyte declines due to burial or to unfavorable irradiance conditions. Hydrologic factors and watershed activities that influence seasonal dynamics and magnitudes of sediment transport in aquatic systems need to be evaluated within the context of their effects on submersed macrophyte growth. The effects (favorable versus unfavorable) of aquatic plants on water quality conditions in aquatic systems need to be considered within the context of basin morphometry, hydrology, and local climate.

Introduction

Submersed macrophytes are unique among rooted aquatic vegetation because they link the sediment with overlying water. This linkage is responsible for great complexities in nutrition, and has potentially important implications for nutrient cycling. It has become clear that, in addition to serving as a base for physical attachment, sediments also provide a source of nutrient supply to submersed macrophytes. Sediment composition exerts an important influence on macrophyte productivity and species composition. Recent attention among aquatic macrophyte ecologists has focused on interactions between aquatic macrophyte growth and sediment nutrient status. Attention is presently being focused also on specific processes in the littoral zone affecting sediment nutrient dynamics.

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During the 1970's, accelerated eutrophication of freshwater systems due to excessive phosphorus loadings in both North America and Europe lead to interest in the role of rooted submersed macrophytes in the nutritional economy of aquatic systems. At that time it was unclear whether submersed macrophytes functioned as sources or sinks for phosphorus. Given potential access by these plants to nutrients in both the water column (foliar uptake) and sediments (root uptake), it has been necessary to evaluate quantitatively nutrient source-sink relationships, involving both soluble and particulate nutrient fractions. Evaluations have necessitated combinations of laboratory and field studies. Results of these studies conducted by the USACE Waterways Experiment Station (WES) are highlighted here, as reported during the Austin Peay State University 1995 symposium, "Water Quality and Exotic Biota."

Macrophyte Nutrition and Growth

For many years controversy has persisted regarding the role of roots versus shoots and sediment versus open water in the nutrition of submersed aquatic macrophytes (reviewed by Sculthorpe 1967; Denny 1980; Smart and Barko 1985; Agami and Waisel 1986; Barko et al. 1986, Barko et al. 1991). Quantification of the relative contribution of sediment and water to nutrient uptake by submersed macrophytes remains critical to improved understanding of littoral nutrient cycling and littoral-pelagic nutrient exchanges. Based on a variety of information sources (references above and personal knowledge), a generalized synthesis of sources of nutrient uptake by rooted submersed macrophytes is provided here.

Phosphorus and nitrogen have been studied most extensively, and for these nutrients sediment is the primary source for uptake. Sediment appears to be the principal site for uptake of iron, manganese, and micronutrients as well (Barko and Smart 1986). These latter elements tend to coprecipitate and are usually present in extremely low concentrations in oxygenated surface waters. Dissolution products of relatively abundant salts are taken up principally from the open water. Among these ions, potassium and calcium are potentially most important in affecting submersed macrophyte growth. Potassium can be obtained from the sediment, but is taken up by submersed macrophytes most abundantly from the open water (Barko 1982; Huebert and Gorham 1983; Barko et al. 1988). Under some conditions this element may be exchanged by submersed macrophyte roots for ammonium ions in sediment (Barko et al. 1988). Calcium is a component of the carbonate system and plays an important role in photosynthetic bicarbonate utilization (Lowenhaupt 1956; Smart and Barko 1986).

Given the significance of sediment in supplying N and P to submersed macrophytes, it has been important to evaluate the effects of macrophyte growth on sediment nutrient availability. Evidence indicates that rooted submersed macrophytes, even with relatively diminutive root systems, are capable of significantly depleting pools of N and P in sediments (Prentki 1979; Trisal and Kaul 1983; Short 1983; Carignan 1985; Barko et al. 1988; Chen and Barko 1988). Under many circumstances, depletion of sediment nutrient pools by aquatic macrophytes can be expected to influence macrophyte growth. From fertilization experiments involving sediments from a variety of locations, macrophyte growth on nutritionally-depleted sediments has been shown to be limited by the availability of sediment N, but not P (Barko et al. 1991).

Macrophytes on Sediment P Release

Great attention to the P economy of submersed aquatic macrophytes reflects the unparalleled importance of this nutrient in the eutrophication of lacustrine systems (Schindler 1974, 1977). Given the demonstrated capacity of submersed macrophytes to take up P directly from sediments, vegetation of the littoral zone needs to be viewed as a direct source of this nutrient to the water column (Barko and Smart 1980; Carignan and Kalff 1980; Smith and Adams 1986; Moeller and Wetzel 1988; Barko et al. 1991). In addition, submersed macrophytes can influence P dynamics in aquatic systems by altering the pH of the water column (James and Barko 1991).

The process of photosynthesis in submersed macrophytes results in oftentimes dramatic increases in the pH of the surrounding water column. For example, studies conducted with the U.S. Geological Survey in *Hydrilla* beds in the Potomac River revealed diel changes in pH between values of about 7.0 and 10.0 (Carter et al. 1988). Since pH values are logarithmic, these changes indicate large (nearly one thousand-fold) variations daily in hydroxyl and hydrogen ion concentrations. From concurrent studies conducted at Eau Galle reservoir in Wisconsin, it became apparent that elevated pH beyond about 9.0 can result in significant increases in rates of phosphorus release from surficial sediments (James and Barko 1991). In simple terms, the mechanism for release appears to involve chemical exchange of hydroxyl ions for phosphate ions in sediment.

Phosphorus released from sediments under conditions of elevated pH can account for a significant portion of the total mass of this element loaded internally in aquatic systems. For example, (James and Barko 1994) demonstrated that about 25 percent of the total seasonal internal phosphorus load into Eau Galle reservoir derived from littoral sediments. Phosphorus released from littoral sediments (i.e., within macrophyte beds) tends to be transported directly into the upper mixed layer of lakes and reservoirs (see below). Thus, contributions to the phosphorus economy of algal communities in surface waters can be significant. The seasonal periodicity of internal phosphorus loadings into the upper mixed layer of Eau Galle reservoir appears to influence not only the nutrient budget, but phytoplankton productivity and the vertical migratory behavior of phytoplankton populations as well (James et al. 1992).

Macrophytes on Hydraulic Circulation

On a daily basis, shallow near-shore regions of aquatic systems typically heat and cool more rapidly than deep open-water regions, due primarily to differences in mixed volume (Stefan et al. 1988). The presence of submersed macrophytes in shallow regions contributes to the development of thermal gradients in both the vertical and lateral planes, as foliage near the water surface converts solar irradiance to heat. Thermal gradients give rise to density gradients that promote hydraulic circulation. In Eau Galle Reservoir in Wisconsin and Guntersville Reservoir in Alabama, dye studies (James and Barko 1991, 1994) have been conducted for several years in combination with close-interval thermal monitoring in an attempt to evaluate the seasonal dynamics of convective transport phenomena. Owing to the eutrophic nature of these impoundments, studies have focused primarily on phosphorus transport. However, the results obtained from these reservoir studies apply to other dissolved constituents as well.

Implications of hydraulic circulation driven by convection are potentially far-reaching, since many kinds of dissolved constituents can be moved with water. Dissolved constituents may include contaminants or herbicides in addition to nutrients. As emphasized above, hydraulic transport from the littoral zone in combination with nutrient (P) release from sediments can contribute significantly to nutrient cycling in aquatic systems.

Macrophytes on Sediment Resuspension

In addition to effects on soluble constituents of the water column in aquatic systems, submersed macrophytes also play an important role in mediating the resuspension and transport of sediment and associated particulate constituents. Sediment resuspension and discharge of sediment downstream in Marsh Lake (Minnesota) were examined during 1991 and 1992 under a variety of wind conditions (James and Barko 1994b). Based on a theoretical wave model, nearly the entire sediment surface area of this reservoir (81-100%) can be disturbed by wave activity at wind velocities as low as 15 km/h blowing from any direction. However, as an apparent result of dense submersed macrophyte beds that in 1991 covered nearly the entire lake, measured sediment resuspension was much less frequent than expected from wave theory.

Critical thresholds of wind velocity required to resuspend sediment in Marsh Lake were much higher in 1991 than in 1992 when plants were essentially absent. The presence of dense submersed macrophyte populations in 1991 resulted in a much lower frequency of resuspension events than in 1992. In addition, discharge of resuspended sediment downstream was much less in 1991 when submersed macrophytes were abundant than in 1992 when macrophytes were absent.

In Marsh Lake submersed macrophyte populations appear to significantly influence water quality conditions (e.g., chlorophyll concentrations and turbidity) by mediating sediment resuspension. Reduced discharges of sediment from this system may result in water quality improvements downstream as well.

Sedimentation provides an important means of nutrient renewal to the littoral zone, and in large part may balance nutrient losses due to macrophyte uptake. Factors affecting sedimentation have been studied extensively in the open water (e.g., Hakanson, 1977; Kamp-Nielson and Hargrave, 1978), but to a much lesser extent in the littoral zone of lakes. Aquatic macrophyte beds serve as effective traps for inflowing dissolved and particulate materials (Wetzel, 1979; Patterson and Brown, 1979; Carpenter, 1981). Moeller and Wetzel (1988) have suggested that sedimentation of algae from macrophyte leaf surfaces may provide an important link for transfer of nutrients absorbed from the water (by algae) to the sediment surface. Similarly, it has been reported that under conditions of nutrient enrichment, decomposing filamentous algae can provide major inputs of N and P to sediment (Howard-Williams, 1981). The vigor of submersed macrophyte beds is likely maintained by nominal inputs of sediment providing a nutritional subsidy. However, excessive inputs of sediment can result in macrophyte declines due to burial or to unfavorable irradiance conditions.

Management Considerations

It is apparent that submersed aquatic macrophytes, through a variety of mechanisms, can have important influences on sediment and water quality in aquatic systems. The significance of these influences can be expected to vary with climate, basin geomorphology, macrophyte density and species composition. In systems where sediment and water quality are of concern, resource management practices should be devised and implemented with consideration for the influence of submersed macrophyte beds.

Nitrogen is a key element in the growth of rooted aquatic macrophytes. Thus, attention to this particular element needs to be elevated to the same level as for P. Advances in our understanding of factors regulating sediment N availability may be prerequisite to the development of innovative aquatic plant management approaches via ecological means. Towards this end, the role of submersed macrophytes in the N economy of aquatic systems needs to be more thoroughly investigated. A variety of physical, chemical, and biological processes (e.g., sedimentation, mineralization, and particulate movement by benthic invertebrates) that potentially contribute to sediment N availability need to be evaluated within the context of macrophyte nutrition (Barko et al. 1991).

Studies of nutrient cycling and hydraulic transport in macrophyte beds are of great value in providing information on rates and volumes of nutrients being exchanged with the open water of aquatic systems. Information on littoral-pelagic nutrient fluxes needs to be expanded in assessing direct effects (i.e., through uptake) and indirect effects (i.e., through reconfigured thermal structure) of macrophyte stands on water quality. Interactions between macrophytes and phytoplankton in aquatic systems need to be examined more fully through consideration of littoral-pelagic hydraulic interactions.

The goal of aquatic plant management in many cases may be an increase, rather than a decrease, in the distribution of submersed aquatic macrophytes. For example, results of studies conducted in Marsh Lake (see above) suggest that the development and maintenance of stands of submersed aquatic macrophytes may be an effective management tool for limiting wind-driven sediment resuspension and sediment discharge in shallow impoundments and lakes. Thus, macrophyte growth in some lakes (particularly shallow wind-swept basins) should perhaps be encouraged, rather than discouraged.

Acknowledgments

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Water Quality Evaluation for In-Lake Enhancement Techniques at Alamo Lake, Arizona

by
Steven L. Ashby,[1] *John L. Myers,*[1] *and Robert J. Stuart*[2]

Introduction

Water quality problems at Alamo Lake are manifested in the degassing of hydrogen sulfide (H_2S) in releases. The water of the lake is relatively high in concentrations of sulfate and organic matter and low in concentrations of nitrate, iron, and manganese, a condition which favors hydrogen sulfide production during anoxia (see Wetzel, 1983, e.g., for a thorough discussion of the sulfur cycle and Gunnison and Brannon, 1981, e.g., for a discussion of anaerobic processes in reservoirs). Hypolimnetic anoxia during the summer facilitates the processes which form the hydrogen sulfide. The low-flow releases (approximately 0.3 m^3 sec^{-1}) of anoxic hypolimnetic water during the summer season are conducive to degassing of hydrogen sulfide within the dam and throughout the outlet works. The hydrogen sulfide gas, which is frequently noticeable outside the dam, is potentially hazardous to project personnel when they are required to be the lower portions of the gate chamber start of the chemical and biological in beneath the control house. The gas causes corrosion to electrical connections in the control house and gate chamber structure. Contact with concrete parts of the outlet works also causes etching to occur. While the production of hydrogen sulfide negatively impacts operations at Alamo Lake, it is the seasonal loss of dissolved oxygen in the waters which results in the chemical and biological processes causing the formation of hydrogen sulfide.

Study objectives were to evaluate the establishment of anoxic conditions in the hypolimnion and determine the feasibility of an in-lake enhancement technique such as aeration or oxygenation. An assessment of water quality of Alamo Lake was conducted to describe the oxygen depletion and associated chemical changes during stratification. In addition to routine monitoring of physicochemical parameters, surveys of oxygen depletion rates and sediment quality were conducted. A hydrologic and physical assessment was included in the study since the relatively long residence time of hypolimnetic water during summer low-flow is likely a contributing factor. Physical assessments such as thermal patterns and lake stability were also conducted to provide additional information for evaluating in-lake enhancement techniques to improve water quality.

Site Description

Alamo Lake is located in the Bill Williams River Basin, a mountainous area in west-central Arizona and part of the Colorado River Basin (Figure 1). The lake is located at the confluence of the Big Sandy and Santa Maria Rivers, about 63 kilometers upstream from the confluence of the Bill Williams River with the Colorado River in Lake Havasu. The drainage area of the reservoir comprises approximately 12,354 km^2, and normal water surface elevation is 335 meters National Geodetic Vertical Datum (NGVD) (Corps of Engineers 1990).

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[2] U.S. Army Engineer District, Los Angeles; Los Angeles, CA.

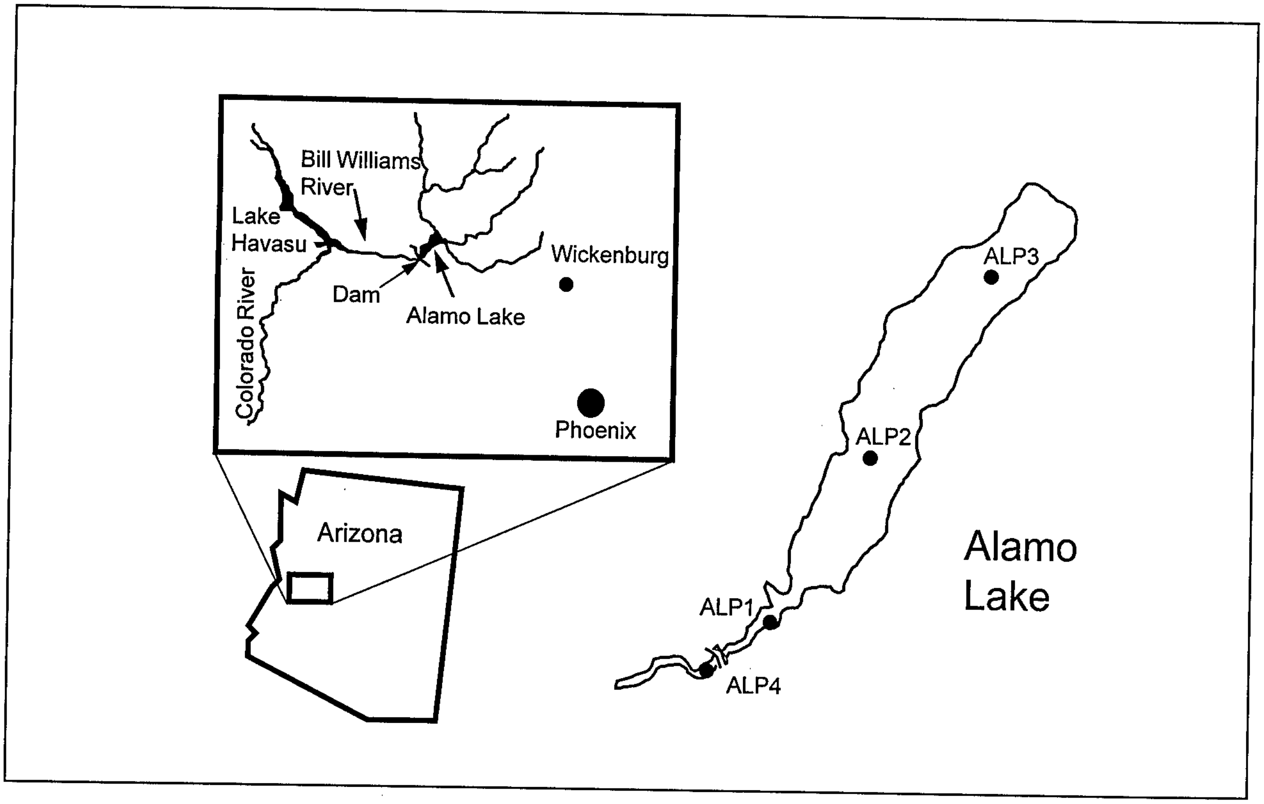

Figure 1. Location of Alamo Lake and water quality sampling stations

Alamo Dam is a rolled earth filled structure with a crest length of 297.2 meters, a crest width of 9.1 meters, and a maximum height of 86.3 meters above the stream bed. A detached, broad-crested spillway with crest elevation of 376.4 meters is located on the right abutment. The outlet works of Alamo Dam consist of a concrete-lined tunnel through the left abutment of the dam, an approach channel, a concrete intake structure at the upstream tunnel portal, a concrete shaft and gate chamber just upstream from the axis of the dam, and an outlet channel discharging directly into the stream bed. The tunnel has an inside diameter of 3.7 meters and a length of 393.2 meters. The outlet channel is an unlined trapezoidal channel 5.5 meters wide and approximately 70 meters long. The gate section contains three passages, each with 1.7 by 2.6 meter slide gates in tandem and a 0.46 meter conduit for low-flow discharges (Corps of Engineers 1964). Pertinent hydrologic features of the project are presented in the general design memorandum (Corps of Engineers 1964).

Methods

Water quality sampling was conducted by the US Fish and Wildlife Service at three stations in Alamo Lake and at one station in the tailwater region: station ALP1 located near the dam, ALP2 in the midlake region, ALP3 in the upstream region of the lake, and ALP4 in the outflow (Figure 1). Physical parameters of temperature, dissolved oxygen, and specific conductance, pH, Secchi depth, oxidation reduction potential, and turbidity were measured approximately once a month before, during, and after thermal stratification with a multiparameter water quality meter (Hydrolab Surveyor, Hydrolab Corp.®). Data are collected at five meter intervals from the surface to near the bottom of the impoundment. Discrete samples were collected on a similar schedule for water quality analyses of nutrient species, iron, manganese, sulfate, calcium, solids, total organic carbon, sulfide, and chlorophyll *a*. Water chemistry for most analyses were conducted using standard procedures (U.S. Environmental Protection Agency (USEPA) 1993 and American Public Health Association (APHA) 1992). Sulfide analysis was conducted using a Hach® HS-C Test Kit. Although more intensive data collection was conducted in 1994, data collected from 1991 through 1994 were evaluated.

Sediment samples were collected at the three stations using a ponar dredge and a sieving bucket for the removal of excess water. Ten replicate samples were collected at each station for analysis of iron, calcium, manganese, sulfate, total organic carbon, alkalinity, total phosphorus, ortho-phosphorus, Kjeldahl nitrogen, ammonia nitrogen, sulfide, and nitrate. Bulk sediment samples were analyzed using standard procedures (USEPA 1993, APHA 1992, and American Society of Agronomy 1982).

Oxygen depletion rates were evaluated with two methods of estimating oxygen demand. One method of estimating oxygen depletion rates utilized vertical dissolved oxygen measurements at station ALP1 taken periodically during 1992, 1993, 1994 coincident with the development of thermal stratification until the onset of anoxia. Data from these measurements were used as input to the computer program PROFILE (Walker 1987) for the calculation of oxygen depletion rates. Calculations were limited to a period from mid-April to mid-June, coincident with available data prior to anoxia. This method calculates depletion rates prior to anoxia but does not account for any additional oxygen deficit that may result with the formation of reduced metals. The second method employed incubation of water collected from mid- and near-bottom depths at stations ALP1 and ALP2. Samples were collected in 300 mL bottles fitted with ground glass stoppers and incubated at 20 $^{\circ}$C for 10 days. Dissolved oxygen measurements were conducted on triplicate samples at day 0, 1, 3, 5, 7, and 9 for calculating oxygen depletion rates. Measurements were conducted in mid-July and mid-October in 1994 and samples were collected from anoxic water except for mid-depth at ALP2 in October. Information available from this effort was limited since samples were not adequately aerated at the time of collection resulting in low dissolved oxygen concentrations at the start of the measurements. When conducted appropriately, this method provides an estimate of the oxygen deficit due to demand exerted by reduced metals upon reaeration.

Results and Discussion

Hydrology

Hydrologic records of mean daily inflow, outflow, and pool elevation were provided by the U.S. Army Engineer District, Los Angeles for describing the hydrology of Alamo Lake during the study period. Although several high flow events occur typically in winter and spring (flow greater than 2.8 m^3 sec^{-1}), flows during stratification are mostly less than 2.8 m^3 sec^{-1} and often reduced to a nominal flow less than 0.28 m^3 sec^{-1} (Figure 2). Inflows were greatest in 1993 (maximum flows above 250 m^3 sec^{-1}) while flows above 2.8 m^3 sec^{-1} occurred only once in 1994. In all years, a flow above 2.8 m^3 sec^{-1} occurred at least once during late stratification. Discharge was mostly less than 1 m^3 sec^{-1} except during high flows (Figure 2). Of interest is the effect of operations on the residence time of lake water, particularly during stratification. Daily retention times (calculated by dividing daily volume, based on elevation and the area capacity curve, by mean daily outflow) varied from years during low flow periods to days during high flow periods for each study year. Most importantly, increased retention during stratification results in progressively deteriorating water quality in the hypolimnion due to limited exchange with epilimnetic water. The relatively long hydraulic residence time during periods of low release should be a major consideration in the evaluation of enhancement techniques.

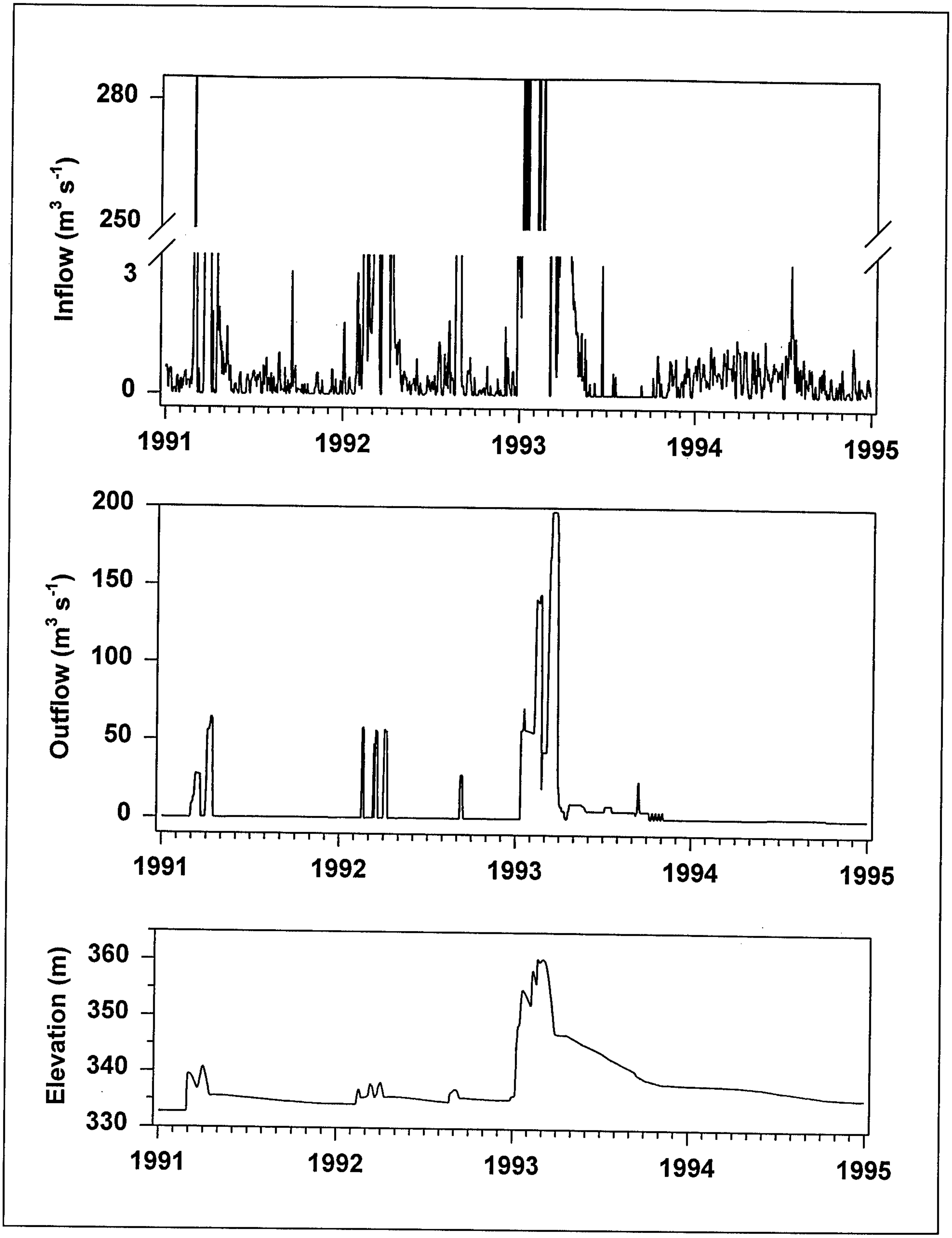

Figure 2. Inflow, outflow, and surface elevation during the study period

Temperature and Dissolved Oxygen

Alamo Lake exhibits the temperature regime of a strongly stratified lake with considerable temperature differences between surface and bottom waters during stratification (Figure 3). Surface temperatures typically increase from April until late August, and bottom temperatures increase well into October. For example, in April 1992, surface temperatures were around 24 °C, with the bottom temperatures slightly below 14 °C. A distinct thermocline developed at a depth of 7 to 8 meters and surface temperatures increased to 26 °C with bottom temperatures increasing to 15-16 °C by mid-June. Thermal gradients became more established as surface temperatures increased to greater than 30 °C until late August. Surface cooling to 25 °C with bottom temperatures increasing to 17 °C in October indicated the onset of autumnal destratification. A similar pattern was observed in 1993, even though flood control operations resulted in a relatively constant decrease in surface elevation from 347 m NGVD to 337 m NGVD. Similarly, in 1994, although surface temperatures were mostly less than 28 °C, bottom temperatures ranging from 12 to 14 °C resulted in a temperature differential between surface and bottom waters of 10 to 15 °C. This temperature differential was observed in all of the study years and may be attributed to not only seasonal surface warming but release of cooler, hypolimnetic water via bottom withdrawal at the project.

In all years, stable conditions of thermal stratification were apparent. Davis (1980) defines thermal stability as the minimum theoretical energy needed to mix a body of water from an initially stratified condition to an isothermal state. In his paper, Davis gives the following equations in explanation:

$$Stability(J) = PEM - PES$$

where

$$\textit{Potential Energy of the Mixed System (PEM)} = g \sum_{i=1}^{n} p_{im} V_i h_i$$

$$\textit{Potential Energy of the Stratified System (PES)} = g \sum_{i=1}^{n} p_{is} V_i h_i$$

where

p_{im} and p_{is} = the mean water densities of each layer for the mixed and stratified cases, respectively, (kg m^{-3})

V_i = the volume of each layer, typically 1 meter deep (m^3)

h_i = the height to the centroid of each layer above the bed of the reservoir (m)

g = the acceleration due to gravity (m s^{-2})

n = the total number of layers

Using this definition, thermal stability was calculated for Alamo Lake to determine the lake's resistance to mixing events. A spreadsheet was employed to calculate the stability of a body of water on a particular day (Meyer 1991). The spreadsheet uses temperature profiles sorted by depth to calculate density. Also, the volume of the lake as it varies with depth must be determined. Temperature profiles and lake volume determinations from area-capacity curves (Corps of Engineers 1964) were imported into the spreadsheet and stability was calculated for each sample date.

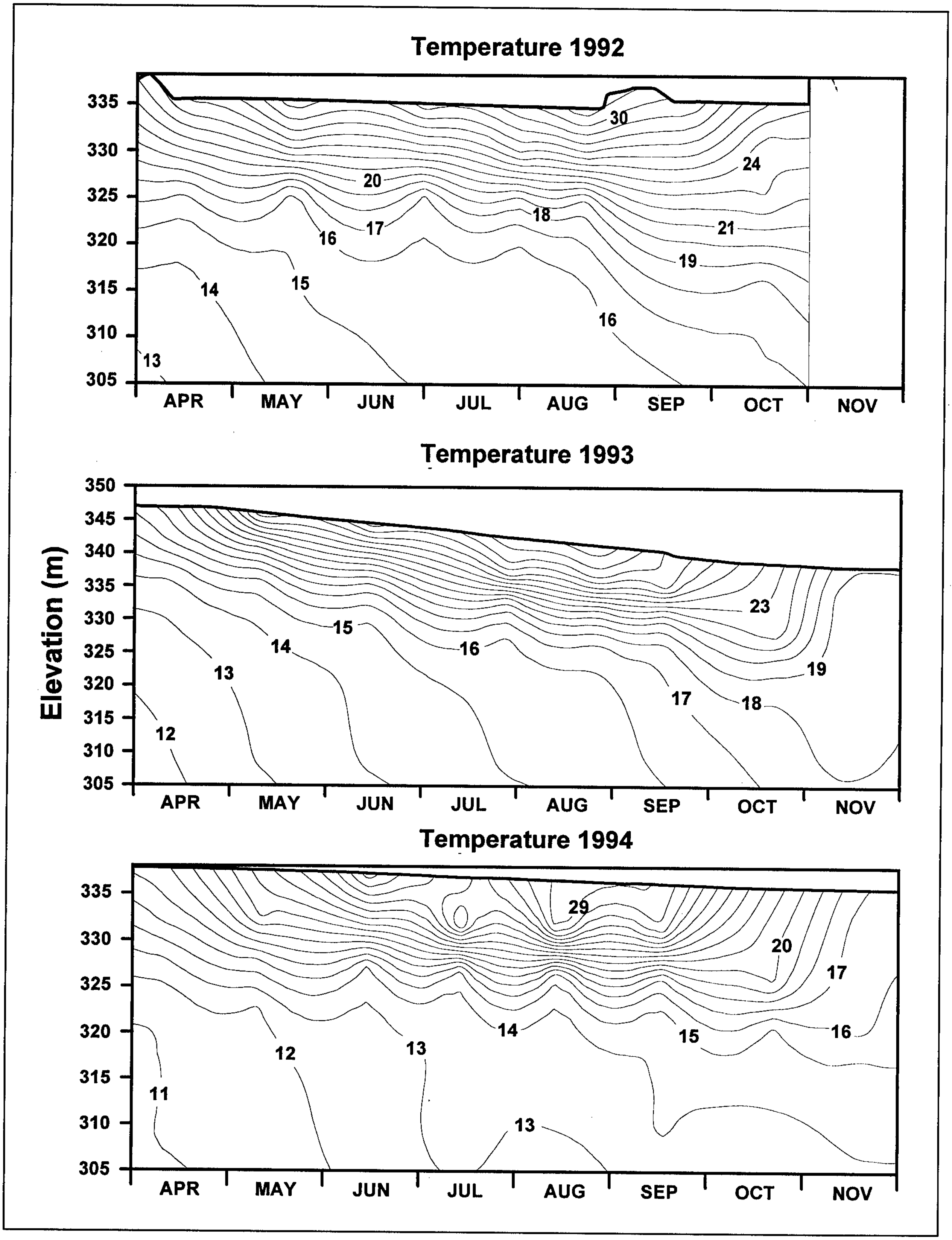

Figure 3. Temperature distribution at the forebay station during the study period

Maximum stability occurred during the summer months of the year with greatest values in 1993 due to the increased volume of warm, surface water during a high-flow year. Maximum stability at the height of thermal stratification suggests energy requirements to disrupt density gradients are quite high (e.g., greater than 2^{10} joules). It is not until late October or November of each year that stability decreases sufficiently, due to cooling of surface waters, to allow complete mixing to occur.

Patterns in dissolved oxygen in Alamo Lake are clearly influenced by the thermal regime and resultant isolation of bottom waters during periods of stability. Prior to the onset of anoxia, dissolved oxygen concentrations in the bottom waters were 2 to 4 mg L^{-1}. Anoxic conditions commonly began in May or June and lasted through October. During anoxia, surface concentrations remained at 7 to 8 mg L^{-1}, while oxygen concentrations of water below the thermocline depleted quickly. The magnitude and duration of anoxia was relatively similar for each year of the study years with anoxic conditions present throughout the hypolimnion (depths greater than 10 meters) and occurring from May through October (Figure 4).

Oxygen depletion and oxygen consumption data are summarized in Table 1. In general, the two methods provided similar estimates of oxygen utilization rates with estimates ranging from 0.01 to 0.11 mg L^{-1} day^{-1}, until anoxia, and an additional deficit rate (based on consumption of oxygen following reaeration of anoxic water) ranging from 0.06 to 0.51 mg L^{-1} day^{-1}. These data provide an indication of oxygen necessary to maintain aerobic conditions in a localized area as well as additional oxygen necessary to meet deficits transported into the aerobic area from upstream.

Water and Sediment Chemistry[1]

In general, the lake exhibits relatively high alkalinity (mean values near 180 mg L^{-1} as $CaCO_3$), as would be expected with mean calcium concentrations near 40 mg L^{-1}. Mean sulfate concentrations near 66 mg L^{-1} and mean total dissolved concentrations near 350 mg L^{-1} (which represent about 95 per cent of the total solids) are additional indications of a well-buffered (pH mostly greater than 7.0 standard units), alkaline system. Mean values were calculated by sites for all years to describe spatial patterns. Concentrations of total ortho-phosphate were relatively high with mean values ranging from 0.127 mg-PO_4 L^{-1} in the outflow (ALP4) to 0.512 mg-PO_4 L^{-1} in the upstream region of the lake (ALP3). A similar pattern was observed for chlorophyll *a* concentrations with maximum mean values in the upstream region. A dissimilar pattern was observed for nitrogen compounds, ortho-phosphate, manganese, and dissolved sulfide with greatest mean concentrations occurring in the downstream region of the lake (ALP1). Iron concentrations ranged from 0.4 mg L^{-1} to 1.3 mg L^{-1} with mean concentrations in the mid and downstream region of the lake near 0.6 mg L^{-1}. Although total organic carbon data were limited, concentrations ranged from detection limit (0.5 mg L^{-1}) to 30 mg L^{-1} with a relatively high mean of 9.6 mg L^{-1}. Water clarity and turbidity were variable with Secchi disk depths from less than 1 meter to greater than 2 meters and turbidity ranging from low values (less than 5 to greater than 100 nephelometric turbidity units (NTUs)) with minimum water clarity and maximum turbidity occurring during high flow periods.

Since the seasonal presence of hydrogen sulfide is a concern, dissolved sulfide data

[1] Water and sediment data were summarized from Ashby and Myers. 1996. Water Quality Analysis of Alamo Lake, Draft Report.

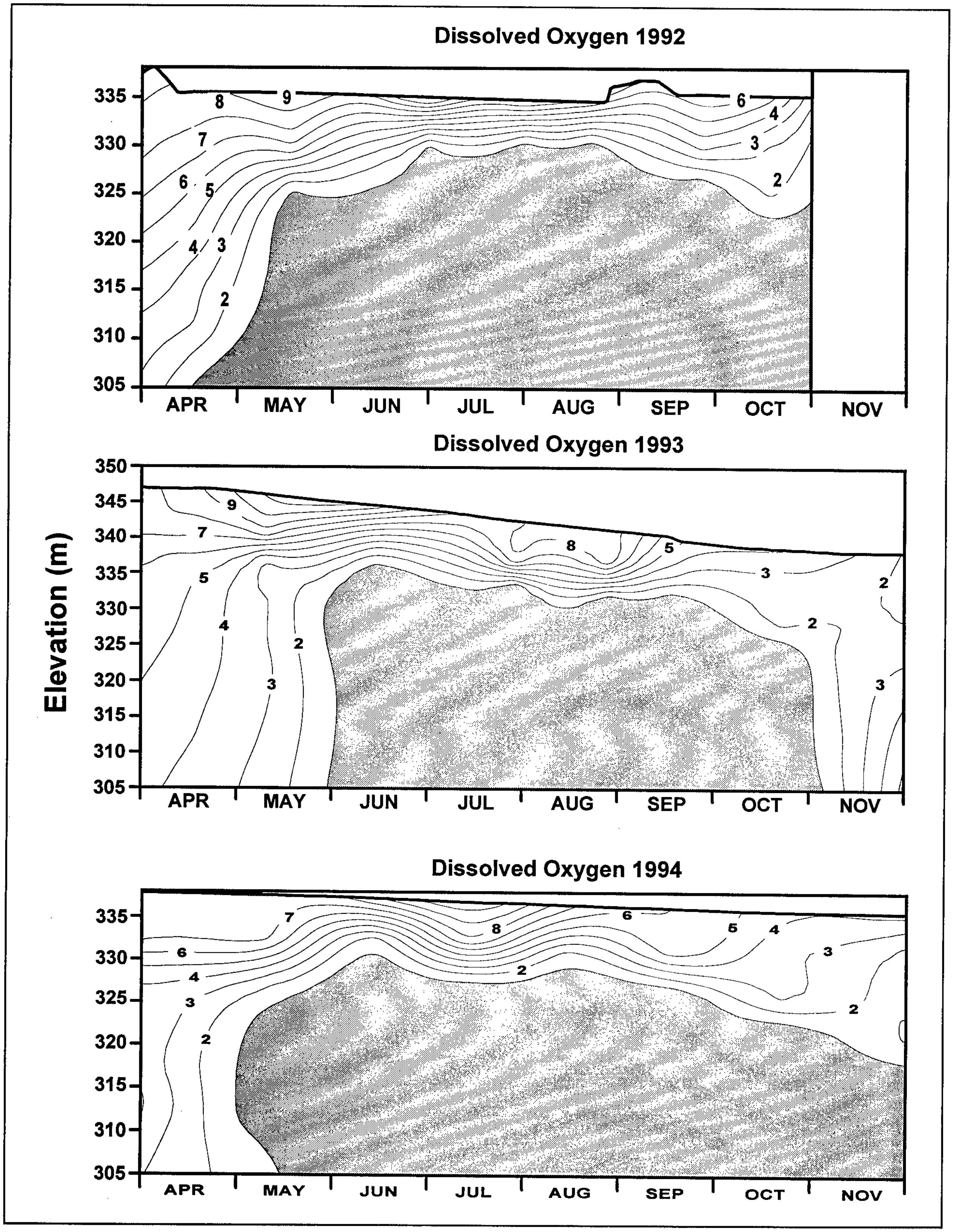

Figure 4. Dissolved oxygen concentration distribution at the forebay station during the study period

Table 1
Oxygen Utilization Assessment

Date	Oxygen Depletion Rate, Hypolimnion mg L^{-1} day^{-1}*	Oxygen Depletion Rate, Metalimnion mg L^{-1} day^{-1}*	Oxygen Depletion Rate, Both mg L^{-1} day^{-1}*	Oxygen Consumption Rate, mg L^{-1} day^{-1}**
4/14-5/18, 1992	0.11	0.05	0.075	
4/12-5/10, 1993	0.07	0.01	0.035	
5/10-6/14, 1993	0.085	0.07	0.077	
4/10-5/9, 1994	0.08	0.03	0.04	
5/9-6/13, 1994	0.02	0.08	0.06	
July 1994				ALP1, mid-depth 0.08
				ALP1, bottom 0.06
				ALP2, mid-depth 0.24
				ALP2, bottom 0.19
October 1994				ALP1, mid-depth 0.14
				ALP1, bottom N.A.***
				ALP2, mid-depth 0.15
				ALP2, bottom 0.51

* Denotes data calculated using PROFILE.
** Denotes data calculated from incubated bottles.
*** Denotes data not available.

greater than the detection limit were evaluated by month for all study years. Although detectable concentrations were observed coincident with seasonal anoxia, maximum concentrations were observed primarily in August, September, and October (Figure 5). Concentrations near 1 mg L^{-1} were observed in April, June, and November indicating that obnoxious odors may be present during early and late stratification. The temporal distribution of measurable hydrogen sulfide concentrations from April through November can be used in evaluating enhancement techniques and determines the period of time necessary for application of a selected technique.

Except for total phosphorus and nitrate/nitrite nitrogen, variability in sediment quality among stations were minimal and longitudinal gradients were not apparent. Total phosphorus and nitrate/nitrite nitrogen concentrations in the sediment decreased from the upstream station to the near-dam station.

Conclusions and Recommendations

Alamo Lake exhibits strong thermal stratification which begins in early April and persist until October. As a result of well-defined thermal structure and increased retention time due to operations, hypolimnetic oxygen depletion establishes throughout the hypolimnion of the lake. Chemical and biological processes during anoxia result in production of hydrogen sulfide which degasses during periods of low discharge causing noxious odors and creating hazardous and corrosive conditions within the outlet works of the project. The establishment of anoxic water

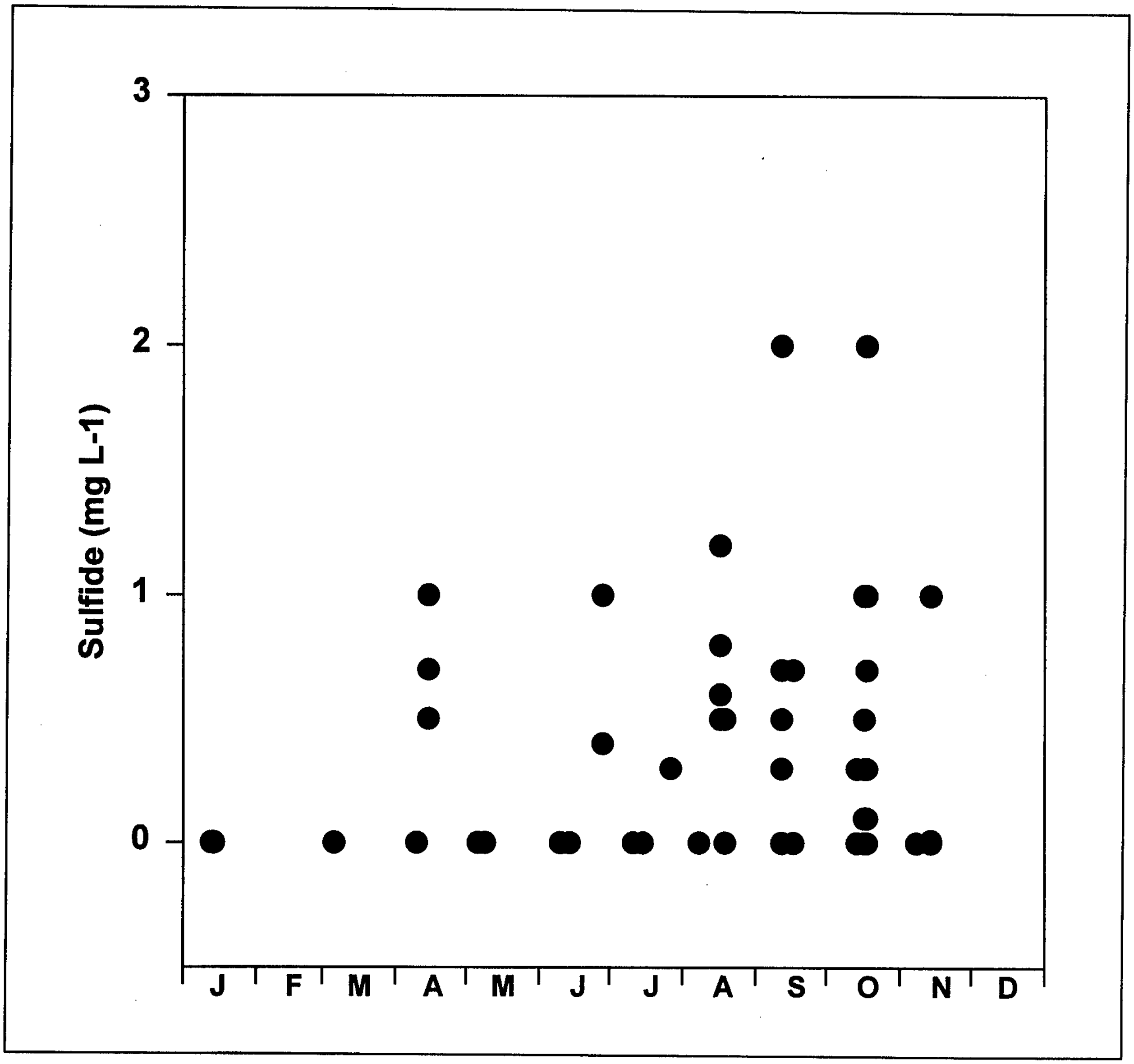

Figure 5. Dissolved sulfide concentrations for all in-lake stations during the study period

available for release has been determined to be the basis for adverse water quality conditions.

Two possible methods were proposed for consideration, each dealing with improving release water quality to minimize adverse impacts in the outlet works and downstream. One method that should be considered is to develop a means to provide surface water to the outlet works for low flow releases or for supplementing bottom withdrawal. A second method would utilized aeration or oxygenation to maintain an aerobic hypolimnion in the vicinity of the withdrawal zone during stratification.

Altering the way by which water is released from the project can result in desired release water quality without addressing the in-lake processes which result in adverse water quality. Installing a siphon or by-pass tube is designed to withdraw surface water for discharge over the dam or rerouting to existing bottom withdrawal outlet works. The technique utilizes withdrawal of epilimnetic water to avoid the release of anoxic water and the subsequent degassing of hydrogen sulfide. The technique is limited to minimum releases and feasibility of structural modifications to the outlet works for the by-pass tube or dam and flood gates for use of the siphon. While the problems associated with the degassing of hydrogen sulfide are avoided, the adverse water quality in the lake remains on a seasonal basis.

Altering the in-lake water quality prior to discharge can also result in improved water quality in the releases. Techniques applicable to Alamo Lake for in-lake water quality enhancement include methods of hypolimnetic aeration. The main purpose of hypolimnetic aeration is to increase the oxygen content of the hypolimnion without warming the hypolimnion or destratifying the lake. Fast and Lorenzen (1976) reviewed 21 different hypolimnetic aerators and were able to group them into three categories: mechanical agitation, injection of oxygen, and injection of air. Mechanical agitation consists of drawing water from the hypolimnion, aerating it on shore, and then returning the water to its original depth. This method has been used successfully, but is unpopular because of poor efficiency of gas exchange (Pastorak et al. 1982). More recent aeration applications have employed gas transfer via diffusion with the use of various diffusers at or near the bottom of the lake. This technique utilizes a distribution system of diffusers for dispersing air bubbles similar to destratification systems but relies on gas transfer properties of the rising bubble column for supplying dissolved oxygen to the water column. While much less efficient than oxygen injection, this method is much less expensive if gas transfer is sufficient to meet oxygen demand. Hypolimnetic aeration is highly effective for increasing dissolved oxygen. The hypolimnion will remain aerobic as long as the aerators are in operation. Negative features include unintentional destratification, supersaturation with N_2 gas, and the creation of a metalimnetic oxygen minimum.

A pilot study using guidance provided by evaluations of enhancement techniques should be conducted to attempt aeration of the forebay region using diffusion of compressed air or oxygen. Rental of a compressor should be considered to reduce the cost of the pilot study if compressed air is used. The diffuser system should employ the use of porous garden hoses (commonly available soaker hoses) with a flexible design for the addition of diffusers as necessary to accomplish sufficient aeration. Comparisons of costs and efficiencies should be conducted using compressed air and oxygen.

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Application of Department of Defense Groundwater Modeling System to Columbus Air Force Site, Columbus, Mississippi

by
Mansour Zakikhani[1]

Introduction

Several studies have been conducted to study the flow and transport of selected chemicals at a test site in Columbus Air Force Base, including the macro dispersition experiments (MADE). MADE-1 involved with injection and monitoring of bromide and three flourobenzoate tracers between October 1986 and 1988 (Boggs et al. 1988). MADE-2 was conducted to study the physical, chemical, and biological processes affecting transport of two conservative chemicals and four reactive hydrocarbons. The MADE-2 study was initiated with a two-day injection of the dissolved chemicals beginning June 26, 1990. Chemical concentration distributions were monitored using an extensive network of saturated zone multilevel samplers at one to three month intervals over a period of 15 months. Detailed information on field monitoring may be found in Boggs et al. (1993). Although extensive data collections and field monitoring have been conducted at the site, few numerical modeling studies have been reported (Gray and Rucker 1995).

As part of a groundwater model development at U.S. Army of Engineers, Waterways Experiment Station, selected numerical models including FEMWATER and MODFLOW/MT3D have been applied to the site using MADE-2 data. Some of the modeling approaches conducted at WES to study the site are discussed.

Site Description

The study site is located at the Columbus Air Force Base near Columbus in northeast Mississippi. The site is about 6 km east of the Tombigbee River and 2.5 km south of the Buttahatchee River (Figure 1) and it lies on 100-year floodplain of both rivers. The land surface has slops gently from north to south. The subsurface geology consists of thin terraced fluvial deposits with average thickness of 11 M. The terrace aquifer is composed of poorly to well-sorted sandy gravel and gravelly sand with minor amounts of silt and clay.

Several techniques have been used to measure the aquifer hydraulic conductivity. Among these techniques, the borehole flowmeter test has shown to provide some reasonable estimates of conductivity for the site (Rehfeldt et al. 1989a). The hydraulic conductivity varies from 10^{-4}cm/sec to 10^{-1}cm/sec. The sample mean bulk density of 84 minimally-disturbed soil cores was 1.77g/cm^3. The mean value and standard division of the porosity measurement were 0.31 and 0.08, respectively.

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

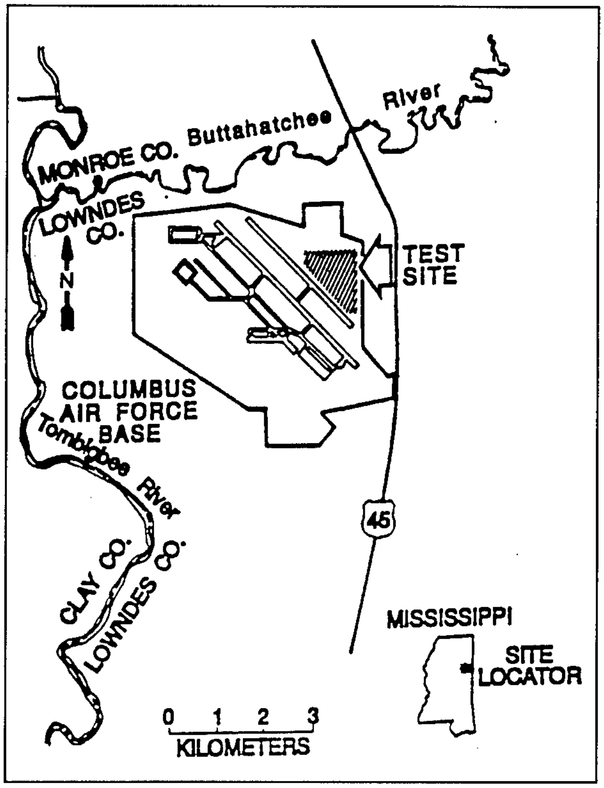

Figure 1. Schematic of site location

Numerical Approach

Conceptual Model

The borehole flowmeter hydraulic conductivity data were used to generate a 3-dimensional mesh system using the DOD Groundwater Modeling System (GMS). The site domain was divided into nine vertical layers using the borehole lithologic data. The horizonal distribution of hydraulic conductivity was estimated using the borehole flowmeter data and a kriging interpolating tool. Borehole flowmeter test data come from the saturated zone at the site. The saturated domain was divided into 14 different geologic materials having different hydraulic conductivities. The upper unsaturated zone was assumed to have geologic characteristics similar to the first upper layer of saturated zone. The unsaturated hydrogeologic characteristic of site soil was estimated from published data of similar soil type. Figure 2 shows a numerical representative of the site.

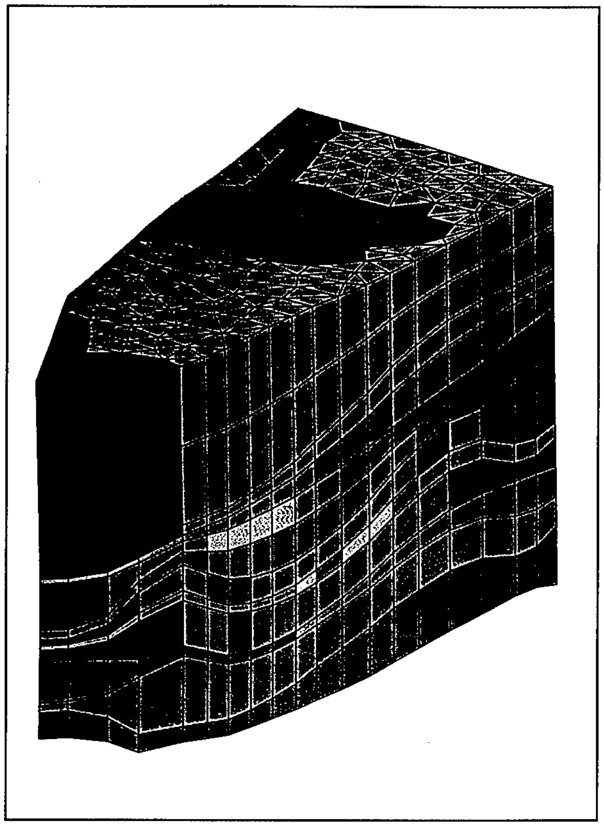

Figure 2. Numerical representation of the site

Infiltration Rate Estimation

Rain water percolation (infiltration) is one of the major hydrologic parameters affecting the movement and dilution of chemicals at the site. A reasonable estimate of the infiltration rate was obtained using the Pesticide Root Zone Model (Mullins et al. 1984). This estimate then was used as input to FEMWATER to calculate the flow characteristics at the site. Figure 3 shows an estimated infiltration rate for the site using PRZM2 model.

Boundary and Initial Conditions

The transient flow boundary conditions were estimated from a series of flow measurements

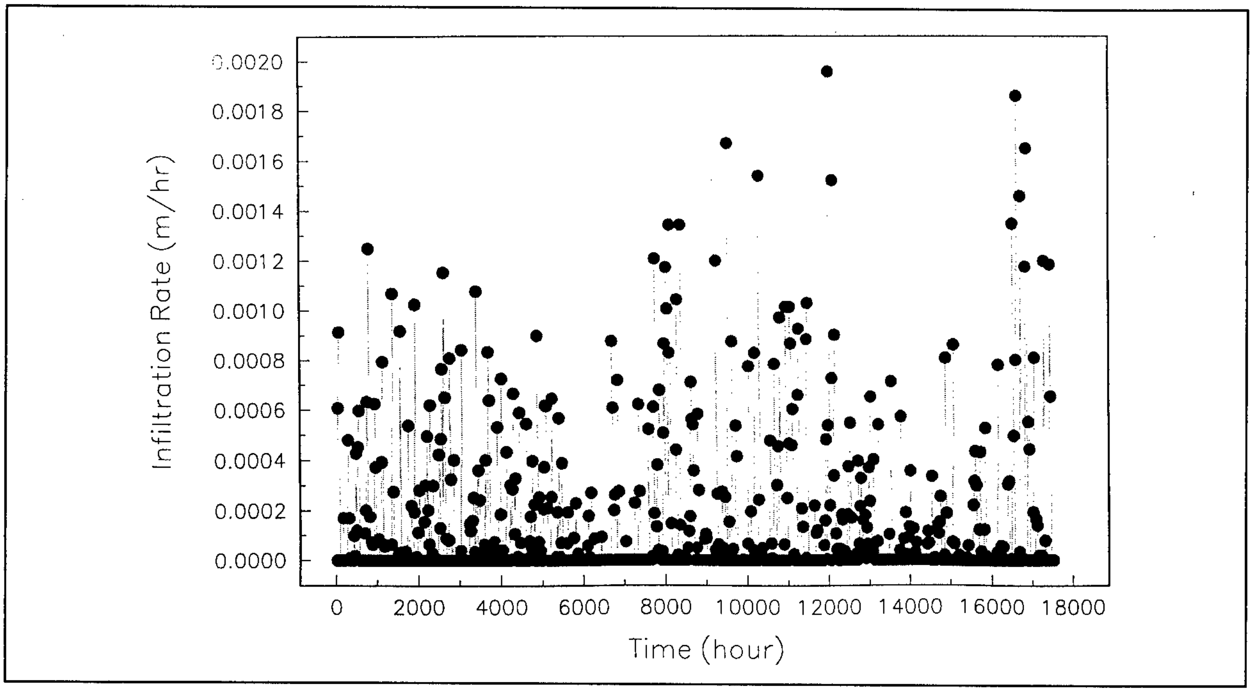

Figure 3. Estimated infiltration rate at the site

during 1990-1991. A graphic package was used to generate contours of the hydraulic heads for each day of measurements. From these contours, the hydraulic heads for boundaries were assigned. Flow data measured at the beginning of simulation are used as initial condition. For mass transport simulations, boundary and initial conditions were not available. Therefore, we assumed that at the boundary concentration is zero and initial condition of concentration was also zero.

Results and Discussion

The developed conceptual model was simulated using FEMWATER unsaturated-saturated model. A result of flow simulation as distribution of flow velocity at 450 days, time of simulation, is shown in Figure 4. The flow model included infiltration rates, boundary and initial conditions and did not have injection rates. The coupled flow and transport simulation was done for tritium assuming initial concentration zero. A preliminary result of concentration for time 450 days is given in Figure 5. The injection zone has low hydraulic conductivity compared to adjacent zones. This caused a slow movent of chemical at the vicinity of injection rate and faster movement at the zone far from the injection zone. The special feature of hydraulic conductivity at the injection zone created an upward movement of the plume. This unique feature of the plume requires more special numerical treatment to produce results comparable to field conditions.

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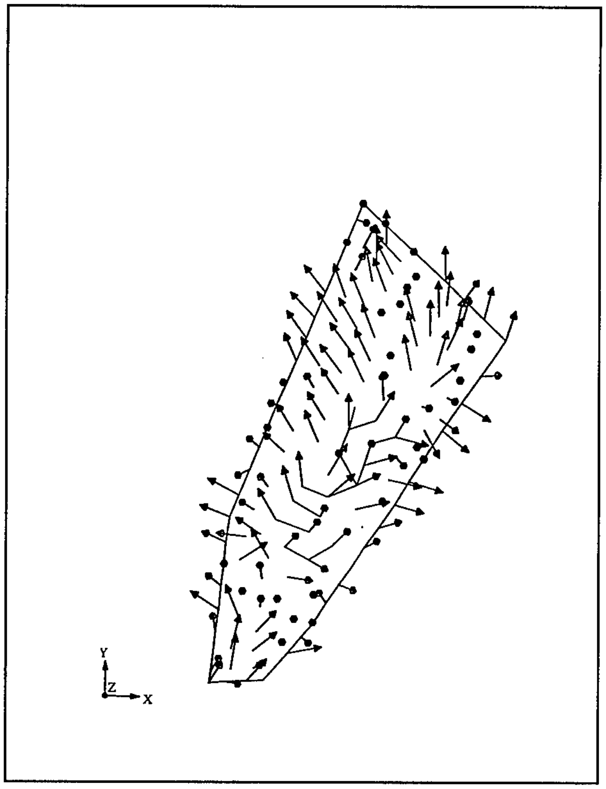

Figure 4. A simulated flow velocity distribution at the site

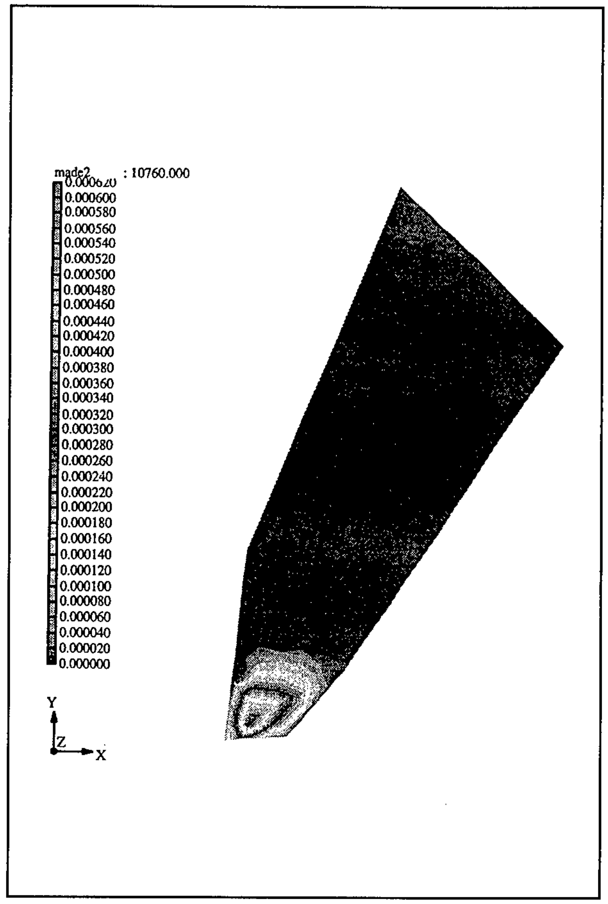

Figure 5. A simulated concentration distribution of tritium at the site

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Determination of Potential Site-Specific and Cumulative Water Quality Effects of Multiple Flood Control Dams in the Red River of the North Basin

by
Dennis D. Holme[1] *and Willis Mattison*[2]

Background

The St. Paul District and the State of Minnesota are jointly preparing an Environmental Impact Statement to analyze the potential cumulative effects of numerous (33) flood damage reduction impoundments proposed for construction in the Minnesota portion of the Red River of the North Basin. The proposed impoundments include 16 with permanent pools (wet dams) and 17 with temporary pools (dry dams). Eleven of them would be constructed off-channel. The dams are variably gated or ungated. Most of the projects would be funded and constructed by local watershed districts with funding assistance from the Red River Watershed Management Board, a regional umbrella water management funding entity. None of the proposed dams are Corps projects.

The concern over cumulative impacts arose during the summer and fall of 1991 when the District Regulatory Office received Section 404 permit applications for several impoundment projects in three of the Red River subwatersheds. It was subsequently revealed that 30 or more projects were being planned. This presented the possibility that local incremental effects of individual projects, which may or may not be significant locally, could have significant cumulative effects at the subbasin or main stem level.

In January 1993 the District Engineer determined that permit actions for certain impoundments would be held in abeyance until a cumulative effects analysis could be completed. In December 1993 the DE issued a notice of intent to prepare an EIS as required under the National Environmental Policy Act (NEPA).

The EIS Process

The environmental review process and scope for this EIS was determined by a task force co-chaired and staffed by the St. Paul District and the Minnesota Department of Natural Resources. That group produced a "scoping decision document" which determined the specific subjects to be analyzed by the EIS. They decided that the environmental review process would be divided into two levels or tiers. Tier 1 would be a *generic* EIS discussing the *cumulative* environmental effects of the different project types and the *general* effects of other flood damage reduction alternatives in northwestern Minnesota. Tier 2 would focus on individual projects and would require preparation of environmental assessments (EA's), environmental assessment worksheets (EAW's), or EIS's variably. The scoping document also provided definitions and guidelines, and assigned tasks to various work groups responsible for the analysis of six resource topics: (1) hydrology and hydraulics, (2) water quality,

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[2] Minnesota Pollution Control Agency, Detroit Lakes, MN.

(3) natural resources, (4) socioeconomic, (5) tribal trust, and (6) archeological and architectural resources. The Water Quality Work Group (WQWG) was chaired by the authors and was composed of experts furnished by interested and or cooperating federal and state agencies and employees or consultants representing the permit applicants. The WQWG convened several times for the purposes of assembling information, for defining the historic and existing condition of the resource, and for developing of methodologies for predicting site specific effects of individual reservoir projects. The group was not involved collectively in the determination of cumulative effects criteria, significance criteria, findings of cumulative effects, and recommendations. This presentation focuses on the procedures, methodology, and findings pertaining to water quality study.

Study Area

The primary study area for the EIS is the Minnesota portion of the Red River of the North basin (MN-basin) shown on Figure 1. The study does not include impoundment projects proposed for the North Dakota portion of the basin. Those projects fall under the regulatory jurisdiction of Omaha District and the State of North Dakota.

The EIS includes analysis at two levels—the subbasin and the MN-basin. Each of the nine subbasins were considered to be logical discrete units within which effects of local projects could accumulate. The MN-basin was evaluated as the largest geographical area within Minnesota where effects of the proposed projects could accumulate.

Study Objective - Cumulative Effects Analysis

The term "cumulative effects" is defined in the National Environmental Policy Act and the Minnesota Environmental Policy Act as the result of incremental effects of the proposed action when added to the effects of all past, present, and other future actions. The significance of cumulative effects, therefore, relates to the determined significance of past actions as well as the incremental effects of the proposed actions. The EIS discusses several ways by which effects can accumulate; by addition, time crowding, space crowding, synergism, indirect action, time lag, and by nibbling.

For each potentially effected resource, including water quality, it was necessary to determine significance criteria and establish thresholds for the incremental cumulative effects. For all of the resources the significance of past effects was used as the context for these determinations. For example, if the effects of past actions on a particular resource are judged to be significant, a low threshold would be used to define the significance of incremental cumulative effects. If the effects of past actions have been judged to be insignificant, the threshold would be set high.

Historic and Existing Conditions

Given the above definition of "cumulative" and significance criteria it was first necessary for each workgroup to determine the historic and existing condition for each resource to provide a base upon which to add incremental effects. The WQWG's Historic and Existing Conditions (H&E) report is the first part of a two part technical appendix. Unfortunately, it presents a historical perspective based largely on data derived from archaic methods and anecdotal information. Long term scientific data descriptive of the effects of cultural development since the beginning of European settlement in the basin do not exist.

The existing condition is based partly on the large volume of water quality data collected over the past 30 years by various agencies. Most of this data, however, applies only to the main stem of the Red River and near the mouths of the tributaries. The determination of existing water quality conditions of the various subwatersheds is based largely on the findings of the

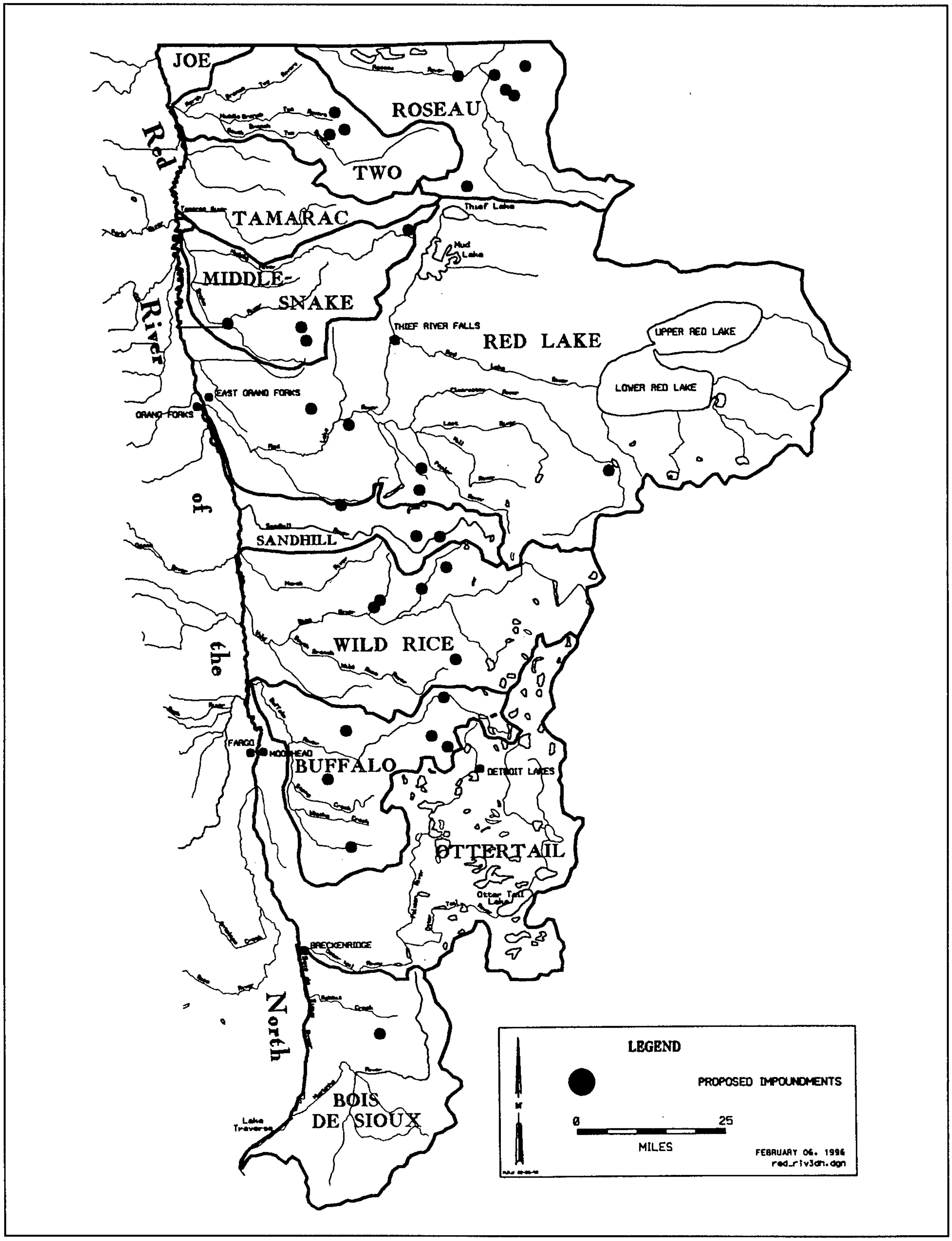

Figure 1. Red River of the north subbasins in Minnesota

1994 Minnesota Water Quality Assessment (305b report), which relies partly on sparse monitoring data and partly on an opinion survey of local water resource managers. The H&E report concluded that the water quality of the lakes and streams of the MN-basin has been significantly and adversely affected by industrial and municipal point-source pollution, agricultural and other nonpoint source pollution, and hydrologic modification. Sections of the Red River main stem and tributaries do not fully support their designated uses as defined in the Minnesota Pollution Control Agency rules. Other sections are either threatened or impaired by these activities.

Water Quality Significance Criteria

The significance criteria for water quality cumulative effects were developed by the WQWG co-chairs in consultation with MPCA staff whose responsibility it is to implement State policy with regard to the Clean Water Act. The criteria is based on four assumptions;

a. The significance of water quality effects may be described as changes in the aquatic environment as they relate to valued (by society) water quality functions described in Minnesota Pollution Control Agency Water Quality Rules, Chapter 7050, Minnesota Beneficial Use Classifications.

b. The body of existing Federal and State laws, rules, policies, goals, and guidelines provides a clear indication of the intrinsic and functional value society has assigned to the quality of the waters of the State and the Nation.

c. The past present, and planned future investment (private and public) in point and non-point source water pollution control efforts provides strong indication that society has decided that water quality degraded by past human activity should be restored and that undegraded waters should be protected and maintained.

d. The designation of certain waters as "outstanding Resource Value Waters" establishes a higher value to certain waters which aims at protecting undegraded lakes, streams and wetlands in their undisturbed state.

Considering that past actions have significantly and adversely affected water quality in the MN-basin; and considering that society has expressed its will, in word and in deed, to reverse that trend; the joint Federal and State agencies set the significance threshold for water quality cumulative effects at zero as reflected in the following statement:

"SIGNIFICANT" water quality effect is defined as those direct or indirect water quality effects that result in an increased probability of an excursion of a water quality statute, law, rule, standard, policy, goal, objective, alert level, or Clean Water Act 404(b) guideline. An "excursion" in this context refers to:

a. A violation of a statute, law, or rule.

b. An exceedence of or non-compliance with a water quality standard, objective, alert level or guideline.

c. The further degradation of a water body which has had its designated uses threatened or impaired.

d. A reversal or departure from, delay in, or interference with achievement of policies and goals.

Future-With Analysis and Effects of Alternatives

For the future-with phase of the study the joint agency EIS Task Force instructed the work group co-chairs to rely primarily on existing data and to use qualitative best professional judgement, if necessary, in conducting their environmental effects analyses. The study schedule and lack of adequate funding did not permit the development of new data. The immediate tasks of the WQWG were to; (1) define the significant resource "water quality" in terms such that some measure of incremental effects could be applied, (2) define the various means by which these effects could accumulate at the subbasin and MN-basin levels, and (3) define thresholds at which cumulative effects become significant.

It is instructive to note that every meeting of the WQWG proceeded in a highly contentious atmosphere in which opposing special interests and viewpoints clashed continually. Predictably the initial discussions did not produce the required definitions. There was agreement, however, that the mechanisms and measures of incremental cumulative effects, whatever they are, would be related to site-specific effects on conventional measures of water quality (i.e., DO, ph, major ions, nutrients, organic contaminants, etc.) and that an analysis of individual project effects would be a useful first step. The lack of data and time to develop it all but ruled out the use of empirical methods. The group eventually agreed to use an approach that would rely almost entirely upon the professional judgement of the participating work group members to determine, qualitatively, the types of effects that might be associated with the individual projects.

We developed methodology by which individuals working alone would systematically follow a procedure using a common set of defined water quality criteria and criterion-specific assumptions to judge the potential effects of the individual projects on each of three potentially effected river reaches; within the impoundment, immediately downstream of the impoundment, and within the Red River main stem. The common thinking was that potential cumulative effects would only include those that extend beyond the immediate downstream reaches of the impoundments.

An evaluation package was assembled and distributed to participants as kits containing an instruction sheet and four components; (1) a list of 14 water quality criteria and their definitions, (2) an index of criterion-specific assumptions, (3) a table containing project-specific pertinent data including site morphometry, operational and design characteristics, location, etc., and (4) a scoring matrix form. The instruction sheet directed the evaluators to (1) read the definitions, (2) read the index of criterion-specific assumptions, (3) visualize each impoundment in terms its design and operational characteristics and location, (4) cite the relevant assumptions by recording the index numbers on the scoring matrix, and (5) rate the impacts using a rating scale of -2 to +2.

Data Reduction

The rating exercise was performed by five members of the study group including three members representing the permit applicants and two members representing the state and federal regulatory agencies. It produced a stack of 33 2-page matrixes from each of the five raters for a total of 6,730 individual judgements. Because the rating system was intended to provide a qualitative rather than quantitative assessment, the scores were not numerically weighted and statistical methods were not applied. Data reduction, was accomplished by counting, sorting, and screening rather than summing and averaging.

The first step in data reduction was to combine the five matrixes into one matrix containing all of the data. The next step was to break the combined matrix into two, showing the collective high and low scores in each cell. All non-integer ratings were treated as 0's. This sorting brought foreword the maximum divergence of opinion among the raters.

The next step was to check for one degree of confirmation for each of the high and low ratings such that all high and low ratings that were not confirmed by at least one of the other four raters was downgraded and the next less extreme rating, if confirmed, took its place. This step resulted in many magnitude 2 ratings dropping to magnitude 1 and many magnitude 1 ratings dropping to zero. This sorting greatly reduced the apparent divergence of opinion among the participants, and none of the members objected to using the results so long as both groupings, high and low, were presented in the technical document.

The results of the analysis revealed that none of the projects had any effects, good or bad, that extended all the way to the main stem of the Red River. A few of the projects had minor effects in the immediate downstream reach. Most of the identified in-pool effects were associated with on-channel dams with permanent pools.

Finding of Significant Cumulative Effects - Assimilative Capacity

Assimilative capacity was included as one of the 14 water criteria considered in the site-specific analysis and was found in each case not to be a problem that extended very far downstream. All of the evaluators, however, had overlooked a consideration brought out later by the Hydrology and Hydraulics Work Group regarding the effect of evapotranspiration. The point was made that when water evaporates from an impoundment it is lost, and the loss contributes incrementally to reduced flows caused by other consumptive uses in the subbasins and MN-basin. It is, thereby, a cumulative effect. During low-flow seasons evaporative loss from reservoirs (unless compensated for from storage) can significantly reduce the magnitude of flows, extend the duration of critical low-flow periods, and increase the frequency of critical low-flow and no flow events. The assimilative capacity of streams can be limited by flow during extreme low-flow periods. By that reasoning we determined that 14 of the proposed impoundments with permanent pools could potentially cause significant adverse cumulative effects by reducing the assimilative capacity of downstream waters at both the subbasin and MN-basin level.

The findings of potential significant adverse cumulative effects for water quality and other resources in the tier 1 EIS mean that certain information regarding those effects will have to be provided by the applicants in the tier 2 environmental review documents. Permit decisions by the regulatory agencies will be made based on that information as well as site-specific effects, and information regarding mitigation and feasible alternatives.

Water Quality Monitoring for Sea Turtle Migration During Dredging Operations

by
Bruce Baird,[1] *Ed Creef,*[1] *and Burnell Thibodeaux*[1]

Seasonal restrictions on dredging are not uncommon to the Corps' dredging community. Commonly seasonal restrictions center around protection of some aquatic species which can be adversely impacted by dredging operations, e.g., migrating penaeid shrimp in Wilmington and Galveston Districts, migrating juvenile chinook in Walla Walla District, and overwintering shortnose sturgeon in the Philadelphia District.

In the New Orleans District, three authorized navigation projects in coastal Louisiana, which require maintenance dredging by hopper dredge, may impact threatened and endangered species. These projects are the Mississippi River, Baton Rouge to the Gulf of Mexico, Louisiana, Southwest Pass lower jetty and bar channels (Southwest Pass); the Mississippi River-Gulf Outlet, Louisiana, bar channel; and the Calcasieu River and Pass, Louisiana, bar channel. The project which is the focus of this paper is the Southwest Pass portion of the Mississippi River, Baton Rouge to the Gulf of Mexico, Louisiana, project (Figure 1) which includes the range of five species of sea turtle.

National Marine Fisheries Service (NMFS) has previously listed five species of whales and five species of sea turtles that may occur in coastal Louisiana. All five species of whales are primarily confined to deeper waters of the Gulf of Mexico and would not be expected to occur, except as a rarity, within the areas of work. In 1995 and prior years, NMFS has determined that listed whales are unlikely to be adversely affected by hopper dredging in the Gulf of Mexico. However, there is a potential to adversely affect sea turtles during dredging operations.

Project Description

The New Orleans District performs annual dredging of navigational channels to maintain authorized project dimensions. The Mississippi River, Baton Rouge to the Gulf of Mexico, Louisiana, channel requires hopper dredging in the Southwest Pass reach where cutterhead dredges would impede navigation.

Annual work in this reach of the navigational channel includes maintenance dredging of shoal material from various locations between mile 4 above Head of Passes (AHP) to mile 22 below Head of Passes (BHP), primarily by hopper dredges. Hopper dredges are also used to maintain the Mississippi River crossings at upriver locations, where turtles are not expected to occur. Authorized channel depth is 55 feet; however, the channel is currently maintained to a 45 foot depth. Channel widths are maintained to 750 feet in the Cubits Gap area and 600 feet in the lower jetty and bar channel area.

Maintenance dredging of Southwest Pass usually takes approximately eight months; the dredging season is variable and during some years dredging may occur for 12 months. For 1995, it was estimated that about 8 miles of channel would require hopper dredging; 5 miles in the area between mile 4 AHP to mile 1 BHP, and in the lower jetty and bar channel from mile 18.8 BHP to mile 22 BHP. An hydraulic

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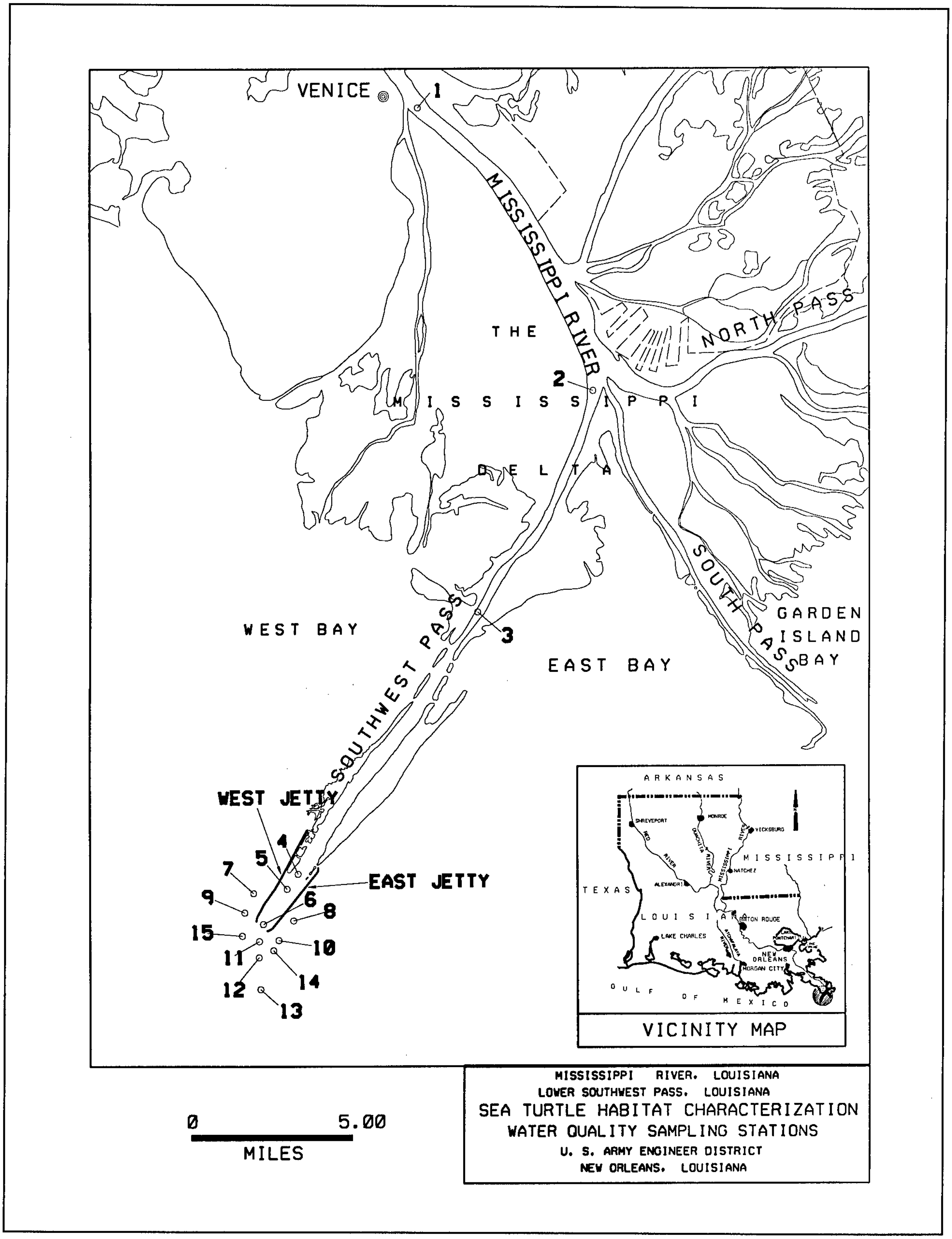
1 U.S. Army Engineer District, New Orleans; New Orleans, LA.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LOWER SOUTHWEST PASS, LOUISIANA
SEA TURTLE HABITAT CHARACTERIZATION
WATER QUALITY SAMPLING STATIONS
U. S. ARMY ENGINEER DISTRICT
NEW ORLEANS, LOUISIANA

Figure 1.

cutterhead dredge would be used for the rest of the pass.

Sea Turtles in the Gulf of Mexico

Kemp's ridley has been labeled the "Louisiana turtle" by Hildebrand (1981) and is thought to be the most abundant turtle off the Louisiana coast (Viosca 1961; Gunter 1981). The highly productive white shrimp-portunid crab beds of Louisiana from Marsh Island to the Mississippi Delta are thought to be the major feeding grounds for subadult and adult ridleys (Hildebrand 1981). The current patterns in the Gulf of Mexico could aid in transport of individuals, where small turtles swimming offshore until reaching sargassum mats would enter the major clockwise loop current of the western Gulf of Mexico carrying individuals north and east along Texas, Louisiana and subsequent coastal areas (Pritchard and Marquez 1973; Hildebrand 1981). The Louisiana Sea Turtle Stranding and Salvage Network (LA-STSSN) registered 373 sea turtles stranded on Louisiana beaches from 1990 through 1994. Of these, 268 were Kemp's ridleys (Koike 1995).

Loggerhead turtle strandings have been reported in Louisiana from Cameron (Fuller 1986) as well as Holly Beach in August and Isles Dernieres in July (SEAN 1980). A tagged loggerhead was recaptured near Grand Isle at Belle Pass (Lund 1974). More recently, LA-STSSN registered 45 loggerheads stranded on Louisiana beaches from 1990 through 1994. This represented 12 percent of the sea turtles stranded, second only to the Kemp's ridley.

Recent capture and telemetry studies of sea turtle movements along the northern Gulf of Mexico showed usage of the nearshore areas near jetties and channels. Kemp's ridleys were captured most frequently, and loggerheads were the second most frequently captured in Texas and Louisiana waters.

Historical sightings of green turtles by fishermen in Louisiana occurred gulfward of Isles Dernieres and Timbalier Islands in spring, summer and fall. Recent sightings have been reported from the northwest areas of Terrebonne Bay in summer and off Belle Pass in fall (Fuller 1986). Green turtle stranding records, as well as turtle fishing records from Louisiana and Texas combined, are one-third that reported from Florida (Fritts et al. 1983). LA-STSSN registered 10 green turtles stranded on Louisiana beaches from 1990 through 1994. This represented 2.7 percent of the sea turtles stranded.

Historical sightings of leatherback turtles have been reported in Louisiana from Terrebonne Bay and Timbalier Bay. Recent sightings were noted by helicopter pilots in National Marine Fisheries Service statistical zones 12, 14 and 17 in January, March, and April (Fuller 1986). These zones include the area off Isles Dernieres and Timbalier Islands (Area 14) and off Cameron (Area 17). Leatherbacks are not usually disturbed by dredges, since they tend to stay on the surface of the water and do not come into contact with the dragheads (Richardson 1988). Only eight leatherbacks were stranded on Louisiana beaches from 1990 through 1994.

While there have been no sightings of hawksbill turtles in the proposed area of work, one was reported from a gillnet catch in Cameron Parish, Louisiana, in the 1986 survey of Louisiana coastal waters by the National Marine Fisheries Survey (Fuller et al. 1987). This supports the general belief that hawksbills are scarce in Louisiana waters. The stranding network data from 1990 through 1994 reported only one hawksbill stranding in Louisiana.

Methods to Reduce Impacts of Hopper Dredging on Sea Turtles

Draghead Modifications

Certain types of draghead dredges, which function by hydraulic erosion, can be modified with cages or deflector systems to prevent turtle entrainment (Joyce 1982). Use of the California type draghead significantly reduced the capture of loggerhead turtles in Florida. This modification was the result of findings of an interagency task force formed to investigate methods for reducing the incidental injuring and/or killing of endangered and threatened turtles in connection with hopper dredging in federal navigation channels (Joyce 1981) (Sea Turtle/Dredging Task Force). In addition to the modified draghead, the overflow is monitored using large mesh baskets designed to retain any turtles or turtle fragments.

Dredging Windows

Aside from physical modification of the existing dredge equipment, dredging only during non-threatening times of the year is another alternative to reduce impacts on sea turtles. Recent research has shown that sea turtles are virtually absent from the nearshore waters of the northern Gulf from December through March (Renaud et al. 1995). In the case of the Southwest Pass navigational channel, hopper dredging is required in the summer months when shoaling occurs, typically as the river stage falls. This is necessary to keep the channel open to shipping, thus dredging only in winter months is not feasible.

Alternative Dredges

Another approach is to use cutterhead dredges instead of hopper dredges. Unfortunately, hopper dredges are necessary in this channel because they are the safest for use in open water where wave energy can be great; they are safest in heavy shipping traffic as occurs in the Mississippi River, particularly near Head of Passes where currents are strong and difficult to navigate; and they are the most cost-effective.

Turtle Relocation

Another method to reduce impacts is trawling in channels to be dredged to remove sea turtles and relocate them to areas away from the dredging. This method has met with limited success, due to the turtles tendency to return to the area quickly (Moulding 1988).

New Orleans District's Conservation Measures

Hopper dredging in the Southwest Pass navigational channel appears to be less likely to impact sea turtles than hopper dredging in tidal dominated channels. The cooler waters of the Mississippi River, the low salinity, the strong currents in the channel, and the presence of sand waves moving along the bottom would all tend to discourage usage of the channel bottom by sea turtles, particularly from November through April, when river water temperatures of 15 °C and lower often prevail. A recently completed (May 1995) two-week netting effort by NMFS personnel at Grand Terre, within 40 miles of Southwest Pass, resulted in no captures of sea turtles. Tidal passes were targeted, including passes with jetties.

The Southwest Pass navigational channel is a very dynamic system, where rapid shoaling can block shipping virtually overnight. During the dredging cycle particularly as the river rises and falls, channel depth is monitored constantly. A rapid response to shoaling is necessary to keep shipping lanes open, and often four or five hopper dredges are needed simultaneously.

There is little evidence that sea turtles are abundant in the Southwest Pass jetty and bar channels, and it is probable that impacts from hopper dredging would be negligible. Turtle presence at the many outlets of the Mississippi River have not been documented, although this may be more a result of the remoteness of the area and the lack of observers on the few beaches in the area. Turtles may avoid this area due to the large volume of fresh water flowing into the area.

Pilot Study

The New Orleans District is currently conducting a pilot study to collect monthly data from the Southwest Pass area (Figure 1), including water temperature, salinity (surface, mid-levels, bottom), suspended sediment and water clarity (Secchi disk). Other observations would include strong currents, flotsam, sargassum weed, water hyacinth quantity, turtles sightings, oil slicks, and any fish or bird concentrations. Samples would be taken at miles 12 AHP, at Head of Passes, mile 10 BHP, and at the base of the jetty in the channel. Samples are also taken at the halfway points inside the jetty, at the end of the jetty, and at points in the bar channel 0.5, 1 and 2 miles from the end of the jetty. Each of these locations would have corresponding sampling sites on either side of the channel, 0.5 miles from the channel.

Conclusions

Southwest Pass of the Mississippi River navigation channel provides the habitat least likely to attract sea turtles of the three channels discussed. Having qualified observers available on standby to monitor dredging on numerous dredges in a dynamic and unpredictable system would be costly and possibly wasteful in a channel where evidence of turtle usage is lacking. Better information can be obtained by conducting studies specifically designed to assess potential impacts to turtles from hopper dredging in this channel.

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Water Quality Constraints to Implementing the Endangered Species Act on the Columbia River

by
Bruce Glabau[1]

Introduction

In 1991, Snake River sockeye salmon were designated by the National Marine Fisheries Service (NMFS) as endangered. In 1992, NMFS designated the Snake River fall and spring/summer chinook salmon as threatened, then reclassified them as endangered in 1994. Under provisions of the Endangered Species Act (ESA) profound changes in the manner in which the Columbia River reservoir system is managed have been made to aid migrating juvenile salmon on their journey to the Pacific Ocean. Much of the focus of these new actions has been on increasing the level of flow and spill during spring and summer at the mainstem reservoir projects on the lower Snake and lower Columbia Rivers. With increased spill, concern over water quality and the effects on resident and anadromous fish has risen. In 1995, a new spill program was implemented in which Corps of Engineers projects on the lower Snake and lower Columbia Rivers were required to spill vast quantities of water in an effort to provide improved fish passage conditions at each project (Biological Opinion 1995). Most of the projects were not able to spill the amounts required for meeting fish passage goals due to limitations caused by total dissolved gas (TDG).

Spill Program

The 1995 Biological Opinion (BiOp) issued by NMFS prescribed spilling lower Snake and lower Columbia River projects to achieve a fish passage efficiency (FPE) of 80 percent at projects in the lower Snake and Columbia Rivers (Figure 1). FPE is defined as the percentage of the total number of fish passing a project which do not pass through the turbine units. These fish either pass through the fish bypass facilities or over the spillway. FPE is a function of the amount of spill, the diel nature of fish passage, efficiency of fish passage over the spillway, and the fish guidance efficiency (FGE) associated with the screens which guide fish from the turbine intake and into the juvenile bypass facility. The spill required to reach the 80 percent FPE target for 1995 is shown in Table 1. Spill requirements from the 1994 Biological Opinion are also shown to demonstrate the scale of change represented by the new 1995 spill program.

In addition, the 1995 Biological Opinion required spill be limited, if necessary, to control the production of TDG. Dissolved gas is produced by the plunging action of water as it passes over the spillway, entraining atmospheric gases, especially nitrogen, which can result in supersaturation above the normal range of a free flowing river. According to NMFS acceptable levels of TDG are a 12 hour average concentration of 115 percent of saturation in the forebay and 120 percent in the tailrace. Spill is also limited to prevent instantaneous TDG levels from exceeding 125 percent saturation for any two hours per day.

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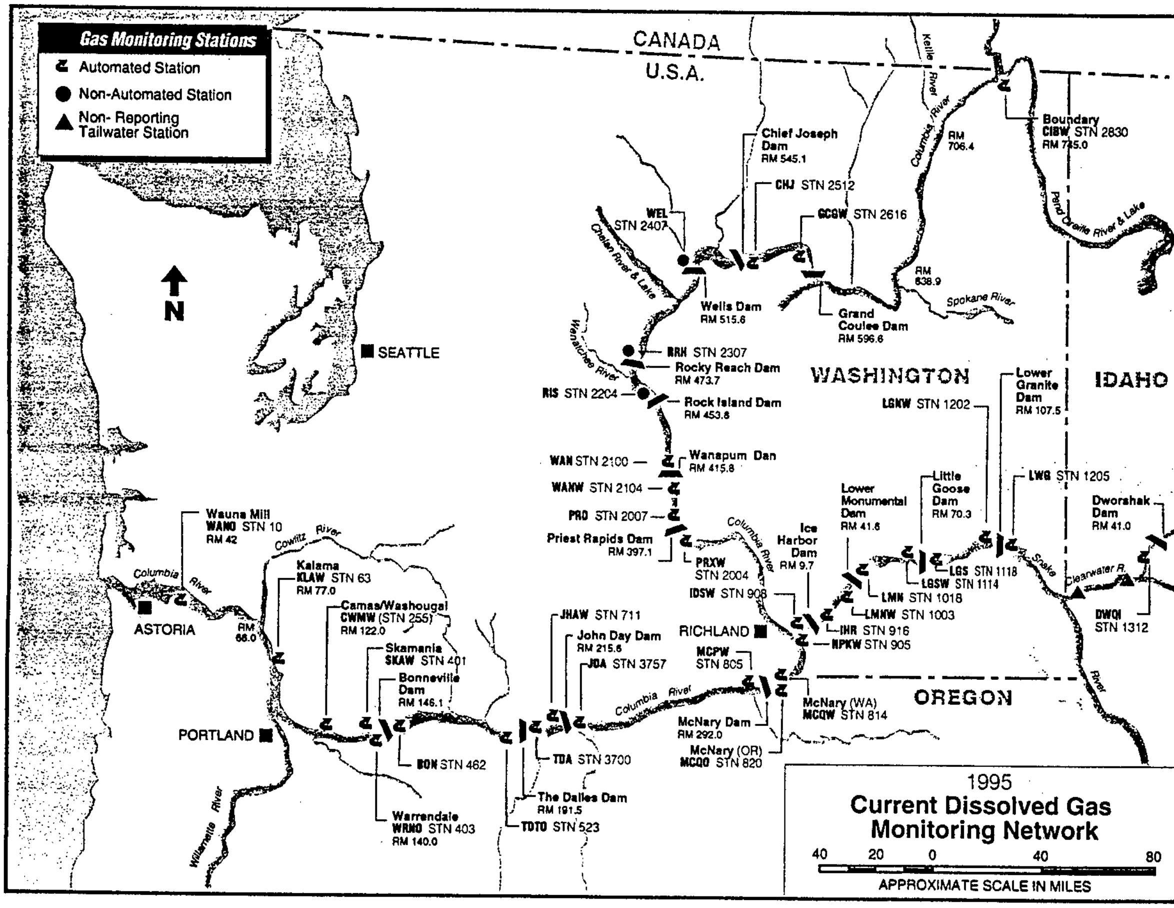

Figure 1. Columbia River projects

High levels of dissolved gas supersaturation are dangerous to migrating salmonids, and can cause gas bubble disease, a potentially lethal condition where dissolved gases in the blood come out of solution and form bubbles in various external and internal tissues. Sublethal exposure of dissolved gas can decrease swimming ability and stamina, impair sensory ability, affect growth and blood chemistry, which may result in susceptibility to predation and disease (Biological Opinion 1995).

Actual Operation

The BiOp established the following flow levels as targets to be met through flow augmentation for the 1995 spring and summer period: 95 and 51.4 kcfs, respectively, at Lower Granite on the Snake River; and, 249 and 200 kcfs, respectively, at McNary on the lower Columbia River. These targets were based on a April-July water supply forecast of 19,300 acre-feet (89 percent of normal) in the lower Snake River at Lower Granite and a January-July forecast of 99,600 acre-feet (94 percent of normal) in the lower Columbia River at The Dalles.

As specified in the BiOp, the spill program began in early April with Ice Harbor spilling to meet a 80 percent fish passage efficiency (FPE), subject to limitations caused by the Idaho standard for total dissolved gas. In mid-April, Bonneville, Little Goose and Lower Monumental began spilling for fish passage. Dworshak also began spilling in mid-April to provide more water for flow augmentation at Lower Granite. On April 20, NMFS received waivers from Oregon and Washington to allow TDG levels

Table 1
Spill Requirements

	1994 BiOp Spring	1994 BiOp Summer	1995 BiOp Spring[1]	1995 BiOp Summer[1]
Lower Granite	0	0	80	0
Little Goose	0	0	80	0
Lower Monumental	0	0	81	0
Ice Harbor	25kcfs nighttime	25kcfs 24hrs/day	27	70
McNary	0	0	50	0
John Day	0	20 percent daily, nightly spill	33	86
The Dalles	10 percent daily, nightly spill	5 percent daily, nightly spill	64	64
Bonneville	2	2	3	3

[1] Spill is in percent of instantaneous outflow. Spill periods are 24 hours at Ice Harbor, The Dalles, and Bonneville and 12 hours per day (1800-0600) at all others.
[2] Bonneville 1994 - no spill percent indicated, required to spill to provide juvenile fish passage efficiency of 70 percent in the spring and 50 percent in the summer is possible.
[3] Bonneville 1995 - no spill percent indicated, required to spill to provide juvenile fish passage efficiency of 80 percent if possible.

prescribed in the BiOp (12 hour average of 115 percent in the forebay and 120 percent in the tailrace), clearing the way for higher levels of spill and fish passage efficiency. By late April, BiOp spill was underway at all the lower Columbia River projects, McNary, John Day, The Dalles, and Bonneville and in early May, spill was initiated at Lower Granite.

Throughout the spring and summer seasons, TDG levels were held as closely as possible to the levels specified in the state waivers. This limited the ability to meet spill targets specified in the BiOp (Table 2). Only at Ice Harbor were spill targets met, but this was a result of limited turbine capacity which caused uncontrolled spill through much of the spring season. TDG levels in late May and for much of June exceeded the state standards, reaching upwards of 130 percent in the tailrace and between 115 and 120 percent in the McNary forebay. Efforts were taken to lower dissolved gas levels below Ice Harbor by reducing the flow in the lower Snake River and reducing spill at upstream projects to lower the dissolved gas concentration in the Ice Harbor forebay. Of the remaining projects, only The Dalles reached spill levels near the BiOp targets. Summer spill at The Dalles would have reached the target but for a spill reduction made in late August at the request of NMFS due to concerns over the cost of spill in lost power revenues and the low numbers of fall chinook subyearlings in-river migrating to the ocean.

Fish Passage Efficiency

The seasonal average FPE's reached during the spring and summer period are shown in Table 2. The FPE's were calculated using FGE's supplied by NMFS (Fredricks 1995) and used for 1995 FCRPS opinion modeling to determine spill levels for 80 percent fish passage efficiency. At Ice Harbor and McNary, the FPE goal of 80 percent was actually exceeded during the spring season. At Ice Harbor, this can be attributed to spill above the BiOp criteria due to high river flows and a lack of turbine capacity. At McNary, exceeding the FPE target

Table 2 Spill for Juvenile Fish Passage							
	Flow Target (kcfs)	Actual Flow (kcfs)	Spill Target Percent	Actual Spill Percent	FPE Target Percent	Actual FPE Percent
Spring						
Lower Granite	95	100.4	80	26.6	80	56.8
Little Goose			80	34.0	80	66.9
Lower Monumental			81	23.9	80	63.2
Ice Harbor			27	37.5	80	82.5
McNary	249	252.1	50	35.1	80	82.4
John Day			33	5.0	80	73.1
The Dalles			64	54.9	80	73.5
Bonneville				33.8		59.7
Summer						
Lower Granite	51.4	56.2				
Ice Harbor			70	43.7	80	66.3
McNary	200	166.3				
John Day			86	8.1	80	29.9
The Dalles			64	60.6	80	77.1
Bonnesville				49.6		61.0

was due to spill that occurred during daytime hours which was caused by flows higher than turbine capacity. All other projects failed to meet FPE goals of 80 percent. In the summer season, only The Dalles was able to spill close to the target level and reached a season average FPE of 77 percent. The inability to spill to meet FPE targets is clearly demonstrated at John Day, where spill was limited to only 8.1 percent which resulted in a FPE of only 29.9 percent. The spill limitation at John Day is amplified in the summer due to sharply lower FGE values for migrating juvenile chinook subyearlings as opposed to the higher value for spring and summer chinook migrating in the spring.

Flow Augmentation

Actual runoff for the April-July period at Lower Granite was 21 million acre-feet (97 percent of normal). At The Dalles the runoff for the January-July period was 104 million acre-feet (98 percent of normal). Seasonal targets were met at Lower Granite for the spring and summer and during the spring at McNary (Table 2). The ability to meet target flows was hampered somewhat by need to control the production of TDG. Dworshak releases were limited for much of the spring and summer by the production of TDG and was not able to release as much water as some would have liked to meet flow augmentation targets. Also, flows in the lower Columbia River had to be carefully managed during the spring period to avoid flows which would cause TDG to be exceeded at McNary and John Day.

Other Impacts

The BiOp required the lower Snake River projects, Lower Granite, Little Goose and Lower Monumental to operate at minimum pool during the 1995 fish passage season. These projects were filled during high river flows in May and June to reduce inflows at Ice Harbor to lower the production of TDG. These projects were not drafted back to minimum pool until late June when river flows receded and TDG levels at Ice Harbor dropped below 120 percent saturation.

The BiOp also requires turbine units to operate within 1 percent of peak efficiency during the fish passage season. For much of the spring turbines at Ice Harbor, and for a limited time at McNary and John Day, were allowed to exceed 1 percent peak efficiency and run at full overload to reduce the level of TDG production and avoid exceeding state standards.

Conclusion

With the need for higher spill for fish passage and stricter observance of state standards for TDG, the nature of managing river operations has changed. More than ever before water managers are forced to look at how to operate the reservoir system to control the level of TDG. Efforts are underway in many areas to overcome the problems associated with the production of TDG at the lower Snake and lower Columbia River projects. At the same time work continues on improving fish passage facilities to aid migrating fish and to reduce the need for excessive spill. By looking at the 1995 system operation it becomes apparent how much of an impact the production of TDG has on meeting fish passage criteria as specified in the BiOp. Each element, from flow augmentation, unit operation, reservoir management and spill for fish passage are effected.

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Hydrology, Environment, and the Shared Use of Water

by
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Hydrology, (streamflows, stages, lake levels) is related to environmental factors (fish and wildlife habitat, biodiversity, spawning and propagation of species). Methodologies have been developed to portray these interrelationships. Increased emphasis on the "environment" has elevated its importance in the analyses of alternatives in water resources decisionmaking related to the shared use of water.

Introduction

Since 1970 environmental awareness and concern has grown in sophistication and intensity. Federal laws such as the National Environmental Policy Act and the Endangered Species Act mandate informed decisionmaking and environmental protection. Litigation has reinforced the need for sound technical evaluations, appropriate coordination and adequate documentation.

Previously, environmental implications of water resources activities received limited consideration. Other uses such as flood control, hydropower, navigation, water supply, and irrigation were the principal focus. Many of the complex interrelationships between hydrology and the environment were not well understood. Increased environmental emphasis promoted the need for better ways to analyze and display environmental ramifications. Improved technology and scientific knowledge led to the development of methodologies including sophisticated computer models which simulate, predict, measure and promote a fuller comprehension of the consequences and tradeoffs involved. The focus is now moving toward a more holistic approach to addressing water resource issues. The State of Florida, for example, adopted "Ecosystem Management" to guide environmental planning and regulatory activities (Minasian 1994). Watershed planning, environmentally sustainable development, ecosystem analyses and system-wide management are now common terms.

Two studies by the U.S. Army Corps of Engineers provide insight into hydrologic and environmental relationships and illustrate various methodologies being used. The Missouri River Division (MRD) is conducting a study to review and update the Master Water Control Manual for the Missouri River (MRD 1994) and the Mobile District, in partnership with the States of Alabama, Georgia and Florida, is conducting a comprehensive water resources study of the Alabama-Coosa-Tallapoosa (ACT) and the Apalachicola-Chattahoochee-Flint (ACF) basins. An interdisciplinary approach synthesizing engineering and life/natural sciences skills, in a collaborative fashion, creates mutual understandings and tools that support more informed and balanced water resources planning and management. The two studies illustrate the breadth and complexity of the issues and highlight methodologies being applied.

MRD

The report containing output data from models on the array of alternatives considered in the Draft Environmental Impact Statement states:

"Understanding system hydrology, defined in terms of total system storage

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volume, lake levels and water releases, is crucial for evaluating the impacts that the operational alternatives will have on system beneficial uses such as fish, navigation and recreation." (MRD 1994)

Results are presented in the following resource categories: system hydrology (lake levels and river flows), wetland habitat, riparian habitat, tern and plover nesting habitat, young fish production in mainstem lakes, lake coldwater fish habitat, river coldwater fish habitat, river warmwater fish habitat, and native river fish physical habitat.

The hydrology and predicted responses to alternative water control plans were analyzed using the Long Range Study (LRS) Model. The LRS model was supplemented and revised to aid in the study (Patenode and Wilson 1994). An Environmental Impacts Model assesses key environmental resources for each alternative (MRD 1994). For example, wetlands and riparian habitats are influenced by reservoir and river hydrology. The model evaluates vegetative response to changes in monthly water tables for 13 cover types. Predictions of the general trends in endangered least tern and threatened piping plover nesting habitat employed a special model which tracked annual acreage of nesting habitat. The relationship between acres of habitat and flow was estimated using field verified aerial video-tapes (Tressler et al. 1994).

The analysis of young fish production in the main-stem lakes used a statistical regression model relating production and abundance indices of young fish to hydrologic factors, i.e., lake elevation, inundated area, inflow, discharge and flushing rate (MRD 1994). The analyses of lake cold water fish habitat was based on the extent of cold water retained in the reservoirs through the summer and fall seasons. A temperature regression model predicted effects on cold water river habitat downstream of the mainstem lakes. The habitat was measured in number of miles of river with a suitable temperature regimen. Warm water fish habitat was predicted in a similar fashion. Native river fish evaluation utilized an examination of physical habitat based on cross-sectional flow pattern comparisons.

ACT/ACF Study

The ACT is in Georgia and Alabama and the ACF also involves Florida, terminating in Apalachicola Bay (McClure and Griffin 1993). The hydrological analyses will use HEC-5 models developed for each basin. The models predict the hydrologic responses associated with reservoir operational scenarios and predicted water demands. Environmental analyses address wetlands and riparian water needs, protected species, riverine fauna water needs, reservoir fisheries water needs, and water needs of fish and wildlife management facilities.

Research conducted in the MRD study was utilized, especially in the fishery area. Wetlands and riparian resources are being evaluated using an ecological characterization of the resource (biological, physical, and hydrological). Potential effects to protected species will be evaluated by identifying potentially limiting ecological criteria and habitat requirements, followed by predicting potential impacts from stage/flow alterations based on the known life histories and habitat requirements.

The Freshwater Needs Assessment of the Apalachicola River and Bay is designed to determine what is needed to preserve the existing environmental diversity and productivity of this valuable ecological resource (TCG 1993). An interdisciplinary team is conducting physical, biological and chemical studies to assess the system's needs. A three dimensional model will be used to determine circulation, flushing and transport (nutrients) characteristics of the bay.

Conclusions

Water use and management strategies affect the hydrology and the environment. Analytical tools used to predict the biophysical ramifications of these activities are evolving toward a more holistic/ecosystem approach.

Decisionmaking should utilize this evolving technology in establishing environmentally sustainable and balanced water management strategies.

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Applicability of the General Conformity Rule to Corps of Engineers Projects

by
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Introduction

The 1990 amendments to the Clean Air Act (CAA) (42 United States Code 7401 et seq.) require Federal agencies to ensure that their actions conform to the appropriate State Implementation Plan (SIP). A SIP is a state plan that provides for the implementation, maintenance, and enforcement of the National Ambient Air Quality Standards (NAAQS), and includes emission limitations and control measures to attain and maintain the NAAQS. Conformity to a SIP, as defined in the CAA, means conformity to a SIP's purpose of reducing the severity and number of violations of the NAAQS and of achieving attainment of such standards. The General Conformity Final Rule, effective January 31, 1994, ensures that emissions from Federal actions conform to each host state's SIP.

History of Conformity

Conformity provisions first appeared in the CAA Amendments of 1977. Although these provisions did not define conformity, they did address the association between Federal activities in states with a SIP. The 1977 provisions stated that no Federal agency could engage in, support in any way or provide financial assistance for, license or permit, or approve any activity that did not conform to a SIP after its approval or promulgation.

Section 176(c) (42 USC 7506c) of the CAA Amendments of 1990 expanded the scope and content of the conformity provisions by defining conformity to an implementation plan. Specifically, the language requires that a Federal agency cannot approve or support an action that:

a. Causes or contributes to new violations of any NAAQS.

b. Increases the frequency or severity of existing violations of any NAAQS, or.

c. Delays the timely attainment of any NAAQS or any required interim emission reductions or milestones.

The purpose of Section 176(c) is to ensure that emissions from Federal actions are consistent with the CAA's air quality planning goals. The intent of the provisions is to foster long range planning for the attainment and maintenance of air quality standards by evaluating air quality impacts of Federal actions before they are undertaken. Federal actions are divided into transportation projects and non-transportation related projects. The "transportation conformity" regulations (40 CFR Part 51, Subpart T) govern projects developed or approved under the Federal Aid Highway Program or Federal Transit Act. Non-transportation projects, such as the McCook Reservoir and Indiana Harbor and Canal Confined Disposal Facility, are governed by the "general conformity" regulations discussed above.

States must draft General Conformity regulations at least as stringent as the Federal regulations. Therefore, a Federal agency should review their host state's conformity regulations, but if no state regulations exist, the agency can

[1] U.S. Army Engineer District, Chicago; Chicago, IL.

rely on the guidance contained in the Federal regulations.

General Conformity Determination Process

The General Conformity Rule consists of three major parts: applicability, analysis, and procedure. The information included in the following sections pertains to the requirements established by the Federal General Conformity regulations.

Applicability

Applicable Locations

The General Conformity Rule applies to Federal actions occurring in air basins designated as nonattainment or as attainment areas subject to maintenance plans (maintenance areas). The designation of nonattainment is based on exceedances or violations of air quality standards. A maintenance plan establishes measures to ensure that an area that previously was categorized as nonattainment maintains attainment status.

The conformity regulation does not apply to Federal Actions that are specifically excluded or are subject to transportation conformity. Additionally, Federal actions occurring in air basins that are in attainment with criteria pollutants are not subject to the conformity rule. A criteria pollutant is a pollutant for which an air quality standard has been established under the CAA.

De Minimis Emissions Levels

Threshold (*de minimis*) rates of emissions were established in the final rule, so that conformity requirements would focus on Federal actions with the potential to significantly impact air quality. With the exception of lead, the *de minimis* levels are based on the CAA's major stationary source definitions for the criteria pollutants (and precursors of criteria pollutants) and vary by the severity of the nonattainment area. A conformity determination is required when the annual net total of direct and indirect emissions from a Federal action, occurring in a nonattainment or maintenance area, equals or exceeds the annual *de minimis* levels. Table 1 lists the *de minimis* levels applicable to Federal actions.

Table 1
***De Minimis* Threshold Levels for Nonattainment Areas**[1]

Pollutant and Area Designation	Tons/Year
Ozone (VOCs or NO_x):	
Serious Nonattainment Areas	50
Severe Nonattainment Areas	25
Extreme Nonattainment Areas	10
Other Ozone Nonattainment Areas Outside an Ozone Transport Region	100
Marginal and Moderate Nonattainment Areas Inside Ozone Transport Region	
VOC	50
NO_x	100
Carbon Monoxide: All Nonattainment Areas	100
SO_2 or NO_2: All Nonattainment Areas	100
PM-10:	
Moderate Nonattainment Areas	100
Serious Nonattainment Areas	70
Pb: All Nonattainment Areas	25

[1] Source: 40 Code of Federal Regulations (CFR) Section 93.853(b)(1).

Regional Significance

A Federal action that does not exceed the *de minimis* levels of criteria pollutants may still be subject to a general conformity determination. The direct and indirect emissions from the action must not exceed ten percent of the total emission inventory for a particular criteria pollutants in a nonattainment or maintenance area. If the emissions exceed this threshold, the Federal action is considered to be a "regionally significant" activity, and thus, general conformity rules apply. The concept of regional significance is to capture those Federal actions that fall below the *de minimis* emission levels, but have the potential to impact the air quality of a region.

Analysis

The conformity analysis must be based on the latest and most accurate emission estimation techniques and must examine the impacts of (1) the direct and indirect net emissions from mobile and stationary sources, and (2) the emissions from any reasonably foreseeable Federal action. Indirect emissions include (1) those emissions the Federal agency can practicably control and has the responsibility to control, and (2) those caused by the Federal action that are later in time and/or still reasonably foreseeable. Reasonably foreseeable emissions are emissions from future Federal actions that can be quantified at the time of the conformity requirements.

Procedure

No documentation or public participation is required for an applicability analysis that results in a *de minimis* and regionally insignificant determination. However, the Freedom of Information Act requires Federal agencies to release *de minimis* and regionally insignificant determinations upon public request. Although the Federal General Conformity regulations do not address review of *de minimis* and regionally insignificant determinations, a state may require a formal review.

For actions that produce emissions that are not *de minimis* and/or regionally insignificant, the Federal agency must show that the emissions conform to the host state's SIP. For these projects, the Federal agency must provide a 30-day notice of the Federal action, and draft conformity determination to the appropriate EPA Region, and State and local air control agencies. Public participation is required for Federal actions that are not *de minimis* but are shown to conform, and the Federal agency must also make the draft determination available to the public to allow opportunity for review and comment.

Chicago District Projects

The Chicago District has completed General Conformity Determinations for two projects: the Indiana Harbor and Canal Confined Disposal Facility, and the McCook Reservoir Flood Control Project. The McCook Reservoir General Conformity Determination is the subject of the remaining discussion. The Chicago District used the Federal Conformity regulations as guidance, because the Illinois Conformity regulations had not yet been approved.

Project Description

The Tunnel and Reservoir Plan (TARP) is a regional plan developed in the 1960's and early 1970's to reduce flood damages and improve water quality in the Chicago metropolitan area watercourses. The goals of TARP include the following: 1) to prevent back flows to Lake Michigan, 2) to reduce waterway pollution caused by combined sewer overflows, 3) to provide an outlet for flood waters, and 4) to accomplish these goals in the most cost effective manner.

TARP includes three major features: 1) near surface collector and drop shaft systems that are connected to 2) underground conveyance tunnels that transport the floodwater to 3) storage reservoirs. The water stored in the conveyance tunnels and storage reservoirs will be pumped to and treated by existing water reclamation plants operated by the local sponsor, the Metropolitan Water Reclamation District of Greater Chicago (MWRDGC).

The McCook Reservoir is one of three reservoirs included in TARP. The reservoir will have the capacity to store 32,000 acre-feet of combined sewage up to a depth of 150 feet. A diffused-air aeration system will be used to maintain aerobic conditions in the combined sewage.

Analysis and Results

The Determination included a comprehensive analysis of the emissions resulting from the construction and operation of the McCook Reservoir. The emissions calculated in the study were those which were quantifiable based on the construction schedule. The draft analysis shows that the Federal action conforms to the Illinois SIP through a determination that the emissions from the Federal action are *de minimis* and regionally insignificant. A discussion of the overall analytical methodology, emissions by source, and conclusion of conformity are presented in the following sections.

Analytical Methods

The methodology used to complete a the general conformity determination consists of the following steps: (1) determining pollutants of concern based on the attainment status of the air quality control region; (2) defining the scope of the Federal action which includes timing and location; (3) calculating emissions based on the scope; (4) reviewing net emissions for threshold levels and regional significance; and (5) determining if the action conforms to the host state's SIP. The analysis was based on emissions that will be generated from the proposed reservoir design that is considered to produce the largest emissions estimate. The analysis includes estimates for the reservoir's construction and operation emissions. The construction schedule was used to determine the activities (1) that will produce emissions and (2) when emissions will occur.

The emission factors and equations applied in the analysis were taken from USEPA sources. The primary documents were for PM-10 calculations, Compilation of Air Pollutant Emission Factors, Volume I, 5th edition (1995); Compilation of Air Pollutant Emission Factors, Volume II (1985); Control of Fugitive Dust Sources (1988); and for NO_x calculations, The Compilation of Air Pollutant Emission Factors, Volume II: Mobile Sources, 4th edition (1985).

Pollutants of Concern

The area surrounding the proposed reservoir is a moderate nonattainment area for PM-10 (particulate matter with a diameter of less than ten micrometers), and a severe nonattainment area for ozone. Consequently, direct and indirect emissions of the following criteria pollutants are subject to a general conformity analysis: PM-10, and the precursors of ozone: volatile organic carbon (VOCs) and nitrogen oxides (NO_x).

If emissions for a criteria pollutant do not exceed the threshold limits used to determine *de minimis* emissions as specified in the final conformity rule, the Federal action conforms. The threshold limit for VOCs and NO_x is 25 tons/year per pollutant and for PM-10 is 100 tons/year. As summarized in the following sections, the McCook Reservoir conforms to the Illinois SIP because the emissions for each criteria pollutant are below their corresponding threshold levels and are not regionally significant.

Timing of Analyses

As specified in 40 CFR §93.159(d), emission analyses must be performed for the following years:

a. The year in which the state is to attain the National Ambient Air Quality Standards (NAAQS), or if applicable, the last year in which emissions are projected in a state maintenance plan.

b. The year in which the Federal action's total direct and indirect emissions are the greatest.

c. Any year in which the SIP for Illinois specifies an emission budget.

PM-10

The emissions resulting from the following phases of construction were estimated:

1) tunnels, 2) hydraulic structures, 3) reservoir main lobe: trim blasting and reservoir floor construction, 4) groundwater control system, and 5) aeration and wash down systems. The activities that produced these emissions were 1) blasting and drilling, 2) transporting cement, concrete, sand, and waste rock on unpaved roads, 3) scooping and dumping aggregates, and cement, and 4) grading and compacting aggregate and clay.

The McCook Determination included a PM-10 emission rate for the year in which the PM-10 emissions were expected to be greatest. The remaining two requirements as outlined in the above section did not apply.

PM-10 Results

The greatest rate for PM-10 emissions was expected to occur during construction. All of the identifiable PM-10 emissions were calculated using equations contained in USEPA document *AP-42: Stationary Sources*. The emissions from each phase of construction were calculated, and the results are tabulated in Table 2. The McCook Reservoir construction and operation is considered to be a *de minimis* source of PM-10 emissions because the total emissions are below the 100 tons of PM-10 per year limit.

Ozone

As required by the General Conformity regulation for severe ozone nonattainment areas VOC and NO_x emissions must be estimated. Internal combustion engines are the source of NO_x emissions during construction and operation of the reservoir. NO_x emissions were estimated using equations contained in USEPA document, *AP-42, Mobile Sources*. VOC emissions will occur only when the reservoir is in operation. The VOC emissions occur when the aeration system strips VOCs contained in the combined sewage and they are released into the atmosphere.

Table 2
PM-10 Emissions Results

Phase	Maximum Year Emissions (ton/yr)
Hydraulic Structures	10.6
Main Lobe	40.6
Groundwater Control	4.1
Aeration and Washdown	0.2
Total	55.5

NO_x Emissions

Based on requirements established by the regulations, NO_x emission estimations were calculated for the years (1) when the emissions were expected to be the highest, and (2) when construction or operation of the reservoir coincided with the years Illinois had established an emission budget. The largest yearly NO_x emission rate was estimated to be 17.1 tons of NO_x per year, below the 25 ton per year *de minimis* limit.

VOC Emissions

VOC emissions were estimated using the WATER8 model, a mathematical model developed and recommended by the USEPA. The reservoir was modeled as an aerated impoundment. The model uses the following pathways for VOC removal: biodegradation, surface volatilization, adsorption to biomass, and stripping by diffused air. The amount of VOCs stripped from the combined sewage is directly related to the influent concentration of VOCs. The influent VOC concentration was estimated by using VOC data contained in 1) a 1992 study of VOCs in the influent to the wastewater treatment plant receiving flows from the deep tunnel, 2) results from a monitoring program of organic priority pollutants in influent to the treatment plan, and 3) a 5-month industrial waste monitoring study. An air flow rate through the diffused-air aeration system of 25 tons/hour was used. The modeling estimated that the maximum VOC emissions would be 14.3 tons per year, below the 25 tons per year *de minimis* limit.

Conclusions

The criteria pollutant emissions from the construction and operation of the McCook Reservoir were estimated to be *de minimis* and not regionally significant. Therefore, the McCook Reservoir project conforms to the Illinois SIP.

For Corps Districts required to perform these types of analyses, it is recommended that the District coordinate with the appropriate regional and state environmental agencies. In some cases it may be desirable to include the results of the general conformity analysis in the environmental impact statement performed for the project.

References

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Our CE-QUAL-W2 Experience

by
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Introduction

The Water Quality Team of the U.S. Army Corps of Engineers, Louisville District, initiated a water quality modeling program in 1992. The CE-QUAL-W2 water quality model is being implemented as a tool to help evaluate current water quality and predict changes thereof under varied environmental conditions in our District lakes. Taylorsville Lake was selected as one of the first of our lakes to be modeled because of increasing problems with eutrophication and declining lake fishery. In 1990, the Taylorsville lake-Upper Salt River Basin was selected as a USDA "Non-point Source Hydrologic Unit Area." Partnerships were established between the Soil Conservation Service (now the Natural Resource Conservation Service), Kentucky Division of Water, U.S. Geological Survey, Kentucky Division of Conservation, Kentucky Department of Fish and Wildlife Resources, Kentucky Department of Health Services, University of Kentucky Cooperative Extension Service, Agricultural Stabilization and Conservation Service, and U.S. Army Corps of Engineers to develop protocol for Best Land Management Practices to reduce erosion and nutrient loading. The primary role of the Corps' is to use CE-QUAL-W2 to determine the extent to which nutrient loading must be reduced before seeing a positive effect on water quality over a given time period.

The objective of this paper is to share our initial experiences in the calibration and use of CE-QUAL-W2 with those already well established in the field and others whose expertise is as little as or less than ours. Material discussed can then be a sweet reminder of challenges overcome through experience, or a preview of those to come while experience is gained, respectively.

The Model

The CE-QUAL-W2 model is a 2-dimensional, laterally averaged hydrodynamic and water quality model developed by J. E. Edinger Associates, Inc. and the U.S. Army Corps of Engineers Waterways Experiment Station (Environmental and Hydraulics Laboratories 1986; Cole and Buchak 1995). The CE-QUAL-W2 model was designed to be used in conjunction with actual field measurements and data collection as an aid in data interpretation. The value of the model lies in its integrative use of physical, chemical, and biological information to describe potential changes in water quality. Model results may simulate natural conditions as long as certain specified assumptions hold true. Deviations of actual observed from model predicted values may indicate changing environmental conditions. Spatial and temporal variations of 20 water quality constituents in addition to temperature can be modeled. These include a tracer, inorganic suspended solids, coliform bacteria, total dissolved solids (salinity), labile DOM, refractory DOM, algae, detritus phosphorus, ammonium-nitrogen, nitrate+nitrite-nitrogen, dissolved oxygen, sediment, total inorganic carbon, alkalinity, pH, carbon dioxide, bicarbonate, carbonate, iron, and organic matter decay.

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CE-QUAL-W2 References

The CE-QUAL-W2 manual referred to in this paper is the most recent documentation available (Cole and Buchak 1995). The model version used for this study is CE-QUAL-W2 Version 2.05, 1994.

Sample Collection

Extensive sample collection was conducted in Taylorsville Lake in 1989 to show spatial and temporal variations in the distribution of nutrients and phytoplankton. Samples were collected every two weeks from 24 January to 24 October for 3 sites. The sample site located in a major tributary was designated an inflow site. The sample site located near the dam was designated a lake site. The sample site located in the upper reach of the main branch was designated as both a lake and inflow site.

Geometric Data/Bathemetric Input File

The first task to be completed was the mapping of the lake and contoured depth intervals of 1.52 m each. The lake was then divided into segments along its length approximating 1,000 m. Each layer within each segment represents one cell. Each cell has associated with it an average cross-sectional width which is used in conjunction with the layer height to determine cell volume. The volume-area table generated by the model is compared to the project volume-area table to assess the accuracy of the bathemetric data. Deviations in volume-area curves between model generated and standard project tables were corrected for by adjusting average cell cross-sectional widths.

When the bathemetric data file was completed, comparisons were made between model generated and observed water elevations as a means of evaluating the accuracy of the inflow and outflow data. Because outflows are more reliably determined based on reservoir pool elevation and incremental gate openings for water release through the dam than are inflows based on streamflow, runoff, and precipitation, deviations in temporal changes in lake elevation between predicted and observed curves were corrected for by adjusting total inflow water volume.

Model Constituents

The primary purpose of our study is to determine the extent to which various environmental factors, mainly light, nitrogen, and phosphorus, affect algal biomass and dissolved oxygen in Taylorsville Lake. We restricted our inflow constituent parameters of interest to suspended and dissolved solids, algal biomass, total inorganic phosphorus, dissolved nitrate+nitrite nitrogen, ammonia nitrogen, dissolved oxygen, alkalinity, pH, and total iron. Because of certain discrepancies between types of data we had available to us and those suggested in the manual, certain estimates and/or alternative choices had to be made for suspended solids, phosphorus, and detritus.

Suspended Solids

The constituent suspended solids consists of inorganic matter alone in the model program, however, suspended solids data collected in Taylorsville Lake in 1989 consisted of inorganic and organic fractions. We assumed an overestimation of inorganic suspended solids with the use of our data. Numerous model runs were conducted in which suspended solids concentrations were reduced by increments to determine the effect on other constituent model results. Effects were minimal, therefore, overestimation of suspended solids was of no major concern for this particular project at this time.

Phosphorus

SRP (soluble reactive phosphorus) is described as the form of phosphorus closest to that used in the model. However, we found that calibration for SRP may not always be possible. In one case, SRP data may not be available, e.g., in 1989, samples were not analyzed for SRP. In another case, SRP may not be detectable, e.g., SRP has been consistently below detection in all District lakes since we began its analysis in 1992. As an alternative, we substituted TIP (total inorganic phosphorus). We estimated TIP as TP (total phosphorus) minus TOP (total organic phosphorus). TP data were available, however, TOP was not. TOP was estimated based on measurements of TOC (total organic carbon) and stoichiometric relationships between carbon, nitrogen, and phosphorus as related to organic matter (Cole and Buchak 1995). Sources of inorganic phosphorus in the model include decomposition of algae, detritus, and organic sediments; and SRP entering the system in inflow waters. Inorganic phosphorus sources other than SRP are neglected. Inclusion of TIP rather than SRP may be more realistic in terms of phosphorus dynamics and associated algal biomass. We will be looking at this further.

Detritus

Inclusion of detritus was established later in the calibration process when it appeared necessary to have an additional organic matter source as an oxygen sink through decay. Oxygen uptake associated with algal decay and sediment and biological oxygen demands were not sufficient to result in timely oxygen depletion in the hypolimnion. The lake lies in a wooded and agricultural area. Resuspension of organic materials is evident upon destratification and mixing. Detritus seemed the best candidate for additional organic matter. Detritus was not collected in the original data set but was calculated as TOM (total organic matter) minus algal biomass. TOM was estimated based on the stoichiometric relationship between TOC and organic matter. In this case, detritus may have been overestimated because of the lack of distinction between dissolved and particulate organic carbon. Considering we did not have any measurements of dissolved organic matter, and that rates of decay for both are designated in the control file, combination of total and dissolved organic matter sources as detritus was acceptable to us.

Temperature/Constituent Calibration

Predicted and observed temperatures were well matched from the onset of the calibration process. Calibration for dissolved oxygen, algal biomass, nitrate+nitrite and ammonia nitrogen and total inorganic phosphorus proved to be much more complex. Much of the calibration process for Taylorsville Lake involved manipulation of rate constants associated with the algal component. Because all algal species, and their different responses to environmental conditions, are lumped together in one group, model calibration for algae and all constituents affected by or affecting algae will depend on average values over space and time. It is of great benefit to have some knowledge of temporal variations in algal composition in your system. Plans are being made to categorize total algae into green, blue-green, and diatom algal sub-groups. These sub-groups are in themselves broad but will provide greater refinement in model calibration.

Conclusions

What we learned so far in our CE-QUAL-W2 experience will be used in all our future modeling efforts. Certain points to be highlighted include:

a. Construct a flow chart showing a composite of all constituent interactions described in the manual to be used as a ready, easy reference.

b. Obtain a hard copy of the program to be used in conjunction with that on your computer to go through model mechanics step by step. This is especially helpful for understanding subroutines.

c. Compare equations and diagrams generated using the manual as reference to equations and/or diagrams generated based on the model program. Inconsistencies between information in the manual and what actually goes on in the model program can occur, e.g., the version of the model you may be using may be different from that on which the manual is based. Inconsistencies may even occur between different elements in the manual itself. In other words, know your program. If you have difficulty in the calibration of any constituent, an internal problem is a viable option. Check it out.

d. Keep a journal describing changes made to the control file and underlying reasons for those changes. Label and save all control files for future reference. You may decide to start again from a particular point in your calibration strategy. To have the actual control file is much easier than trying to backtrack based on information in your journal.

e. Create a graphics command program that allows you to compare predicted values for 2 or more runs simultaneously with the observed values. This greatly facilitates evaluation for improvements in fit.

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Cole, T. M., and Buchak, E. M. (1995). "CE-QUAL-W2: A two-dimensional, laterally averaged, hydrodynamic and water quality model, Version 2.0," Instruction Report EL-95-1, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

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Total Dissolved Gas Studies on the Lower Columbia: Instrument Performance Evaluation

by
Nicole A. Flint[1]

Introduction

As a component of a number of studies conducted on the lower Columbia River, the intent of the Transect Study portion of the Gas Abatement Study performed in the summer of 1995 was to quantify bias between fixed monitors and variability in cross sections of the river at fixed monitors. The Transect Study also provides general water quality and dissolved gas information to be expounded upon by further gas abatement studies. In an attempt to resolve questions concerning the bias and variability of data being collected throughout the transect study, a protocol was adopted, and adjusted over the summer, in order to respond to a continually evolving project. Quality assurance and quality control measures were employed in order to address concerns regarding calibration and response time of instruments and repeatability and comparability of measurements within and between instruments. The total dissolved gas data are presented in units of percent saturation. In the hopes of narrowing the variability affecting field data collection and representation, an attempt was made to develop a protocol that would support a higher level of confidence in the equipment used and, as a result, the data that evolved from this study.

Methods

Weekly calibrations involved adjusting meters in collaboration with USGS personnel at their Portland lab. Barometers used for calibration were first calibrated to weather stations located at the Troutdale airport on a weekly basis. Adjustments were made to the TDG instrument barometric reading if a 2 mm Hg difference from the standardized barometer or greater was observed. Calibration of the TDG probe commenced with visual inspection of the membrane for defects. Discoloration of the membrane's silicon tubing, or a release of air bubbles when submersed in water resulted in replacement of the membrane. For calibration in water, the probe was placed in water bath saturated with air. At this time, two or more instruments were calibrated to the water bath standard and adjustments were made if a difference in pressure of 2 mm Hg or more was observed.

In addition to weekly calibrations in the lab, daily adjustments were performed in the mornings before beginning sampling to ensure that there was minimal bias between the instruments. Barometric pressure was calibrated to a hand held standardized barometer, and air and water comparisons of TDG pressures were performed for all the instruments used that day. Instruments were compared with one another in river conditions similar to those expected to be encountered that day. Instrument calibration logs were kept in order that drift could be assessed for the instruments.

When measuring TDG pressure in the river, equilibration time was also a concern. Originally, readings that did not change by more than two millimeters of mercury over a one minute period were recorded, but this was amended to a more rigorous criteria. Based on continuing

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observations in the field, the new criteria for total pressure readings was to record the pressure when no measurable change occurred over two minutes. Equilibration usually was achieved within twenty to thirty minutes when the probe was changed from an air to a water environment. After a little experience in the field, it was determined that by leaving the probe in the water at a shallow depth while traveling at low speeds to nearby sites, a reduction in equilibration time of ten to fifteen minutes could be achieved. Multiple readings were also recorded to document the stability of the reading. This included three additional readings at two minute intervals immediately following the initial stabilized reading. This recording of additional pressure readings was implemented in the second synoptic sampling of transects at every seventh transect.

Sequential readings using one instrument and concurrent readings using two instruments were also recorded to quantify bias between instruments and variability over short time periods. Sequential readings consisted of taking readings at each of the five sites on the transect and then returning to one site for a second reading before moving on the next transect. The site to be repeated was predetermined so that the repeat reading was taken with as much time separating it from the first reading as logistically possible. A limited number of concurrent readings were taken. In this case, two probes were physically connected side by side, deployed in the water, and read simultaneously. This method usually utilized two different models, as new instruments and models become available and were incorporated in data collection.

Results

All results of total dissolved gases are discussed in units of percent. All readings were taken at a depth of fifteen feet, well below the compensation depth suggested by the manufacturer. No recommendations for specific instruments are being presented in this analysis. Rather, the integrity of the data and the matter in which the data were collected is the focus of this report.

Daily field calculations generally resulted in minimal adjustments to instrument barometric and total pressure settings, as exhibited in Figure 1 and 2. Most adjustments fell within 2 mm Hg, with a few at ± 4 mm, and a random 10 to 12 mm change occurring on occasion. One thing to note, however, is that with each positive adjustment, there was a complimentary negative adjustment of similar magnitude. In simple terms, most adjustments tended to cancel each other out. No changes were made to barometric pressure settings (BAR) 87% of the time, and 91% of all calibrations of total pressure (P_t) resulted in no change. From these observations, it seems apparent that drift within instruments is minimal and not significant.

Equilibration time per site was determined by noting the time lapse between the point at which the probe had been positioned on site and when the final reading was taken. In order to facilitate initial equilibration at each transect, the probe was deployed to a safe depth (as close to the prescribed fifteen feet as possible) while adjustments in location were made. The data represents the time elapsed from the initial reading upon arrival at each particular site to a time when no change in total pressure was measured for two minutes or more. The maximum amount of time necessary for equilibration was 25 minutes, but 95% of the 1,364 observations fell within an efficient two to eleven minutes (Figure 3). The number of minutes needed for a probe to come to equilibrium in relation to the distance below dams can be seen in Figure 4. Readings taken closer to dams and in areas in which mixing within the transect is not yet achieved, such as downstream of tributaries and near islands, took longer to come to equilibration. Weather (wind especially) also plays a part in mixing and may affect equilibration time. When fifteen or more minutes are necessary for equilibration, change of depth and initial equilibration after being the air influence the time span. For example, if the probe had

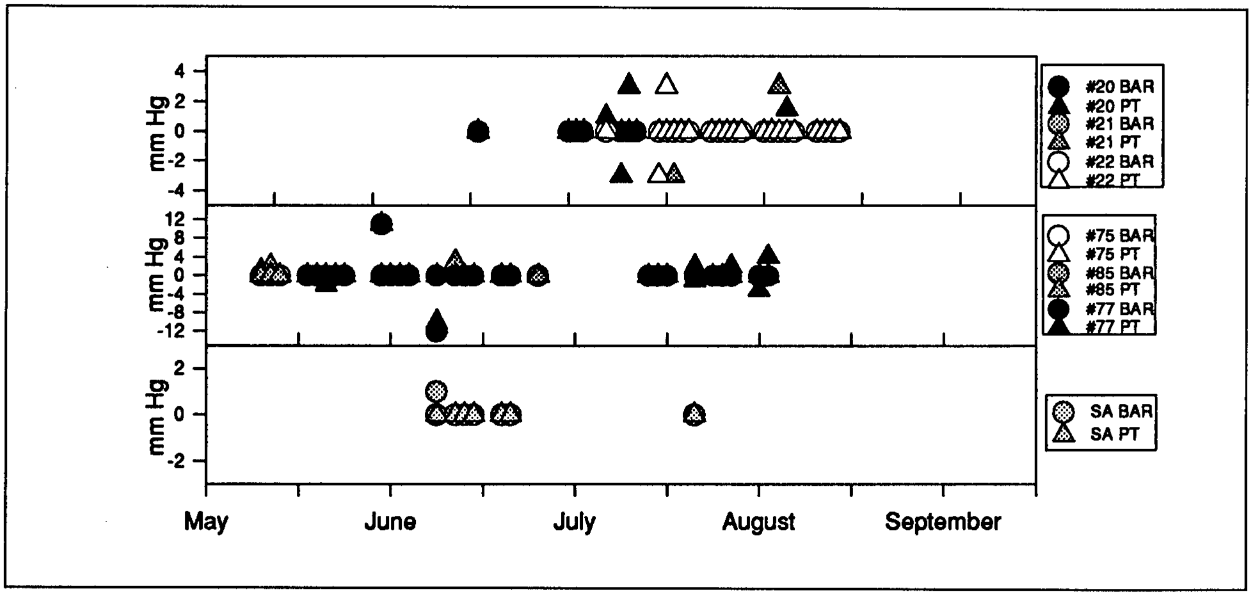

Figure 1. Instrument calibration. Changes made to instrument barometric and total pressure settings in mm Hg on various dates. Data is grouped according to model, with numbers assigned for each instrument

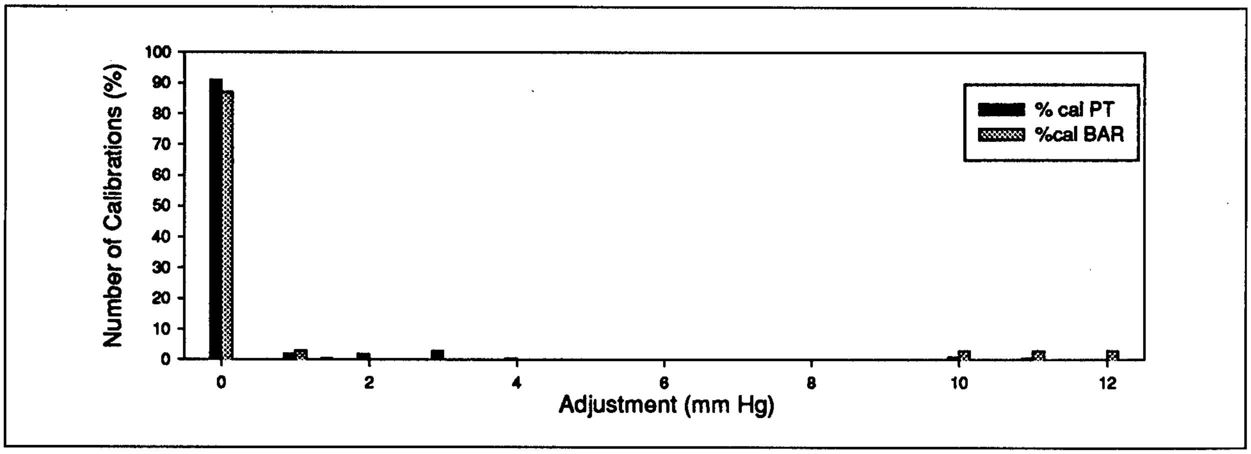

Figure 2. Instrument calibration. Adjustments in barometric pressure and total pressure of all instruments

been deployed and allowed to equilibrate at one depth and then was moved to another depth, more time often would be required to equilibrate. Similarly, if the probe had not been deployed from air to water early enough to achieve initial equilibration, additional time would be reflected in the reading for the first site of the transect.

Figure 5 shows the variation in sequential readings in two minute intervals following the initial reading with no measurable change in two minutes. This figure shows about 80%, 75%, and 65% of the first, second, and third sequential readings had 0.1% saturation or less change in reading. This indicates the variability in sequential readings at two minute intervals and also shows that by waiting for two minutes to take an environmental reading, the variability is low a majority of the time. Once again, these observations suggest that enough time was given for the instruments to equilibrate.

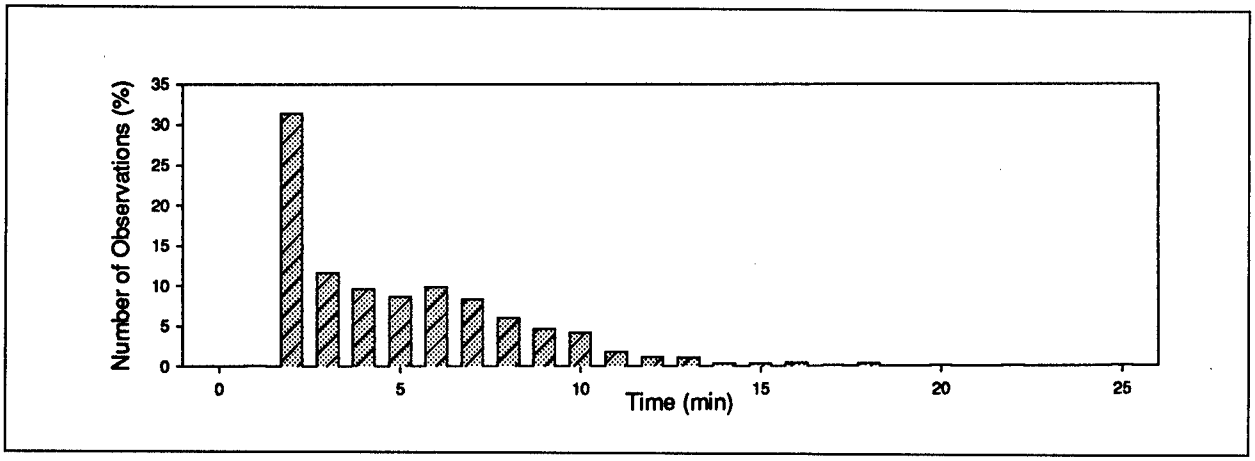

Figure 3. Instrument equilibration time. Minutes elapsed for equilibration versus number of observations expressed as percentage

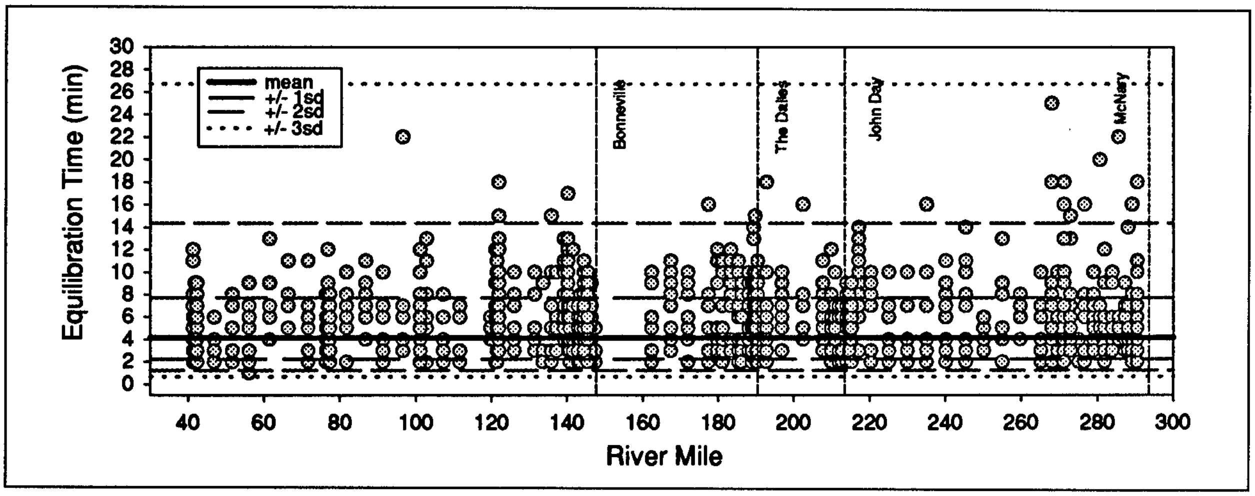

Figure 4. Instrument equilibration. Time elapsed between initial and final readings per site. River mile increases in upstream direction. Dams are represented by vertical lines

Sequential pressure readings of TDG were collected and evaluated in determining reproducibility of the data collected on each transect in the second set of data. As Figure 6 reveals, a difference of 0.1% TDG saturation or less between Replicates #1 and #2 was observed 29% of the time. Of the 92 total observations, 95% fell within 2.2% difference in saturation. Variability in readings was greater closer to dams in some instances, such as below Bonneville Dam. This reflects the turbulence and premixing conditions that are expected in these regions (Figure 7). Other areas of variation may correspond to confluences of tributaries and changes in weather conditions (wind) between replicate readings. An additional concern, as noted in several field observation sheets, is the influence of photosynthetic activity. This may express a need for further investigation. Repeat data was also evaluated in order to investigate repeatability at various levels of saturation (Figure 8). Between 104% and 109%, it appears that variability between repeat readings is for the most part relatively low. At higher saturation levels, though, there appears to be a less precise distribution. This also points to greater variability and higher saturation levels beneath the dams.

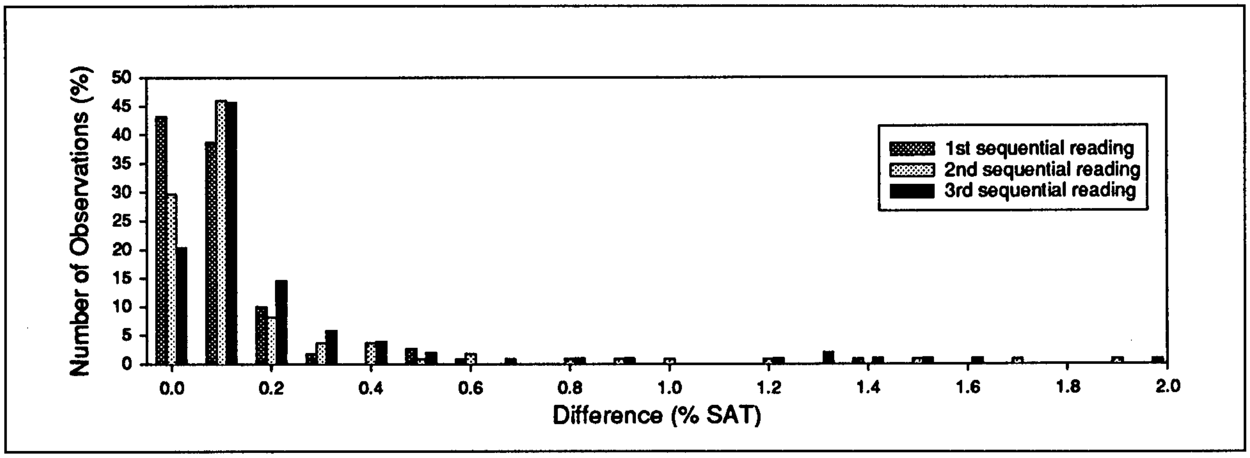

Figure 5. Sequential readings in two minute increments at the same location. Differences in percent saturation

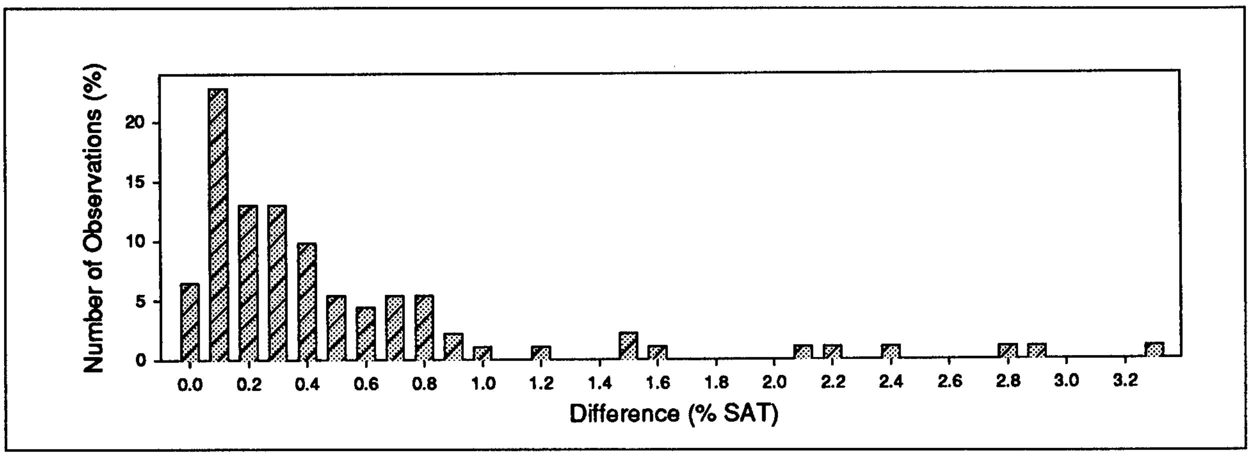

Figure 6. Repeats evaluation. Difference in total dissolved gas percent saturation between repeat readings over short time periods expressed as percentage versus number of occurrences

Concurrent readings were taken to further quantify the bias among instruments. Figure 9 shows that 95% (two standard deviations) of the readings fell within 0.9 of each other. No difference between the instrument readings was observed 11% of the time. These data show a low level of bias between instruments and support a high level of confidence in the readings that were taken.

Conclusion

Though direct analysis of error due to bias is difficult, variability among and drift within a number of instruments can be analyzed in order to minimize concern in regards to their performance. On the whole, the instruments performed exceptionally well, once a suitable protocol was established. Calibrations required minimal adjustments in most cases, indicating that drift within instruments was negligible. Equilibrations was adequately achieved once a stabilization for two minutes was reached. Replicate readings demonstrated a general high degree of repeatability and low variability. The analyses presented support the protocol that was adopted and the data that were collected.

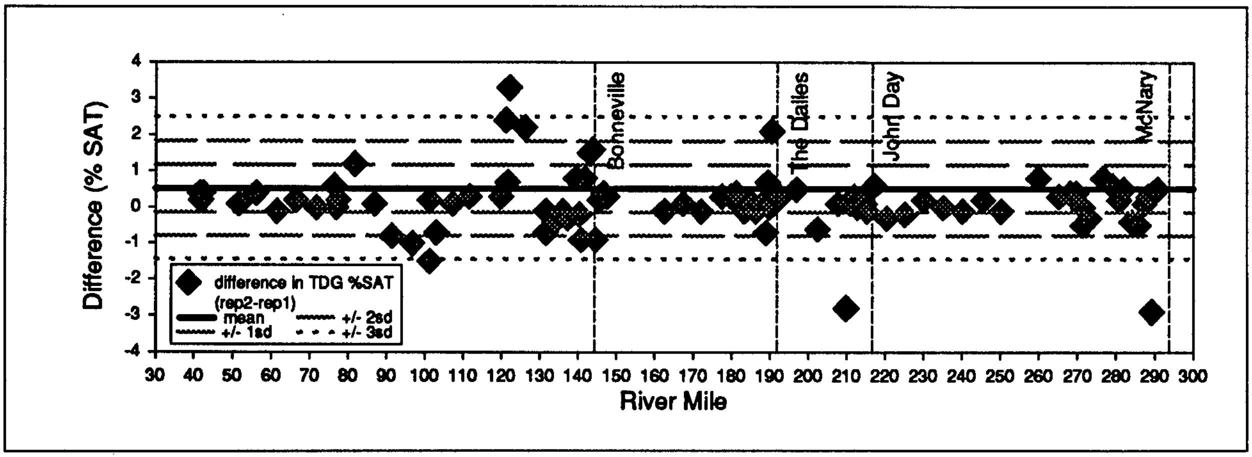

Figure 7. Sequential pressure readings. Difference in percent saturation between repeat readings compared to river mile. Dams are shown as vertical lines

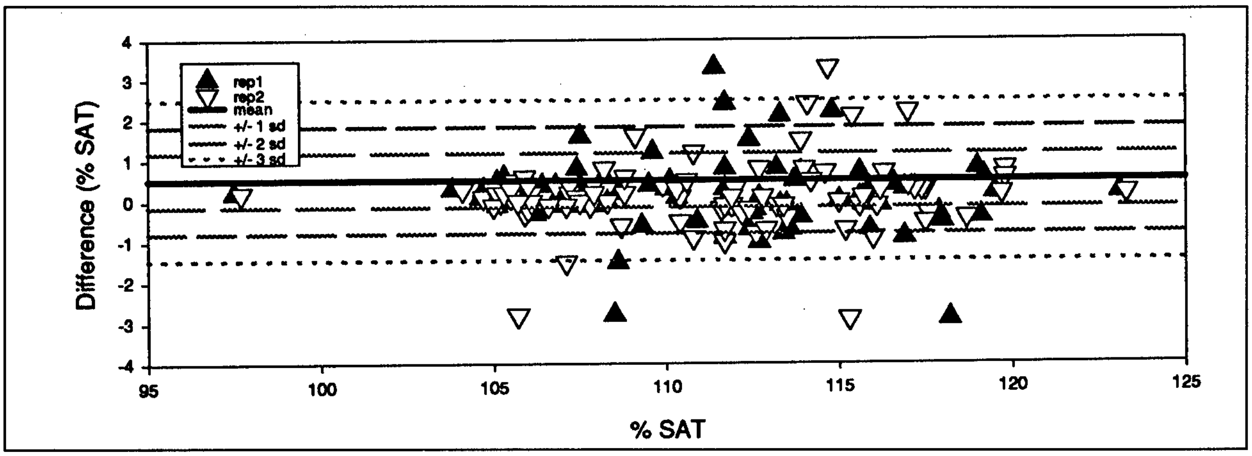

Figure 8. Sequential pressure readings. Difference between repeat readings plotted against the percent saturation of the initial reading

Another prime factor is that the environment we attempted to characterize is a dynamic, flowing parcel of water, subject to the progression of time and changing meteorological conditions. Using multiple readings over time and space, it was possible to get a picture of the conditions that existed for this parcel of water. This evaluation provides a measure of variability due to instrumentation in lending credence to the study.

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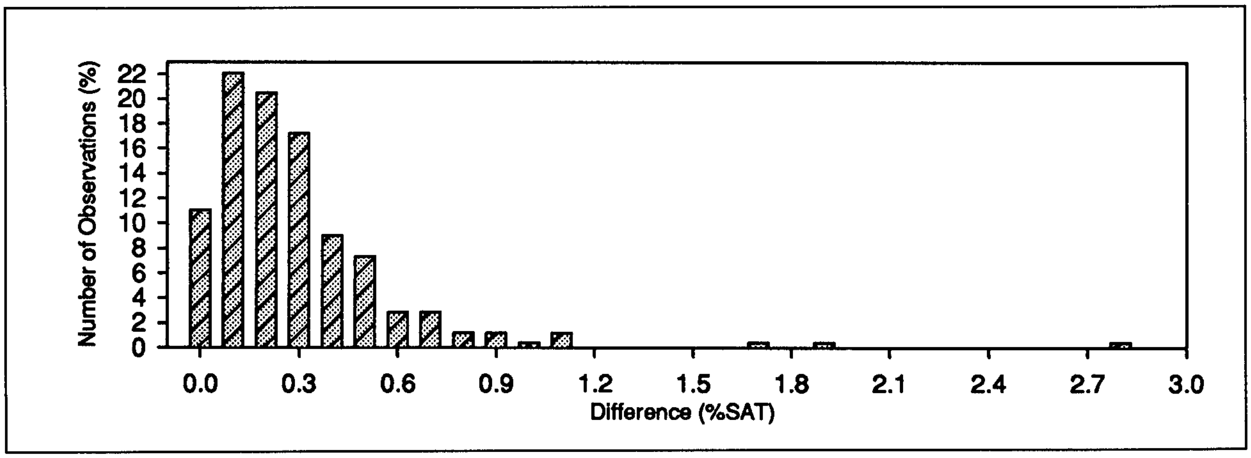

Figure 9. Replicate readings. Difference in total dissolved gas percent saturation among two or more instruments at the same site

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Assessment of Monitoring Total Dissolved Gas at Tims Ford Dam

by
Helen G. Rucker[1] *and John M. Higgins*[2]

Introduction

Total dissolved gas (TDG) monitoring has become a concern at several Tennessee Valley Authority (TVA) hydro facilities. The releases of dams have historically been known as a source for elevated TDG levels or dissolved gas supersaturation; therefore the Environmental Protection Agency (EPA) has established a TDG guideline of 110 percent (Water Quality Standards Handbook 1992). At some hydro facilities, reservoir releases can become supersaturated with atmospheric gases during force aeration or flood control spill operations. Monitoring is needed to understand and avoid adverse effects of these operations.

Historical operating conditions were changed at several TVA facilities as the result of the Lake Improvement Plan approved by the TVA Board of Directors in 1991. One of the goals of this plan was to improve DO concentrations in the releases from 16 dams. Methods to increase DO concentrations included downstream weirs, turbine venting, reservoir diffusers, and forced air and oxygen injection equipment in the penstock, scrollcase, turbine, and draft tubes. TDG monitoring was identified as a need for projects such as Tims Ford where forced air injection created the potential for adversely affecting aquatic life. TDG is important because in this situation a single supersaturated gas such as oxygen or nitrogen may not result in detrimental effects on aquatic life. Particularly at Tims Ford, there is potential for elevated TDG levels because air is injected in pressurized water from the low level intakes, then this water is released where it is no longer under the hydrostatic pressure in the reservoir.

Site Description

Tims Ford Dam is located on the Elk River in middle Tennessee. It is a multipurpose dam which provides flood control, water supply, and hydropower generation. The hydroelectric plant has two units: a generating unit rated at 3890 cu ft per second (cfs) and a small unit rated at 74 cfs. The small unit is used to maintain minimum flows when the large unit is not in use. Releases from Tims Ford Dam typically drop below six milligrams per liter (mg/L) in mid-April (see Figure 1) and continue to decline to less than 1.0 mg/L by August and stay below 1.0 mg/L until late December. The low DO concentrations are caused by a combination of factors, including reservoir stratification, the oxygen demand of allochthonous or autochthonous organic materials, and low level penstock intake. The data in Figure 1 show the historical DO values from 1972 through May 1993. The 10 percent-line indicates that 10 percent of all DO data is below that line and the 90 percent-line indicates that 10 percent of all DO data is above that line. These two lines yield a band which encompasses 80 percent of all historical DO data.

The combination of air and oxygen injection equipment is used to meet the target DO

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Reservoir Release Dissolved Oxygen Concentrations in 1995

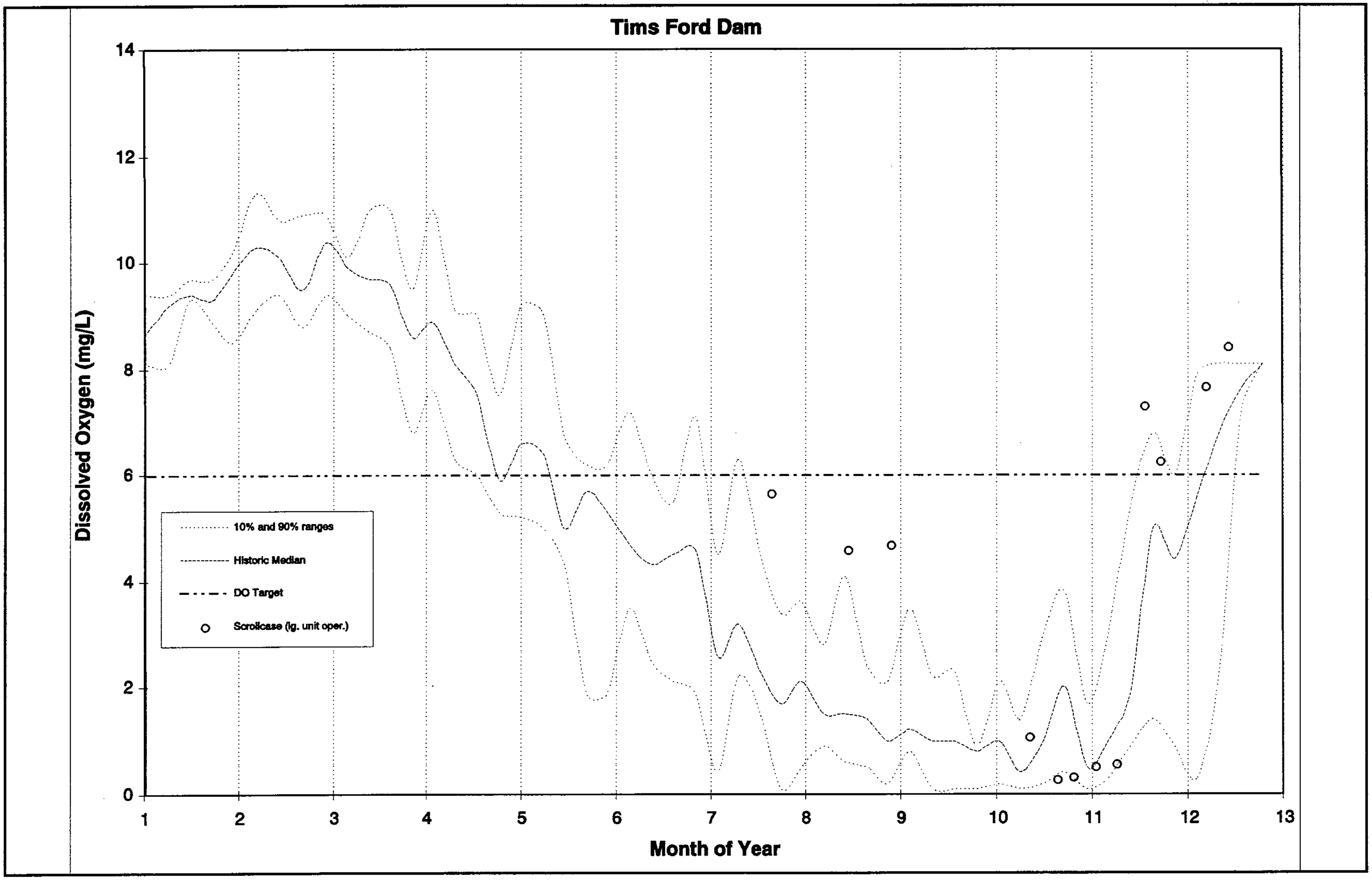

Figure 1.

concentration of 6 mg/L in the releases from Tims Ford Dam. Two blowers and three compressors are available to inject air into the large and small hydroturbine units, respectively. An oxygen injection system supplies oxygen to the penstock for the large unit and the sluice line for the smaller unit (Harshbarger et al. 1995). The oxygen injection system is used when target DO concentrations can not be met by air injection. A performance system monitors the effectiveness of the aeration system. The system collects date, time, unit discharges, oxygen flow rates, air flow rates, headwater and tailwater elevations, DO, and water temperature data every five minutes. The data are used for operational decisions and to track system performance.

During weekly site visits, water quality grab measurements are made to verify the accuracy of the performance monitoring system and standardize field monitors as needed. DO and water temperature measurements are made to standardize the probes for the continuous field monitor and to validate water temperature and DO data collected by the performance monitoring system. TDG measurements are made because the forced air injection creates a potential for adverse impacts on aquatic life.

TDG Monitoring

'Dissolved gas supersaturation has been recognized since the turn of the century as a potential threat to aquatic organisms due to a resulting condition classically known as gas bubble disease' (Marking 1987). Excess gas in the water forms gas bubbles which tend to collect on the surfaces in the water, including fish. Osmotic pressures on both sides of the gill membrane tend to equalize and because blood and water have nearly the same saturation point, the gas excesses are about equal in the blood stream and water. The cause of death in gas bubble disease is usually asphyxiation, caused by gas emboli in the gill filaments or heart or both (Marking 1987). The effects of higher TDG levels are a concern for the aquatic life below Tims Ford Dam particularly because the State of Tennessee is developing a trout fishery immediately downstream of the dam to develop and promote recreation and economic development.

Continuous monitoring is desired because of the wide range of variability of operating conditions at the site and the variable TDG results from these conditions. 'Continuous monitoring may be desirable if gas supersaturation varies by more than 2 percent from day to day or if treatment is to begin automatically whenever a preselected level of saturation occurs.' (Marking 1987). Sufficient historical TDG data is not available to determine operational policies for the wide range of varying conditions. Continuous monitoring also helps reduce potential operator error made by inexperienced operators using the saturometers for grab measurements as observed by D'Aoust and Clark (1980) and identified in a TVA comparative study between the saturometer and gas chromatography.

Weekly TDG grab measurements at Tims Ford Dam indicated the EPA limits could be approached. Special tests were conducted in 1994 under different operating conditions. These tests indicated that during certain times of the year and under certain conditions, the potential exists to exceed the EPA guidelines. To avoid potential problems, the small unit compressors are not operated until the system can be evaluated further.

TVA did not have capabilities for monitoring TDG continuously. A literature review was conducted to determine current industry standards for continuous monitoring. TVA wanted to evaluate a continuous TDG monitor that could be integrated with the existing water chemistry (DO and water temperature) monitoring system.

In order to evaluate the reliability of a continuous TDG sensor, TVA conducted a six week study in the tailwaters of Tims Ford Dam. In addition to Tims Ford Dam having air and oxygen injection equipment, another reason for the selection of this particular site for the study is that the water discharged from the hypolimnion of Tims Ford Reservoir contains an organic allochthonous or autochthonous slime material that occasionally clogs and fouls the DO sensor probe, thus giving erratic and erroneous results. Theoretically, fouling of the TDG probe should have less effect on TDG data because the fouling should not inhibit the gases from moving through the tubing. Therefore daily DO trends could be compared with the percent saturation trends for accuracy and minimize site visits for probe maintenance. Operation of the aeration system could be maximized to reduce cost for liquid oxygen.

Study Design

The purpose of this study was to identify and test a reliable continuous TDG monitoring system that would collect reliable information for further evaluation of the aeration system. A six week time period from July 20 - August 31, 1995 was selected for the study. This time period covered the low DO season and time when excessive fouling normally occurs. Gas chromatography laboratory testing was considered but not included because previous comparative studies indicated that the membrane diffusion method employed by the saturometer was sufficiently reliable (Fickeisen et al. 1975 and TVA field study). The selected tensiometer also employed the membrane diffusion method.

Several criteria were considered when selecting the instrument for evaluation. The instrument would need to provide and record continuous TDG data unattended. The data obtained would have to compare with the accuracy of the saturometers, even under excessive fouling during slime buildup on the instrument. The performance monitoring system requires a 4-20 milliamp signal input. The instrument chosen for evaluation would need capabilities for integration with this system. Commercially available TDG monitoring equipment was assessed and a sensor was selected for evaluation that could be integrated with the existing DO and water temperature field monitor.

The TDG monitor was installed on July 20, 1995, and integrated with the existing DO and water temperature monitoring system. Since the study was temporary, the TDG monitor was not permanently added to the performance monitoring system. The data was logged with a commercial data logger and retrieved on a weekly basis. The monitoring location was off the taildeck at a four foot minimum depth to reduce bubble formation on the tubing. The monitor was checked on a weekly basis, data retrieved from the unit, and comparison checks with saturometers were performed by field personnel. The tensiometer was compared to the saturometers for field standardization purposes. For quality assurance, field Quality Assurance (QA) procedures for measuring TDG with two WEISS® Saturometers were followed to ensure the accuracy of the instruments and proper use by field personnel. Afterwards, the tensiometer was allowed to dry and equilibrate with the atmosphere to see if it returned to the original set point. The tensiometer was also inspected for any physical damage, deterioration, and/or fouling. The DO and water temperature probes were maintained and standardized at this time also.

Results

On July 20, 1995, prior to installing the system in the monitoring location, TDG measurements were taken downstream, where two saturometers and the tensiometer could be placed concurrently in situ to obtain benchmark readings. On August 21, 1995, a portable tensiometer of the same model was used to obtain more benchmark readings with the saturometers. During all comparisons, readings from both instruments were within two percent or less TDG percent saturation (see Figure 2) and the tensiometer reached equilibrium more quickly than the saturometers.

Continuous data collected from the monitor included time, date, water temperature (°C), DO (mg/L), Delta P (mm Hg), and local barometric pressure (mm Hg). In comparing the DO values to the Delta P measurements,

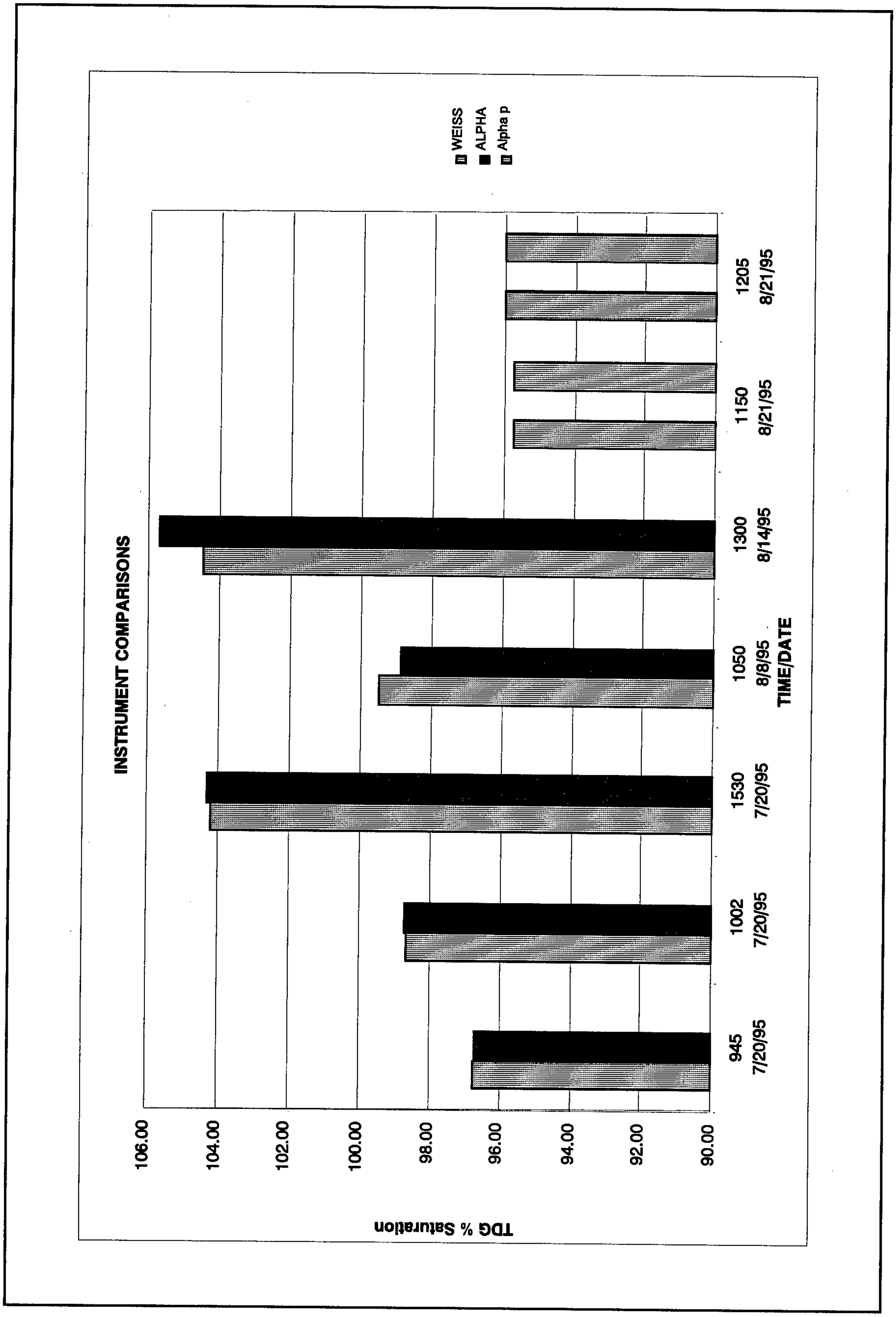
INSTRUMENT COMPARISONS
TDG % Saturation
106.00
104.00
102.00
100.00
98.00
96.00
94.00
92.00
90.00
945 7/20/95
1002 7/20/95
1530 7/20/95
1050 8/8/95
1300 8/14/95
1150 8/21/95
1205 8/21/95
TIME/DATE
WEISS
ALPHA
Alpha p

Figure 2.

peaks indicating a change in DO are relative to peaks in the Delta P, as shown in Figure 3. The changes are contributed to the different operating conditions at the dam, e.g., the big unit generating with blowers injecting air into the discharges.

Data was imported into a spreadsheet where TDG percent saturation, nitrogen and argon percent saturation, and oxygen percent saturation were calculated. Data are presented in line charts with each parameter of TDG percent Saturation, N_2 + Ar percent Saturation, and O_2 percent Saturation plotted separately against the DO data curve for comparison (see Figures 4, 5, and 6). In comparing the DO values to the nitrogen and argon percent saturation, the distinct peaks on the latter curve represent the higher nitrogen values from the blowers during the big unit operation which inject air into the discharges (see Figure 5). A line chart representing all data curves is presented in Figure 7.

The tensiometer provided consistent data with the saturometers within two percent of TDG percent saturation. No adjustments to the monitor were necessary during the study. The data logger recorded data unattended every thirty minutes and stored data which was retrieved weekly. An analog output module allows permanent integration capabilities into the current performance monitoring system.

Heterotrophic conditions were not present during the evaluation and therefore performance under these conditions could not be evaluated during the study. The absence of these conditions is probably because the DO deficit in the reservoir was much delayed due to low inflows. Most likely the low inflows reduced organic/nutrient loading in the reservoir. Reduction in organic material in the reservoir should have reduced the severity of the DO depletion and the allochthonous-produced slime material that contribute to the fouling normally seen in the releases.

Conclusions

The tensiometer successfully monitored TDG at Tims Ford Dam during this evaluation. Variations seen in the TDG data are closely associated with the variations in DO concentrations caused by the air and oxygen injection. The continuous monitor agreed closely with the saturometer field checks. Adjustments to the monitor and/or data were not necessary.

Based on the results of this study, TVA is continuing the evaluation of this equipment at Tims Ford Dam and extending the evaluation to the releases of Hiwassee and Kentucky Dams. Portable units are being used for grab measurements in other valley reservoirs.

Acknowledgments

The authors greatly acknowledge the assistance of Jackie A. Fowler, TVA Environmental Technician, for providing quality field data and Ed Heidel, AMJ Equipment Inc, for assisting with system integration.

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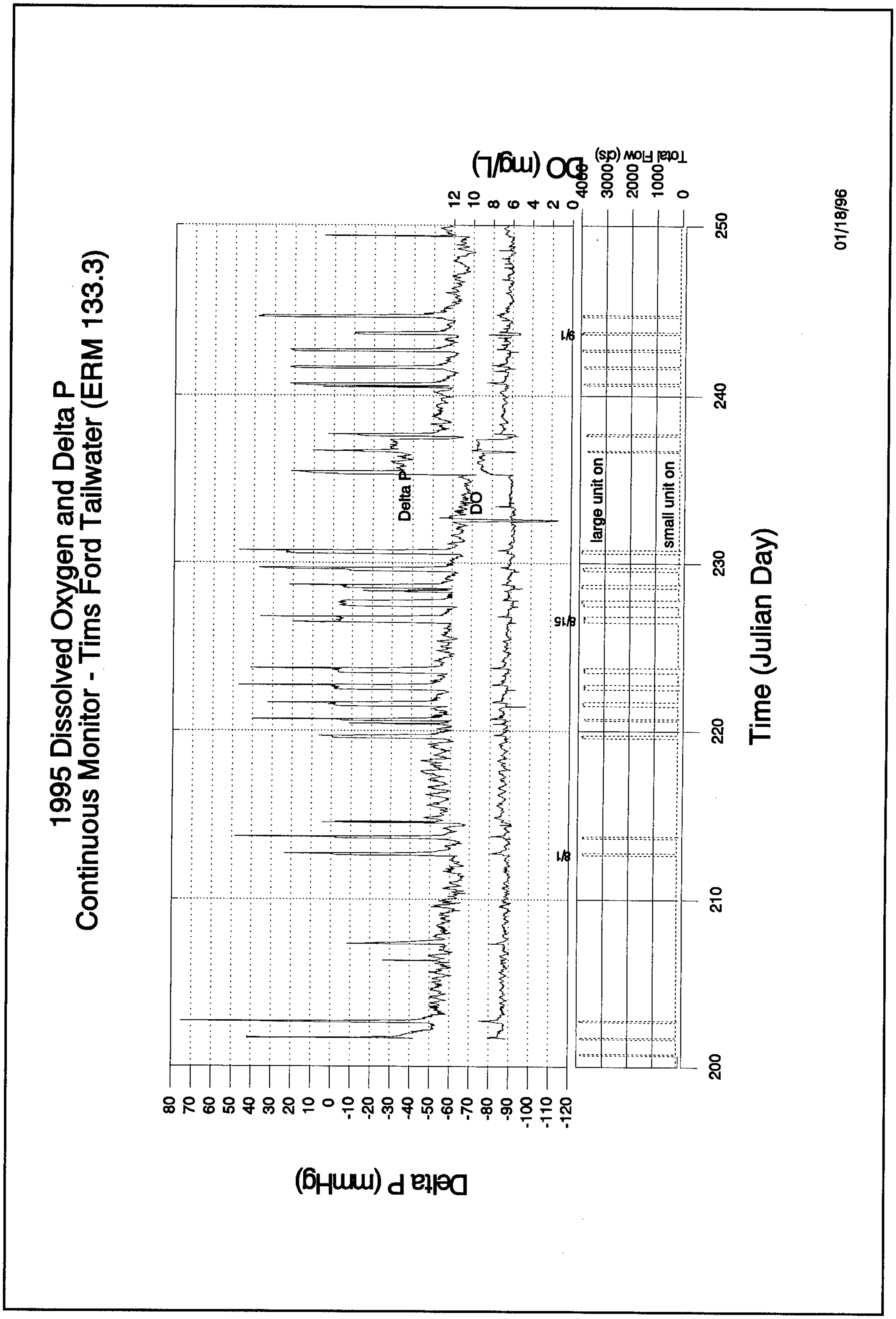

1995 Dissolved Oxygen and Delta P
Continuous Monitor - Tims Ford Tailwater (ERM 133.3)
Delta P (mmHg)
DO (mg/L)
Total Flow (cfs)
Delta P
DO
large unit on
small unit on
8/1
8/15
9/1
Time (Julian Day)
01/18/96

Figure 3.

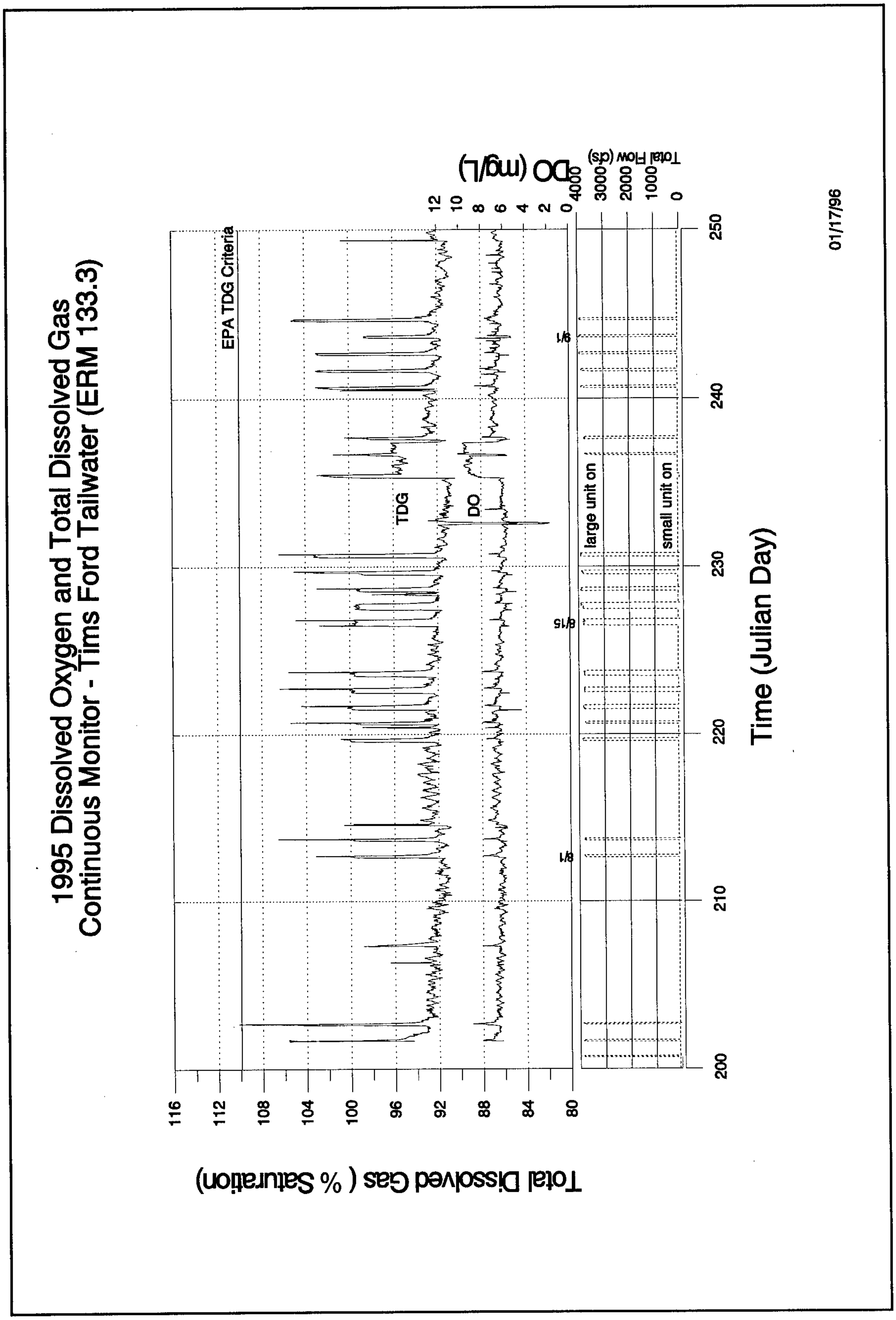
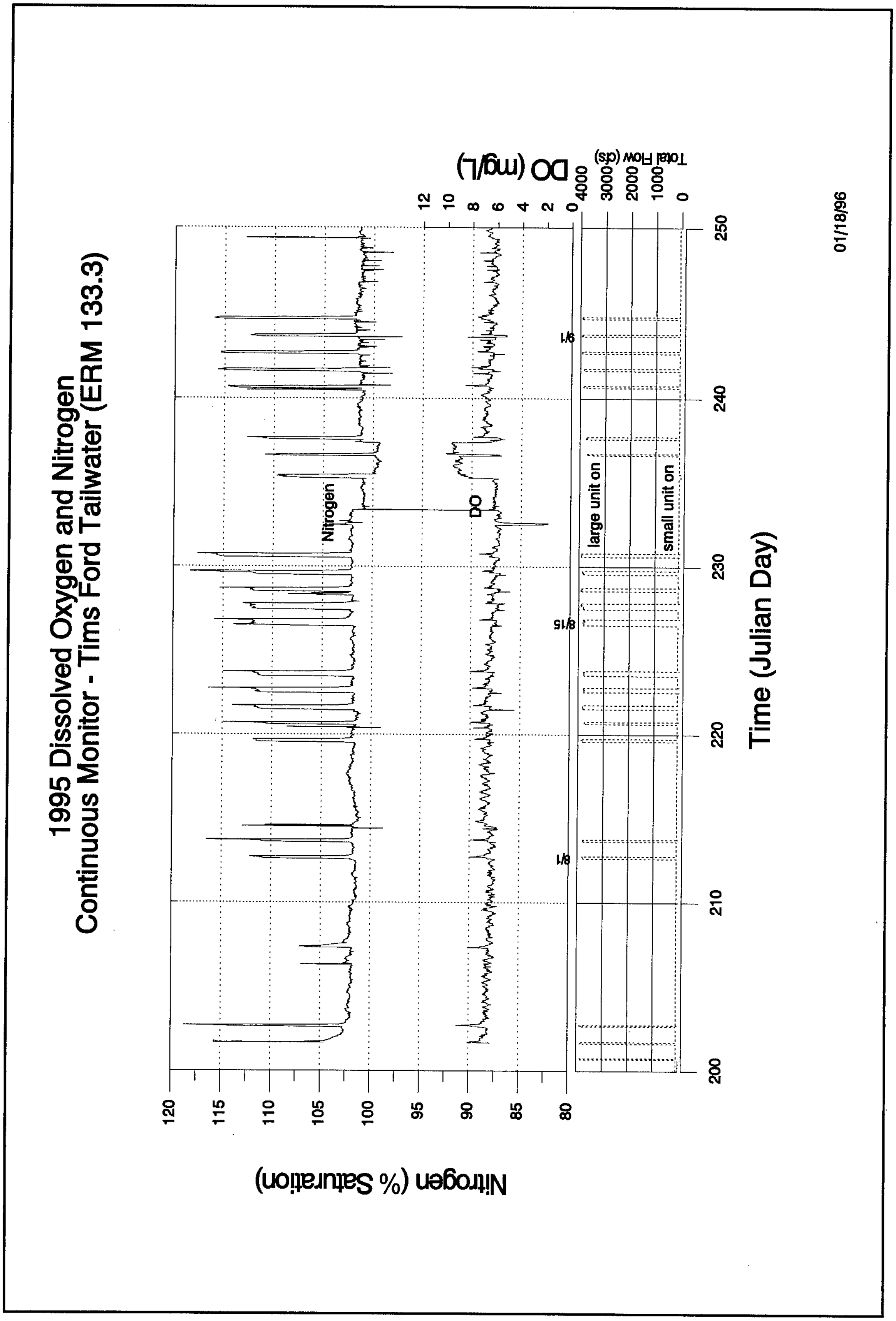

1995 Dissolved Oxygen and Total Dissolved Gas
Continuous Monitor - Tims Ford Tailwater (ERM 133.3)
Total Dissolved Gas (% Saturation)
116
112
108
104
100
96
92
88
84
80
EPA TDG Criteria
TDG
DO
DO (mg/L)
12
10
8
6
4
2
0
Total Flow (cfs)
4000
3000
2000
1000
0
8/1
8/15
9/1
large unit on
small unit on
200
210
220
230
240
250
Time (Julian Day)
01/17/96

Figure 4.



1995 Dissolved Oxygen and Nitrogen
Continuous Monitor - Tims Ford Tailwater (ERM 133.3)
Nitrogen (% Saturation)
DO (mg/L)
Total Flow (cfs)
Time (Julian Day)
Nitrogen
DO
large unit on
small unit on
8/1
8/15
9/1
01/18/96

Figure 5.

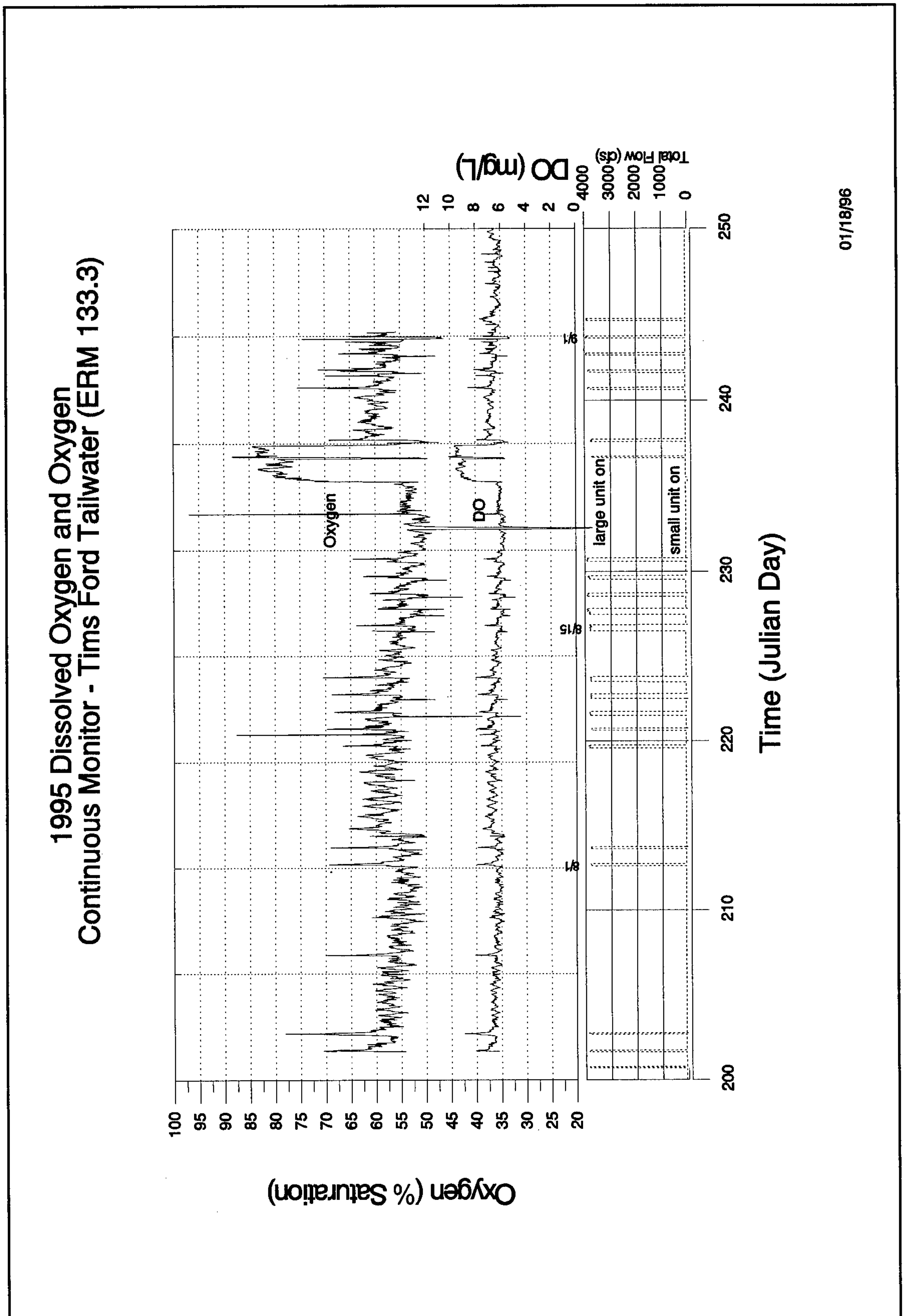

1995 Dissolved Oxygen and Oxygen
Continuous Monitor - Tims Ford Tailwater (ERM 133.3)
Oxygen (% Saturation)
100 95 90 85 80 75 70 65 60 55 50 45 40 35 30 25 20
DO (mg/L)
12 10 8 6 4 2 0
Total Flow (cfs)
4000 3000 2000 1000 0
Oxygen
DO
large unit on
small unit on
8/1
8/15
9/1
200 210 220 230 240 250
Time (Julian Day)
01/18/96

Figure 6.

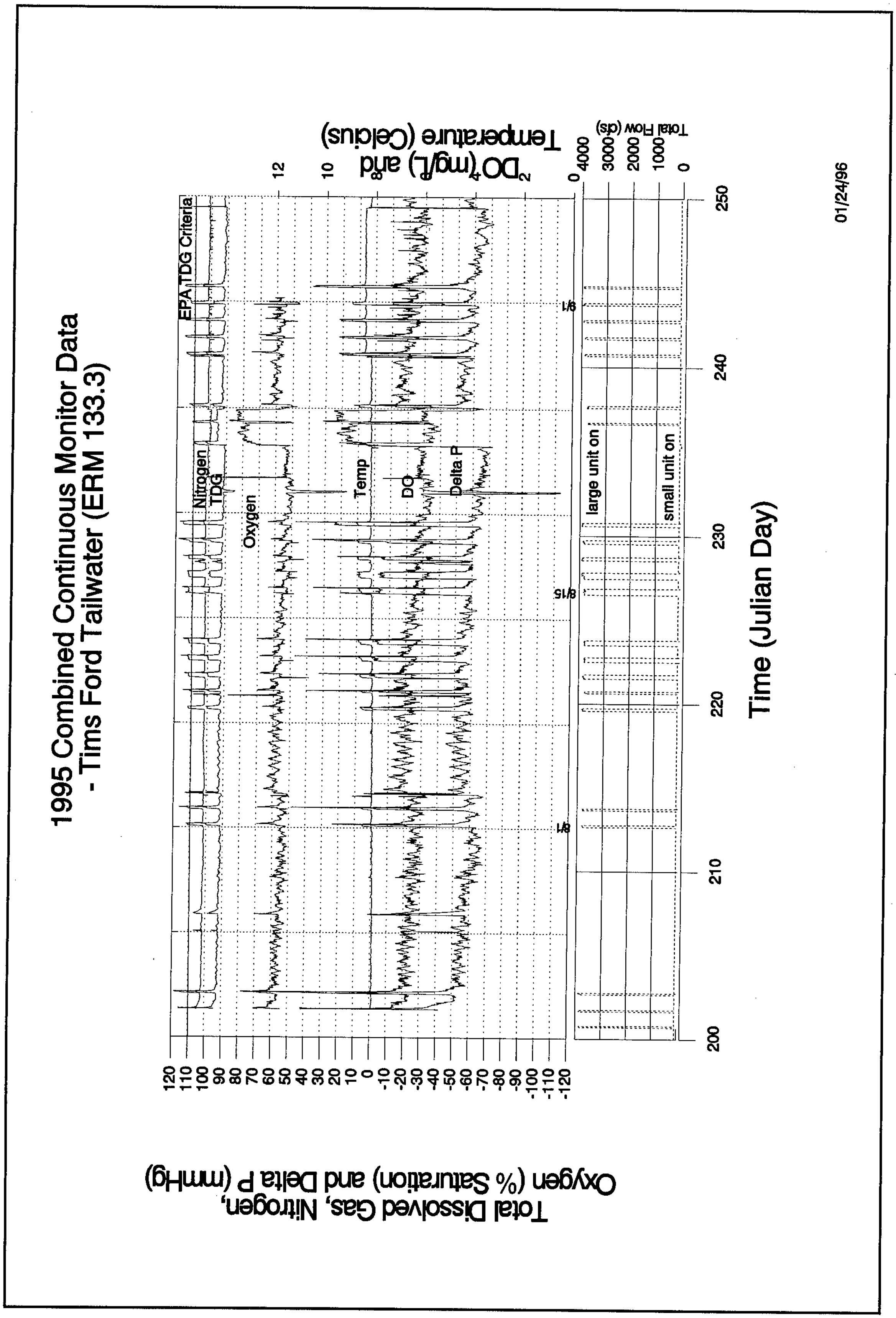

1995 Combined Continuous Monitor Data
- Tims Ford Tailwater (ERM 133.3)
Total Dissolved Gas, Nitrogen, Oxygen (% Saturation) and Delta P (mmHg)
DO (mg/L) and Temperature (Celcius)
Total Flow (cfs)
Time (Julian Day)
EPA TDG Criteria
Nitrogen
TDG
Oxygen
Temp
DO
Delta P
large unit on
small unit on
8/1
8/15
9/1
200
210
220
230
240
250
120
110
100
90
80
70
60
50
40
30
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
-110
-120
12
10
8
6
4
2
0
4000
3000
2000
1000
0
01/24/96

Figure 7.

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Dissolved Gas Measurement Uncertainty

by
Darrin A. Geldert[1] *and John S. Gulliver*[1]

Background on Uncertainty Analysis

Measurements' inaccuracies arise from many sources. The position of the instrument, the operation of the instrument and the instrument itself all lead to errors. The error in any one measurement is a fixed, given value. The possible value of that error is called uncertainty, and is a statistical variable arrived at through a process of uncertainty analysis. Typically, the measurement reported is taken to be the mean estimate and the uncertainty describes the variation of the measurements about the mean. In any measurement, all sources of uncertainty should be identified and quantified. The uncertainty of any measurement is a combination of precision (random) uncertainty and bias (fixed or systematic) uncertainty (Abernathy, Benedict, and Dowdell 1985). Precision uncertainty is introduced in any repeated measurement due to the variability of the instrument. Bias uncertainty will effect each measurement in the same manner resulting from a calibration or positioning error.

The analysis of measurement uncertainty provides a method to assess the quality of the measurement procedure and the results. The most common technique for analyzing measurement uncertainties is a first order-second moment analysis (Kline 1985). The total uncertainty (U), as given by this technique can be expressed as

$$U^2 = W^2 + B^2 \qquad (1)$$

where

W = precision uncertainty

B = bias uncertainty and each term must be considered in any application

Application To Total Dissolved Gas Measurements

General

To analyze the uncertainty in total dissolved gas measurements, a clear picture of what is being measured and how the various bias and precision uncertainties are related must be presented. Typically the total dissolved gas concentration is reported as the percent saturation, as given by the equation

$$\%Satn. = \frac{TDGP}{Bar} \times 100 \qquad (2)$$

where

$\%\ Satn.$ = percent saturation of total dissolved gasses

$TDGP$ = total dissolved gas pressure

Bar = barometric pressure

The total dissolved gas pressure and the barometric pressure each have precision and bias uncertainties. Therefore, the precision

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uncertainty, $W_{\%\ Satn}$ is a combination of the uncertainties that result from determining *TDGP* and *Bar*. It is described by the equation

$$W^2_{\%Satn} = \left[\frac{\partial\%Satn}{\partial TDGP}\times W_{TDGP}\right]^2 + \left[\frac{\partial\%Satn}{\partial Bar}\times W_{Bar}\right]^2 \quad (3)$$

where

W_{TDGP} = precision uncertainty in *TDGP*

W_{Bar} = precision uncertainty in *Bar*

Similarly, the total bias uncertainty, $B_{\%\ Satn}$ is a combination of the uncertainties due to calibration and other systematic errors.

$$B^2_{\%Satn} = \left[\frac{\partial\%Satn}{\partial TDGP}\times B_{TDGP}\right]^2 + \left[\frac{\partial\%Satn}{\partial Bar}\times B_{Bar}\right]^2 \quad (4)$$

where

B_{TDGP} = bias uncertainty in *TDGP*

B_{Bar} = bias uncertainty in *Bar*

Precision Uncertainty of Total Dissolved Gas Pressure Measurements

The precision uncertainty in the Total Dissolved Gas Pressure (W_{TDGP}) is due to the accuracy or precision of the probe and the sampling precision. In still water, the accuracy of the total dissolved gas pressure measurement taken by the TBO-L probe is given as ± 0.3 percent or ± 2.4 mmHg at typical pressures found on the Columbia and Snake River system (Common Sensing 1994). The ability of the probes to measure pressure changes in a moving body of water is limited. The river is not in a steady-state situation, but continuously mixing and changing.

In 1994, on the Snake River below the Ice Harbor Dam three probes were placed in the river within approximately 10 m of each other. These probes expectedly gave different values for *TDGP*, due the accuracy of the instrument and calibration errors (measurement precision and bias) and due to the location of the measurement station with regard to large eddies in the flow (sampling precision) and upstream conditions (sampling bias). The bias should be consistent and be represented by a steady average difference between the measurements. The variance about the average difference must then be explained by precision uncertainty. As shown in Figure 1, the observed difference between any two probes had a mean standard deviation of 2.6 mmHg or there was a 95 percent confidence that difference measured between the two probes was within 5.2 mmHg of the average difference. Therefore, the total precision uncertainty is, W_{TDGP} = 5.2 mmHg. If the total precision uncertainty is broken down, since W_{manf} = 2.7 mmHg for a typical Ice Harbor pressure, the sampling precision can be calculated as

$$W_{Samp} = \sqrt{W^2_{TDGP} - W^2_{Manf}} = 4.4mmHg \quad (5)$$

The average difference is simply the calibration error which grows with time as seen on Figure 1. It is also apparent that the precision error increases slightly. This should not happen unless, as the probe becomes dirty, the membrane may not function as well and the precision drops. These two uncertainties are combined at typical pressures to give,

$$W_{TDGP} = \left(W^2_{Manuf} + W^2_{Sample}\right)^{1/2} = 5.03mmHg \quad (6)$$

Precision Uncertainty of Barometric Pressure Measurements

The precision uncertainty associated with the barometric pressure (W_{Bar}), is the accuracy of the TBO-L meter which is again given as, ± 0.3 percent or ± 2.2 mmHg. The sampling precision is negligible as the barometric pressure is constant over a given area.

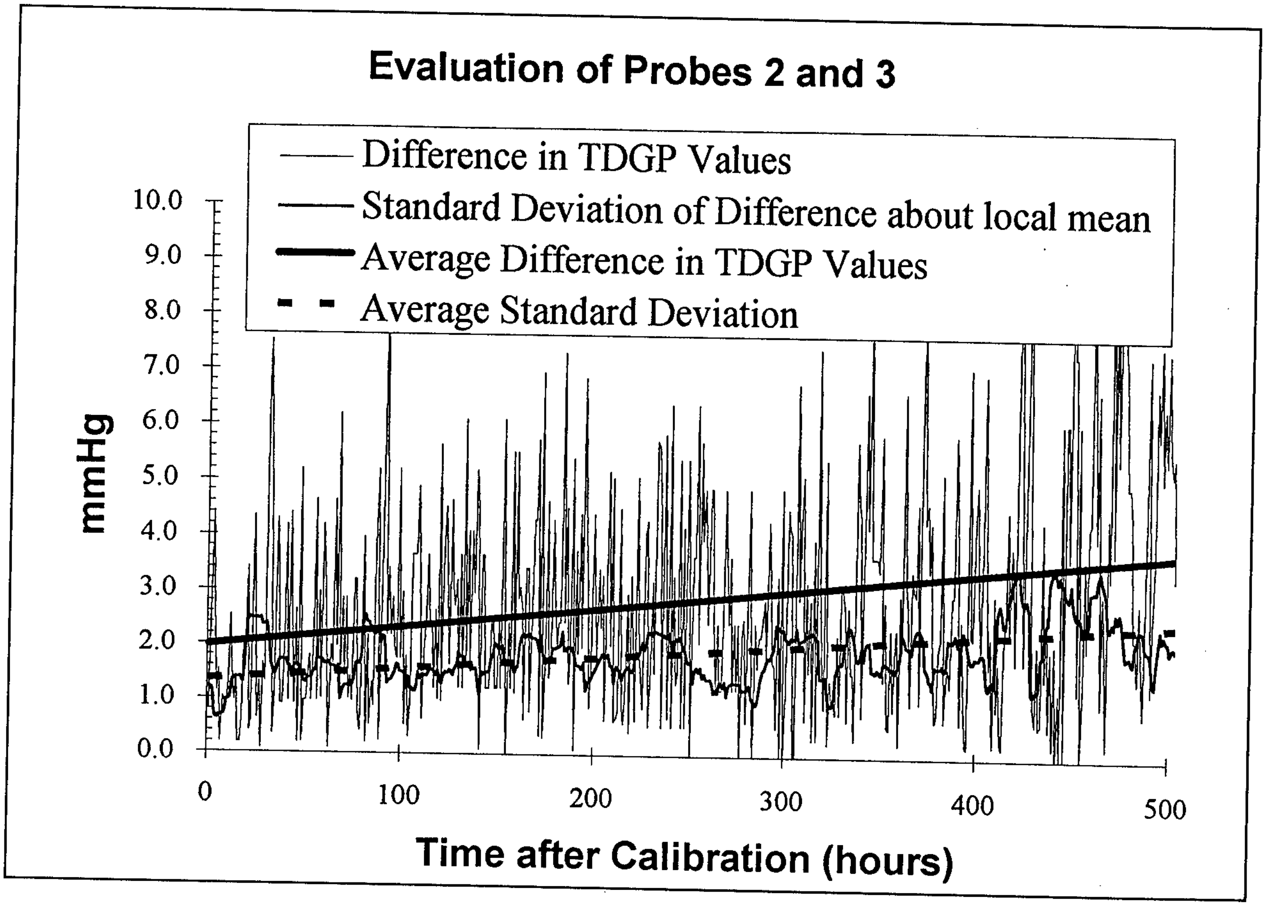
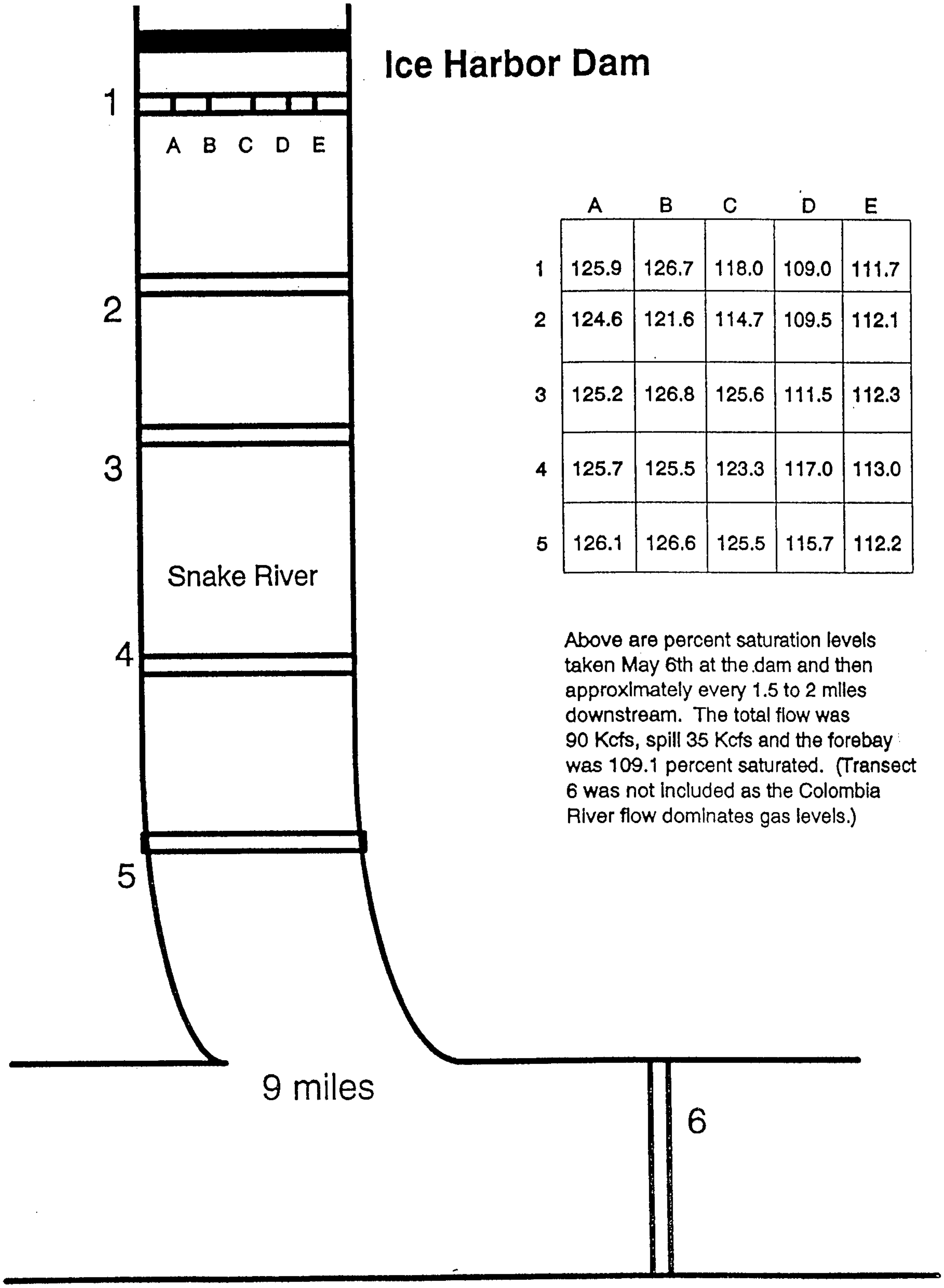


Figure 1. Sampling precision calculated as mean standard deviation between probes 2 and 3

Bias Uncertainty of Total Dissolved Gas Pressure Measurements

The bias uncertainty (B_{TDGP}) is determined from the USACE "1993 Dissolved Gas Monitoring For The Colombia and Snake Rivers." During 83 calibrations of *TDGP* probes a mean error of 6.7 mmHg was reported. This corresponds to a standard deviation of 8.4 mmHg. Standard scientific reporting (Abernathy, et al. 1985) requires a 95 percent confidence level or approximately two standard deviations if sufficient data is available. This is equivalent to a bias uncertainty of $\pm$ 16.8 mmHg. There is also a sampling bias which depends on the position of the probe in the river which is assumed to be zero in this application. However, sampling bias may actually have the greatest impact on the percent supersaturation measured.

The sampling bias would be an error due to the position of the probe in the river. At a dam the portion of the flow that is carried over the spillway becomes enriched with gases while the portion flowing through the powerhouse does not. Therefore, after a dam the river consists of a minimum of two streams, and large differences in supersaturation are possible. In addition, not all of the gates are open equally, and there are fish attraction discharges near upstream passage facilities. This makes placement of a probe downstream of the spillway quite difficult. Total dissolved gas transects taken downstream of the Ice Harbor Dam in 1994 (Figure 2), provide an example of the variation that may be expected. The left or



Ice Harbor Dam

	A	B	C	D	E
1	125.9	126.7	118.0	109.0	111.7
2	124.6	121.6	114.7	109.5	112.1
3	125.2	126.8	125.6	111.5	112.3
4	125.7	125.5	123.3	117.0	113.0
5	126.1	126.6	125.5	115.7	112.2

Above are percent saturation levels taken May 6th at the dam and then approximately every 1.5 to 2 miles downstream. The total flow was 90 Kcfs, spill 35 Kcfs and the forebay was 109.1 percent saturated. (Transect 6 was not included as the Colombia River flow dominates gas levels.)

Figure 2. Dissolved gas transects taken downstream of Ice Harbor Dam, May 6, 1994

powerhouse side of the river has a saturation level of approximately 112 percent throughout the 9 miles below the dam. The right or spillway side of the river is approximately 125 percent from the spillway to the 9 mile point. It is then obvious that care must be taken in placing the probe in the river. We must first ask ourselves, in which of these multiple discharges do we want to measure *TDGP*? Our presumption is that the highest *TDGP* in the downstream reach would be the most appropriate measure for protection of fish species. It is possible however, that high *TDGP* in one location would be acceptable if no fish are found in that habitat. The question of where is the best position should be left to personnel familiar with each facility and what is desired. If the decision is that the highest levels of *TDGP* found downstream of a spillway should be measured, transects of *TDGP* under various conditions will identify this location.

Bias Uncertainty of Barometric Pressure Measurements

The bias uncertainty for the barometric pressure (B_{Bar}), is also a calibration error. In 1993 the barometers were calibrated 77 times and had a mean error of 0.86 mmHg. Assuming that the distribution of calibration errors is Gaussian, the mean error may be expanded to two standard deviations, or a 95 percent confidence interval. The 95 percent confidence interval of bias uncertainty for barometric pressure is therefore, 2.16 mmHg. There is no significant sampling bias as barometric pressure is relatively constant.

Total Precision Uncertainty

The total precision uncertainty can now be determined by substituting the above values into Equation 3.

$$W_{\%Satn} = \left[\left(\frac{100}{Bar}\times W_{TDGP}\right)^2 + \left[\frac{TDGP\times 100}{Bar^2}\times W_{Bar}\right]^2\right]^{1/2} = 0.73 \qquad (7)$$

This means any measured value of percent saturation may be off by $\pm$ 0.73 due to the precision of the instrument, to the 95 percent confidence interval. Thus, 19 out of 20 measurements will be within 0.73 of the mean measurement.

Total Bias Uncertainty

The total bias uncertainty is determined from Equation 4:

$$B_{\%Satn} = \left[\left(\frac{100}{Bar}\times B_{TDGP}\right)^2 + \left[\frac{TDGP\times 100}{Bar^2}\times B_{Bar}\right]^2\right]^{1/2} = 2.24 \qquad (8)$$

This means any measured value of percent saturation has an associated bias uncertainty of $\pm$ 2.24. This does not include a sampling bias, which can be substantial, because there is insufficient information available to determine how the point measurements represent the total dissolved gas level of the river.

Total Uncertainty

The total uncertainty associated with any given percent saturation U, as given by Equation 1, is therefore

$$U^2_{\%Satn}=W^2_{\%Satn}+B^2_{\%Satn}=0.73+5.04$$

or

$$U_{\%Satn}=2.36$$

This is the uncertainty associated with a given probe's measurement of total dissolved gas pressure. Sampling uncertainty, or the ability to use specific probe locations to measure the characteristics of an entire river, warrants further investigation.

Summary and Conclusions

The United States Army Corps of Engineers measures dissolved gas levels on the Colombia and Snake Rivers. For many years the data collected near dams have been used to make decisions regarding river operations. Currently the total dissolved gas levels are reported as percent saturation. The percent saturation measurements have never had any estimate of the uncertainty in the data. A detailed uncertainty analysis has been performed on the data.

Results of the analysis show that the dissolved gas instruments are generally well calibrated and maintained. Evidence of this is that the uncertainty to a 95 percent confidence interval in any given percent saturation measurement is $\pm$ 2.36. In other words, there is a 95 percent confidence that the measurements taken will be within 2.36 of the true value. Thus, a 120 percent saturation measurement would be given as 120 $\pm$ 2.36 percent saturation. It is difficult to maintain and calibrate total dissolved gas instruments in the field to an accuracy greater than this.

There is still the possibility of a considerable sampling uncertainty. A probe may be placed in a location that is not representative of the river or the highest dissolved gas concentration in the river reach. It is recommended that downstream dissolved gas sampling be undertaken with a Loranz type of location finder to pinpoint the optimum location for upstream and downstream total dissolved gas probes at each dam.

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Remote Monitoring of Hydroprojects: Design, Installation, and Verification of Remote Monitoring Systems

by
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Introduction

Remote monitoring systems are important tools for lake managers, hydropower operators, and others concerned with hydroproject influenced water quality. Remote, automated water quality monitors provide temporal data sets that are utilized for determining water quality trends under various operational and seasonal conditions. Data collected via remote monitors may be used for identifying areas of management concern and are valuable for developing and calibrating predictive models.

The usefulness of data collected by remote monitors is dependent on how effectively the sampled water represents the parameters of concern for the area. Many variables affect the representativeness of monitoring locations including lateral, longitudinal, and vertical heterogeneities in the water; the equilibration times of the water quality instruments; and hydrological, biological, and physico-chemical processes within the sample areas.

This technical note describes the processes involved in designing and deploying automated, remote monitoring systems and analyzing the data they generate. It is not intended as an exhaustive review of the subject, but highlights the more critical steps in the development of monitoring systems, citing case studies where appropriate. Although the primary purpose of this paper is describing the installation and maintenance of automated remote monitoring systems, the ideas presented apply to manual sampling programs as well. The ultimate goal of any monitoring program should be the collection of pertinent, representative data. The following flow diagram is presented as a generic guideline for implementing a monitoring program. It is meant to organize some of the ideas that will be further developed in this paper, and not as a "recipe" for designing and installing automated monitors.

Pre-Installation

Goals

The first step in implementing any monitoring program is the determination of its goal. Potential questions may include the following:

a. Why are the data needed?

b. Who will need access to the data?

c. Are the data needed real-time or at some other level of frequency?

d. What type of sample interval will be required?

e. What is the time frame from data collection to data reporting?

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The objective at this stage is to determine what will be expected of the monitor program. The answers to these questions determine subsequent equipment and location decisions and are necessary to prevent the implementation of what has been characterized as a "data-rich but information-poor" (Ward et al. 1986) monitoring program.

Additionally, the answers to questions such as these determine if automated monitoring is needed to attain the goal of the program. Grab sampling may be better suited to a temporary monitoring program or one having a long sampling interval. In a lengthy or permanent installation or one requiring a short sampling interval, grab sampling quickly becomes cost prohibitive and automated, remote monitors are both more appropriate and more effective.

Site Characterization

Once the decision has been made to install an automated monitor and the water quality parameters to be measured have been identified, the next step in the pre-installation process is to characterize the study area. This may be accomplished with short-term manual sampling. A working knowledge of the parameter(s) to be measured is essential to identify the most representative deployment site. Additionally, the hydrology, morphometry, flow patterns, climate, chemistry and biology of the site determine the optimum monitor location. Characterization of the area should include the identification of any lateral, longitudinal, and vertical heterogeneities that may be present. Sampling should be conducted under the conditions that will be experienced by the monitor, that is, if the monitor is to measure hydropower release water quality, then pre-installation sampling should be conducted during release periods.

Four general areas for consideration in the deployment of hydroproject monitors are the forebay, within the hydroproject's physical structure, the tailrace, and the tailwater. Preliminary areas for study would depend on the monitoring objective. For example, the preliminary study area for a release water quality monitor for a hydropower dam may be the tailrace. An installation for monitoring the effectiveness of water quality improvement measures may be located upstream for pre-treatment conditions and downstream for post-treatment conditions. A monitor for evaluating hydroproject operation on downstream habitat may be located in the tailwater some distance downstream of the project. Regardless of the monitoring program's goal, certain locations should present themselves as logical starting points for deployment consideration. Secondary consideration may focus on accessibility for calibration and maintenance; however, the most convenient location is not always the most representative one and greatest emphasis should be placed on data quality.

Many relatively inexpensive water quality instruments capable of internally storing data are commercially available, which allows project planning to include experimentation with various site locations via short-term deployments. These data may then be combined with grab data to provide temporal and spatial representations of the daily and seasonal variations for the area. Careful analysis of the available data is crucial during the preliminary stages of the monitoring program's development to prevent problems concerning data validity and defensibility later. Often, a logical location for the monitor may present itself; however, peculiarities of the site, particularly with respect to flow patterns, may preclude monitor installation in this area. The logical location provides a starting point for the validation stage of the pre-installation process.

Conservative water quality measures not easily affected by biota, such as temperature or specific conductance, may be used as "tracers" to track parcels of water. Comparing conservative parameters cannot conclusively validate the representativeness of a potential location, but can eliminate a non-representative one. Several case studies will be presented to further develop these ideas.

Ice Harbor Example

Ice Harbor Dam is located on the Columbia River immediately upstream of McNary Dam and immediately downstream of Lower Monumental Dam along the Oregon/Washington border (Figure 1). Spilling operations conducted in conjunction with the COE fish passage program, as well as flood control, often lead to dissolved gas concentrations that are supersaturated with respect to the atmosphere. Supersaturation of dissolved gasses in water may have severe detrimental impacts on fish and, as a result, extensive studies to measure dissolved gas concentrations and dynamics have been conducted at the COE projects in the Columbia River Basin.

Data gathered during transect studies in support of the total dissolved gas monitoring program serve as an example of how data gathered for other purposes may be used to plan an automated monitor installation. The results from these lateral transects are displayed in Figure 2. Two monitors (indicated by arrows and labeled by river mile in Figure 2) were previously installed in the Ice Harbor Tailwater; however, they were neither designed nor intended to reflect the extent of the variation in total dissolved gas concentrations in the area.

The goal of the monitoring program dictates the deployment design. If the goal for the program were to measure critical total dissolved gas concentrations, then a single monitor near the area of highest total dissolved gas concentrations may be sufficient (Figure 2A). If concerns were for the mean total dissolved gas concentrations for the area, then a single monitor located near mid-channel may be appropriate (Figure 2B). If, however, the program's goal were to map the total dissolved gas concentrations for the tailwater, then a single fixed monitor would be inappropriate and an alternate plan would have to be developed involving numerous fixed positions (e.g., Figure 2 C, D, and E). This example serves to emphasize the need for good planning and pre-installation sampling in the early stages of the monitoring program's development.

Monitor Equipment

Hardware

Data requirements and available funding will dictate the hardware selected for the monitor installation. Water quality instruments equipped to measure most parameters of management concern are commercially available. These instruments vary with respect to accuracy, precision, data presentation, and expense. Consideration should be given to the design limitations of the instrument when selecting water quality equipment. For example, if the purpose of the monitor were to record dam release dissolved oxygen concentrations for mitigation and the requirement was to remain within 0.5 mg/ℓ of a target dissolved oxygen concentration of 5.0 mg/ℓ, then oxygen probes with an accuracy of less than $\pm$ 0.5 mg/ℓ would be inadequate.

Deployment/retrieval monitoring is utilized for thermal monitoring and special studies at Richard B. Russell Reservoir on the Savannah River, whereby water quality instruments with data logging capabilities are deployed and the data retrieved later. If data are needed real-time, a computer/modem system may be used. Relatively inexpensive, reliable water quality sonde interfaced with a personal computer/ modem may be obtained for less than $5,000 (1996). Commercially built data collection platforms are available and most may be tailored to fulfill the design requirements of the site. With computers and other data platforms, the operator achieves greater flexibility with respect to how the data are stored and accessed. As a general rule, equipment should be selected based on the following factors:

a. Instrument accuracy, precision, and resolution desired.

b. Instrument deployment requirements.

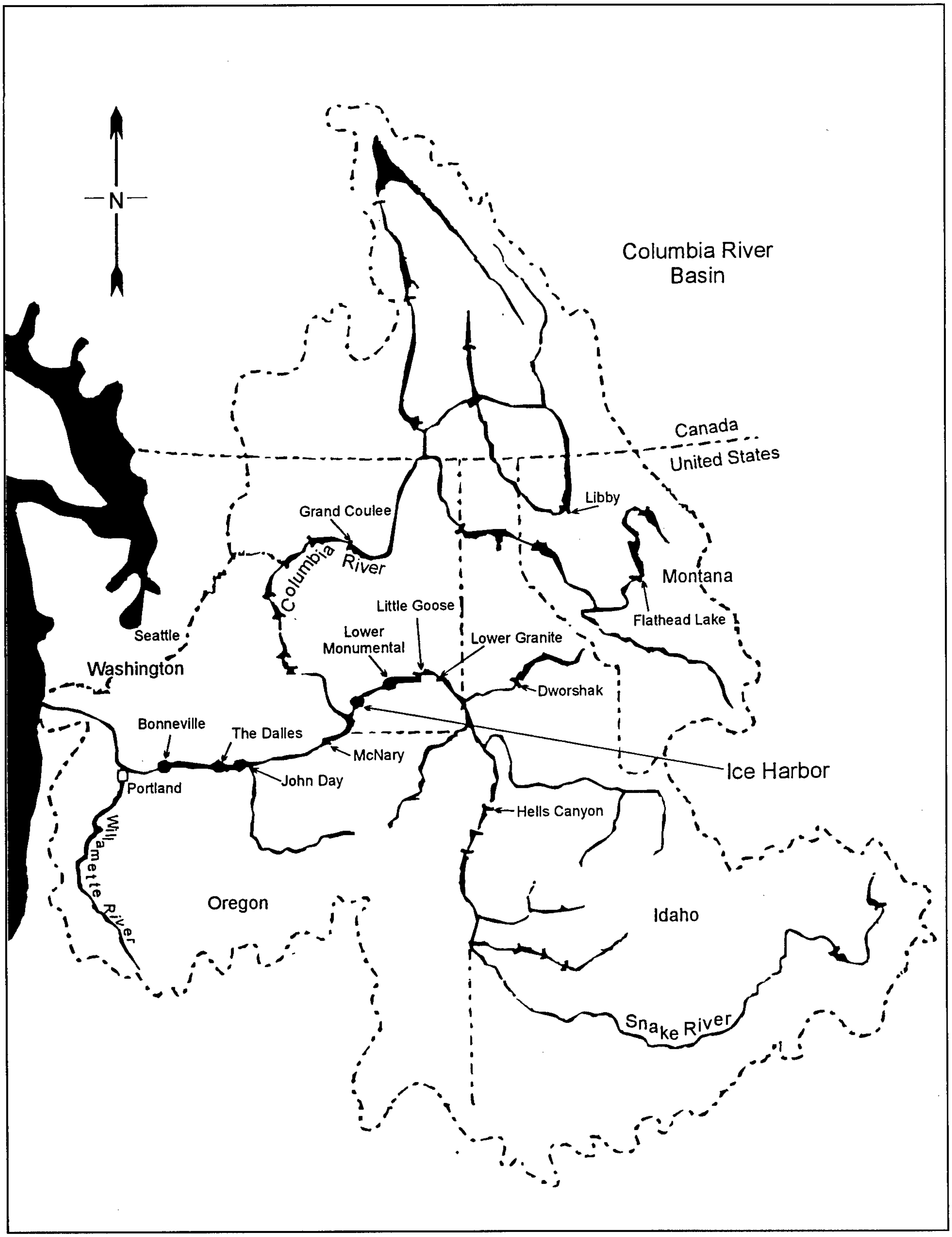

Figure 1. *Columbia River basin*

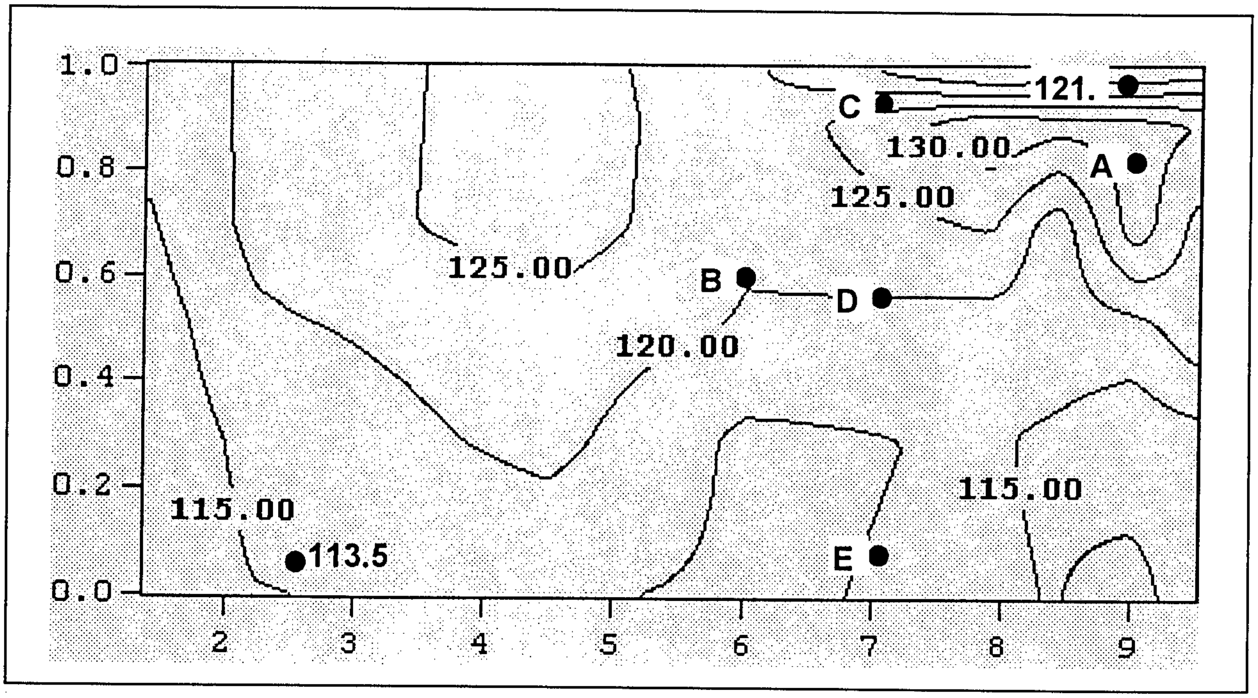

Figure 2. Contour plot of Ice Harbor total dissolved gas transect data

c. Deployment method, i.e., deploy/ retrieval, computer/modem, incorporation with existing equipment, etc.

d. Fouling concerns and required calibration and maintenance regimens.

e. Instrument expense and monitoring program budget constraints.

Software

Off the shelf data collection platforms include software or programming instructions that allow them to be configured to communicate with a variety of instruments. Additionally, personal computer communications packages may communicate with water quality equipment and store and transmit data; however, design flexibility is generally less. Basic software programs (Microsoft Corporation) may be developed as an alternative to off the shelf communications packages and afford the user control over communication protocol and data storage format (Vorwerk 1996). The data storage format is an important design consideration as it facilitates incorporation of the final monitor data set with other pertinent data sets, e.g., hydroproject operation data, and allows real time data presentation to better fit project requirements.

Location Validation

Post-deployment data validation is an extremely important final step in the monitor installation process as this evaluates the monitor location's representativeness. Although post-validation may seem unnecessary if care was taken during pre-installation sampling, the installation itself may have a measurable impact on how the water quality is represented by the equipment. A dam release monitor could be installed in the tailrace of a project with water pumped to it from an area determined to reflect the area of management concern during generation periods. Subsequent calibration visits may confirm that the sensors are operating well within the manufacturer's specifications. From this it may be assumed that the monitor is accurately representing the parameters of concern. If, however, the water were being warmed as it passed from the tailrace through the pipe to the monitor, it would actually reflect the water within the sample chamber and not the tailwater. Likewise, changes in the physical structure of a site or introduction of water quality improvement measures may alter the representativeness of an established monitor. These concerns must be addressed via post-deployment verification studies.

Data Interpretation

After the monitor is in place and recording representative water quality data, the next concern is how the data should be utilized. Raw monitor data are of little use if they are not presented in a manner that facilitates interpretation. Off the shelf spreadsheet and database programs such as Excel (Microsoft Corp.), SAS (SAS Institute Inc.), and SPSS (SPSS Inc.) expedite data analysis and reporting by facilitating monitor data linkage with other project data. Data must undergo vigorous error detection and filtering processes prior to analysis. Raw monitor data must be edited to remove machine characters, usually artifacts of the data collection software, before they may be properly imported into analysis software packages.

Water quality sensors typically exhibit some degree of response drift as a result of the sensors' chemical reactions (e.g., oxidation of dissolved oxygen probes) or to biological activity (e.g., algal growth on dissolved oxygen probes may decrease the reported dissolved oxygen concentrations by inhibiting oxygen diffusion across the sensor's membranes). Routine calibration may reduce the degree of sensor drift; however, post-deployment corrections for sensor drift can further improve data accuracy.

For the Savannah River monitors where dam release dissolved oxygen concentrations are the primary concern, frequent calibration visits (at lease once/week) during summer months reduces the degree of drift resulting from biological activity. Calibration drift is assumed to be linear which allows corrections to be based on the degree of drift per hour for the period between calibrations. Each reading is then corrected for drift by adding or subtracting this value to it with the drift at the time of the first calibration being equal to zero. The causative factors leading to drift vary depending on the site, the parameters being measured, and the equipment being used (the instruments used for monitoring the Savannah River hydroprojects have a resolution of ± 0.2 mg/ℓ, therefore drift must be >0.2 mg/ℓ before corrections are made). Drift must be determined for each site and should be factored into the data set prior to its incorporation with other project data (Whitfield and Wade 1993).

Data should be incorporated with other project data prior to final analysis. By combining the available data into a comprehensive project data set, "windows of reflectiveness" may be better identified and data interpretation made more accurate. For example, the release monitor at Hartwell Dam, a COE project located on the Savannah River (Figure 3), is deployed in the tailrace (Figure 4). It consists of a submersible pump and pipeline to pass water from the tailrace to a water quality sonde in a nearby building. Because it samples water from the tailrace, the monitor represents release water quality only during periods when Hartwell Dam is releasing water. Data for periods of non-release reflect the tailwater conditions only in the area localized around the sample intake line. Representative periods are readily apparent when the monitor data and operations data are incorporated (Figures 5a and 5b). Data falling outside of the "window" that defines representative periods, generally resulting from changes in project operations, are not included in final reporting as they are not reflective of the parameters of concern.

A large (>20 minutes for some instruments) equilibration period may be required by some instruments before accurate measurements are possible. This is especially true for gas measuring instruments such as dissolved oxygen or total dissolved gas sensors. Instrument and design limitations such as these should be considered during the final analysis, particularly

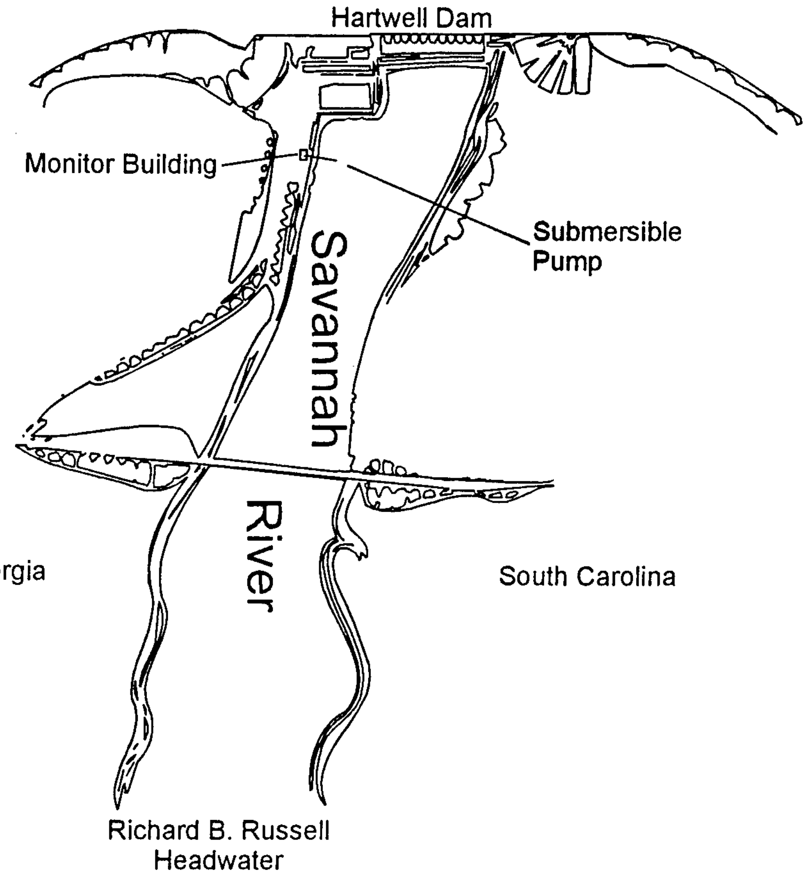

Figure 3. Hartwell Dam release monitor

in situations where rapid changes are experienced.

Case Studies

Continuous, automated monitors are presently being used by the COE to monitor the release water quality of the hydropower projects on the Savannah River forming the Georgia/South Carolina border, the tailwater conditions during periods of no release at St. Stephen Dam on the Cooper River in South Carolina, the effectiveness of turbine venting procedures at Bull Shoals Dam on the White River in Arkansas, total dissolved gas concentrations at various projects throughout the Columbia and Snake River systems, as well as other applications at other projects throughout the United States. While the monitoring goals, parameters of concern, and available funding vary significantly from project to project, the overall goal, the collection of representative data, is common to all. The case studies that follow demonstrate some of the techniques that have been used to ensure sample reflectiveness at various projects.

Richard B. Russell Dam

Richard B. Russell Dam is a COE generation/pumped storage project located on the Savannah River between the COE reservoirs of Hartwell and J. S. Thurmond (Figure 3). The Russell monitor measures release water

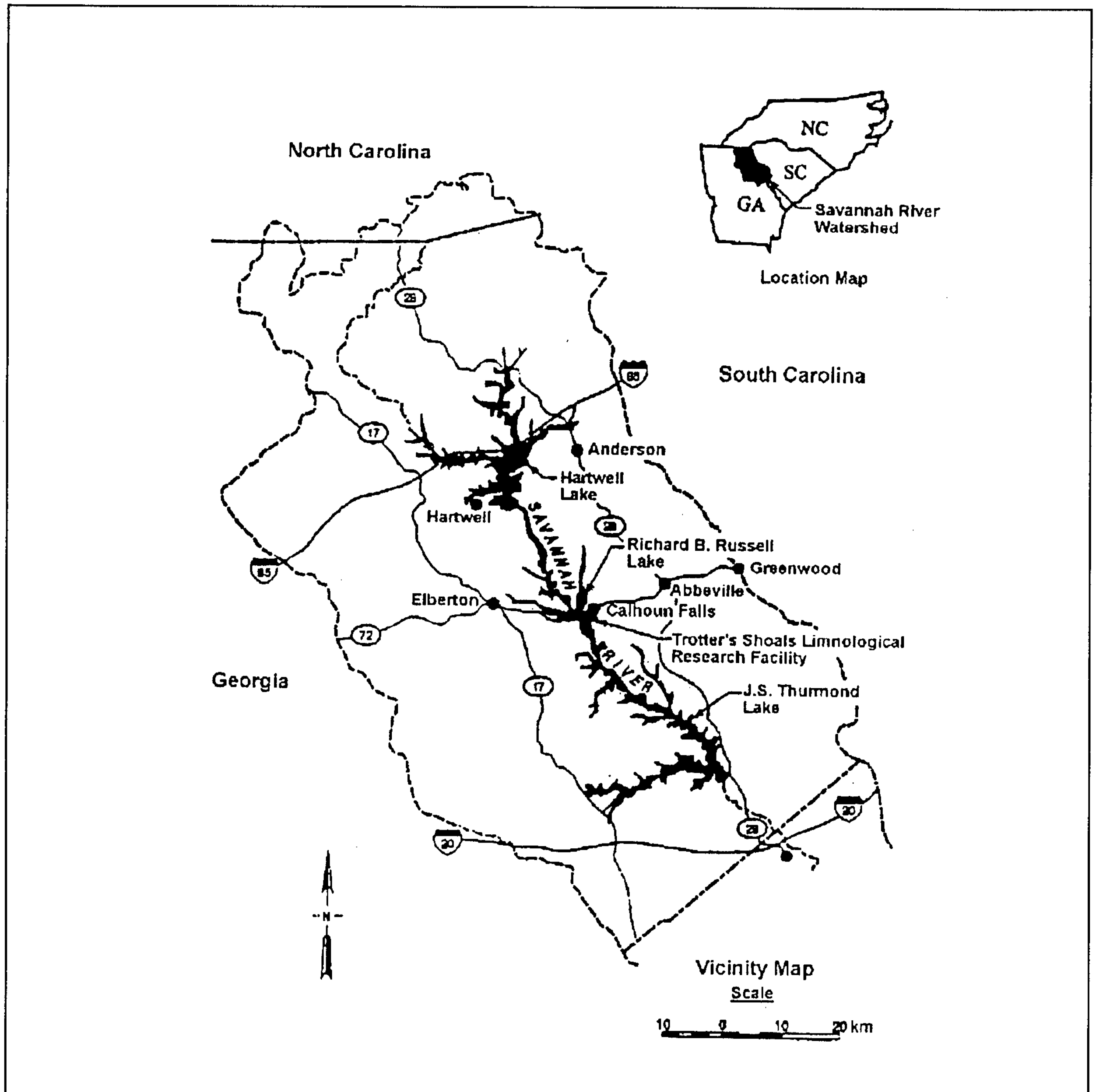

Figure 4. Savannah River basin

quality for the purpose of maintaining a release dissolved oxygen concentration of 6.0 mg/ℓ. The COE operates an oxygen injection system in Russell forebay to maintain this concentration during the summer months when hypolimnetic dissolved oxygen concentrations approach anoxia. The 6.0 mg/ℓ dissolved oxygen concentration requirement applies to the release and not to the tailrace or tailwater conditions; therefore, the sampled water must reflect the Russell release and not the conditions of the Thurmond headwater.

The monitor was originally located in the tailrace where follow up studies later demonstrated that flow patterns caused the monitor to be less reflective of Russell Dam release water than the ambient tailwater conditions (Figure 6).

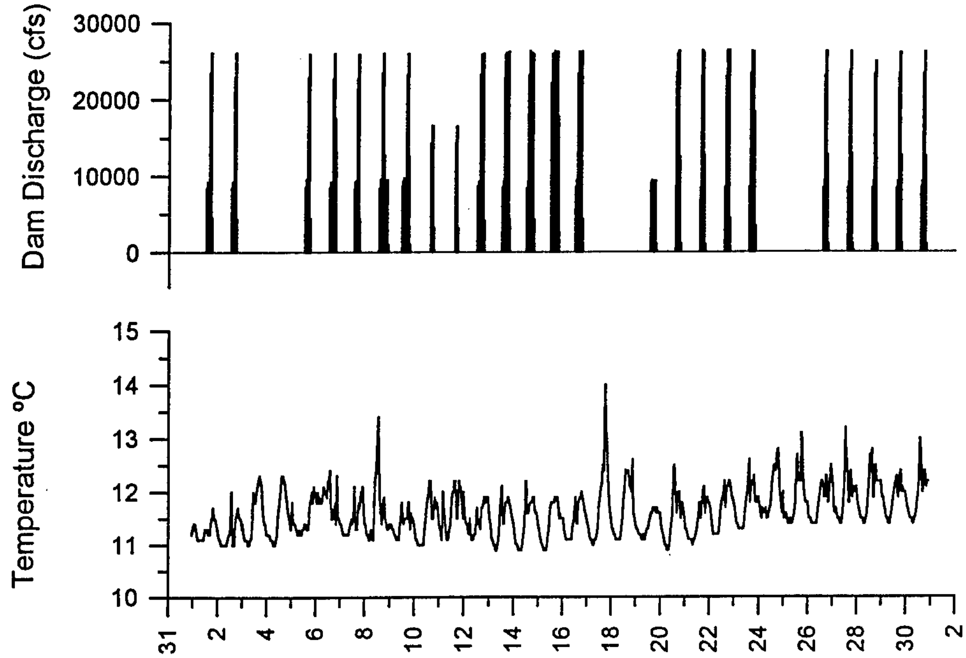

Figure 5a. Hartwell Dam operation and release temperatures 31 May to 30 June, 1995

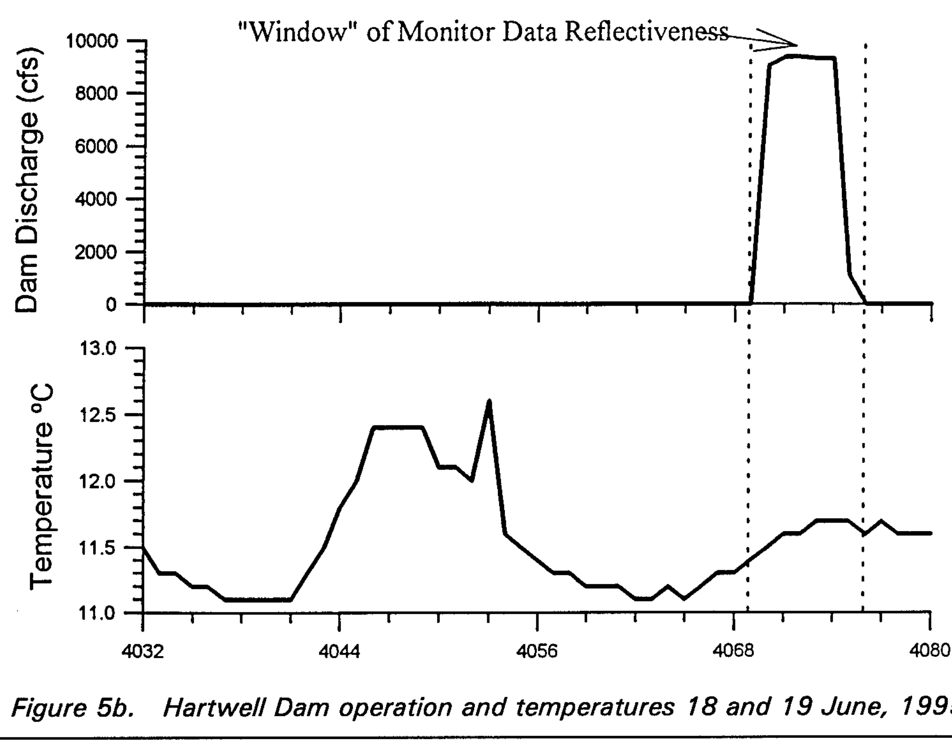

Figure 5b. Hartwell Dam operation and temperatures 18 and 19 June, 1995

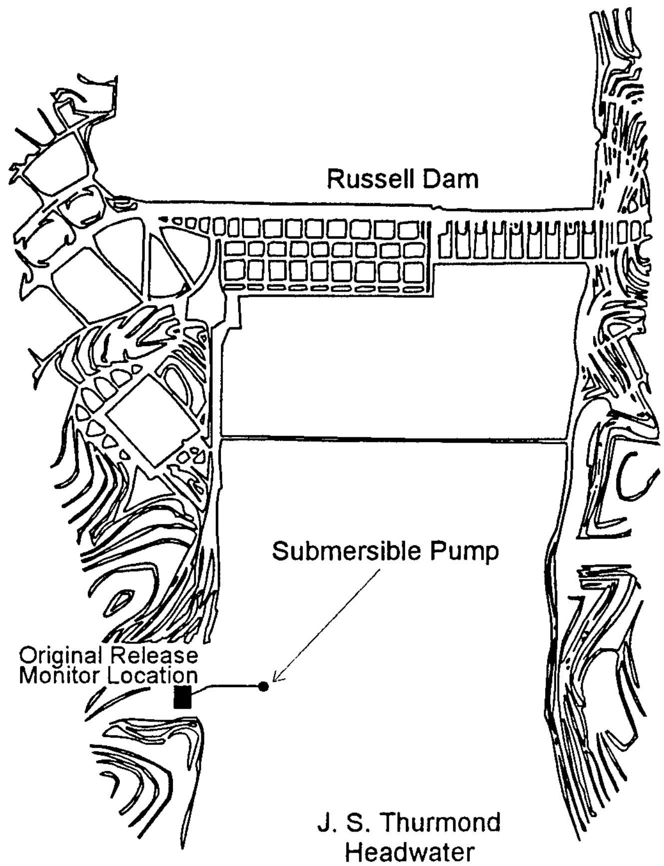

Figure 6. Richard B. Russell original downstream monitor

Temperatures and dissolved oxygen concentrations were measured at various points in the tailrace and the dam and were compared to the temperatures of the water sampled by the original tailrace monitor. For comparison, temperature was selected over dissolved oxygen as it was a more conservative parameter and as such was deemed to be less susceptible to exterior influences (Vorwerk and Carroll 1994).

The lacustrine tailwater region at the Russell project prevented the deployment of the tailrace monitors that had been successful for other Savannah River monitors. A mixing chamber system containing a water quality sonde was implemented such that water passage was controlled by solenoid switches. The switches were configured to restrict water passage to periods of turbine operation. This system (Figure 7) allowed representative water to be sampled with a single in-dam unit. While the monitoring goal (to measure release temperatures and dissolved oxygen concentrations) was the same for the Savannah River monitors, the individual characteristics unique to each site had to be included in the location determination process.

Bull Shoals Dam

At Bull Shoals Dam on the White River, AR (Figure 8), the goal for the monitoring program was to determine the efficiency of turbine venting operations conducted for increasing downstream dissolved oxygen concentrations. Two of the seven Bull Shoals units had turbine venting capability and penstock monitors had previously been installed to measure the pretreatment water quality. In situ sampling demonstrated that locating the post-treatment monitors in or near the draft tube exits would best represent the release water quality. The draft tube access ports were chosen for their proximity t the draft tube exits and the easy accessibility they

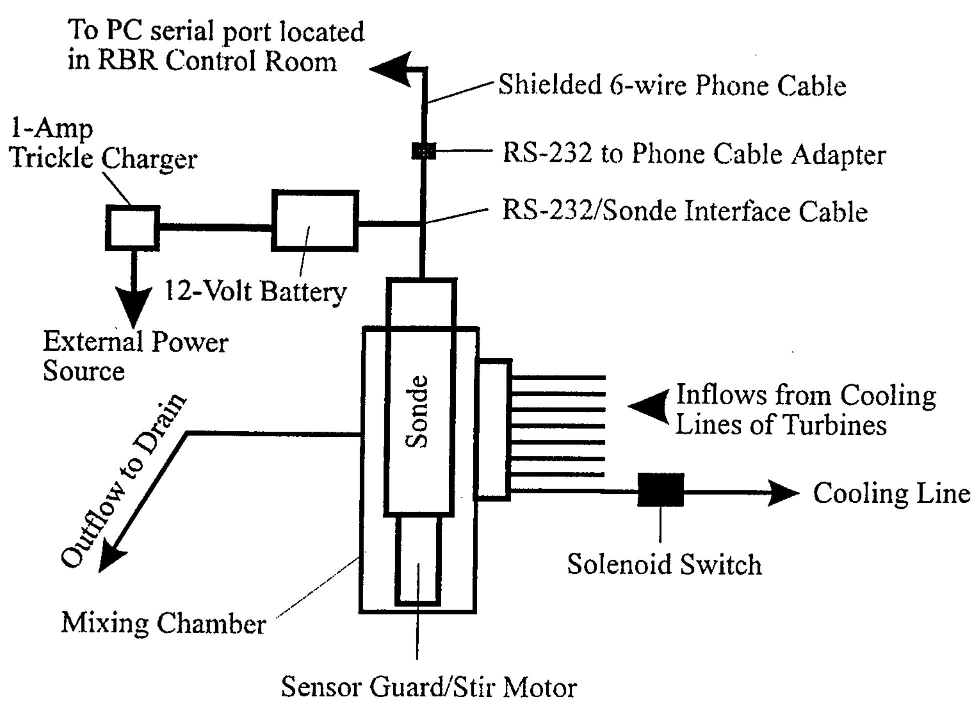

Figure 7. Richard B. Russell piping gallery monitor

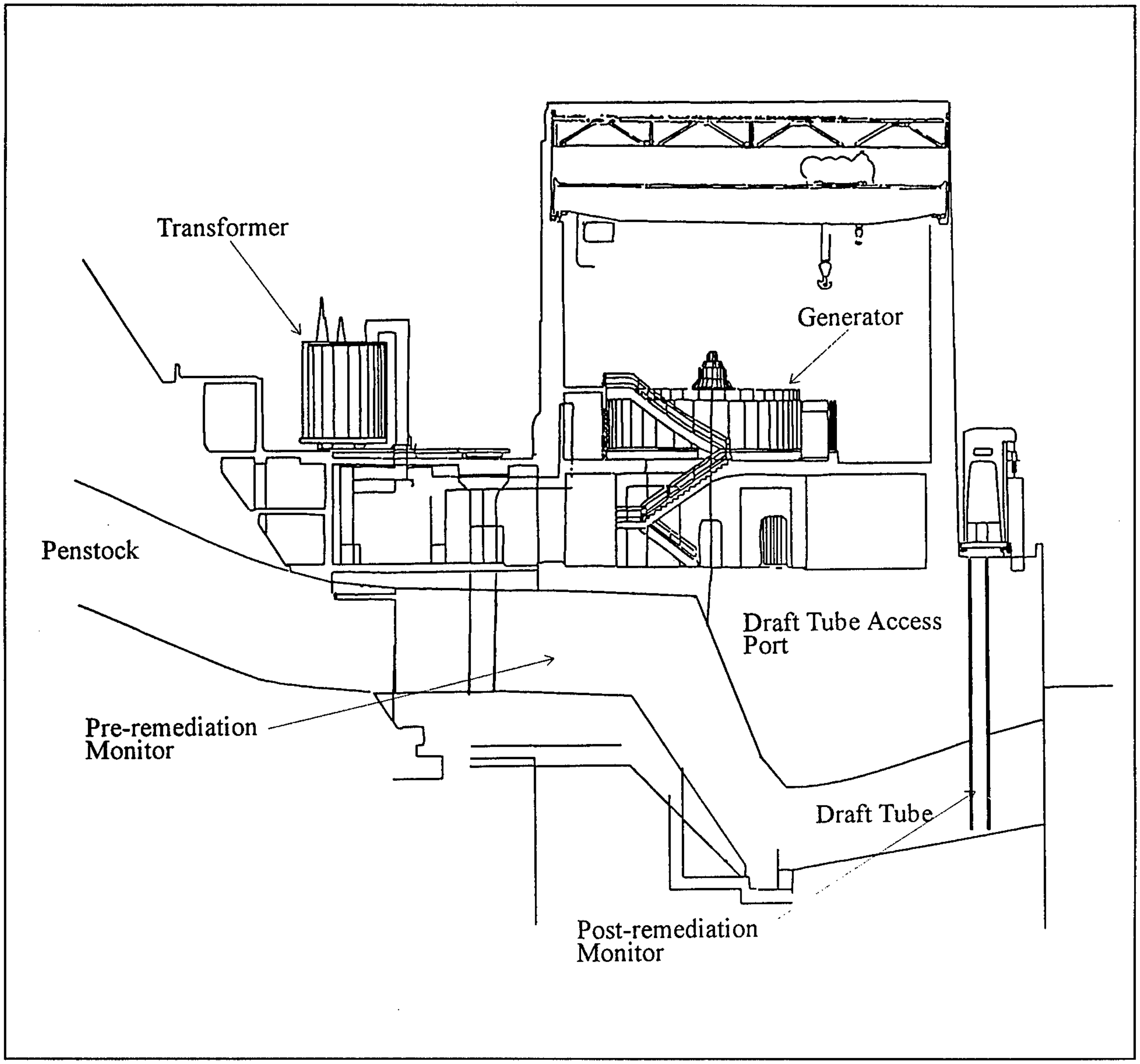

Figure 8. Bull Shoals powerhouse White River, Arkansas

provided for calibration and maintenance. The concern was to isolate the monitors from the release of the other units to allow for accurate identification of the dissolved oxygen increase resulting from individual turbine venting.

St. Stephen Dam

St. Stephen Dam is a COE powerproject located near St. Stephen, SC that rediverts water from Lake Moultrie back to the Santee River (Figure 9). A fish kill attributed to insufficient dissolved oxygen concentrations during non-release periods in the spring of 1991 prompted evaluation of the dissolved oxygen dynamics surrounding the project. It was determined that releasing water when the dissolved oxygen concentrations were low diluted the poorly oxygenated canal water with well oxygenated reservoir water and prevented dissolved oxygen related fish kills. The monitor program implemented at St. Stephen was designed to measure the tailrace dissolved oxygen concentration during periods of no release. Real time monitor

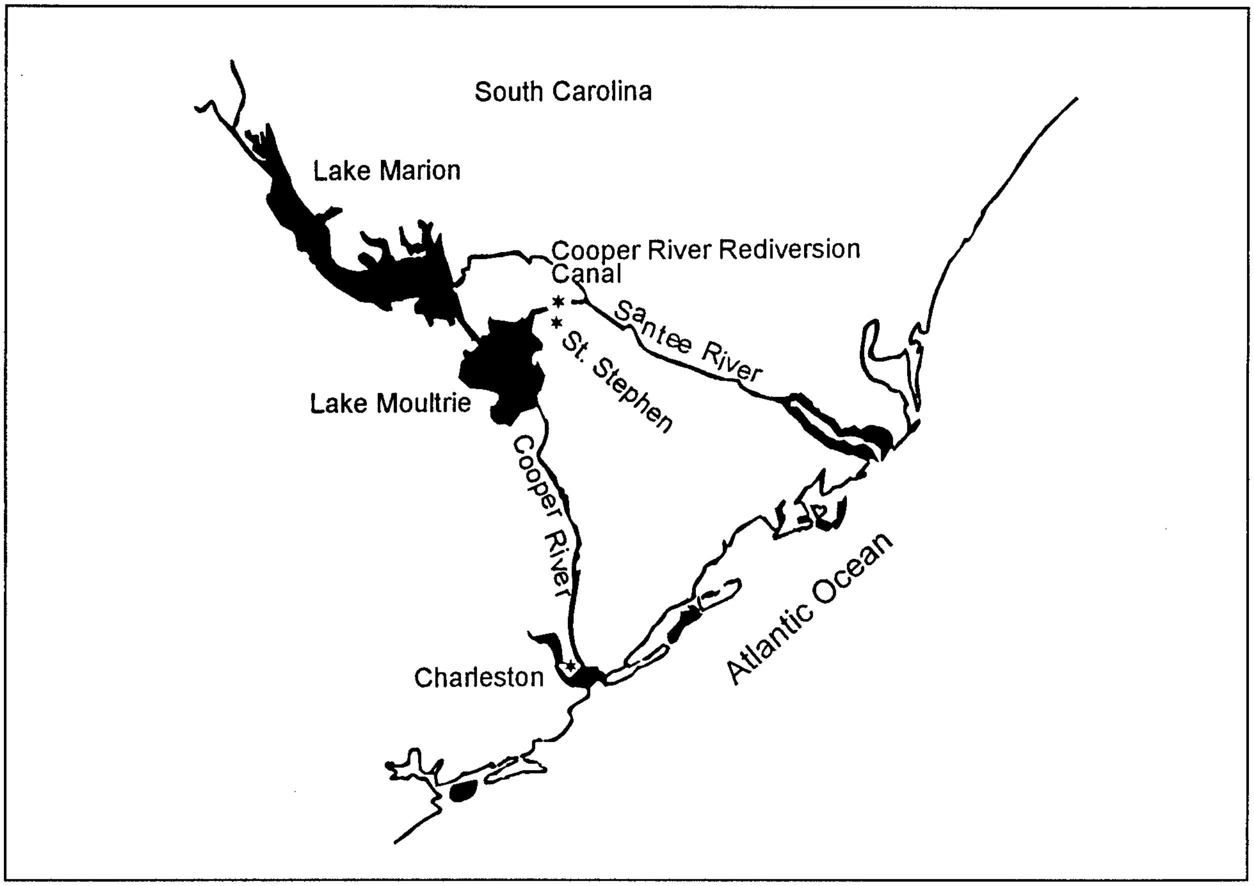

Figure 9. St. Stephens vicinity map

data were used to indicate when critically low dissolved oxygen concentrations occurred so water could be released, thus minimizing the potential for a fish kill. Manual sampling indicated that the monitor should be placed near the bottom of the canal and near the dam because anoxic conditions were realized in these areas first. A monitor attached to the wingwall downstream of the dam (Figure 10) represented "worse case" conditions (Vorwerk and Carroll 1995).

Conclusions

Remote, automated monitors are valuable water management tools. Large gains continue to be made with respect to water quality instrumentation that reduce the need for costly equipment and labor intensive sampling regimes. Too often, however, the assumption is made that deployment of a fixed monitoring system is sufficient for generating the desired data with little (if any) fore-thought devoted to outlining the goals of the monitoring program. Without clear goals, it is impossible to design a pre-installation program for determining the most appropriate location for the fixed monitor. Data density without data quality is of no use to project managers.

By clearly defining the objectives of the monitoring program prior to beginning data collection, characterizing the study site with respect to the physicochemical and biological attributes of the system, it becomes possible to design and install an automated, fixed location monitor that supplies data representative of the parameter(s) of management interest. Data

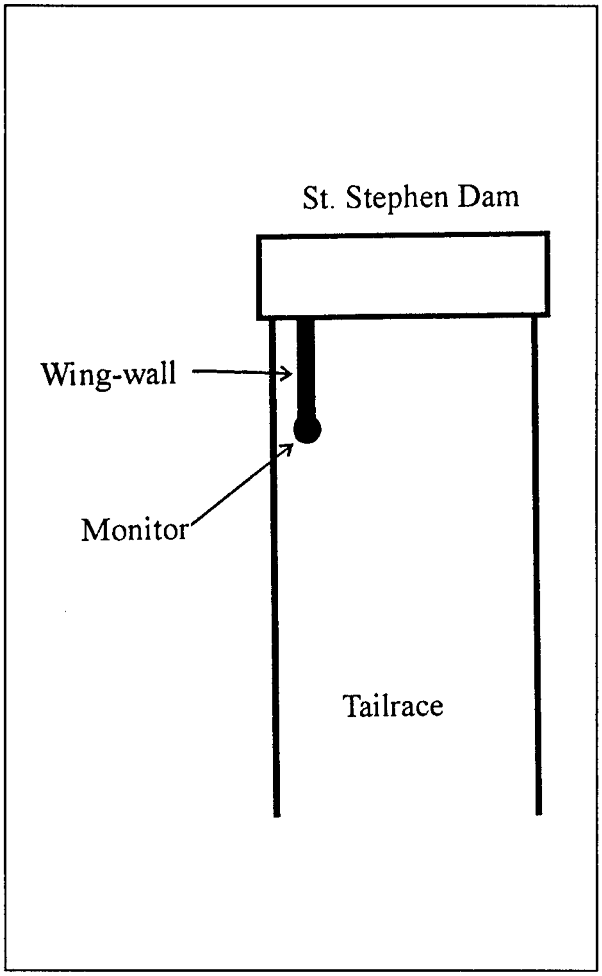

Figure 10. Schematic diagram of the St. Stephen Dam tailrace monitor

should be analyzed as they are collected, especially during the critical pre-installation sampling, as it may be necessary to redesign the sampling approach to better address the questions to be answered or address new questions that arise during the study.

Incorporation of all the available data, e.g., project operations data, meteorological data, all historical data for the project of concern, etc., aid in addressing issues that may require intensive sampling efforts to obtain. Valuable information may be realized from historical data sets that may have been neglected otherwise. The monitoring program should remain focused on the objectives that were outlined at its inception. Periodic evaluation of the monitor's performance, especially when there are structural and/or operational modifications to the project or monitor, should be a routine component of the analysis process. Re-evaluations of this nature are imperative for ensuring representative data collection.

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Statistical Verification of Mean Value Fixed Water Quality Monitor Sites in Flowing Waters

by
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Introduction

Water quality managers must carefully choose locations for fixed water quality monitors to ensure that the data they collect accurately reflect water quality conditions of the water of interest. Often, a monitor site will experience some spatial or temporal bias and data collected there will not represent the release or river in question.

For rivers and hydroproject releases, bias may be the result of the combined spill and generation releases (Lemons et al. 1996a), releases into lacustrine tailwaters (Vorwerk and Carroll 1993), generation drawing water from a forebay with heterogeneities (Lemons et al. 1996b), point sources of pollution, or other processes (Vorwerk and Jabour 1996). A monitor system intake may be located in some portion of a flow and accurately measure its water quality, while not reflecting the quality of other portions (Figure 1). Thus, to provide useable data for operation, regulatory, or background monitoring needs, a manager must verify the representativeness of monitor sites with regard to the monitoring program goals.

This verification must include a quantification of the spatial and temporal similarity between water quality data gathered at the monitor site and in the stream or river in question. Flowing water monitor systems can be designed to create temporal records of either of two kinds of water quality information, means or extreme values (Ward 1979). Different verification techniques are necessary for each of these designs. This paper will discuss techniques necessary to verify mean-value monitor systems.

To obtain mean values of water quality parameters in flowing water, the analyst must have some knowledge of the mixing processes present. In situ data are needed for the verification. If the stream is turbulent and well-mixed, it may be the case that any location can accurately represent the quality of the water. If the stream is not well-mixed and has heterogeneities in water quality, the data must be flow-weighted. Flow-weighted data allows the calculation of the mass transport of parameters through the cross section of the stream in time. Some examples of flow-weighting include temporal quantification of dissolved oxygen mass or average dissolved oxygen concentration moving down a river, a record of average total dissolved gas saturation, mass transport of nutrients, or a record of average temperature. The important aspect is that the value of the parameter of interest is averaged across the area of the channel cross-section with respect to velocity.

Any verification must be both qualitative and quantitative. This paper will formalize approaches at statistically quantifying and verifying the adequacy of monitoring sites for measuring the average water quality at river transect. Total dissolved gas data collected from the

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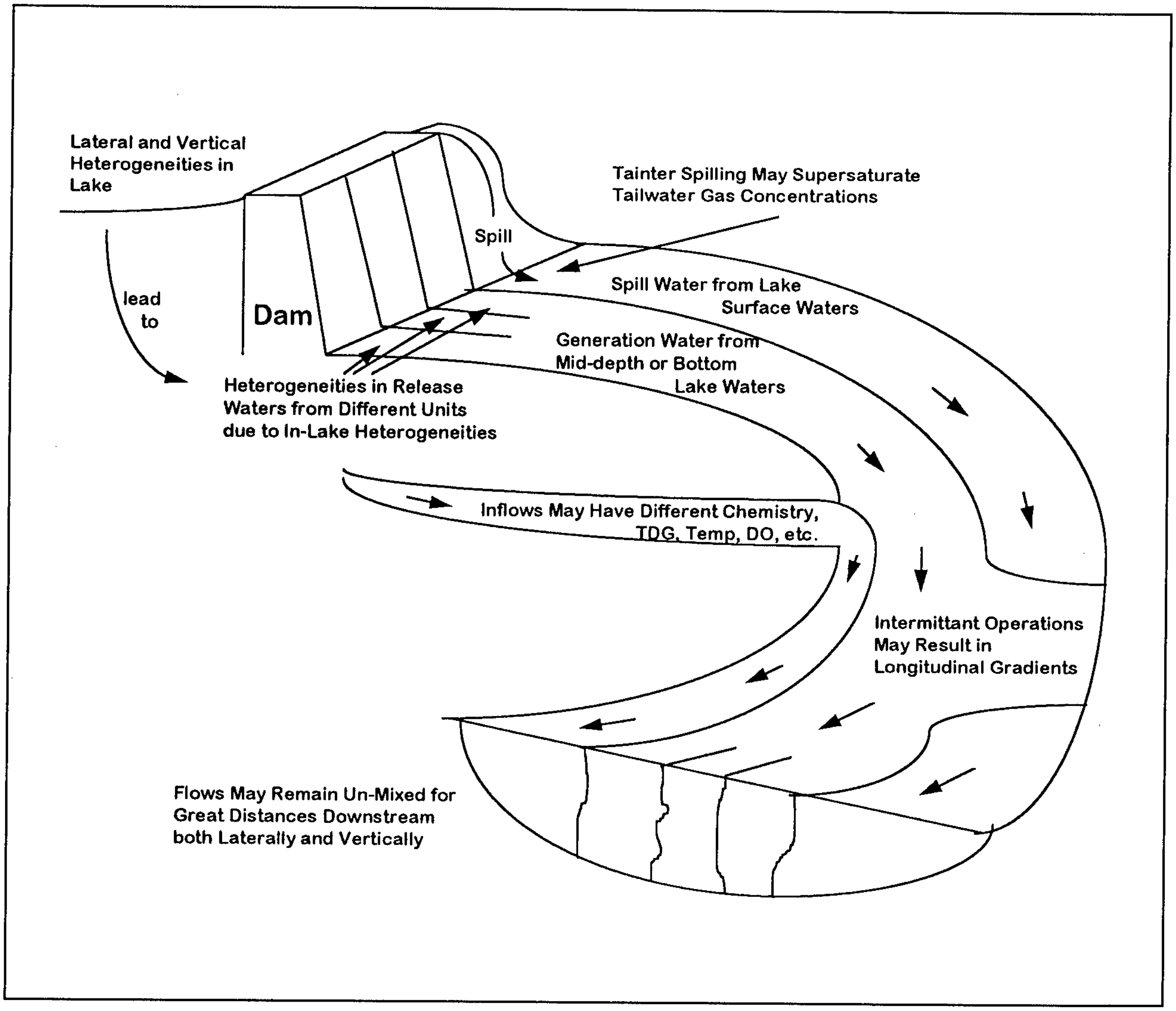

Figure 1. Some possible sources of heterogeneities in flowing water

Columbia and Snake Rivers will illustrate these techniques. The statistical methods provided will allow users with a basic knowledge of statistics to design and carry out studies to verify the representativesness of their own monitor locations. A review of statistics with water quality applications can be found in Gaugush 1986. Though this paper is based on the use of automated fixed water quality monitors, the procedure described should be applied to manual monitoring as well.

Approach for Mean Data

Data Collection and Preparation

The basic approach to verifying the representativeness of a monitor site is to compare matched pairs of observations from the monitor and averaged from the flow (Figure 2). These pairs must be taken over as many different times, flow conditions, and water quality variations as possible.

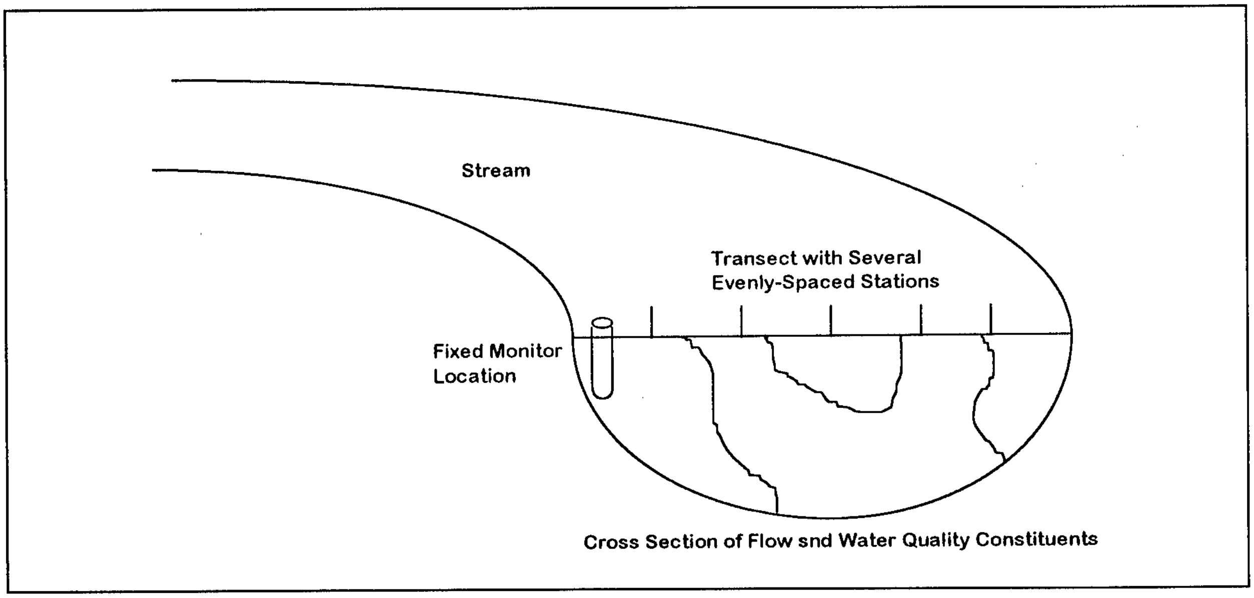

Figure 2. Cross-section of flow with evenly-spaced sample stations along a transect

The observations in the flow must be distributed so they adequately describe water quality conditions across the stream. For wider streams and rivers or for more highly variable water quality conditions, more sample locations are necessary. The sample values from the stream are averaged with an area-weighted average. If velocities vary greatly in the stream cross-section, the data averaging must also be flow-weighted. The next section provides details on this weighting.

In practice, data are often limited and the only available option is averaging the transect data with a simple arithmetic average, and then carrying out the statistical comparison. However, if the stations are not evenly spaced or if the water column has lateral or vertical heterogeneities in water quality or velocity, then a flow-weighted average should be calculated.

Flow-Weighting Data

The following method can be used to calculate a flow-weighted average. For each sample station i and depth z with water quality parameter value $P_{i,z}$, and velocity $U_{i,z}$, assign and area $A_{i,z}$, that the information gathered at that location represents (Figure 3). The area can be difficult to calculate and is most often approximated from depth soundings, maps, surveying techniques, global positioning equipment, and "best-guest." The transect flow-weighted average of the parameter P can then be expressed as:

$$\overline{P} = \frac{\sum_{all\,i} \sum_{z=0}^{z\max} A_{i,z}\, U_{i,z}\, P_{i,z}}{\sum_{all\,i} \sum_{z=0}^{z\max} A_{i,z}\, U_{i,z}} \qquad (1)$$

Averaging should be carried out for each sample time. Again, only in the most carefully designed and executed studies is such information available. More typically, an analyst may have three to seven lateral measurements along a transect to compare to fixed monitor information. In this case, the analysis can be performed, but the analyst must be aware that those limited data lessen the weight that may be given to any conclusions.

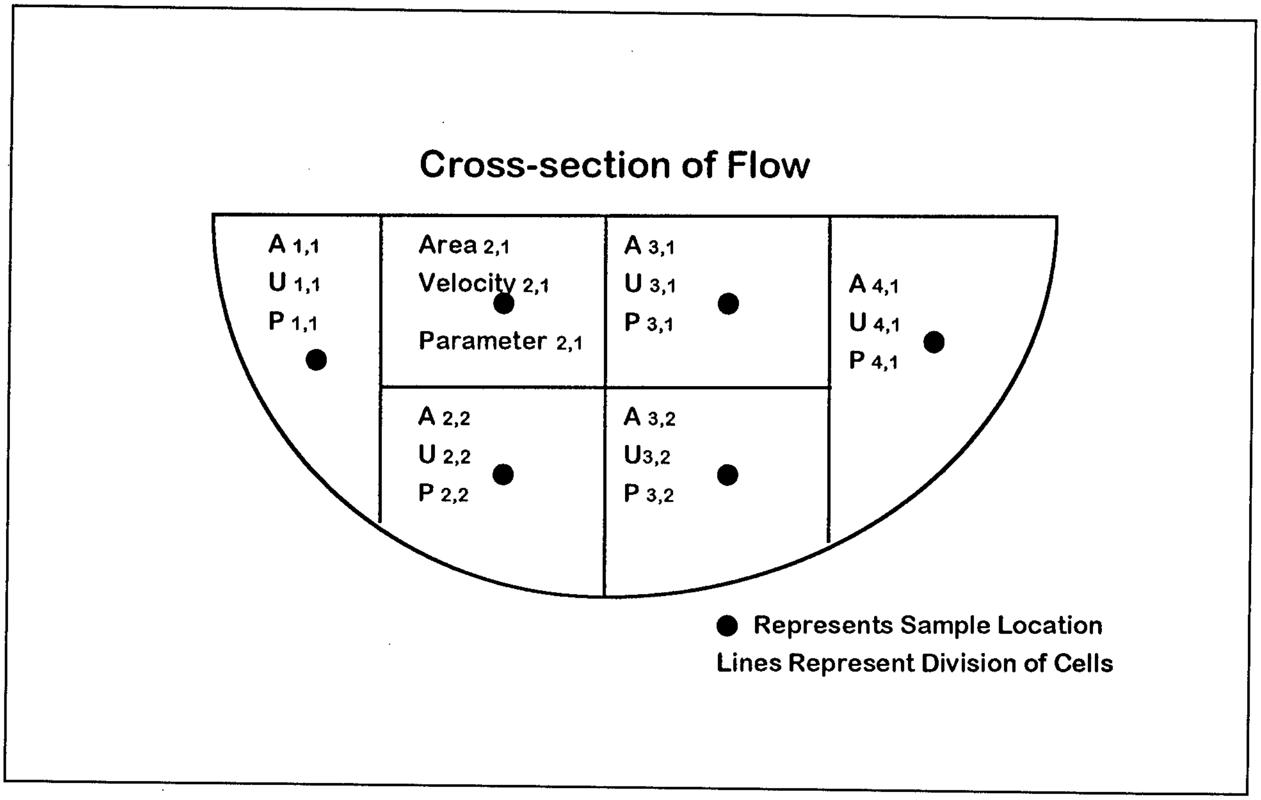

Figure 3. Hypothetical sample scheme for flow-weighting data

Statistical Comparison

At this point, the verification dataset should contain n pairs of data $(X_{m,j}, X_{s,j})$; each containing a monitor observation X_{mj} and an average stream value X_{sj} for time j and where m and s indicate that the observation that the observation came from the monitor or stream, respectively. We next test the relationship between the two locations using a paired t-test (following Hines and Montgomery 1980). This test assumes that the samples each come from a normally distributed, independent distribution. However, moderate departures from normality should not adversely affect the analysis (Pollard 1977). The difference between each pair of observations, $D_j = X_{mj} - X_{sj}$, should come from a normally distributed independent distribution.

To verify that the data come from a normally distributed population, either of two methods can be used. The easiest method is to plot the data on normal probability paper or use a statistics software package to generate a normal probability graph. A second method is to use a quantitative test such as the Kolmogorov-Smirnov test or Lillefore's test. Further details of these tests can be found in Hines and Montgomery 1980, and Pollard 1977. This paper will use normal plots generated using SPSS (SPSS Inc., Chicago, IL), a statistical analysis software package.

Once we have determined that the data come from a normal, or nearly-normal distribution, we begin the comparison by stating the hypotheses. The null hypothesis is that the mean of the differences between pairs, μ_D, is zero. This implies that monitor values agree with stream values and is representative. The alternative hypothesis is that the mean of the differences is not zero, that is, that monitor values do not agree with stream values and is not representative. This is stated as follows:

$$H_0{:}\mu_D = 0 \tag{2}$$

$$H_1{:}\mu_D \neq 0 \tag{3}$$

These hypotheses are tested with the following statistic:

$$t_0 = \frac{\frac{\bar{D}}{S_D}}{\sqrt{n}} \tag{4}$$

where

$$\bar{D} = \frac{\sum_{j=1}^{n} D_j}{n} \tag{5}$$

$$S_D^2 = \frac{\sum_{j=1}^{n} D_j^2 - \frac{1}{n}\left[\sum_{j=1}^{n} D_j\right]^2}{n-1} \tag{6}$$

and we reject H_0 if $t_0 > t_{\alpha/2,n-1}$ or if $t_0 < -t_{\alpha/2,n-1}$. The confidence level, α, is typically taken to be 0.05 and is the type I error, or the probability of rejecting H_0 when H_0 is true. If H_0 is rejected, we conclude that the fixed monitor system does not represent the water quality of the stream at the α confidence level. If H_0 is not rejected, we conclude that "we have not found sufficient evidence to reject H_0," (Hines and Montgomery 1980). This may be because the monitor site accurately represents the stream or because the sample size (i.e., number of comparisons) is so small that not enough data are available to make the stronger conclusion to reject H_0. So, for verification, we need a large enough sample size to minimize the type II error, i.e., the probability of accepting H_0 when H_0 is false.

Similarly, one sided hypotheses can be tested as follows:

$$H_0{:}\mu_D \leq 0, H_1{:}\mu_D > 0, \text{reject } H_0 \text{ if } t_0 > t_{\alpha,n-1} \tag{7}$$

$$H_0{:}\mu_D \geq 0, H_1{:}\mu_D < 0, \text{reject } H_0 \text{ if } t_0 > -t_{\alpha,n-1} \tag{8}$$

Determining the Power of the Test

The rejection of the null hypothesis is considered a "strong" conclusion because we control the type I error (choice of α), or probability of rejecting H_0 when H_0 is true. The acceptance of the null hypothesis, on the other hand, is considered to be a "weak" conclusion because we don't control the type II error (β), or probability of accepting H_0 when H_0 is false.

Thus, to determine the meaning of our conclusion when we accept the hypothesis that a monitor represents the flow, we must determine the type II error. For the monitor location to be acceptable, the type II error must be acceptably small.

To estimate the type II error, or β, a statistic d is calculated, and with α and n, β can be determined from operating characteristic charts available in statistics books (Hines and Montgomery 1980, page 604). Using Equations 5 and 6, we calculate d as follows

$$d = \frac{|\bar{D}|}{S_D} \tag{9}$$

Once β is found, the probability of correctly accepting H_0 is the power, namely $P = 1 - \beta$. Because we want only to correctly accept H_0, we desire the power to be as close to one as possible. The question then becomes "what's good enough?"

Since we typically choose α to be 0.05, it seems reasonable to attempt to hold β to a similar probability. However, because we have no direct control over β, probabilities less than 0.2 are probably sufficient. Thus, we consider comparisons with the power greater than 0.8 to be acceptable.

If we are designing a verification study, pilot studies, such as the one described in Examples 1 and 2, provide *a priori* knowledge of $\bar{D}$ and S_D. This information can be used to design the verification study with a sample size large enough to insure that the power is as great as desired. This is accomplished through increasing the sample size until the desired value for β is achieved on the operating characteristic curve.

Example 1: Columbia River Camas/Washougal Station, Hand Calculation

The following example illustrates this method with data from the Camas/Washougal total dissolved gas monitoring station (CWMW) on the Columbia River. To assist smolt in their downstream migration, the U.S. Army Corps of Engineers spills surface water from projects on the Columbia and Snake Rivers. This spillage causes air to be driven into the water column to depths where it causes gases in the water column to be supersaturated with respect to surface saturation. This supersaturation can be detrimental to fish, so the Corps monitors spill gas concentrations in the rivers. Thus, this system is designed to determine the extreme total dissolved gas concentrations resulting from spilling water. This information is used for compliance and in project operations. To determine if these monitors could be used to determine the flux of total dissolved gas in the river, the statistical verification studies presented in this paper were carried out. The verification is based on comparing monitor data with data collected at eight transects near the CWMW monitor site (river mile 122) on three different days (Table 1). The stations on the transects were approximately evenly-spaced, so the data for each transect was simply averaged together to obtain an average total dissolved gas concentration at that transect.

Figures 4 and 5 show normal probability plots of the transect and fixed monitor system data, respectively. Ideally, the data would be randomly distributed along the normal distribution line, with points close to and on either side of the line. Though the transect data in Figure 4 do not appear to be completely random about the normal line, they are sufficiently normal for this analysis. The data essentially fit the normal distribution line, but show a trend to

Table 1
Average Total Dissolved Gas as Percent Saturation at Columbia River Transects and CWMW Fixed Monitor

Date	Transect Mile	Tran. Ave, Percent Sat	Monitor, Percent Sat	Samples
18-May-95	119.9	115.1	113.4	5
25-May-95	121.2	118.1	115.5	5
25-May-95	121.6	119.0	117.3	5
25-May-95	122.1	119.4	118.5	5
25-May-95	119.9	117.0	113.4	7
27-Jul-96	121.2	112.1	109.8	32
27-Jul-96	121.6	116.0	111.9	15
27-Jul-96	122.1	112.9	109.5	15

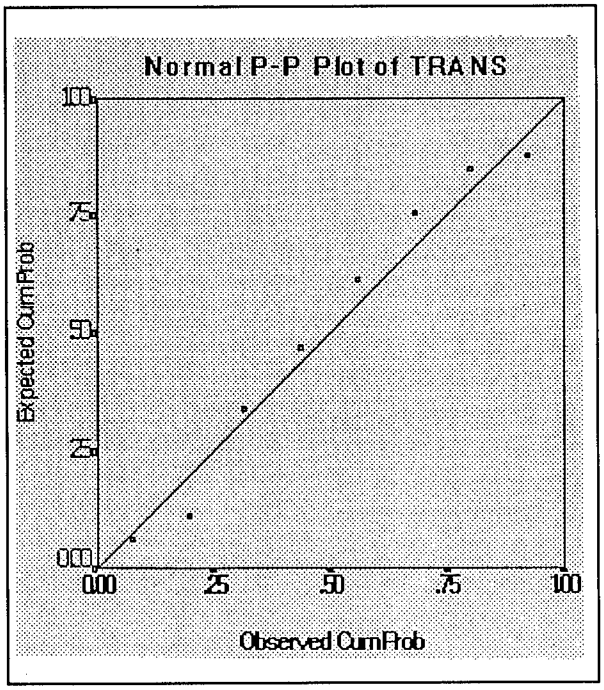

Figure 4. Normal probability plot of transect data. The straight line plots the normal distribution. The square symbols are the observed data

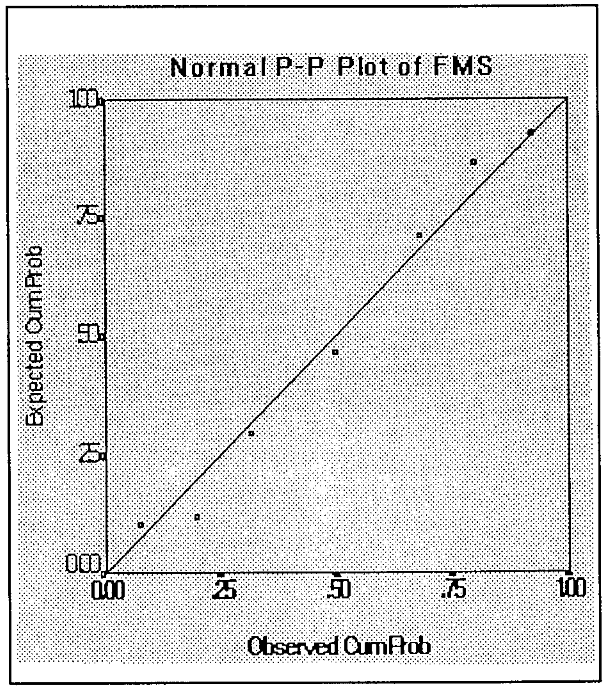

Figure 5. Normal probability plot of fixed monitor station data. The straight line plots the normal distribution. The square symbols are the observed data

be above the line for higher cumulative probabilities and below the line for lower cumulative probabilities. We conclude that the data are approximately normally distributed. Figure 5 suggests that the fixed monitor data are also normally distributed. Note that the data in Figure 5 are somewhat more randomly distributed on each side of the normal line, with fewer "runs" or continual observations on one side or the other of the normal line. This graphically-based determination is subjective. To lessen subjectivity, test (Kolmogorov-Smirnov, Lillefore's) as discussed above can be used, but the analyst is often forced to use what data are available.

Because the data were collected for another project and not specifically for monitor verification, the transect locations did not coincide exactly with the monitor location. For our comparison, all transects that were with 3.5 km of the fixed monitor station were selected. The number of samples varied with transect mile and date. May samples had five or seven evenly spaced measurements at a constant depth of 4.6 m. July samples had multiple depths and 5-7 sample locations. The calculations of the differences, the square of the differences, and the totals of the two sites are depicted in Table 2.

Figure 6 is a normal probability plot of the differences of the transect and fixed monitor data pairs. Though the data show some tendency to be lower than the normal plot for low probabilities and higher than the normal plot for high probabilities, the data appear to be approximately normally distributed.

Equations 2 and 3 were used to test whether the data collected at the fixed monitor site represents the water quality within the river. First, the parameters necessary for the test statistic were calculated. The mean difference (Equation 5) was:

$$\bar{D} = \frac{\sum_{j=1}^{n} D_j}{n} = \frac{20.1}{8} = 2.5 \qquad (10)$$

Table 2
Differences, Squares of Differences, and Totals for Data in Table 1. The Sample Size for the Comparison, *n*, is 8

Date	Transect Mile	D	D^2
18-May-95	119.9	1.7	2.9
25-May-95	121.2	2.6	6.8
25-May-95	121.6	1.7	2.9
25-May-95	122.1	0.9	0.8
25-May-95	119.9	3.6	13.0
27-Jul-95	121.2	2.3	5.3
27-Jul-95	121.6	4.1	16.8
27-Jul-95	122.1	3.4	11.6
Sum		20.3	60.1

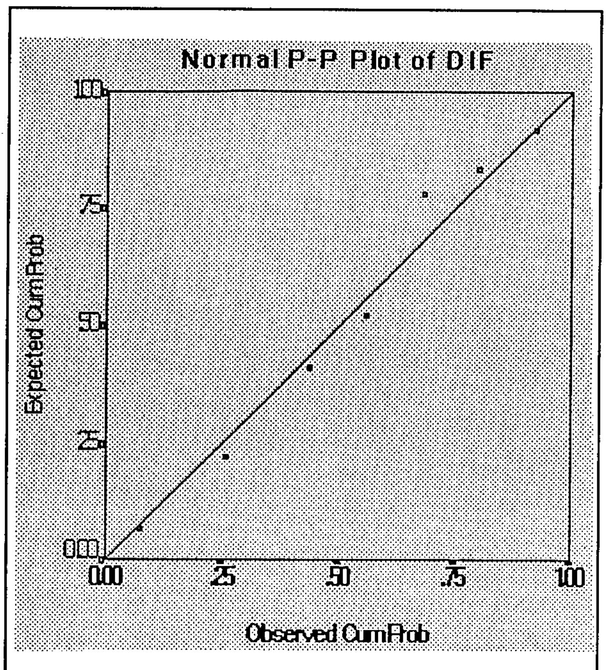

Figure 6. Normal probability plot of the differences between transect and fixed monitor station pairs of observations. The straight line plots the normal distribution. The square symbols are the differences

The variance was estimated using Equation 6:

$$S_D^2 = \frac{\sum_{j=1}^{n} D_j^2 - \frac{1}{n}\left[\sum_{j=1}^{n} D_j\right]^2}{n-1} = \frac{60.1 - \frac{1}{8}(20.1)^2}{8-1} = 1.4 \qquad (11)$$

The test statistic t_0 was then calculated using Equation 4:

$$t_0 = \frac{\frac{\bar{D}}{S_D}}{\sqrt{n}} = \frac{\frac{2.5}{1.2}}{\sqrt{8}} = 5.9 \qquad (12)$$

Next the test statistic calculated in Equation 12 was compared with $t_{\alpha/2,n-1}$. This value can be found in various statistics books in the students t table or t distribution table (Hines and Montgomery 1980, p. 596). For $\alpha = 0.5$ (our choice) and $\upsilon = n - 1 = 7$ (determined by the sample size of 8), the value of $t_{\alpha/2,n-1} = t_{0.025,7} = 2.365$ (from tables).

Since:

$$5.9 = t_0 > t_{\alpha/2,n-1} = t_{0.025,7} = 2.365 \qquad (13)$$

we rejected H_0 and concluded that the difference between the transect values and the fixed monitor station values was not zero. The fixed monitor did not adequately represent the water quality in the river at this location.

We tested the hypothesis that the transect percent TDG values were greater than the fixed monitor percent TDG values using Equation 7. We hypothesized that $H_0{:}\mu_D \leq 0$ with alternative $H_1{:}\mu_D > 0$. We rejected H_0 if $t_0 > t_{\alpha,n-1}$. Again using $\alpha = 0.5$ and $\upsilon = n - 1 = 7$ (determined by the sample size of 8), the value of $t_{\alpha,n-1} = t_{0.05,7} = 1.895$.

Since:

$$5.9 = t_0 > t_{\alpha,n-1} = t_{0.05,7} = 1.895 \qquad (14)$$

we rejected H_0 and concluded that the difference between the transect values and the fixed monitor station values was greater than zero. The fixed monitor consistently recorded total dissolved gas percent saturation values that were less than the average of those actually present in the river at this location during this study. Thus, we conclude that the fixed monitor system does not accurately represent the flux of total dissolved gas in the river.

To avoid tedious hand calculations, we typically use various software packages which calculate the paired t-test statistics for our data sets. Two commonly used packages are SPSS (SPSS, Inc., Chicago, IL) and SAS (SAS Institute, Inc., Cary, NC).

Example 2: The Columbia and Snake Rivers Fixed Monitoring System

Introduction

The technique employed in the above example can be used to look at an entire monitoring system. Though the fixed monitoring system is designed to determine extreme concentrations of total dissolved gas in spill waters, here we explore the potential of each station for use in monitoring the average total dissolved gas concentration in the river. The system consists of monitors at 26 sites. As in Example 1, these fixed monitor sites were compared to transect data collected during 1995.

Again, because the transect study was designed to aid modelers and not strictly to verify the fixed monitor system, adequate data were not available for each location. The analysis shown here was intended only to provide insight into the representativeness of the monitoring system. Details, such as verifying normality, have been omitted. The results presented here might best be used to design future, more rigorous verification studies.

Data Collection

Transects within 3.5 km of each fixed monitor site were used for comparisons to fixed monitor data. This created a larger dataset than if only transects that were adjacent to the monitor sites were used. Larger datasets reduce the type II error, i.e., the probability of accepting H_0 when H_0 is false. The paired test requires at least two pairs of data for each site. This constraint eliminated 3 stations, leaving 23 for further possible analysis.

Results

Because of the number of comparisons that were desired, SPSS was used to analyze the data. A paired t-test was run on each of the 23 fixed monitor sites and their comparable transect data. The results of these analyses are shown in Table 3.

The "relationship" column was created by comparing the "T Value" column (t_0) with values from a student-t table using the degrees of freedom in the "d.f" column. First, we tested to see if the difference was zero. If this was not rejected, we labeled the "Relationship" column "Accept Null Hypoth."

If the null hypothesis was rejected, Equations 7 and 8 were used with the appropriate values from the table to determine whether the transect data were greater than or lesser than the fixed monitor data. These results were labeled in the "Relationship" column as "Transect > FMS" or "FMS > Transect," respectively.

For eleven of the 23 stations, the statistical tests rejected the hypothesis that the FMS and transect data were equal. This means that data collected at these FMS sites did not reflect the water quality conditions occurring across the river.

These stations, which had non-equivalent FMS and transect comparisons, are marked with

Table 3
Verification of Fixed Monitor Station Location with Transect Data

Station	FMS Mean[1]	Trans. Mean[1]	Dif. Mean[1]	Dif. s. d.[1]	T Value	d. f.	2-Tail Sig.	Relationship[2]
BON[3]	108.5	111.3	-2.8	0.4	-16.1	3	.001	FMS > Transect
CWMC[3]	113.7	116.2	-2.6	1.1	-6.6	7	.000	Transect > FMS
HPKW[4]	113.7	116.0	-2.3	4.0	-.8	1	.565	Accept Null Hypoth.
IDSB[3]	126.8	120.9	5.9	6.8	3.1	12	.009	FMS > Transect
IDSW[3]	126.9	120.9	6.1	6.0	3.7	12	.003	FMS > Transect
IHR[4]	111.8	111.9	-0.1	0.5	-0.3	2	.794	Accept Null Hypoth.
JDA[3]	107.8	106.0	1.8	0.2	15.2	1	.042	FMS > Transect
JHAW[3]	109.8	106.5	3.3	3.7	2.4	6	.053	FMS > Transect
KLAW[4]	109.8	110.2	-0.5	0.5	-2.0	3	.152	Accept Null Hypoth.
LGNW	109.3	108.9	0.4	1.8	1.0	13	.362	Accept Null Hypoth.
LGS[3]	107.3	108.1	-0.7	0.0	-33.7	1	.019	Transect FMS >
LGSW	110.7	113.7	-3.0	7.8	-1.5	13	.169	Accept Null Hypoth.
LMNW[3]	117.6	113.9	3.8	0.8	13.7	7	.000	FMS > Transect
MCPW[3]	117.9	115.4	2.4	2.6	3.7	15	.002	FMS > Transect
MCQO[4]	113.9	112.7	1.1	1.4	1.6	3	.202	Accept Null Hypoth.
MCQW[4]	112.0	112.7	-0.7	2.2	-0.6	3	.569	Accept Null Hypoth.
SKAW[3]	112.9	114.1	-1.2	1.2	-2.8	7	.026	Transect FMS >
TDA[4]	106.0	106.2	-0.1	0.8	-0.2	1	.852	Accept Null Hypoth.
TDAB[4]	105.8	106.2	-0.3	0.7	-0.7	1	.621	Accept Null Hypoth.
TDTO[3]	112.0	115.5	-3.5	1.3	-8.6	9	.000	Transect FMS >
WANO[4]	106.5	106.6	-0.1	0.2	-0.5	2	.682	Accept Null Hypoth.
WRNB	113.5	114.0	-0.5	0.9	-1.4	6	.227	Accept Null Hypoth.
WRNO	114.4	114.0	0.5	0.8	1.7	6	.147	Accept Null Hypoth.
AGGR. FILE	114.8	113.9	0.9	4.6	2.6	156	.012	FMS > Transect

[1] Variable is total dissolved gas percent saturation.
[2] Decision made at alpha = 0.05 significance level.
[3] Additional study recommended.
[4] Additional data collection recommended.

an ampersand. We would recommend further analysis on these stations to determine if the fixed monitor system needs to be moved, modified, or increased in scope. It is possible that the differences detected occur uniformly, allowing a simple addition or subtraction from the FMS data to then accurately represent river conditions. If the variance is large temporally or spatially, these stations should be relocated. To ensure the validity of these conclusions, it is generally accepted that a sample size of at least seven is necessary.

At the remaining 12 stations, the null hypothesis that the FMS and transect data were equal was accepted. This may be because the FMS adequately represents the transect, or simply because the limited data did not provide sufficient evidence to reject the null hypothesis. Thus, further analysis is needed to determine whether or not the monitors represent to flow.

Determining the Power of the Test

Using Equation 9, we calculated the statistic d for each station where the null hypothesis was accepted. These results are shown in Table 4. The table shows that in no case is the power greater than 0.32. Thus, we conclude that in each case where the conclusion of the test was to accept the null hypothesis (fixed monitor data represents water quality conditions in the river), there are insufficient data to make a reasonable statistical decision.

We next calculated the necessary sample size for each of these twelve stations to obtain the desired a target power of 0.8. These values are shown in Table 5. With the exception of KLAW and MCQO, the sample sizes are somewhat unrealistic. This occurs because of the relationships between the sample means and standard deviations. From Equation 9, $d = \frac{|\underline{D}|}{S_D}$. The power of the test H_0 relies, in addition to the sample size n, on this relationship. In these other stations the variance is so large in comparison to the mean that sample sizes are not reasonable. This implies that the fixed monitors are not located in such a way that their values change uniformly with the flow values. Thus, a first step at improving these monitors would be to place them in locations experiencing more uniform changes with flow

Table 4
Calculation of Parameters Necessary to Determine β, β, and the Power of the Test

Station	$\bar{D}$	S_D	$d = \frac{\lvert\bar{D}\rvert}{S_D}$	n	β From Table	Power
HPKW	-2.3	4.0	0.58	2	0.94	0.06
IHR	-0.1	0.5	0.20	3	0.96	0.04
KLAW	-0.5	0.5	1.0	4	0.74	0.26
LGNW	0.4	1.8	0.22	14	0.90	0.10
LGSW	-3.0	7.8	0.38	14	0.76	0.24
MCQO	1.1	1.4	0.79	4	0.79	0.21
MCQW	-0.7	2.2	0.32	4	0.93	0.07
TDA	-0.1	0.8	0.13	2	0.97	0.03
TDAB	-0.3	0.7	0.43	2	0.95	0.05
WANO	-0.1	0.2	0.50	3	0.92	0.08
WRNB	-0.5	0.9	0.56	7	0.72	0.28
WRNO	0.5	0.8	0.63	7	0.68	0.32

Table 5
Determination of Sample Size Necessary to Obtain Desired Power of 0.8

Station	$d = \frac{\lvert\bar{D}\rvert}{S_D}$	n
HPKW	0.58	28
IHR	0.20	300
KLAW	1.0	10
LGNW	0.22	300
LGSW	0.38	75
MCQO	0.79	15
MCQW	0.32	75
TDA	0.13	400
TDAB	0.43	50
WANO	0.50	32
WRNB	0.56	30
WRNO	0.63	25

and in increasing the number of fixed monitor locations across the flow.

Conclusions

This paper has demonstrated statistical techniques for verifying the representativeness of fixed monitoring systems that monitor mean-values of parameters in flowing water. These techniques were illustrated with data collected on the Columbia and Snake Rivers. Based on the criteria detailed in this paper, a preliminary analysis of the 1995 Columbia and Snake Rivers fixed monitor system dataset revealed that none of the fixed monitor systems accurately represented the average river total dissolved gas concentrations. This demonstration was, however, based on limited transect data which were not specifically collected for the purposes of monitor site verification.

These examples illustrated the usefulness of the statistical approach in eliminating the subjectiveness involved in determining whether or not a monitoring station accurately represents the water quality in a river. The information presented can be used to guide managers to the most problematic locations, so improvements can be made on a "worst case first" basis. Additionally, pilot studies similar to the ones used to collect the data used in this paper can be used to help design verification studies to control the power of the test, obtaining the desired trust in the results.

Many other factors such as cost, ease of accessibility, and equipment availability contribute the difficulties in monitor system design and installation. The cost of an intensive analysis like the ones described above may be prohibitive to many water quality managers. However, the ideas presented in this paper should make the manager more aware of the difficulties involved in collecting representative data and improve the final system design.

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Total Dissolved Gas Abatement Study on the Clearwater River, Idaho

by
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Russell D. Heaton,[2] and Thomas D. Miller[2]

Introduction

Several strategies intended to aid the outward migration and increase survivability of salmonids in the Lower Snake and Columbia Rivers are currently being evaluated. Flow augmentation increases river flows above levels occurring under normal operations by releasing water from storage reservoirs. Unfortunately, diverting water over spillways creates a potential serious drawback; Total Dissolved Gas (TDG) supersaturation.

This phenomenon occurs when the partial pressure of atmospheric gases in solution exceed their respective partial pressures in the atmosphere (Fidler and Miller 1994). The total dissolved gas concentration at depth becomes supersaturated relative to the conditions at the air-water interface. High levels of TDG can lead to gas bubble disease or trauma. The symptoms can be harmful and are often fatal to fish and aquatic organisms.

During the summer of 1995, 53-mile-long Dworshak Reservoir was lowered for flow augmentation to reduce travel time of outmigrating smolts in the Lower Snake River. Authorized by the U.S. Army Corps of Engineers (USACE), the State of Washington Water Research Center conducted a TDG abatement study on the Clearwater River, Idaho. Dissolved gas was measured starting at the tailrace of Dworshak dam and ending near the confluence with the Lower Snake River (Figure 1).

The intent of this study was to determine TDG concentrations along a natural, free-flowing river and ascertain where the water degassed. Although high gas levels dissipated very quickly from the tailrace area, elevated concentrations remained in solution further downstream. This paper will also confirm TDG concentrations did not exceed the federal and state water quality criteria of 110 percent saturation.

Total Dissolved Gas Supersaturation

Background

A wide variety of natural and anthropogenic causes result in TDG supersaturation. Solar heating of water bodies, oxygen production by algae and aquatic plants, injection of air into pumping systems and warm water discharges from fossil fuel and nuclear power generating plant are recognized sources. High levels of TDG are known to be caused by hydroelectric and impoundment dams. Westgard (1964), Ebel (1969), Beiningen and Ebel (1970), and Ebel and Raymond (1976) recognized a serious problem to anadromous fish in the Columbia River Basin due to the discharge of water through hydraulic structures. The studies

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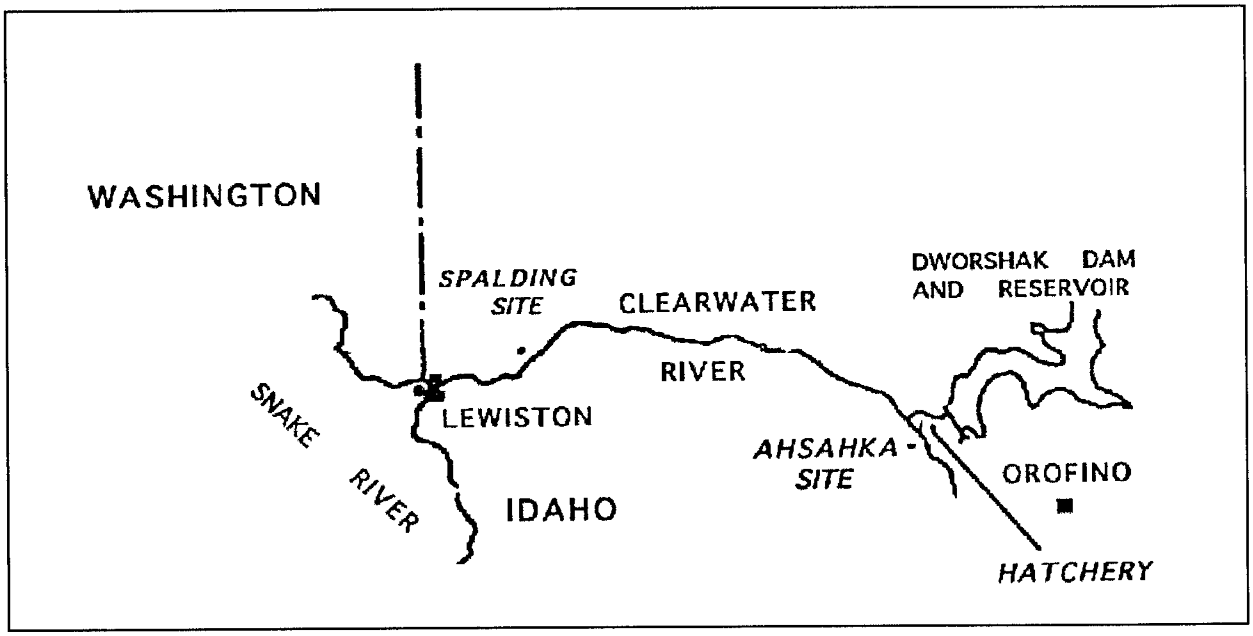

Figure 1. Map of Clearwater River study area

showed large quantities of water plunging over spillways were the main cause of supersaturation. In dam sluice ways and radial gates, air is entrained in falling water which plunges to depth in pools at the base of the dam. There, under elevated hydrostatic pressure, air (in the form of bubbles) is forced into solution at pressures of several atmospheres (Fidler and Miller 1994). Fish and aquatic organisms in supersaturated water may form gas bubbles in or on their bodies, if the gas pressure is not compensated through behavioral or physical processes (BPA 1994).

Marsh and Gorham (1905) established the basic knowledge of gas bubble disease. Bouch (1980) defined gas bubble disease as a non-infectious, physically induced process caused by uncompensated hyperbaric pressure of total dissolved gases. Alderdice and Jensen (1985) proposed the term "gas bubble trauma" (GBT). Trauma implies physical injury. Disease may include trauma as well as subsequent disturbance to function, structure, or systems of the body.

Numerous studies summarized by Weitkamp and Katz (1980) and Colt et al. (1986) have focused on fresh water fish with major emphasis on trout and Pacific salmon species. White et al. (1991) described the major symptoms of GBT in fish as: 1) bubble formation in the cardiovascular system; 2) over-inflation of the swim bladder and the intestinal and peritoneal cavities; 3) subdermal emphysema on body surfaces, including the lining of the mouth; 4) extra-corporeal bubble formation in gill lamella; 5) emphysema in muscles, internal organs and the spinal cord; and 6) exophthalmic (pop eye).

Dissolved gas levels in the Columbia and Snake River systems are of special concern. It is very significant due to recent listings of several sockeye and chinook salmon stocks as threatened and endangered under provisions of the Endangered Species Act.

Study Area

Dworshak Dam stands on the North Fork Clearwater River near Orofino, Idaho. It is the largest dam ever constructed by the USACE and the highest straight-axis, concrete-gravity dam in the Western World. Straight-axis denotes the alignment of the dam is straight from abutment to abutment, rather than concave or convex.

Dworshak Dam operations have been recently adjusted to provide for temperature control of the Snake River during the summer months and for augmenting Snake River discharges during the spring outmigration of juvenile fish. The dam has created a 53-mile-long reservoir.

Approximately two miles from the dam's tailrace, the North Fork flows into the main stem Clearwater. The Clearwater River originates in the Bitterroot Mountains near the Idaho-Montana border and flows west converging with the Snake River at Lewiston, Idaho (Figure 1). Explorers Lewis and Clark navigated the Clearwater River during their expedition to the Pacific Ocean in the fall of 1805.

Methods

TDG is one of the water quality parameters that cannot be measured without direct sampling or in-situ sensing (USACE 1995). Due to its high variability, a fixed monitoring station located at Dworshak National Fish Hatchery was not adequate to investigate the dissipation of TDG downstream.

Sample sites along the Clearwater River were located at one-mile intervals for the first ten miles; at two-mile intervals for the next ten miles; at three-mile intervals for the next twelve miles; and at four-mile intervals until the confluence with the Snake River. TDG data were collected from August 8th through August 22nd. Universal Transverse Mercator (UTM) coordinates located the river mile and the sampling site.

TDG levels were measured using tensionometers produced by Common Sensing, Inc. These instruments employ semi-permeable silicon tubing connected to pressure transducers and associated electronics to directly sense the total gas pressure in the water sampled (USACE 1995). All instruments were calibrated weekly with standard barometers, sphygmomanometers and thermometers. Daily calibration checks and adjustments involved tweaking the barometer and total dissolved gas pressure (P_T) ports with barometric pressure readings from the hatchery monitoring station. Equilibration time was 20 to 30 minutes in the water for the initial reading. During data collection, total pressure readings were considered equilibrated if they did not change more than 1 mm Hg. The TDG probe was lowered approximately six feet except in shallow water. Global Positioning System (GPS) coordinates, water temperature, barometric pressure, P_T and percent saturation, and probe-on-station start and stop times were recorded upon stabilization. A Hydrolab wireless instrument was utilized to record dissolved oxygen and other parameters for future studies. Wireless probes were also deployed at Ahsahka and Spalding stations (Figure 1) to record diurnal background and downstream data.

Results and Discussion

Dworshak Dam produces dissolved gas during operation of the powerhouse, spillway and regulating outlets. Total dissolved gas supersaturation was highest near the tailrace of the dam. It decreased as the water moved downstream, but elevated concentrations remained in solution (Figure 2). Because the rate of equilibration was slow, dissolved gas levels persisted far from the source of supersaturation. Average spill discharge was 3500 cubic feet per second (cfs).

Degassing of the water was attributed to dilution and mixing of the gassed water with the powerhouse water (10,000 cfs) and the confluence with the main stem Clearwater (River Mile 40). The federal and state water quality standard of 110 percent saturation (U.S. EPA 1986) was not exceeded.

Diurnal data collected at Ahsahka and Spalding stations (Figure 3) show a correlation with water temperature. As the temperature of the water rises, percent saturation increases because of the reduced solubility of gases at higher temperatures (U.S. EPA 1986). Weitkamp and Katz (1980) and Colt (1984 and 1986) discussed the rises in dissolved gas levels caused by

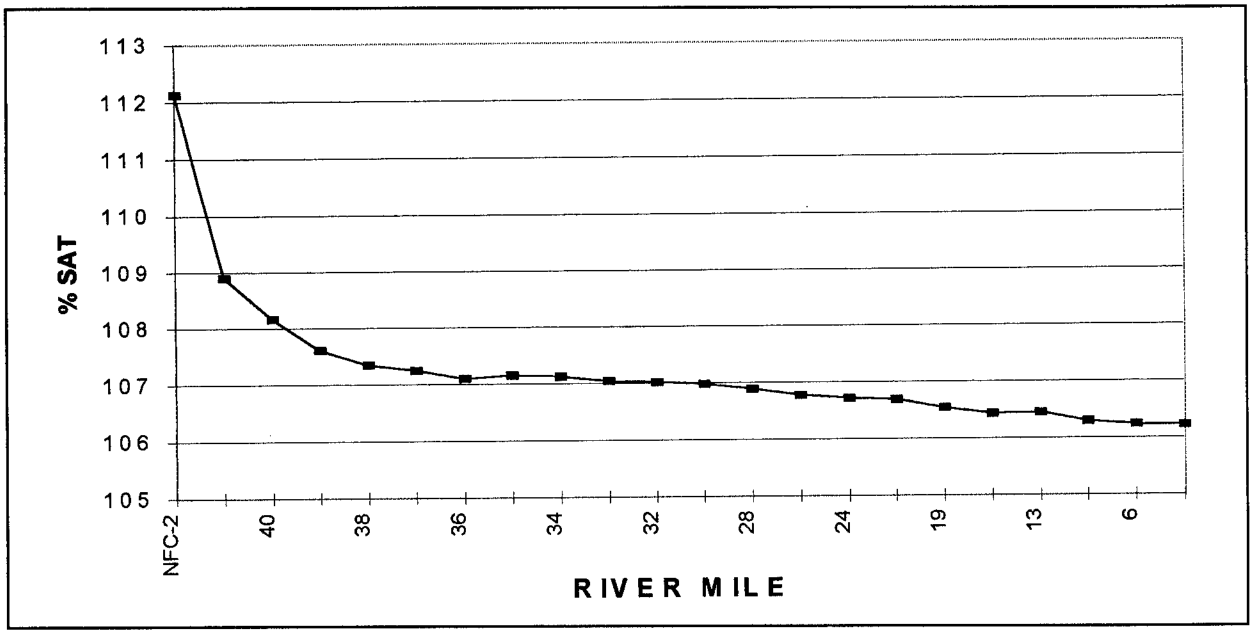

Figure 2. Mean TDG percent saturation on Clearwater River (3.5 Kcfs spill)

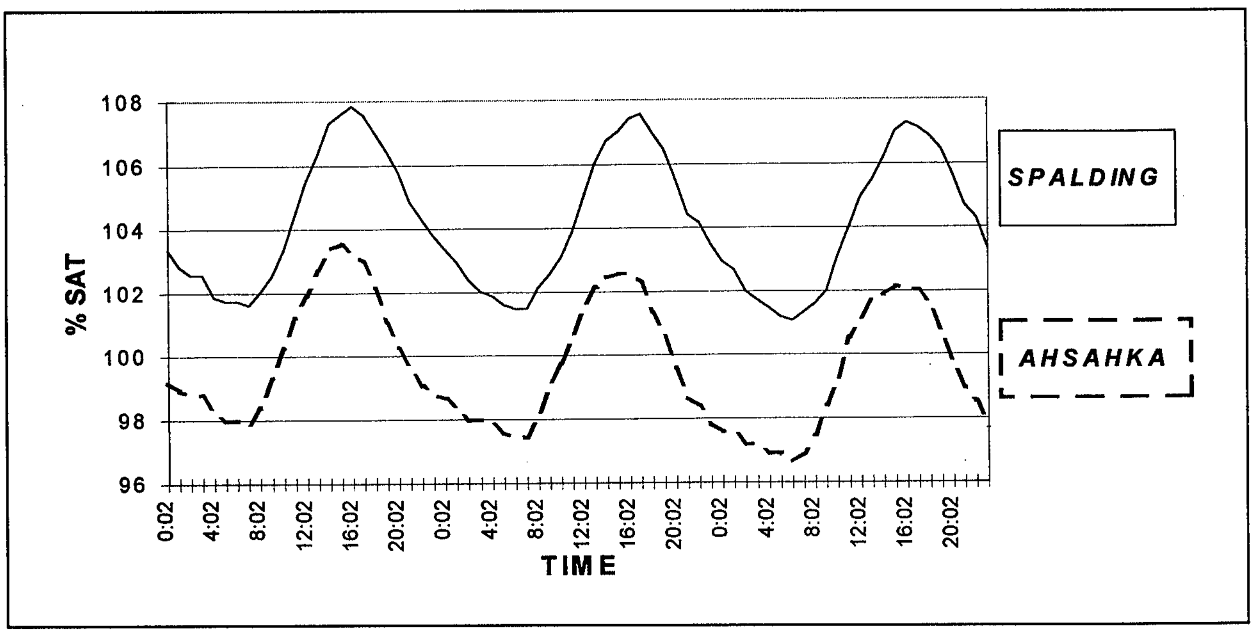

Figure 3. Diurnal TDG percent saturation at Ahsahka and Spalding stations

reduced gas solubility. As the river warmed up due to solar heating, percent saturation increased. Highest gas levels occurred between 16:00 and 18:00 hours. As the water temperature decreased, percent saturation declined.

Future research may be needed to determine the spatial, seasonal, and diurnal changes in total dissolved gas supersaturation along the Clearwater River. At each river mile, a set of sampling sites should be located on a lateral line that transects the river in order to adequately define the transport and mixing characteristics of TDG throughout the study area. A better understanding of the physical processes and biological behavior occurring in the river is extremely important in resolving the TDG supersaturation issue.

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Total Dissolved Gas Studies on the Lower Columbia: An Overview

Faith E. Ruffing[1]

Introduction

Dissolved gas saturation occurs in water when atmospheric gasses are equilibrated with gasses dissolved in water. When the ratio of the pressures of the total dissolved gases (TDG) to the barometric pressure (BAR) exceeds 1, the water is said to be supersaturated. Dissolved gas problems have been identified below Corps projects since the middle of this century. In recent years attention has focused on dissolved gas production below dams in the Columbia River Basin. This paper is an overview of the evolution of policies, regulations, programs, objectives, and observations that led to the total dissolved gas studies on the Lower Columbia conducted by the Portland and Walla Walla Districts.

No Spill Policy

Supersaturation results from the entrainment of the atmospheric gasses into the water as it passes over the spillway of Corps dams in the Columbia River Basin. According to the federal water quality standard the percent saturation (ratio of TDG to BAR $\times$ 100) may reach 110 percent before concern for aquatic organisms becomes an issue. The percent saturation may reach as much as 140 percent below some Corps projects. Corps policy has been to operate the dams in a manner which would reduce the amount of spill and thereby keep the gas saturation at a minimum. Figure 1 summarizes some of the basic concepts for the no spill policy for gas abatement.

Endangered Species Act

In 1994 Snake River spring/summer Chinook salmon and the Snake River fall Chinook salmon were reclassified from threatened to endangered joining the Snake River sockeye salmon listed as endangered since 1991. Figure lists points considered to bring about a change in policy from no spill to spilling for fish survival. The National Marine Fisheries Service in its Biological Opinion of 1995 described the flow levels determined to contribute to the survival and recovery of the listed Snake River stocks. Flows of 85-100 kcfs in the Snake River and 220-260 kcfs in the Columbia from April to June, and 50-55 kcfs in the Snake and 200 kcfs in the Columbia from June to August were recommended for 1995 and future years. Figure 2 summarizes the impact the Endangered Species listing of salmon had on the operations of the Corps Projects.

Cooperation, Interaction, and Integration

The "increased spill to save the salmon" focused the problem on Corps projects. The environmental debate took on "spotted owl" dimensions because of the broad impact on all economic bases in the region. In 1994, the Corps responded to a request from the National Marine Fisheries Service to spill more water in the Columbia Basin to improve the fish passage efficiency of the smolts migrating down the river to the ocean. A regional effort of

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Atmospheric Gases, (Nitrogen, Oxygen, Argon and other gasses in trace amounts) are entrained in the water as it is released over the spillway at Corps Projects on the Columbia and Snake Rivers.

The total pressure from these dissolved gases can be measured and compared to barometric pressure to determine the Total Dissolved Gas (TDG) percent saturation.

Equation for Percent saturation:
Total dissolved gas pressure / barometric pressure X 100.

EPA Standard:
TDG percent saturation will not exceed 110. The Oregon Department of Environmental Quality further defined the standard as not to exceed 105 for water less than 2 feet in depth.

Levels of Gas Bubble disease onset:
Fish and other aquatic organisms are impacted at percent saturation above 110%. Impacts increase with saturation and saturation above 115% may be lethal depending on species and life stage.

Percent Saturation of Total Dissolved Gas in waters below Corps Projects increases with the quantity spilled. The percent saturation exceeds 110 % below all Corps projects in the Columbia and Snake Rivers during peak flows of spring and summer. The maximum saturation ranges from 120 to 140 % depending on the project.

Corps policy has been to reduce spill, thereby reducing TDG % saturation to within the water quality standards.

No spill, no entrainment, no TDG saturation, no fish kills.

Figure 1. No spill policy

Decreasing fish populations in the Columbia and Snake River Basins resulted in the listing of Salmon species as endangered under the criteria of the Endangered Species Act.

Environmental changes over the last century contributing to the population declines included:
loss of habitat, spawning beds and food source;
water quantity reduction and quality degradation;
decreased velocities and increased travel times;
barriers to migration posed by Corps and other regionally owned and operated projects in the Columbia and Snake Rivers.

These barriers prevent free flow of the water and the fish that move through it guided by the currents and the nutrients carried with them. Corps developed juvenile and adult bypass systems and a fish barging program to mitigate impact structures have on migration.

Regionally developed solutions to increase fish survival included increasing the spill release at Corps Projects on the Lower Columbia and Snake Rivers. This would provide current through the pools to the dams and releasing water over the spillway would provide a means of moving smolts past the dams quickly.

The volume of water requested by the National Marine Fisheries Service would surely increase the Total Dissolved Gas Saturation above the Water Quality Standards and to the lethal range of 115 to 140 %.

The Corps was challenged to release water to increase fish survival while at the same time creating water quality conditions known to be lethal to the fish they were intending to save.

In addition to the increased saturation, the loss of water through spill has economic impacts on regional hydropower production, irrigation, recreation, navigation and wetland development and protection. These losses impact just about everyone in the Pacific Northwest.

Figure 2. Endangered species act

cooperation, interaction, and integration involved a number of agencies working together to address the issue of saving the endangered salmon species in the Columbia River Basin. Figure 3 identifies the agencies involved in this effort.

Cooperation between the Corps, National Marine Fisheries Service, Environmental Protection Agencies and interaction and integration with other state and local agencies, tribal entities, private companies and community organizations focused this effort on solutions rather than conflict.

Agencies
U S Army Corps of Engineers
North Pacific Division; NPP, NPW
Consultants: WES, USGS.
U S, Oregon, Washington, and Idaho Fish and Wildlife
National Biological Services
National Marine Fisheries
Fish Passage Center
Tribal Council
Bonneville Power Administration
Northwest Power Planning Council
Washington State Department of Ecology
Oregon State Department of Environmental Quality
Bureau of Reclamation
University of Washington

Figure 3. Cooperation, Interaction, and Integration

Fixed Monitoring Stations in the Columbia and Snake River Basin

A system of permanent monitoring stations, first initiated in 1984, consists of a set of 34 stations. This system was established through a cooperative effort between the Corps of Engineers North Pacific Division, the U.S. Bureau of Reclamation, the Bonneville Power Administration, and the Douglas, Chehlan, and Grant County Public Utility Districts. (Figure 4, Map of Monitoring Stations.)

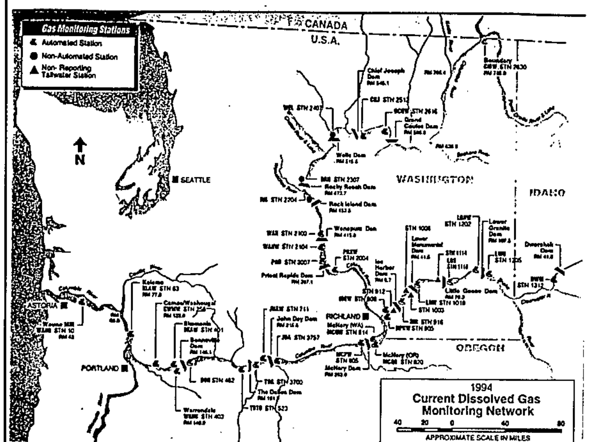

Figure 4. Map of monitoring stations

Originally stations were located in the forebays of the dams below the spilling project. Changes in the program included locating monitors in the tailrace of the spilling projects, on both sides of the river for some of the larger projects and installing duplicate instruments at some of the more critical projects. Data from these stations has been compiled into annual reports prepared by the North Pacific Division. Some of the observations made from the data collected are displayed in Figure 5.

Objectives:

To provide the water quality data needed to schedule spill at Columbia and Snake River Projects.

To monitor for compliance with the existing state water quality standards.

Observations:

TDG Saturation is above Water Quality standards for reaches of the river below the dams during high flows of spring and moderate summer flows.

TDG Saturation is related to spill at some of the projects.

TDG Saturation measured in the tailrace of some of the dams has diel fluctuations when the spill is constant and when the spill is turned on and off on a daily basis.

TDG Saturation is related to the design of the dam.

Figure 5. Objectives and observations made from the fixed monitoring system program

Dissolved Gas Abatement Study

In addition to enhancing the fixed monitoring system, the Corps initiated a planning program called the Dissolved Gas Abatement Study. This program consists of two phases and several elements that are moving forward simultaneously to address dissolved gas supersaturation in waters below the eight Corps projects in the Columbia and Snake Rivers. Figure 6 gives Phase I and Phase II program objectives and elements and identifies the types and objectives for the physical field studies completed in FY95 as part of Phase I. The significant portion of the work included the transect studies. Information from these studies have been used by others outside the district and form the basis for several papers presented at this conference. The data is being used to determine the plan of study to be carried out for Phase II. It is expected that the field work for the planning will be completed by FY98 and Phase II of the Program will be completed in FY01.

Database Development

The fixed monitoring program provided data collected hourly over six month periods over the past several years. This gives a detailed picture of dissolved gas saturation along the longitudinal axis of the rivers. The one dimensional pictures provided a wealth of data on gas saturation in the river under a variety of discharge and spill conditions. The data were limited to stations in the forebay and tailrace of the dams without

Objectives

Phase I

To define, evaluate and recommend potential methods to reduce Total Dissolved Gas supersaturation created during spillway operations of the eight existing U S Army Corps of Engineers projects on the Lower Snake and Columbia Rivers. Operational as well as structural modifications will be analyzed in an effort to lower TDG levels and improve the survival of migratory fish.

Phase II

To further evaluate each alternative as recommended in Phase I

Program Elements.

Alternatives analysis
Prototype Testing
Mathematical and Physical Modeling
Biological Field Studies
Physical Field Studies
Database development

Physical Field Studies

Objectives

To development of a protocol which meets QA/QC criteria

To evaluate the performance of the Fixed Monitoring Stations

To establish a 2 and 3 dimensional picture of TDG saturation in the rivers, lateral, longitudinal and vertical which enhances the data from the fixed monitoring system

To develop a database which includes dissolved gas and all related data for use by the physical modelers

Types of Studies

Transect studies
Drift studies
Spillway performance tests
Surface bypass studies

Figure 6. Dissolved gas abatement study

addressing information on conditions in reaches and pools between the dams. The monitors were usually located on the shore and the one dimension provided no information on lateral and vertical mixing.

The physical data collected for Phase I of the Gas Abatement Study during the transect studies provided 3 dimensional information on conditions between the dams. The data were limited to a few time frames and discharge and spill conditions for those times. The physical/chemical data collected during Phase I of the Gas Abatement Study has been integrated into a multifaceted database. Figure 7 gives a description of the type of data collected for each of these facets of the database. This data can be integrated with the biological data and can be incorporated into the mathematical models. These models can then be calibrated to predict gas saturation under a variety of spill and release conditions throughout the reaches and pools of the river. The last stage of the field studies will be to verify the calibration of the models. The data was made accessible to other agencies and the scientific community through the World Wide Web.

Conclusion

The Total Dissolved Gas Studies on the Lower Columbia have been an exciting environmental challenge and demonstration of the effectiveness of cooperative, interactive, and integrated efforts in achieving economic and environmental goals. Through a cooperative effort between the Walla Walla and Portland districts and the assistance of Waterways Experiment Station, 191 transect locations were identified in the Lower Columbia and Snake Rivers, concentrated at the projects and the fixed monitoring stations and distributed through the pools. In addition to the data collected by the fixed monitoring stations, 70,000 data points were recorded through the efforts of the field teams and incorporated into a database. Related operational and meteorological data will be added for use by the modelers. Four transect study sets were conducted by NPW from May 8 to June 19 and two transect study sets were conducted by NPP between May 18 and August 8. A Total Dissolved Gas study protocol was developed and tested and a performance evaluation of the fixed monitoring station was initiated. Special studies on spillway performance and surface bypass designs added to the knowledge and understanding of the inflow, mixing and gas dynamics of the Columbia and Snake Rivers.

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Fixed Monitoring Station Data

Barometric pressure, water temperature
Total dissolved gas pressure (TDG)
Dissolved oxygen pressure (DO)
Percent Saturation TDG and DO
Q spill, Q release, number of gates open at project being monitored

Physical Field Study Data

Total Dissolved Gas, Dissolved Oxygen, temperature, and other water quality parameters,
Barometric pressure and weather conditions
Historical record, time, date
Geographical, mapping, latitude, longitude, global positioning
Bathymetric, depth of site, travel times
Photographic records

Project Operations data

Q release = Q hydropower generation + Q spill
Detail: Number of bays open and setting for each gate, and number of turbines operating and kilowatts per generator for each time period.

Meteorological data

Weather stations, temperature, precipitation, wind velocity and direction, relative humidity and cloud cover

Figure 7. Database development

Filtering Dissolved Gas Data

by
Steve Wilhelms[1] *and Mike Schneider*[1]

The levels of dissolved gas in the Columbia and Snake Rivers are closely related to project design and operation. This relationship provides a mechanism to influence the location, timing, magnitude, and duration of dissolved gas conditions throughout the Columbia River. The automated dissolved gas monitoring network was expanded to 38 stations in 1995, providing additional insight into the temporal and spacial distribution of dissolved gas throughout the Columbia River systems. This information, when coupled with data regarding project operation, hydraulic structure properties, river morphometry, meteorologic conditions, water quality, and river hydrodynamics, can provide a detailed picture of the dissolved gas characteristics throughout much of this system. The objective of this paper is to review dissolved gas data gathered during 1995 throughout the Columbia River System in light of influential project features. Implications concerning project operation and gas abatement will be discussed.

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Total Dissolved Gas Studies on the Lower Columbia: Transect Studies

by
Faith E. Ruffing,[1] *Nicole A. Flint,*[1] *George A. Kalli*[1]

Total Dissolved Gas (TDG) has been monitored throughout the Columbia and Snake River Basins through a system of fixed monitoring stations located at key sites in the forebay and tailrace of the Corps projects. To enhance the information collected by the Fixed Monitoring System, Transect Studies were conducted in 1995 which consisted of taking TDG measurements along a prescribed series of transect lines located perpendicular to the longitudinal axis of the river. At each location a set of sampling sites were located on a lateral line that transects the river. Transect studies are necessary to define the cross sectional distribution of the TDG downstream from each spilling project for the purpose of assessing fisheries impacts and to assess the representativeness of the fixed monitoring station locations relative to the whole pool. This type of data is required for the numerical modeling efforts and for the gas bubble trauma research efforts. This data collection effort is also required by the Biological Opinion, measure VIIIA.16. The assemblage of the measurements into a data set provides a two-dimensional description of TDG saturation along the longitudinal and lateral axes of the reservoir reaches.

Objectives

The data collection objectives during initial field trips were to determine the location of the transects along the river in relation to the projects and fixed monitoring stations; to measure the width of the river at the transect; to measure bathymetry of the transects; to locate the 20 foot depths on each side of the river thereby locating the ends of the transects; to locate and determine the latitude and longitude of each sampling site along the transect; to photograph left and right bank markers; and to measure the TDG level at 15 foot depths at the sampling sites. The data collection objectives for subsequent field trips were to return to the same transects and sampling sites to measure TDG and dissolved oxygen, to collect vertical profiles at selected sites on each transect, to do repeat sampling in order to determine precision, and collect multiple TDG readings to confirm whether the adopted protocol was appropriate.

Transects

Transects were located at half-mile intervals downstream of the projects for the first two miles, at one-mile intervals for the next four miles, at two-mile intervals for the next eight miles, and at five-mile intervals for the rest of the pool. In addition, sites were established at main tributaries and fixed monitoring stations. This approach concentrated the data points near the dams where the gas gradients are steep and dispersed them further downstream where the TDG levels vary less. Transects are named by pool, location of the transect point as river mile to the tenth of a mile, and transect site location. Numbers following the river mile indicate the sampling site on the transect progressing from left to right looking downstream. For example, BON 185.55 is in the Bonneville Pool at river mile 185.5, site 5. For the transects above McNary on the Columbia and those on the Snake, the sampling sites were located equidistant along the transect line which extended from

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bank to bank. Below McNary the sampling sites were determined as follows.

For each transect the 20-foot depth was located on each bank of the river. Looking downstream, the left bank 20-foot depth was designated the 0 percent point of the transect, the right bank 20-foot depth was designated the 100 percent point of the transect. The 10, 35, 50, 65, and 90 percent distances from the 0 point were determined and named sites 1, 2, 3, 4, and 5, respectively. This was done to concentrate the readings in the main channel. If there was an island in the river at a transect location, data collection sites were located on each side of the island. A combination of maps, electronic distance measuring instruments, compasses, global positioning systems and photographs of distinctive landmarks on the left and right banks were used to establish the river mile of the transect points and the sampling sites along the transect line. This information was translated into latitude and longitude for identification of sites for future studies.

Transect Study Sets

The transect data was collected from 2 May through 20 June below Lower Granite, Little Goose, Lower Monumental, and Ice Harbor. Four sets of data were collected below these projects. The first set was collected during an early spring freshet, the second during high late-spring flows, and the third and fourth during moderate summer flows. Selected data from these studies are displayed in Figures 1-4 for these pools. In the Little Goose, Lower Monumental, Ice Harbor, and McNary pools, transect sets two, three, and four were executed as drift studies where field crews attempted to stay with a parcel of spilled water as it moved downstream. From 19 May through 16 August two sets of data were collected below McNary, John Day, The Dalles, and Bonneville; the first data set was collected during high late-spring flows, and the second set was collected during moderate summer flows. Selected Total Dissolved Gas in percent saturation for these transect study sets are shown in Figure 5-8. Project release ranges and spill settings for each pool varied and these variations are included in these figures.

Protocol

TDG levels were measured using Common Sensing, Inc. instruments which were calibrated weekly with standard barometers and thermometers. Weekly calibrations involved adjusting meters to known standards provided by the USGS. Barometers used for calibration were first calibrated to weather stations located at nearby airports. Daily calibrations with hand-held barometers were performed dockside. Records of the calibration were kept for each instrument to detect continuing problems and drift in the machines. Instruments with continuing problems were replaced. During data collection, total pressure readings that stabilized for two minutes were considered equilibrated. This was usually achieved within thirty minutes after the probe was introduced into the water. Instruments were checked throughout the day in the field to assure that all instruments maintained comparable readings. Data recorded from the instruments included temperature, barometric pressure, total gas pressure in the water, percent saturation of total dissolved gas, and percent saturation of dissolved oxygen. Hydrolab instruments were also used to record time, depth, temperature, and percent saturation of dissolved oxygen. The protocol followed for the Transect Studies was developed by the Corps and USGS.

Total Dissolved Gas Data

The transect studies identified 191 transect locations. Initially, TDG data was collected at 15-foot depths. For transects downstream of McNary where there was a considerable expanse of shallow water beyond the 20 foot depth end points, additional measurements were taken at shallower depths. As study needs developed the "simple" and "full" transect formats were added. The simple format was used at all transects unless otherwise noted. A simple transect consisted of five TDG data sites at 15 foot depth and one vertical profile. The vertical profiles consisted of one TDG reading at the 3-foot depth, one at 15 feet, one at mid-depth, and one at 3 feet from the channel bottom. The mid-depth reading was not taken if the water depth was less than 60 feet. A "full" format was used at transects at the fixed monitoring station, at transects immediately above dams, and at the four half-mile-interval transects below the projects. A full transect consisted of vertical profiles at all five sites on a transect. Multiple and repeat transect format was added in order to answer questions concerning quality assurance and quality control. A "multiple" format was used at every seventh transect from McNary Dam downstream to river mile 41.5. A multiple transect consisted of three additional readings at two minute intervals following the initial stabilized reading for a total of four readings. A "repeat" format was employed for each transect below McNary. This consisted of taking a reading at each of the sites, and then returning to a predetermined site for a repeat reading. Sometimes, especially in high flow areas, readings fluctuated without stabilizing within a half a minute. In such cases a several readings were recorded at one minute intervals and then averaged to give the measurement for that site.

Quality Assurance and Quality Control (QA/QC)

Data from the Transect Studies related to QA/QC include repeat measurements taken of selected sites on the transects, multiple measurements taken of selected transects after instrument equilibration, calibration logs for each instrument, and records of protocol deviations. These data sets were collected to answer questions regarding the repeatability of the measurements, to establish that equilibrium had been reached, and to evaluate the reliability and performance of different instruments. In general, it was possible to collect repeatable data following the protocol set forth for the Transect Studies. The difference between the initial and the repeat samples was less than 1 percent saturation for 97 percent of the comparisons. Similarly, differences of less than 0.5 percent between equilibration and multiple readings were observed for all multiples except for those on transects immediately below Bonneville and The Dalles Projects where extreme turbulence caused fluctuations in the measurements. This confirmed that measurements taken using the prescribed protocol reflected equilibration of the gases across the membrane of the instrument. There is a ± 1 percent variation in the instruments themselves. Review of calibration logs and comparison of dockside calibration showed less variation than $\pm$-1 percent between instruments.

Results

The results of the transect studies are organized according to pool from the mouth of the Columbia to the Snake and up the Snake to Lower Granite Project. The data is displayed in Figures 1-8. Only those measurements taken at 15 foot depths are included in the figures.

Transects done immediately above, at, and just below the permanent monitoring station show a close correlation for most of the stations. TDG readings from those stations in the tailrace of each dam and the forebay of the next dam and locations in the tidal waters are shown on the figures. The average TDG recorded by the fixed monitoring stations for the hours during which the transect sampling was done and the daily average for the day the transect was done are displayed for comparison.

Vertical profiles conducted at select sites and transects did not show significant variation from 15 feet to the deeper depths. Differences documented between 15 feet and 3 feet were reflective of the degassing occurring at the surface of the water.

Several phenomena play a role in the gassing of the water below the dams and degassing as the water flows through the pool. Some of those identified are the discharge, the elevation of the pools, the pattern of the spill, the configuration of the dam, the morphometry of the river below the dam and throughout the pool, the dilution and mixing of the gassed water with the power plant water, and inflow from the larger tributaries. Tributary impacts on total dissolved gas saturation in the river are seen at the confluence of the Snake and the Columbia, and the Willamette and the Columbia. The weather, especially the wind velocity, results in waves and white caps increasing the surface area open to air and agitation of the water which brings the gas to the surface. This appears to be an important part in the degassing of the water in certain stretches of the river.

TDG saturation was highest in the tailrace of the dams and decreased as the water moved through the pools. During the involuntary spill, gas saturation exceeded the standard of 110 percent throughout the pools below The Dalles and Bonneville projects. Below McNary the saturation dropped below the standard about halfway through the pool and did not exceed the standard below the John Day Dam. Below Lower Monumental, Little Goose, and Lower Granite the TDG percent saturation was between 110 and 120 percent for the most part. Those below Ice Harbor were the highest, ranging from about 138 percent below the project to 110 percent at McNary forebay. The number and percentage of sites above 110 percent and above 120 percent for each pool and transect study set are shown in Table 1.

Summary

The Transect studies of 1995 in the Lower Columbia and Snake Rivers established 191 transect line locations throughout the basins with at least 5 sampling sites each. These transect lines were concentrated below Corps Projects and fixed monitoring stations. TDG measurements were taken at these locations through the high flows and spills of spring and the moderate and low flows of summer. These measurements indicate TDG saturation is above the water quality standard of 110 percent for most of the length of these rivers during the spring high flows and spill conditions outlined in the Fish Passage Plan. Decreased flow and spill scenarios of summer results in a decrease in number of miles of standards exceedance but still results in TDG supersaturation in about half of the river miles of these basins. Operational changes in spill pattern and quantity result in longitudinal and lateral changes in the TDG gas saturation distribution in the reaches of these rivers below Corps projects. These changes are specific for each project and are influenced by other considerations such as height of the spillway and the depth of the stilling basin. Studies proposed for future years will provide data that will delineate the characteristics of each project and pool that are important in increasing and distributing the TDG saturation below these projects. From this information the appropriate operational and design changes can be developed to reduce and maintain TDG saturation levels within the established water quality standards.

A better understanding of the physical processes and biological behavior occurring in the river is paramount to resolving the TDG saturation and fish passage issues associated with the Corps' projects. Although data collected from the permanent monitoring stations since 1984 show a correlation between spill and TDG levels at some projects, at others it is not closely correlated. Transect studies conducted in 1995 indicate that factors other than spillway discharge impact the gassing and degassing of the water.

Spills of set time, quantity and duration which reflect different design and operation alternatives to improve fish passage efficiency will be analyzed in Phase II of the GAS Abatement Study. The gassed water will be measured as it moves through the pools below the dams using a combination of field study biologists in boats, the permanent monitoring system, and selectively placed remote monitoring systems. These studies will be scheduled to take advantage of the naturally occurring flows of spring and summer and will be coordinated with other efforts involving spills. Related water quality

Table 1
Water Quality Standards Compliance

Transect	Total Readings	Total >110	Percent >100	Total >120	Percent >120
TID1	176	176	100.0	5	2.8
TID2	169	100	59.2	6	3.6
BON1	95	95	100.0	0	0.0
BON2	84	81	96.4	0	0.0
TDA1	67	5	7.5	0	0.0
TDA2	55	0	0.0	0	0.0
JDA1	77	36	46.8	4	5.2
JDA2	115	50	43.5	0	0.0
MCN1	63	60	95.2	19	30.2
MCN2	49	49	100.0	16	32.7
MCN3	159	131	82.4	0	0.0
MCN4	109	61	56.0	33	30.3
IHR2	35	33	94.3	0	0.0
IHR3	72	71	98.6	3	4.2
IHR4	46	25	54.3	0	0.0
LMA1	48	43	89.6	0	0.0
LMA3	113	61	54.0	0	0.0
LMA4	55	13	23.6	0	0.0
LGS2	65	61	93.8	0	0.0
LGS3	204	1	0.5	0	0.0
LGS4	35	0	0.0	0	0.0

parameters will be measured simultaneously to enrich the database information and enhance the knowledge and understanding of the physical and biological processes occurring in the Lower Columbia and Snake River Basins. Details of these studies can be found in the Phase II Gas Abatement Program Field Study Plan for FY 1996.

Figure Descriptions

These Figures display the Total Dissolved Gas measurements taken during the Transect Studies. The graphs includes the River Mile (RM) location of the projects, the transect sites, the fixed monitoring stations and the mouth of the tributaries for each pool. The data points are shaded from dark to light as the site number increases. The hourly fixed monitor readings taken during the time the transects closest to the monitors were done and the hourly readings for the day these transects were done were averaged and included in the figures.

Little Goose Pool (Little Goose Forebay to Lower Granite Tailrace). Figure 1 display the study data for this pool from RM 71 to RM 110 on the Snake River.

Lower Monumental Pool (Lower Monumental Forebay to Little Goose Tailrace). Figure 2 displays the study data for this pool from RM 41 to RM 71 on the Snake River.

Ice Harbor Pool (Ice Harbor Forebay to Lower Monumental Tailrace). This pool extends from RM 10 to RM 41 on the Snake River (Figure 3).

McNary Pool (McNary Forebay to Ice Harbor Tailrace). This pool starts at RM 292 on the Columbia and continues to RM 325, the confluence of the Snake. The pool is influenced by operations at Priest Rapids on the Columbia and at Ice Harbor at RM 10 on the Snake River (Figure 4).

John Day Pool (McNary Tailrace to John Day Forebay). This stretch of the river from RM 292.0 to 217.2 is wide and shallow immediately below the dam. Blalock Island (which is actually submerged under 5-10 feet of water is located between RM 278 and 265. Selected data are displayed in Figure 5.

The Dalles Pool (John Day Tailrace to The Dalles Forebay). This pool, RM 215.4 to 192.9, is the shortest reach at 22.5 miles. Islands immediately below the Dam separate the spill and the generation water and is reflected in a gradient in the data from the left to the right bank above the transect at RM 204 (Figure 6). This transect is immediately below Miller Island. The high gas water on the right bank goes to the right of the Island (Transect 205) and is mixed as it joins the water from the left channel. The spill at this project is maintained at a low rate because it produces high TDG levels.

Bonneville Pool (The Dalles Tailrace to Bonneville Forebay). This 54 mile reach of river from RM 190.6 to 146.6 starts with a 100-150 foot deep channel immediately below the dam at RM 191.0. The depth decreases to about 50 feet at the first transect below the project at RM 190.6. The river becomes wider as it moves downstream and travels through a gradually steeper canyon as it cuts through the east side of the Cascades to the Bonneville forebay. High winds are encountered in this stretch to the forebay (Figure 7).

Tidal Pool (Bonneville Tailrace to Wauna). This 100-mile stretch starts at the Columbia River Gorge at RM 145.5 and extends downstream past the Portland Metropolitan area to Puget Island on the edge of the Columbia River Estuary at RM 41.5. Major tributaries flowing into the Tidal Pool are the Sandy at RM 122 and the Willamette at RM 101 on the Oregon side; and on the Washington side, the Cowlitz at RM 87, and the Lewis at RM 68. Sauvie Island at the confluence of the Willamette is separated from the mainland by Multnomah Channel which enters the Columbia at RM 87. There are fixed monitoring stations at Warrendale, Oregon and across the river in Skamania, Washington at RM 140.1 downstream from Bonneville Dam. Monitors are also located at Camas Washougal, RM 122, and Kalama, RM 77, on the Washington side and at Wauna, RM 42, on the Oregon side (Figure 8).

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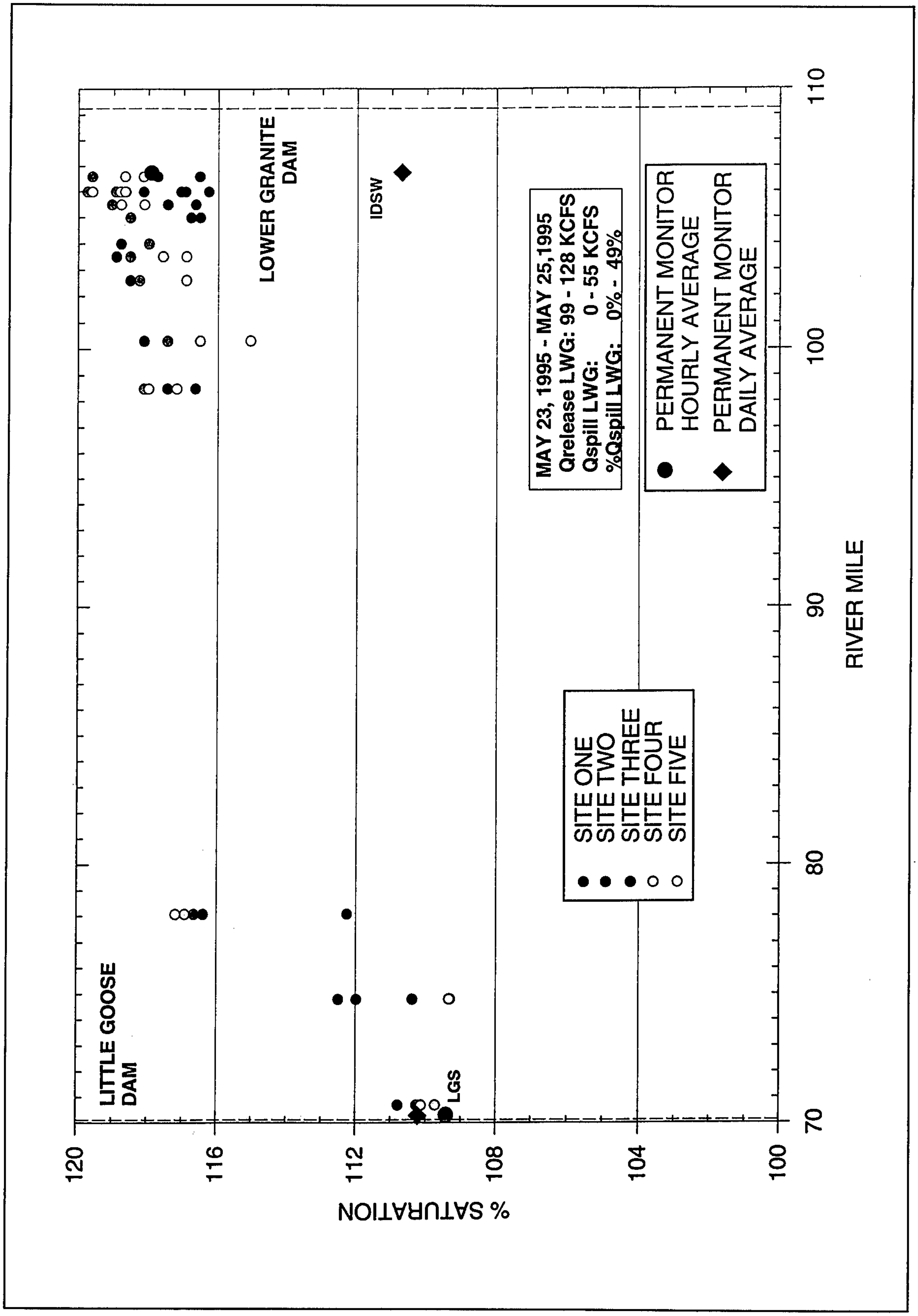

Figure 1. Little Goose pool transect study set two

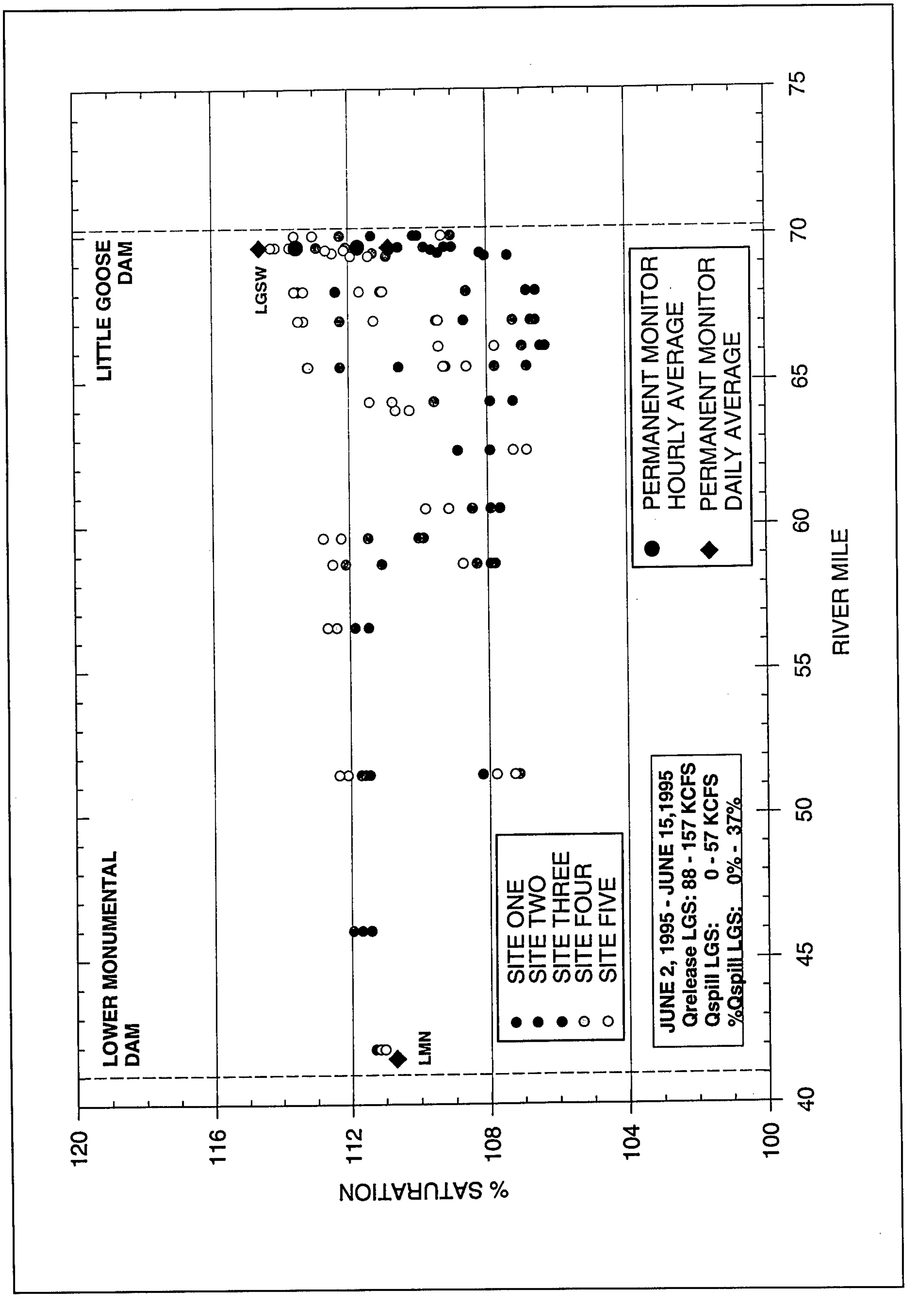

Figure 2. Lower Monumental pool transect study set three

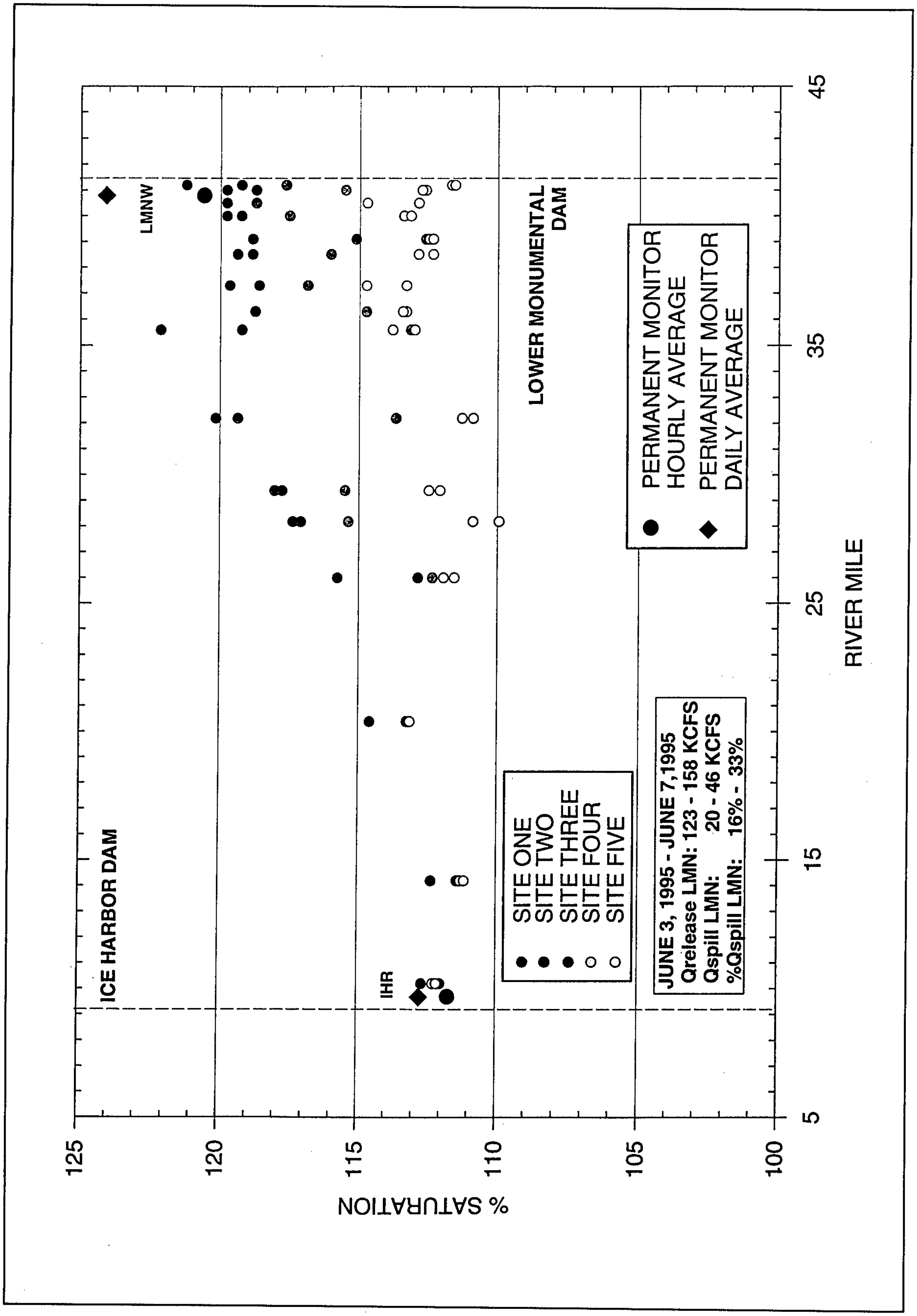

Figure 3. Ice Harbor pool transect study set three

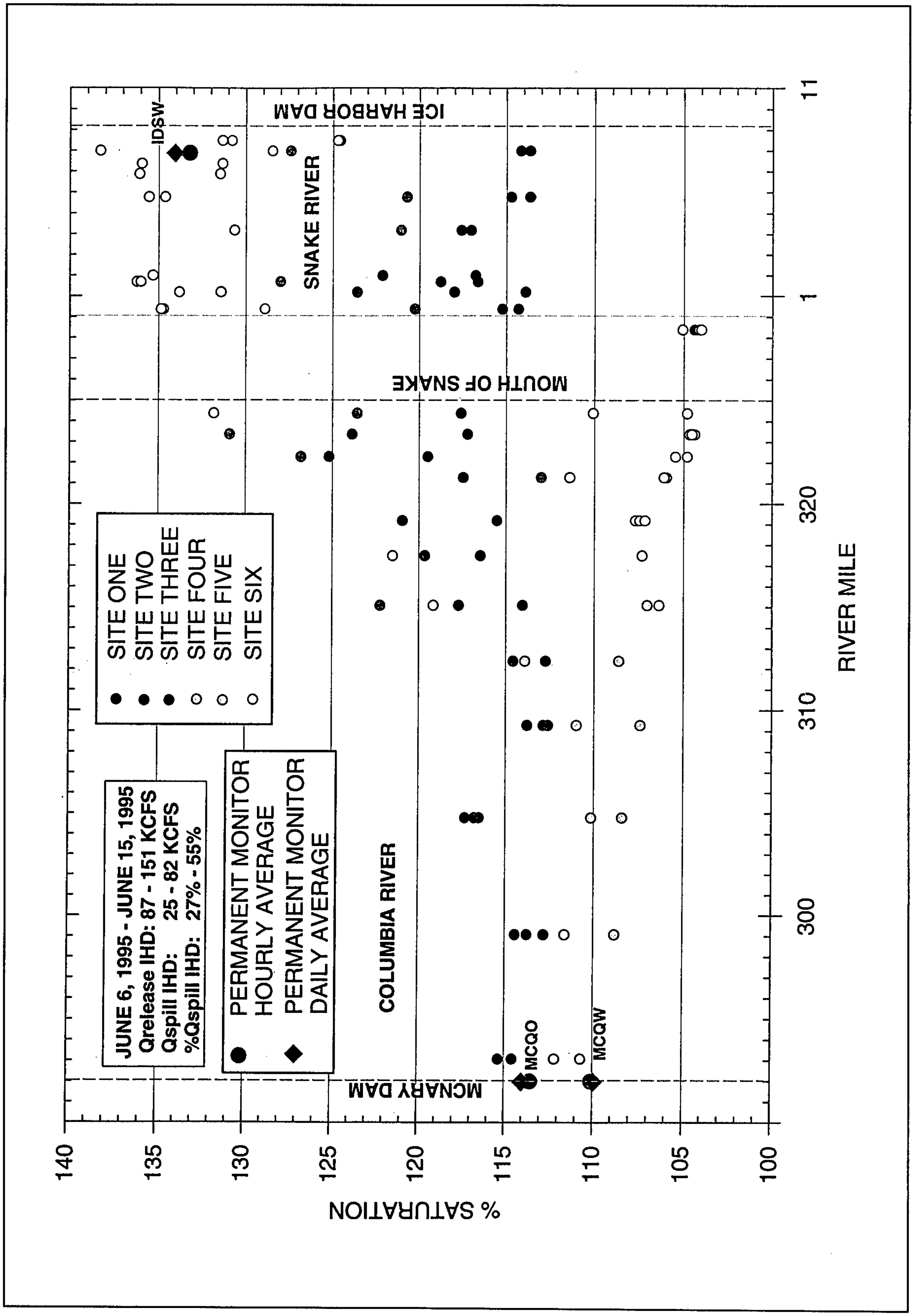

Figure 4. McNary pool transect study set three

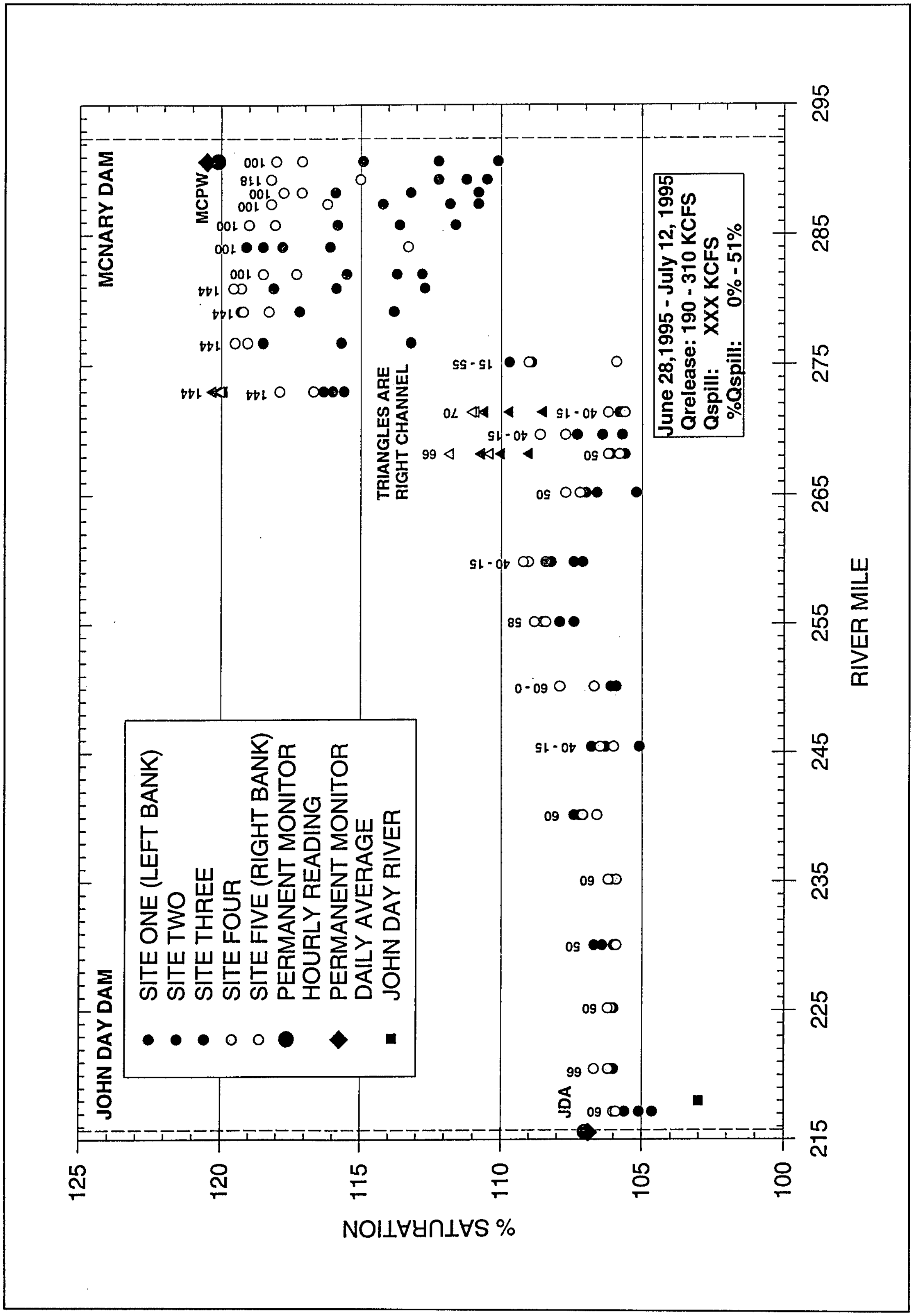

Figure 5. John Day pool transect study set two

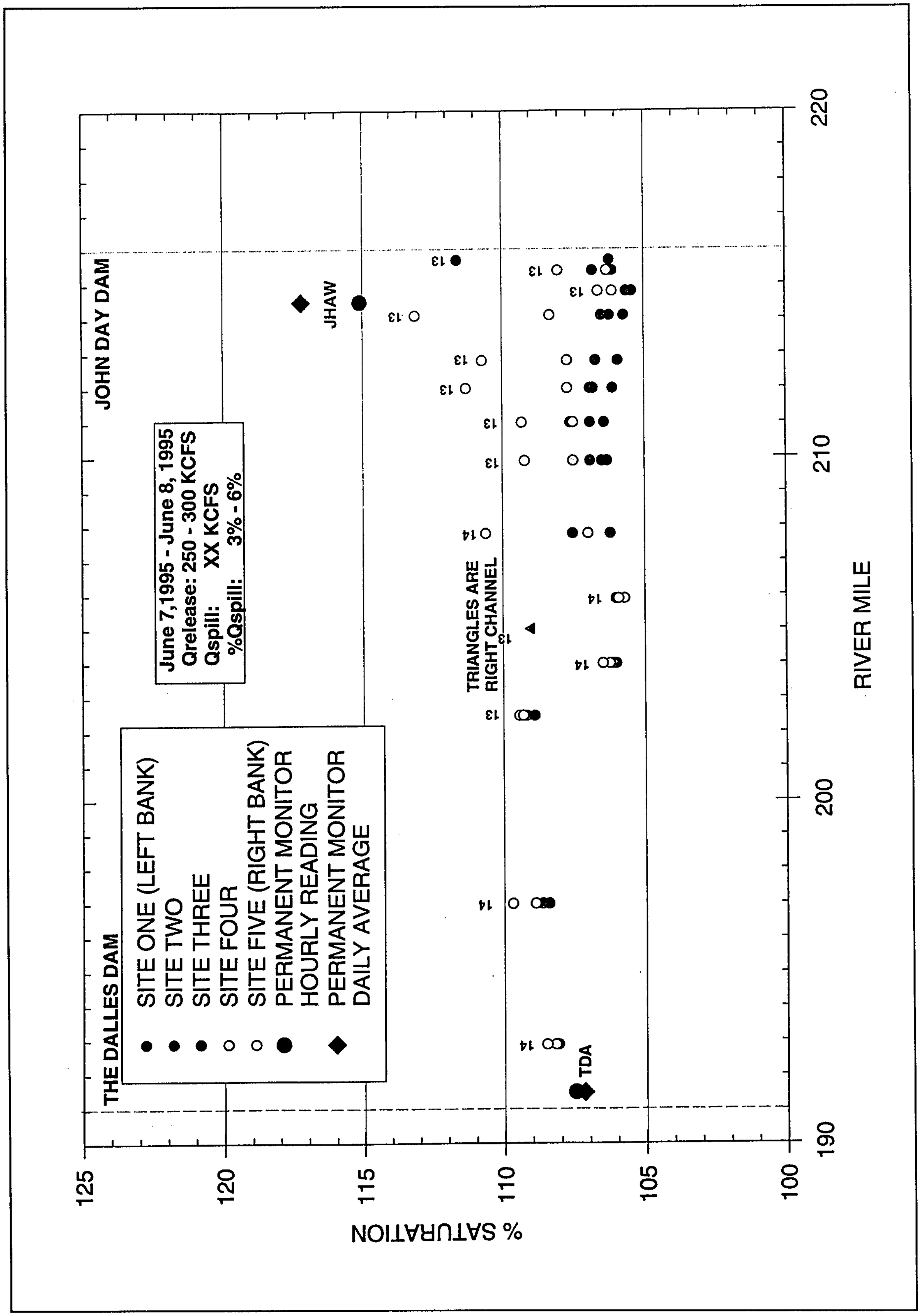

Figure 6. The Dalles pool transect study set one

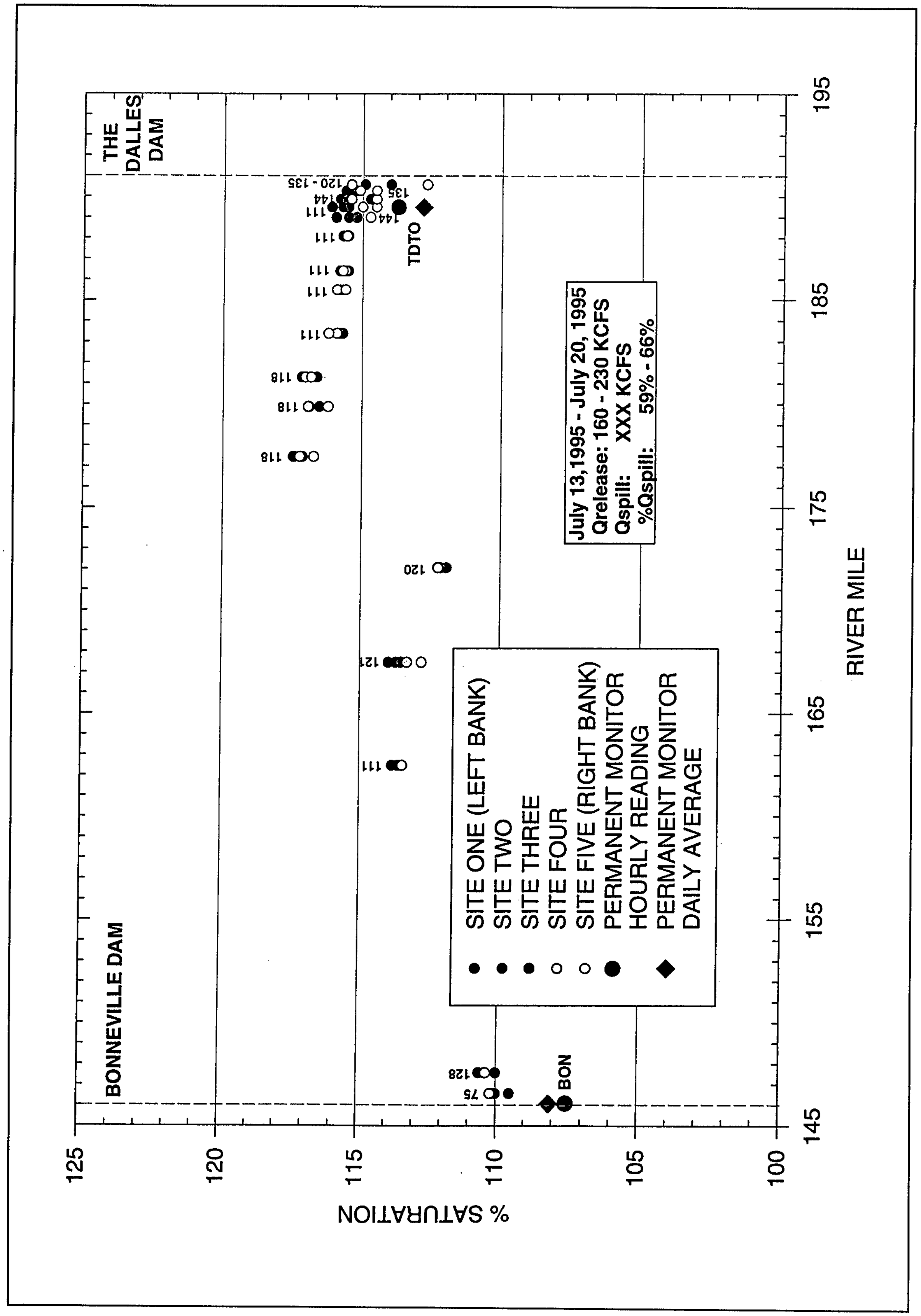

Figure 7. Bonneville pool transect study set two

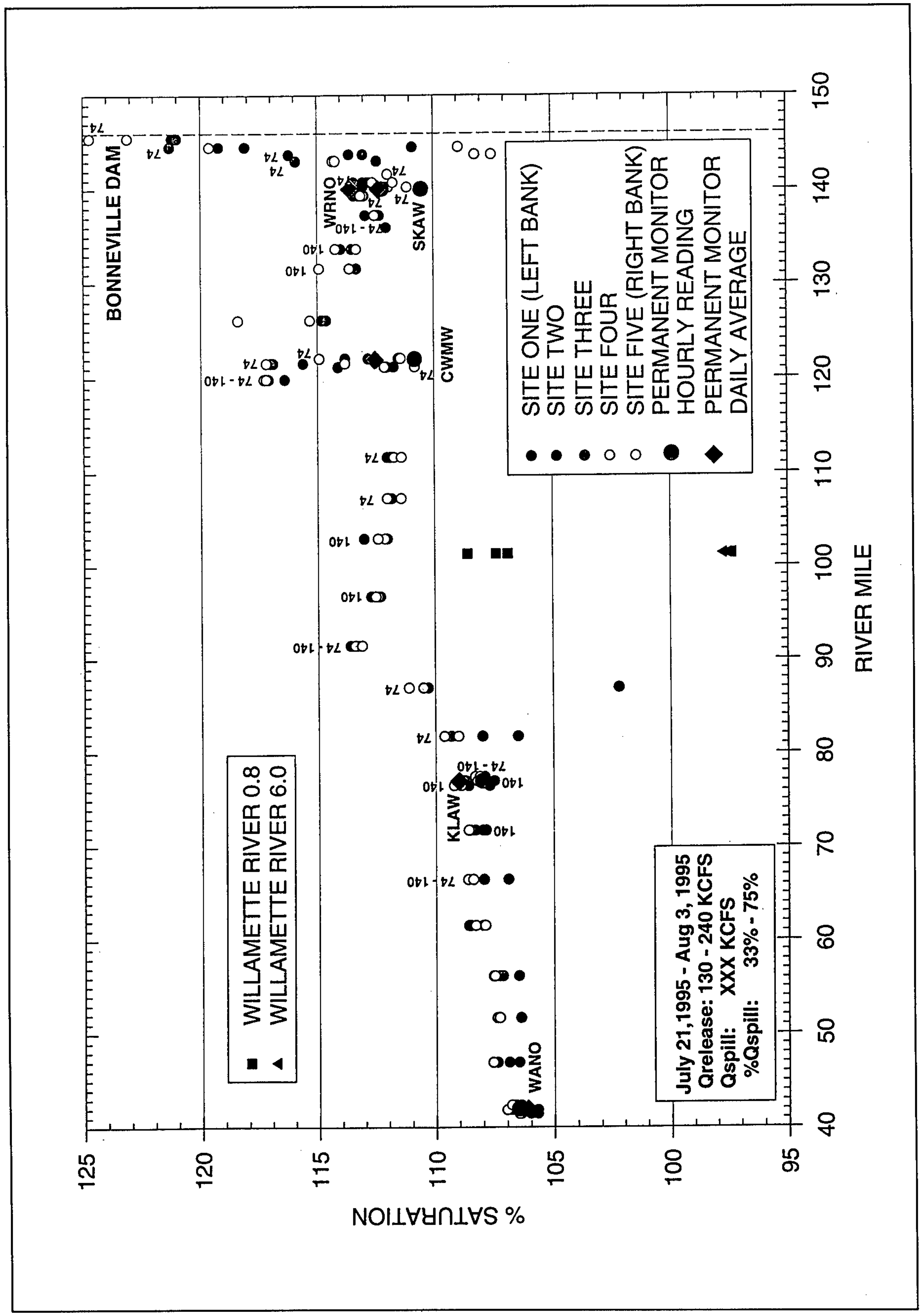

Figure 8. Tidal pool transect study set two

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Use of Methane Gas as a Surrogate Tracer for Reaeration-Rate Measurements at Navigation Structures in the Ohio River Basin

by
George Kincaid,[1] Kimberly Miller,[2] William Cremeans,[1] Vincent Marchese,[1] and Richard Meyer[1]

The reaeration potential of lock-and-dam and hydropower structures located in the middle Ohio River Basin was evaluated by using in situ methane gas as a surrogate tracer for dissolved oxygen. This investigation was the first production-run application of the methane surrogate-tracer technique. Previous investigations involved research or research and development applications.

The work was conducted using a team formed by the U.S. Army Corps of Engineers, the U.S. Geological Survey, a commercial laboratory, and the University of Minnesota. Prior to these efforts, little substantive data existed with regard to the potential of lock-and-dam and hydropower structures to reaerate waters of the Ohio River.

Several major benefits have been derived, including:

- The ability (for the first time) to quantify the reaeration effectiveness of, or amounts of oxygen that can be added by, Ohio River lock-and-dam and hydropower structures.
- Insights for operating lock-and-dam and hydropower structures during low-flow periods and droughts.
- The opportunity to evaluate the correctness of existing standards, regulations, and policies with regard to the ability of river structures to provide reaeration.
- Better definition of an important mechanism that contributes to the dissolved oxygen budget of the Ohio River system.
- Information relating to the abilities of lock-and-dam structures to force the outgassing of volatile spill materials.

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Prediction of Dissolved Gas Supersaturation Downstream of Hydraulic Structures

by
Darrin A. Geldert[1] and John S. Gulliver[1]

Introduction

In a recent effort to aid anadromous fish survival, such as the salmon, in the Pacific Northwest, increased spills have been required during the downstream passage season. It has been hypothesized that the young salmon will move downstream faster with these, "fish spills" making them less susceptible to predation and disease. In addition, the spillways are believed to be a generally safer downstream passage route than through the powerhouse turbines. However, as the water moves down the spillway, it entrains air and the air-water mixture plunges into the plunge pool carrying bubbles deep into the pool. The increase in hydrostatic and dynamic pressures on the bubbles traveling deep into the plunge pool will allow more gas to be transferred from the air to the water than is possible at atmospheric pressure. The maximum amount of gas that can be dissolved into water, the saturation concentration, is linearly proportional to the partial pressure of the gas in the air, as described by Henry's law. Thus the water downstream of the structure is possibly supersaturated with respect to atmospheric pressure (Gulliver 1991).

To better understand and model the dissolved gas levels throughout the Columbia and Snake river systems, improved field data and current gas transfer research have been used as the basis for developing new physically based relationships to model dissolved gas levels downstream of spillways. Independent consideration of air entrainment, a detailed consideration of transfer across the water surface and across the bubble-water interface, and the effect of both the tailwater depth and downstream river depth on saturation will be considered herein. In addition, field data were specifically collected for use in deriving new predictive relationships.

Background

The area over which transfer is occurring will be subdivided into two regions; on the spillway face before the water enters the plunge pool, and in the plunge pool. This allows for the "degassing" effect seen at some structures. In addition, transfer across the water surface and transfer across the bubble surface interface will be considered in each region. At any surface a basic gas transfer equation can be applied

$$C_d = C_{se} - (C_{se} - C_u)\ \exp(K_L at) \tag{1}$$

where

C_d = downstream dissolved gas concentration

C_u = upstream dissolved gas concentration

C_{se} = effective saturation concentration

K_L = transfer rate coefficient

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a = specific surface area

t = residence time

In applying Equation 1 to a region, it must be expanded to allow for different values of K_L, a, t, and C_{se} at each surface. The variables included in the mass transfer equation, K_L, a, t, and C_{se}, allow a complete description of mass transfer under various flow conditions and physical parameters, if the proper physical relationships are determined. While each variable will be affected by temperature and the physical properties of the gas and liquid, changes in the rate of mass transfer are predominantly due to four factors:

a. The level of turbulent mixing. The amount of turbulent mixing is directly related to the rate of surface renewal as described by Danckwerts (1951) description of gas transfer. As the level of turbulent mixing increases, the renewal rate will increase and this will be characterized in the mass transfer equation by K_L.

b. The amount of surface area available for gas transfer. As air is entrained, bubbles of air will greatly increase the surface area per unit volume. The bubble size directly relates to the area and is dependent on the velocity of the plunging jet, the amount of air entrainment, and the intensity of turbulence in the flow. These are all factors that are important to the specific surface area characterized by "a".

c. The residence time of the bubbles in the flow. The time available for mass transfer is represented as "t". The depth and velocity of the flow, as well as the rise velocity of the bubbles and the distribution of the bubbles over the depth are all important to the residence time.

d. The increased hydrostatic pressure experienced by the bubbles carried deep into the plunge pools. The increase in pressure causes an increase in the equilibrium concentration, which accordingly increases the driving force for mass transfer. The depth the bubbles experience and the increased pressure associated, may greatly enhance mass transfer. This is characterized by C_{se}.

The first predictive model for dissolved gas levels downstream of a spillway was developed by Roesner and Norton (1971) for low head, run-of-the-river U.S. Army Corps of Engineers Dams on the Columbia and Snake rivers and is the basis of GASPIL. Their model was later modified by Johnson and King (1975), and applied to a number of Bureau of Reclamation Dams and other sites. In each model the downstream concentration is given by an equation in the general form of Equation 1, however, only transfer across the bubble surface in the stilling basin is considered. They used a saturation concentration at a given depth and K_L a was combined as K, an empirical entrainment/mass transfer coefficient. By calculating the saturation concentration at some depth, the result is essentially the same as using the effective saturation concentration as defined by Hibbs and Gulliver (1995), where

$$C_{se}=C_s\left[1+\frac{d_{eff}\gamma}{P}\right] \tag{2}$$

where

P = atmospheric pressure

γ = the specific weight of water

C_s = 100 percent

d_{eff} = an effective depth that the bubbles experience

Roesner and Norton (1971) and subsequent models used a d_{eff} of two-thirds the tailwater depth. This value was based on the triangle formed by the linear diffusion of a jet which was assumed to have penetrated to the basin floor, the centroid of such a triangle was considered to be the effective depth. For Roesner and Norton, the residence time, t, is the stilling basin length divided by the average flow velocity. They proposed that each dam would have an individual entrainment coefficient (K) would be directly related to the turbulence level in the stilling basin and therefore, the measurable rate of energy loss.

Johnson and King (1975) started with the work of Roesner and Norton and made a few modifications in an attempt to generalize the predictive analysis, so that it could be applied to a number of Bureau of Reclamation Dams. The time used was equal to the shorter of: (1) the travel time to the end of the basin or, (2) the bubble rise time. Johnson and King proposed that the entrainment coefficient, K, was dependent only on the hydraulic performance of the basin. They found by plotting the energy gradient and the ratio of the jet's cross sectional area to its shear perimeter, curves could be determined to predict K that would in turn predict a downstream concentration that agreed with field data.

The large number of variables needed to define the mass transfer occurring at one dam required the gathering detailed field measurements to aid in the analysis. Correspondingly, the model developed by Geldert (1996), incorporates a number a of new ideas that resulted from intensive field measurements of total dissolved gas levels. First, transfer is considered across bubbles and the water surface, both on the spillway face and in the plunge pool. Rindels and Gulliver (1991), have shown that transfer on the spillway face may be significant. In addition, intense surface turbulence indicates that transfer across the surface need to be considered. When considering transfer across the bubble surfaces in the pool, field observations of bubble plumes downstream of spillways show that although the stilling basin may be 60 meters long, the bubbles entrained at the plunge point may not leave the flow for more than 300 meters. Therefore, the river depth, in addition to the stilling basin depth, will be included in a determination of an effective saturation concentration. In addition, estimations of the numbers of bubbles, or the air void ratio, on the spillway and in the pool are now possible due to recent research on air entrainment (Wilhelms and Gulliver 1994), which will allow for independent consideration of "a". Together with improved field data and new methods for estimating K_L, a new physically based predictive relation has been proposed.

Results

The dams for which dissolved gas data was taken, Lower Granite, Little Goose, Ice Harbor and The Dalles, were all used in the model development. Starting with the data from Ice Harbor, the contribution of transfer across the water surface is assumed to be zero, as the deep pool limits surface turbulence. The data can therefore be used to determine the coefficients relating to transfer across the bubble surface only. The predicted and actual measured downstream total dissolved gas concentrations are shown in Figure 1a. The standard error of the predicted to actual data is 1.87 percent of saturation.

The fit coefficients, as determined from the Ice Harbor data relate to transfer across the bubble-surface interface only and will therefore be applicable to all of the test structures. The basic relationships regarding bubble transfer relate the transfer efficiency to the turbulence and physical water properties which will not change with structure or flow pattern.

The Dalles requires a surface transfer coefficient, due to the high flow energy in the 6 m deep stilling basin and the 3 m deep reach downstream. Except for the lowest gate opening, where the actual the surface transfer may approach zero due to reduced surface turbulence, the constant surface transfer provides an adequate fit of the field measurements. The

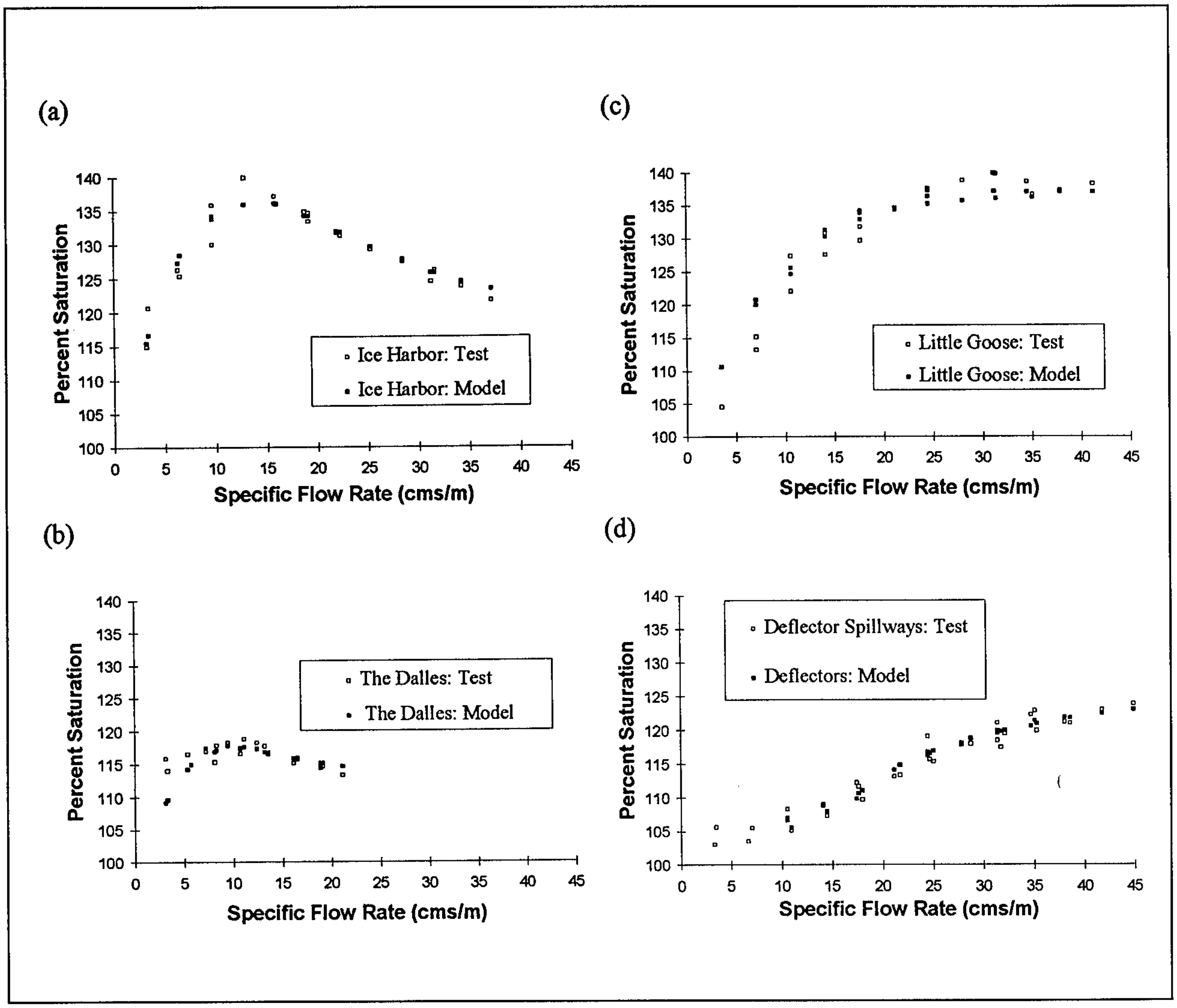

Figure 1. Predicted and test measurements of dissolved gas downstream of Ice Harbor, The Dalles, Little Goose and spillways with deflectors at Little Goose and Lower Granite

standard error of the predicted and measured data in Figure 1b is 2.13 in percent saturation.

In order to fit the field test data from the outside bays of Little Goose, a constant surface transfer is also introduced. The predicted and actual values downstream of the Little Goose outside bays are shown in Figure 1c. The standard error is 1.32 percent of saturation.

Although the flow pattern in stilling basins with deflectors is very different from that without deflectors, the relationships for transfer across the bubble interface will remain the same. The fit of field test data from the stilling basins with deflectors is possible with the inclusion of a surface transfer coefficient, and a fit coefficient for the expansion of the surface jet, used in the determination of an effective saturation concentration. The standard error for both sets of data shown together in Figure 1d, is 1.31 percent of saturation.

The relations outlined here and detailed by Geldert (1996), are able to model the measured downstream dissolved gas concentrations well at all sites. One or more coefficients were fit, however, to the measurements at each structure.

Thus, the only verification possible is to consider the reasonableness of each coefficient. The fitted bubble coefficients are indirectly supported when used in conjunction with a single additional coefficient, for surface transfer, to provide good fits of the data at The Dalles, and Little Goose. While the predictions do not directly confirm the validity of the coefficients, the low standard error of the predicted downstream concentrations over a wide range of flows indicate that the coefficients are reasonable.

The values of the effective saturation concentration and the transfer coefficient for bubble transfer, determined using the fit coefficients, are shown in Figures 2a and 2b. The effective saturation concentration is plotted versus flow rate in Figure 2a. The stilling basins without deflectors have an initial value of C_{se} equal to two-thirds the basin depth and then decrease as the more shallow river depth becomes more important. The effective saturation concentration at the dams with flow deflectors slowly increases as the flow increases and the jet expands.

The values of the bubble transfer coefficient, $K_L a t_B$, are shown in Figure 2b. For the dams without flow deflectors, the value grows with flow rate. Although there is less air by volume at the high discharges, the increased turbulent energy will both increase the renewal rate, increasing K_L, and will shear the bubbles down to smaller diameters causing the specific area of the bubbles to increase. For similar reasons the shallower stilling basin at The Dalles will be more turbulent than the deep Little Goose stilling basin. There is more volume with which to dissipate the energy, and the values of K_L and "a" per bubble will be less at Little Goose, off setting any increase in time due to the greater rise distance the bubbles must travel at Little Goose to leave the flow. For spillway bays with flow deflectors, the depth of the jet is small especially at low specific discharges, creating a high velocity in the jet region. Thus at low specific discharges, K_L and a would be the largest, raising $K_L a t_B$. Then, the competing effects of "a", "t", and K_L balance and $K_L a t_B$ remains relatively constant.

Conclusions

A comprehensive physically based model has been developed which includes of a number of new ideas; (1) Transfer on the spillway face, which can have a significant degassing effect prior to plunging into the stilling basin, (2) An effective saturation concentration based upon both the stilling basin depth and the downstream river depth, (3) Transfer across the air-water interface at the water surface in the stilling basin.

The changes in dissolved gas levels start on the spillway face, especially when the flow becomes self-aerated. However, the majority of gas transfer will occur in the tailwater region immediately downstream of the spillway. The predictive relationship developed herein for this region relies on physical flow and design parameters and well-established theories regarding bubble characteristics and mass transfer. The inclusion of the river depth in the effective depth relation and the inclusion of surface transfer allow for predicted dissolved gas levels that correlate well with the actual measured data at the structures tested. However, without another set of data for dams of this type, verification is not fully possible. In order to provide verification and to add further insight into the problem; (1) Dissolved gas data should be collected at more sites, using the same methods as in the March 1995 Spillway Tests, (2) Dissolved gas data should be collected at sites with high forebay dissolved gas levels.

The dissolved gas levels in a river are largely determined by what happens at the spillway, stilling basin, and bubbly flow immediately downstream. It takes many river miles for the dissolved gas levels to change significantly without a structure because of the lower velocities, reduced surface area and large depth of the

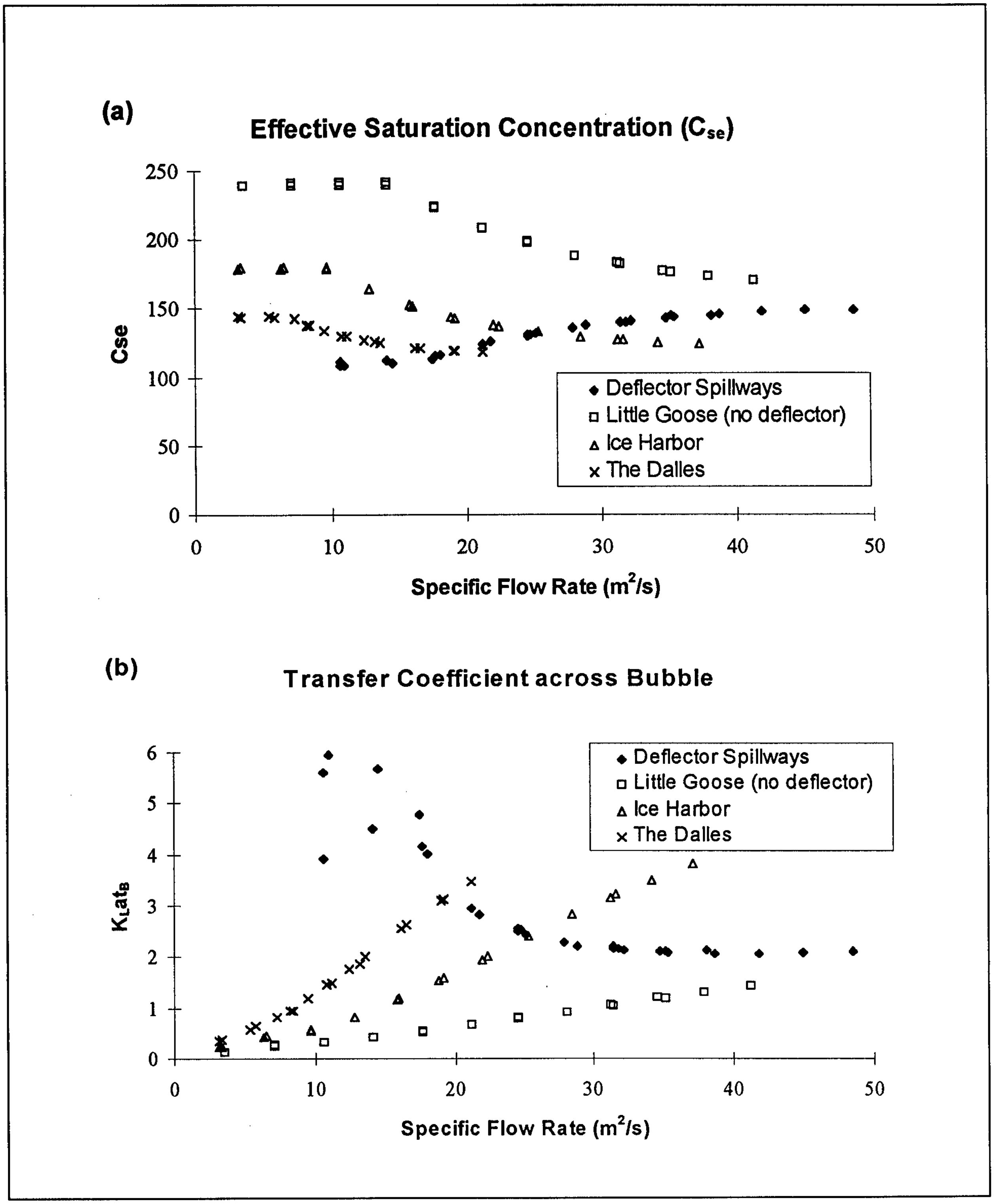

Figure 2. The values of the effective saturation concentration and the bubble transfer coefficient plotted versus the specific flow rate using the fitted coefficients

river-reservoir flows. The accurate prediction of the dissolved gas concentration below spillways is therefore vital to the development of a complete river dissolved gas model. The predictive relations developed herein should substantially improve the accuracy of such work.

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Dissolved Oxygen and Nitrogen Degassing Dynamics In the Lower Columbia and Lower Snake Rivers

by
Joe H. Carroll[1] *and Nicole Flint*[2]

Dissolved gas supersaturation is universal downstream of all COE dams on the lower Columbia and Snake Rivers, especially during the high spill season which coincides with the fish out migration period. High total dissolved gas (TDG) levels typically result from releasing water through the spillways at all the projects. The general purpose of the Dissolved Gas Abatement Study (D-GAS) conducted by CENPP and CENPW is to make recommendations to minimize TDG supersaturation below COE projects on the lower Columbia and Snake rivers and minimize resulting impacts on the fishes. With the increased awareness and attention to the cause and effect relationship of gas supersaturation and gas bubble disease in fishes, a better understanding of the gassing and degassing processes is needed in managing for this critical water quality parameter throughout the Columbia River system.

The physical process of gassing or supersaturation results from passing near surface waters with entrained air over spillways which then may plunge to relatively deep depths in the spilling basin. Hydrostatic pressure forces the air bubbles into solution raising the TDG concentration in the release water. If the water is supersaturated with TDG then degassing results as the releases mix and flow through the receiving pool. Degassing is a combination of physical, biological, and chemical processes. The physical component is driven by exchange at the air/water interface and is highly dependent on factors such as wave action and turbulence. Biological processes, community metabolism, respiration, and photosyntheses, would contribute to the dissolved oxygen dynamics as the releases move through the receiving pools. Chemical oxidation is the third mechanism which may be impacting dissolved gas levels in the river system.

Due to the highly variable operations of COE projects on the Columbia and Snake Rivers, TDG levels vary significantly in time and space throughout the basin. Preliminary data collected for the D-GAS program during 1995 are presented here to describe the degassing dynamics below selected projects. TDG, dissolved oxygen, dissolved nitrogen and their relationships are considered.

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Predicting Dissolved Gas Levels With Neural Network Models

by
Bolyvong Tanovan[1]

Introduction

The Pacific Northwest is actively engaged in the salmon recovery program since the passage of the Electric and Power Conservation Act of 1980. The objective is to provide at least a fish passage efficiency of 80 percent at each main-stem Columbia and Snake Rivers dam. Achieving that objective with the current project configuration and fish bypass facilities requires a significant amount of spill. Not only does spill take water away from power generation, it also creates high total dissolved gas (TDG) saturation that often exceeds state standards. Some believe high TDG levels are also harmful to the migrating fish moving up and down the reservoir system. A delicate balancing act is needed to control spill so that dam passage survival does not outweigh system survival. Selection of the best spill alternatives is based on large part on the results of predictive models. Two of these tools are in use by the Corps North Pacific Division. The first tool is GASSPILL, a one-dimension numerical model of gas entrainment and transport originally developed in the early 1970s by Water Resources Engineers, Inc. The Waterways Experiment Station is now modifying GASSPILL to incorporate the latest advances in TDG simulation techniques and allow shorter time step calculation. While this work is going on, another tool based on the Neural Network. Technique was used with encouraging results. This paper describes this exploratory application in some detail, based on actual work performed on the Columbia and Snake Rivers system.

Spill For-Fish-Passage

The Corps has been conducting experimental spill operation to improve fish passage conditions since the late 1970s. In 1982, spill forfish-passage was included in the Northwest Power Planning Council's Fish and Wildlife Program (Program) to "achieve a level of smolt survival at least comparable to that achievable by the best available collection and bypass systems." This interim measure was revised in 1984 to "guarantee at least 90 percent fish survival for the middle 80 percent of the spring and summer migrations." An index, fish passage efficiency (FPE), was introduced to relate fish survival to spill at each dam. FPE represents the percentage of the juvenile migrants passing a dam via a route other than hydroelectric turbines (see Figure 1). FPE depends on the amount of spill and its efficiency to attract fish to the spillways, the efficiency of the fish screens in guiding fish away from the turbines, the survival rates for each of the fish passage routes, and the diel movement of the fish migrants. The same level of fish survival may require different spill levels depending on the projects, season, targeted fish species, etc.

The spill criteria kept changing over the years in order to provide "the protection levels necessary to prevent further decline for the

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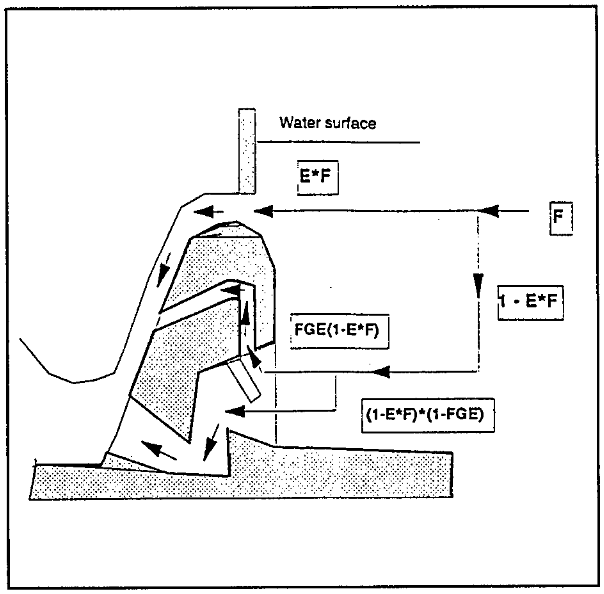

Figure 1. Calculation of FPE

depressed Columbia Basin salmonid stocks." In 1989, the objective was to achieve at least a bypass standard of 70 percent FPE for spring migrants and 50 percent FPE for summer migrants. Under the "share the wealth" concept, higher FPE levels were to be provided in good water years. In 1994, when the National Marine Fisheries Service (NMFS) became the overseer of salmon recovery measures under the Endangered Species Act (ESA), "bypass performance standards equal to an 80 percent FPE at all federal projects in the mainstem Snake and Columbia rivers" became the norm. NMFS conceded, however, that the level of spill needed would have created TDG levels in excess of the state standards of 110 percent saturation. New temporary limits had to be obtained. In 1994 and 1995, these consisted of a TDG cap of 120 percent in the tailrace area and 115 percent in the forebay of the next downstream dam.

Neural Network Simulation

The neural network technique is a relatively new computing paradigm that mimics the human brain's ability to classify patterns or to make predictions based upon past experience. It "learns" patterns from training data and makes its own predictions when presented with new data. This is in many ways similar to the classical multiple regression analysis. The neural network model, however, built itself up automatically, using every interaction between variables; without the no need to specify the type of regression equation or the degree of that equation or polynomial. It can accommodate a huge number of training patterns, even those with "noisy" or slightly incorrect data in them, and generally provides tighter data fits than with regression.

Neural network simulation is performed through several layers of processing elements caused neurons (see Figure 2). The value

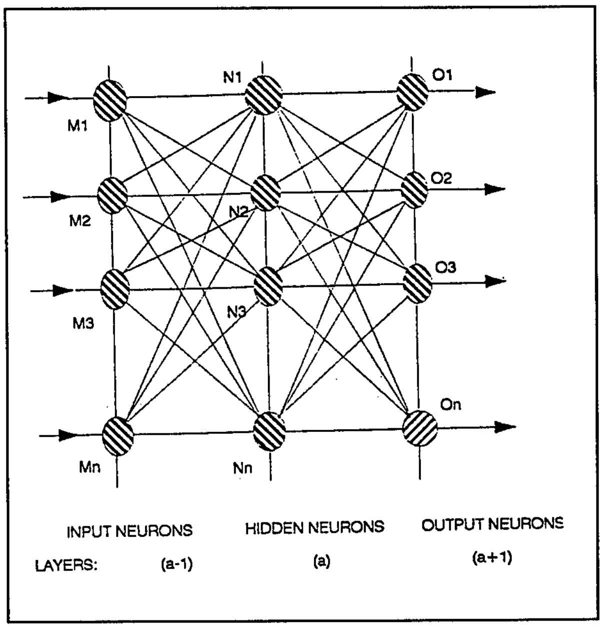

Figure 2. Neural network structure

assigned to a neuron Ni on a layer (a) is the weighted sum of the values assigned to all other neurons M_1, M_2,.... M_n on the previous layer (a-1). The weight reflects the strength of the connection between each connection N_1 to M_1, N_1 to M_2, etc. Initial weights may be assigned random values between -0.5 and 0.5 that are uniformly distributed. This values transformation continues from layer to layer until the layer representing the output is reached. The difference between the observed value of the output and its calculated value is backpropagated from the last layer (a+n) to each of the preceding layers to minimize the errors. This feed-forward back-propagation type network is not the only artificial network architectures available. It is, however, the most widely used because of its ability to generalize.

Connection weights are refined repetitively based on the series of input and output used to train the model. An internal algorithm based on the "gradient descent" optimization procedure guides the calculation of the interconnection weights that minimize the sum of errors calculated for all the training series. A built-in control regulates connection weights adjustment according to a pre-specified learning rate. Too large a learning rate will cause oscillations in the results, while too small a learning rate will slow their convergence. The number of hidden layer neurons influences the optimization process. Too few of them may not provide enough degrees of freedom in reaching the optimum values; and too many of them may prolong the search indefinitely. Normally, a default value is automatically assigned by the model in case it is not provided by the user. The analyst will terminate the training when the global or individual errors between observed and computed values reaches a set target, or after a pre-set number of iterations has been completed. The robustness of the model is determined by running the trained model using inputs data that were not part of the original training data. Overtraining, with very small residual errors, does not necessarily the model's ability to generalize and, therefore, is not a reflect of the model performance with new data. In general, it also takes more computing time to produce small error tolerances.

A commercial software package, NeuroShell2, developed by Ward Systems Group was used in setting up neural network models for the Columbia River mainstem dams. The same inputs as those required by the GASSPILL model (e.g., forebay TDG, water temperature, forebay and tailwater pool elevations, total release and spill discharge) ar used to predict the TDG level at the tailrace of the spilling dam or in the forebay of the next downstream dam. There are three steps involved. The first step consists of reproducing TDG levels in the tailrace area of each dam. The second step is to simulate the dissipation of the tailwater TDG in the downstream pool. The third and final step is to combine the first two model components into a batch so that the entire hydrosystem can be simulated in a batch mode in one pass. Execution of the batch model is just the process of feeding an array of inputs to the network and receiving back the appropriate array of outputs. NeuroShell2 offers a type of backpropagation called a recurrent network that is excellent for time series data. It can automatically create a separate set of data patterns (test set) and uses it to evaluate how well the network is predicting. The test set is approximately 10 percent the size of the training set. By ignoring this option, the user may still choose to use unsupervised (unscontrained) networks. In this case, there are no actual outputs; the network learns to make pattern classifications by making its own clustering scheme for patterns. NeuroShell2 also assigns a default value to the number of hidden neurons, and sets a learning rate that controls the speed of the learning process (change in weight times the error). A momentum factor is used to minimize calculation oscillation caused by too large learning rates and to ensure efficient convergence.

As was done in GASSPILL, the neural network system model starts at the upstream-most project and proceeds in the downstream flow direction. For example, for the Columbia River, the model starts with tailwater TDG calculation at Chief Joseph Dam, followed by calculation of the TDG values in the forebay of Wells Dam. A spill input is specified for Wells so the model can proceed with similar calculation at Wells tailwater and in the Rocky Reach reservoir forebay. The situation at the confluence of the Columbia and Snake Rivers calls for a special summation. Inputs at Ice Harbor are used to predict TDG in the forebay of McNary

Dam (Oregon side); and inputs at Priest Rapids Dam are used to predict TDG in the forebay of McNary (Washington side). The combined inputs from both the Oregon and Washington sites and the spill at McNary are used to predict the TDG levels below McNary Dam and in the forebay of John Day Dam. Sample results of the calibration are produced in the form of 6-hourly plots of observed and predicted TDG on Figure 3 for Lower Granite Dam and the immediately downstream Little Goose Reservoir for the April-July 1995 spill season. Correlation coefficients of the calibration shown in Table 1 below are encouragingly high.

The neural network models do not provide explicit formulaes like in a normal regression. Instead, they are stored in files that can later be accessed via a Dynamic Link Library (DLL) in an Excel spreadsheet. Bar chart plots can be prepared, however, to exhibit the relative contribution of each variable in predicting the network's output (see Figure 4). These plots help determine which inputs are most significant to the outputs.

Conclusions

The results of the work described in this paper show that neural network models can be used to predict dissolved gas saturation levels on the mainstem Columbia and Snake Rivers in response to various spill specifications at each of the dams. The model performances in replicating both 6-hourly training and test data are very encouraging, based on the goodness of fit between observed and predicted TDG levels. The neural network approach to system simulation of TDG appears to be valid and as easy and as efficient as the traditional TDG deterministic modeling.

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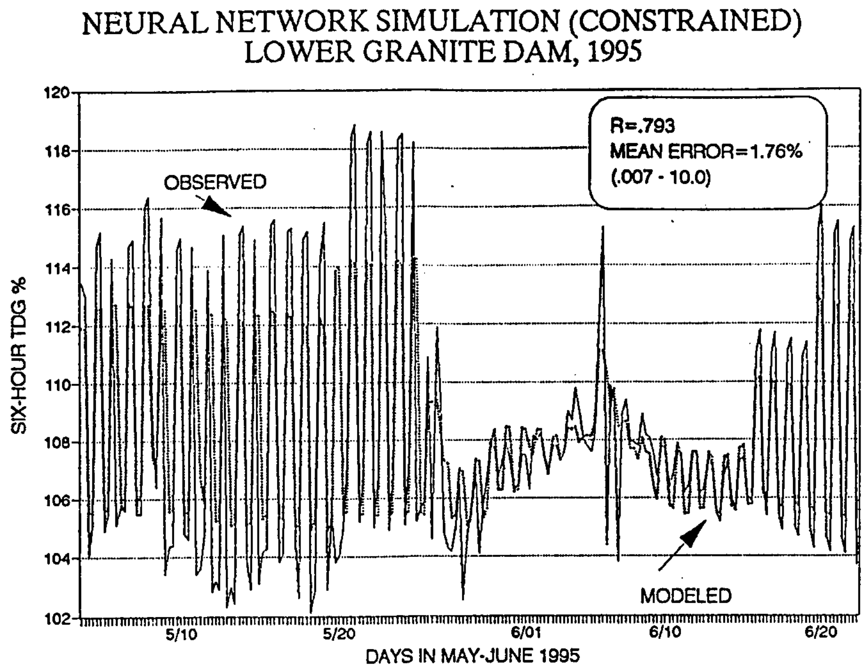

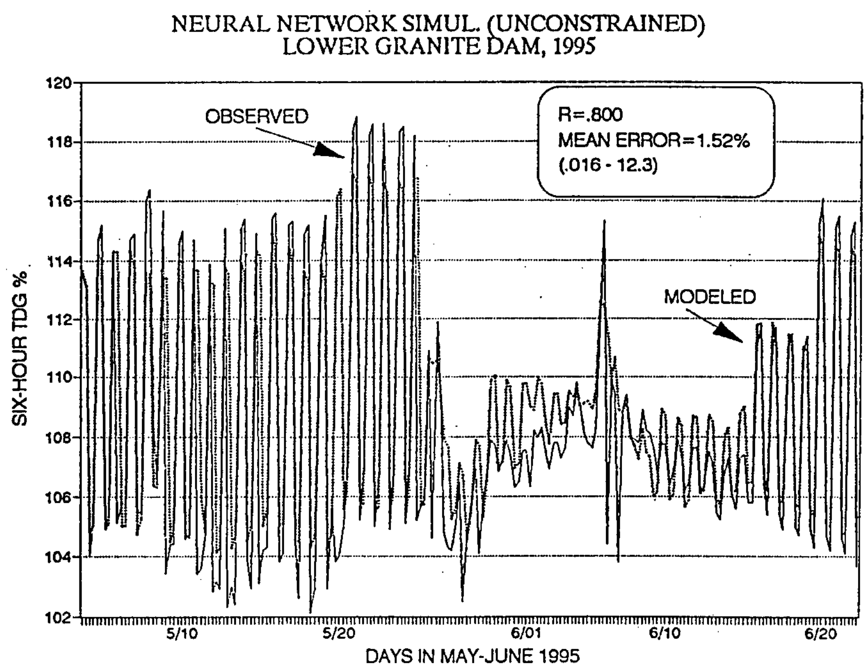

Figure 3. Observed and predicted 6-hr TDG values (Continued)

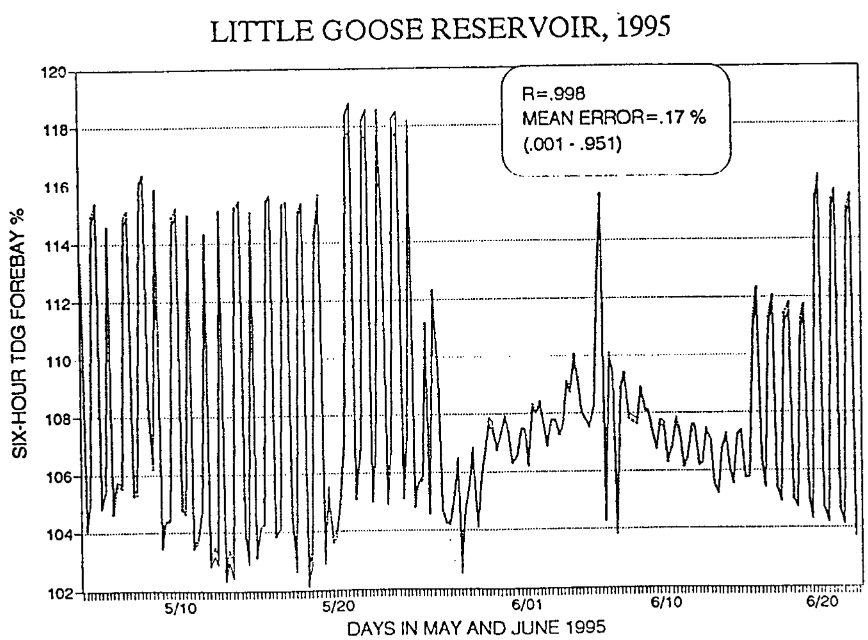

Figure 3. (Concluded)

Table 1
Summary Calibration Results (Correlation Coefficients)

Projects	Basis	
	Forebay	**Tailwater**
Lower Granite	.764	.853
Little Goose	.902	.901
Lo. Monumental	.857	.945
Ice Harbor	.945	.870
McNary	.836	.885
John Day	.712	.840
The Dalles	.887	
Bonneville	.882	.882

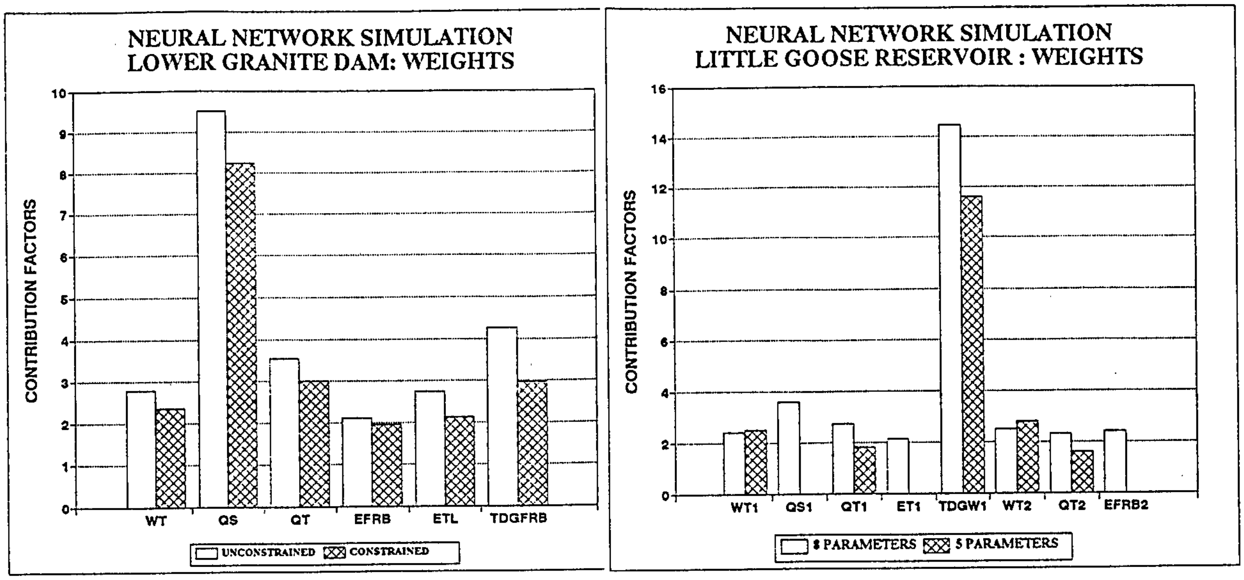

Figure 4. Contribution coefficients—sample output

Modeling of Dissolved Nitrogen in the Columbia and Snake Rivers

by
Mike Schneider[1] *and Steve Wilhelms*[1]

System spills on the Columbia and Snake Rivers have been mandated in part, to provide a safer passage route for fish past mainstem dams. One consequence of system spills is elevated dissolved gas levels that are detrimental to fish. The simulation model entitled GASSPIL was developed to evaluate the effects of project operation on dissolved gas levels throughout the Columbia River system. The objective of this paper is to present a summary of the evaluation and revisions to the GASSPIL model. This was accomplished by refining the spacial and temporal characteristics of the model, refining the current gas transfer descriptions, evaluating the routing technique, and making preliminary estimates of dissolved gas for several system operational scenarios.

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[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

An Empirically Based Numerical Model for Computing Aerator Effectiveness in Reservoirs

by
Robert S. Bernard[1]

Introduction

One might suppose that numerical models should simulate physical processes by solving only equations derived from basic principles such as those for conservation of mass, momentum, and energy (in its various forms). In practice, however, these equations are often so complex as to render modeling efforts intractable without empirical simplification. The flow associated with a bubble diffuser is a case in point. Calculating the turbulent flow about a *single* rising bubble is a problem that taxes the capabilities of existing computers. For the myriad bubbles released by a typical diffuser/aerator, the coupled flow problem is insoluble at the bubble scale. A practical alternative is to parameterize the interaction between the bubbles and the water, and resolve the flow and transport on a scale much larger than the bubble diameter.

The approach taken here is to treat a bubble plume as a cylindrical column of water in which there is an upward buoyant force (created by the bubbles) and a transfer of dissolved gas directly from the bubbles to the water (or vice versa). When these two mechanisms are incorporated into a numerical flow model and adjusted to reproduce test data for small tanks, the adjusted model can be used to predict aerator performance in much larger volumes of water. Model predictions include the overall rate of gas transfer along with the extent and rate of expansion of the aerated (well mixed) region.

Model Description

The MAC3D model (Bernard 1995) is a three-dimensional computer code that solves the Reynolds-averaged Navier-Stokes equations on curvilinear finite-volume grids, assuming that all boundaries are rigid and the flow is incompressible. Bubble plumes are represented as vertical columns in which there is an upward buoyant acceleration g_{plume}, given approximately by

$$g_{plume} \approx \frac{\gamma\, g\, Q_{air}}{A_{plume}\, v_{rise} \left[1 + \dfrac{H}{H_{atm}} \right]}$$

where

γ = dimensionless coefficient

g = gravitational acceleration (32.2 ft/sec^2)

Q_{air} = airflow rate through the bubble diffuser

A_{plume} = assumed cross-sectional area of the plume

H = depth to the diffuser

H_{atm} = depth of water equivalent to one atmosphere (33.9 ft)

v_{rise} = absolute rise velocity of the bubbles

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

The quantity $(1 + H/H_{atm})$ is a correction factor for the compression of the bubbles with increasing depth, and v_{rise} is the sum of the vertical flow velocity and the *relative* rise velocity (assumed to be about 1 ft/sec).

To account for the mixing associated with turbulence, MAC3D assumes that eddy diffusivity and kinematic eddy viscosity are identical, approximating them both with the expression,

$$\nu \approx \lambda \left| \underline{u} \right|$$

where

$\underline{u}$ = vector flow velocity

λ = an empirical length scale, and the viscosity/diffusivity ν has units of area/time

This simple formula is no substitute for an advanced turbulence model, but it does ensure that the diffusivity decreases, as does the velocity, with increasing distance from the diffuser.

There is a subtle connection between γ and λ, in that γ controls the production of energy in the bubble column, while λ controls the dissipation of energy in the flow as a whole. The ratio λ/γ determines the magnitude of the velocity, but λ alone determines the spatial distribution. The steady-state distribution is that which balances energy production with energy dissipation. The physical upper bound for g_{plume} coincides with $\gamma = 1$, in which case *all* the energy generated by the bubble plume goes directly into the computed flow. In a real flow, however, some of this energy is dissipated by turbulent eddies that are too small to be resolved by the numerical model. Thus, it stands to reason that γ should be less than unity in practice, with λ given an associated value that varies in proportion to γ and brings model predictions roughly into agreement with experimental observations.

The length scale λ also plays an important role in preserving numerical stability. Specifically, if δ is the nominal spacing between computational grid points, then λ should to be large enough that $\delta/\lambda < 30$. This criterion ensures that the flow computed by MAC3D will remain stable and well-behaved, regardless of the local magnitude of the velocity.

Trial-and-error comparisons have been made of MAC3D predictions and tank-test velocity data[1] for two different bubble diffusers in a 25-ft-diameter tank with total water depths of 10 ft and 31.3 ft. The results of these efforts suggest a provisional relation between the length scale and the total depth; i.e.,

$$\lambda \approx \gamma(\lambda_0 + 0.016H_{total})$$

where

λ_0 = 0.12 ft

H_{total} = total depth, and a reasonable range for γ appears to be $0 < \gamma < 1/2$

Inside a bubble plume, the rate of transfer of dissolved gas to or from the water is proportional to the difference between the local concentration C and its local saturation value C_{sat}, in units of mass/volume. When this is combined with transport by the flow (advection and diffusion), the governing equation for the dissolved gas becomes

$$\frac{\partial C}{\partial t} + \underline{u} \cdot \nabla C = \frac{K_{plume}}{A_{plume}} \left[C_{sat} - C\right] + \nabla \cdot \left[\nu \nabla C\right]$$

[1] Unpublished test data provided by Gary P. Johnson, November 1995, U.S. Geological Survey, U.S. Department of the Interior, Urbana, Illinois.

where

t = time

∇ = the gradient operator

K_{plume} = the depth-averaged transfer coefficient for the plume (in units of area/time)

A_{plume} = the *assumed* cross-sectional area of the cylindrical bubble column

Note that K_{plume} has to be inferred directly from tank tests for a given bubble diffuser, but A_{plume} need only approximate the average cross section of the real plume within a factor of three or so. In any case, K_{plume} is nonzero inside the bubble plume, and zero elsewhere.

Gas flux through the free surface *into* the water is given by K_{LS} (C_{sat} - C). The coefficient K_{LS} has units of length/time, and it is computed in MAC3D via the relation,

$$K_{LS} \approx 0.00185|u|$$

where u is the horizontal flow velocity along the surface, and the empirical coefficient 0.00185 was chosen to match computed surface-exchange rates with those observed in small tank tests conducted by Wilhelms and Martin (1992). The linear dependence on velocity was suggested by J. Gulliver[1] based on the work of Broecker et al. (1978).

Computed Results

Figure 1 shows comparisons of observed velocity magnitudes[2] and MAC3D predictions made for H_{total} = 10 ft and Q_{air} = 35.5 ft^3/min (cfm) with γ = 1/16, 1/8, and 1/4. The radius of the cylindrical tank was 12.5 ft, the grid spacing was 0.5 ft in the radial and vertical directions, and A_{plume} was assumed to be 6.28 sq ft. In the physical tests, the bubble diffuser was placed at the center of the tank, 3 ft from the bottom. In the MAC3D simulations, the model bubble column extended vertically from this location to the water surface. It is evident from Figure 1 that the overall accuracy of the velocity predictions is about the same for all three values of γ.

Gas transfer tests[2] were performed with the same tank and diffuser to determine the influence of H_{total} and Q_{air} on the total exchange coefficient for dissolved oxygen, K_LA, defined by

$$K_LA = -\frac{1}{t}\ln\left[\frac{C_{eq} - C}{C_{eq} - C_i}\right]$$

where

C_{eq} = the equilibrium concentration

C_i = the initial concentration

These tests were conducted at three different depths and four different airflow rates. *One* of the tests (11-ft total depth and 60-cfm airflow) was used as a benchmark for assigning a value to K_{plume}. All of the remaining test conditions were then simulated with K_{plume} proportional to Q_{air}. The initial concentration was zero, and K_LA was computed using the volume-averaged concentration after one hour. The entire volume was fully mixed within 1 to 2 hours in all cases, at which time the computed variation in C was less than 0.5 mg/l throughout the tank. Figure 2 compares predicted and observed equilibrium concentrations with the saturation

[1] John S. Gulliver, personal communication, February 1996, University of Minnesota, Minneapolis, MN.
[2] Unpublished test data provided by Charles W. Downer, Laurin I. Yates, and Calvin Buie, Jr., September 1995, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

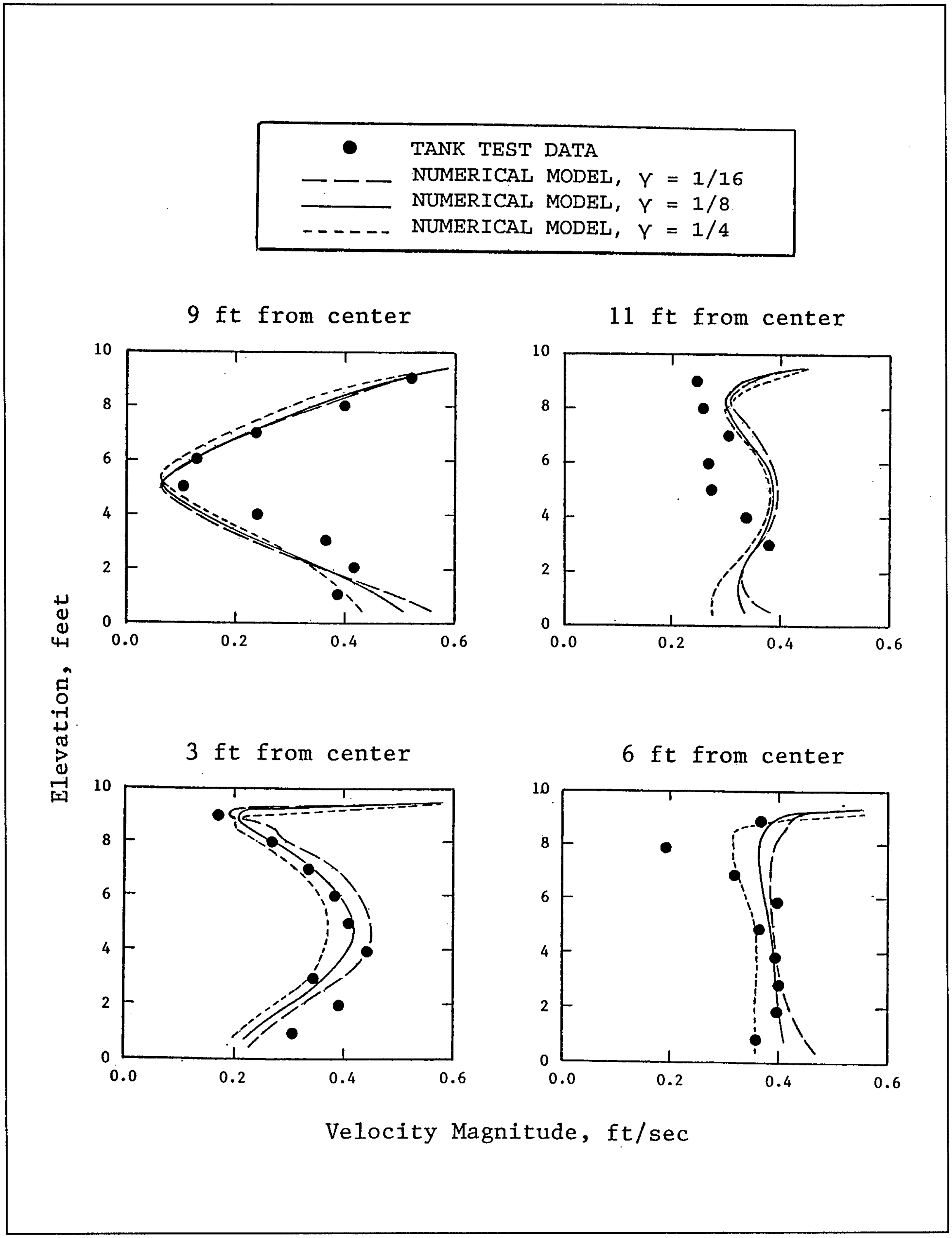

Figure 1. Flow velocities for diffuser with 35.5-cfm airflow in 10-ft-deep, 25-ft-diameter tank

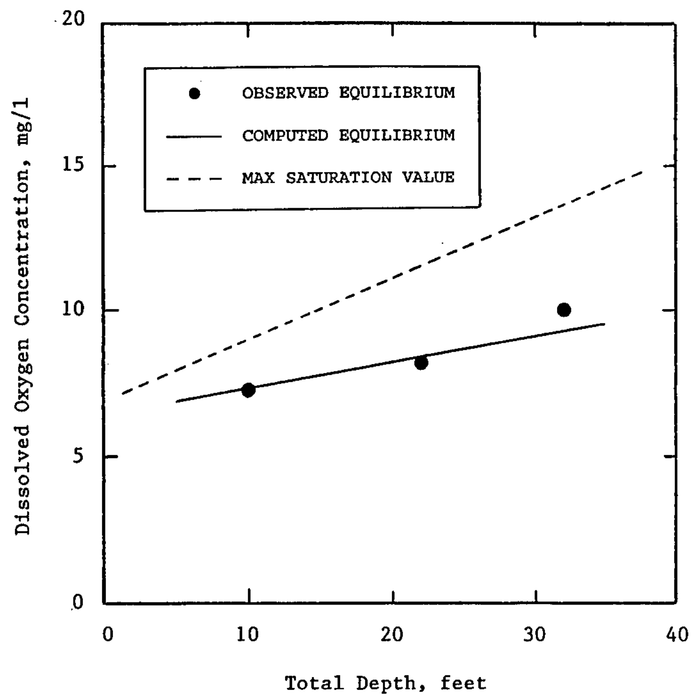

Figure 2. Variation of dissolved oxygen concentration with total depth

concentration at the bottom of the tank, and Figure 3 compares predicted and observed K_LA values for the entire battery of tests.

Conclusion

This investigation has shown that a numerical flow model can be appended with equations for dissolved-gas exchange and transport, and then empirically tuned to reproduce approximately the results of controlled tank tests for a given bubble diffuser. The accuracy of the computed velocities is fairly good in view of the complexity of the flow, and the predicted influences of depth and airflow rate on the exchange coefficient, K_LA, are roughly in agreement with the tests. Future work will determine whether the empirical coefficients derived from these and other tank tests will enable the MAC3D numerical model to predict diffuser performance in much larger volumes of water such as reservoirs.

Acknowledgement

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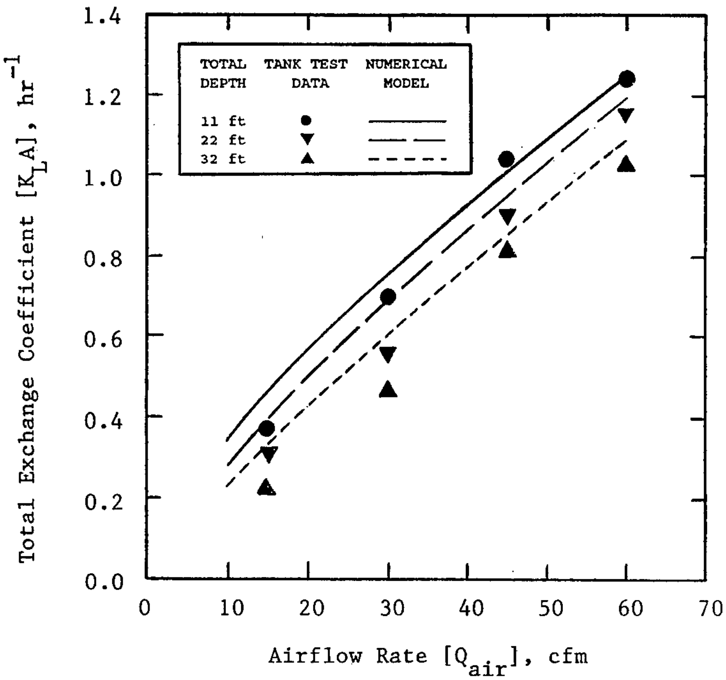

Figure 3. Influence of airflow rate and total depth on exchange coefficient

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Total Dissolved Gas Distribution Data Collection Methods: Modelers' Needs

by
Thomas D. Miller,[1] *Chris A. Pinney,*[1] *and Russell D. Heaton*[1]

With the advent of requests for spill for juvenile anadromous salmonid passage at all eight Corps dams on the Lower Snake and Lower Columbia rivers comes a need for improved understanding of distribution of total dissolved gas (TDG) downstream from spilling projects. The Dissolved Gas Abatement Study (D-GAS) also sprang from the increased emphasis on spill for fish, and is the driving force behind TDG distribution data collection efforts. To facilitate evaluation of fishery benefits from reduced TDG brought about by operational and structural modifications proposed by the D-GAS a system-wide numerical smolt survival model is being implemented. Two dimensional (spatial) TDG prediction capabilities are fundamental to this modeling effort. The production and distribution of TDG supersaturation from these dams are highly variable and dependent on many factors. This dynamic nature complicates collection of TDG data needed to attain an adequate conception of the spatial and temporal distribution of supersaturation downstream of the mainstem projects. In this situation, where a 'snap shot' of the reservoir or system is needed for proper evaluation, the use of satellite imagery and other remote sensing methods would be ideal. However, TDG is one of many water quality parameters that can only measured by direct sampling or in-situ sensing. Therefore, in-situ studies of the distribution of TDG in the tailwaters and pools of the eight Corps projects on the mainstem Columbia and Snake rivers were performed. It is essential to understand the spatial distribution and dynamics of TDG supersaturation downstream of spilling projects to both describe the existing system conditions and to develop, calibrate, and validate the numerical model that is currently under development.

Historically, TDG has been monitored in the study area through a system of fixed monitoring stations located at key sites in the forebay, and more recently, tailwaters of the Corps projects. Due to their single-point spatial nature, measurements from the fixed monitoring stations were deemed inadequate for the purposes of this study. To enhance the information collected by the fixed monitoring system, transect studies were conducted in 1995. These consisted of prescribing a series of locations following the longitudinal axis of the rivers and taking TDG measurements at each of a set of sampling sites located on a lateral line that transects the river at those prescribed locations. Though the transect studies were developed to attempt to define the transport and mixing characteristics of TDG throughout the study area, early in the 1995 season it became apparent to the study team and modelers that this approach could not address the hydrodynamics that occur therein. Alterations were made to the study plan including temporal and longitudinal continuity achieved by following parcels of water and running transects around the clock, and drift studies accomplished by 'drifting' with the flow and taking continuous readings. The modelers found these changes did not adequately alleviate the data analysis difficulties and gaps presented by the original plan. During 1996 thirty self-contained wireless dissolved gas loggers will be deployed in a grid below each studied project. The grid will be determined based on 1995 work and physical

[1] U.S. Army Engineer District, Walla Walla; Walla Walla, WA.

model evaluations. This remote monitoring approach will provide the modelers with orders of magnitude more data points and address both the temporal and spatial dynamics of TDG distribution in the system at approximately the same cost as the previous methods.

Assessing the Impact of Channel Deepening in Delaware Bay Through Numerical Modeling

by
Keu W. Kim,[1] *Billy H. Johnson,*[1] *and J. Gebert*[2]

Abstract

A field data collection program provided the necessary data for model verification. These data include water-surface elevation, velocity, salinity, and temperature at three depths at several long-term stations. In addition, two different two-week surveys provided similar data at multiple locations across several ranges. Results from application of the 3D model using October 1992 data as well as results illustrating the impact of the channel deepening on salinity intrusion are presented.

Introduction

Delaware Bay extends approximately 215 km from its mouth at the Cape May-Cape Henlopen transect to the head of tide at Trenton, New Jersey (Figure 1). The estuary has a relatively wide and shallow lower bay and a narrow upstream channel with the conveyance area decreasing with increasing upstream distance.

Delaware estuary tides are predominantly semi-diurnal. Mean tidal ranges vary from approximately 1.32 m near the mouth to 2.55 m at Trenton. On the Chesapeake side of the C&D Canal, the mean tidal range is 0.65 m, and the high water occurs approximately 1 hour earlier than on the Delaware side. Approximately 6.5 hours are required for the tide to propagate from the mouth of Delaware Bay to the head of tide at Trenton, New Jersey. Overall, the ratio of the tidal-mean volume flux through the bay entrance to the mean freshwater inflows is approximately 300 to 1 (Galperin and Mellor 1990). Thus, under mean-inflow conditions, Delaware estuary reaches have been classified as vertically homogeneous (Biggs 1978) with typical vertical salinity variations of 1 ppt (Garvine et al. 1992). However, under high-flow conditions, vertical stratification develops (Sharp et al. 1986). Thus, depending on the freshwater inflow, the Delaware Estuary can be either well mixed or partially stratified.

Since 1910, the main navigation channel of Delaware Bay has been dredged often and the U.S. Army Corps of Engineers, Philadelphia District is now proposing to deepen the existing 12.38 m navigation channel to 13.93 m below mean low water. One factor being considered in the evaluation of the feasibility of deeper draft access to the ports of Philadelphia and Camden is the impact of a deeper channel on salinity intrusion in Delaware Bay. This issue is addressed through the application of a three dimensional (3D) numerical hydrodynamic model called CH3D-WES (Johnson et al. 1991) that makes computations on boundary-fitted curvilinear grids. Since studies such as those by Wong (1991) have shown that a substantial fraction of subtidal volume exchange in Delaware Bay is associated with the exchange through the Chesapeake and Delaware canal (C&D Canal) which connect the Delaware and Chesapeake Bays, the numerical model must include not only Delaware Bay but also the C&D canal and at least a portion of the Chesapeake Bay.

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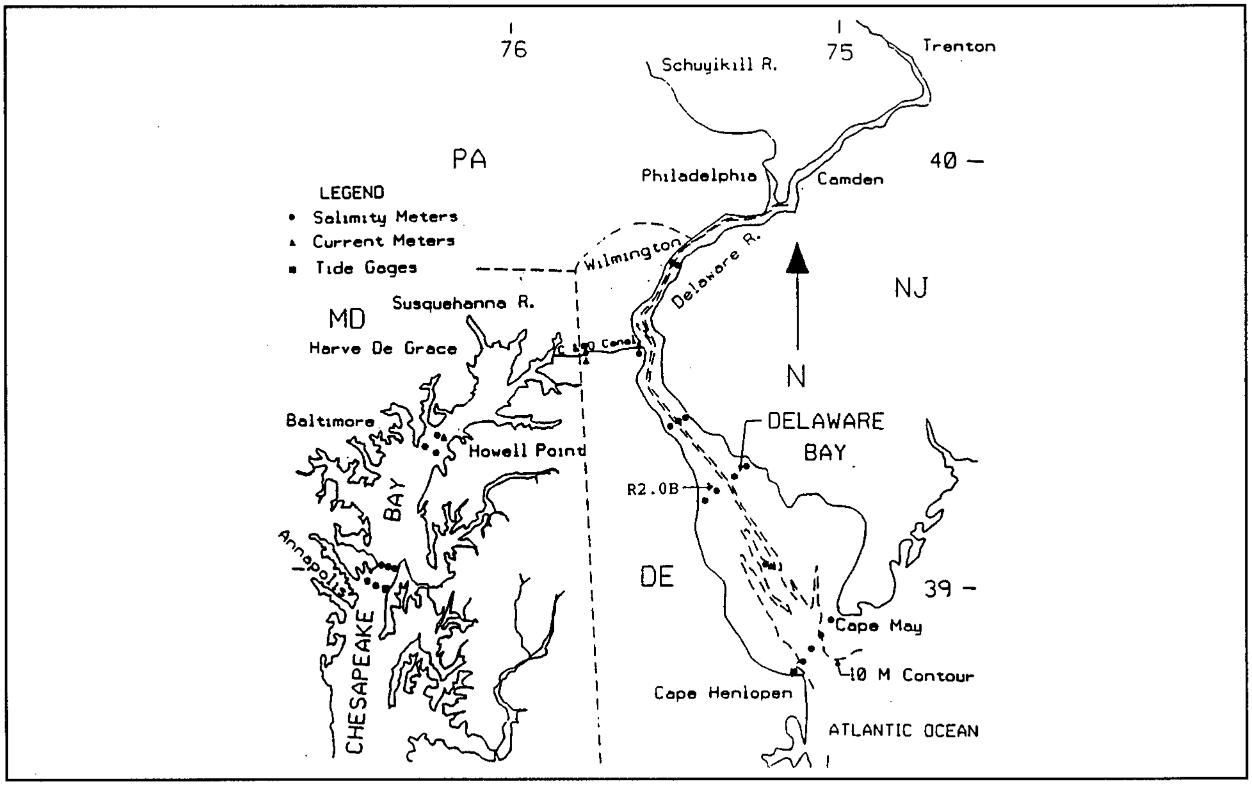

Figure 1. Delaware Bay and Upper Chesapeake Bay

The numerical grid employed in the 3D model is shown in Figure 2. There are 3,052 active horizontal cells and a maximum of 16 vertical layers, resulting in 12,568 computational cells. To capture the important features of the hydrodynamic processes, grid resolution is 1.52 m in the vertical (layer thickness) and ranges from 1 to 10 km in the horizontal.

Data Requirements

Basic input data required are time-varying water surface elevations at the ocean entrance and freshwater inflows at the head of tributaries or rivers. In addition, time-varying salinity and temperature data must be prescribed at all inflow boundaries. Also, time-varying wind must be prescribed at one or more locations. These are then distributed over the entire grid. Boundary data, as well as interior data required for model verification, are being provided by a detailed field data collection program that began in October 1992 and concluded in October 1993 (Fagerburg 1995). The field data collection program consisted of both short-term and long-term continuous recording of tide, current, and salinity data. The short term (2 weeks) field data sets cover 12-25 October 1992 and 19-30 April 1993. The data collection stations were positioned at various locations from Wilmington, Delaware to the entrance of Delaware Bay, as well as within the C&D canal, and in upper Chesapeake Bay. A total of seven data collection ranges with 2 to 4 stations per range were monitored for current and salinity at 3 to 5 depths. There are a total of ten stations through out the Delaware Bay, C&D Canal, and the upper Chesapeake Bay monitoring tide, velocity, temperature, and salinity at an interval of 15 minutes. The locations of these stations are shown on Figure 1.

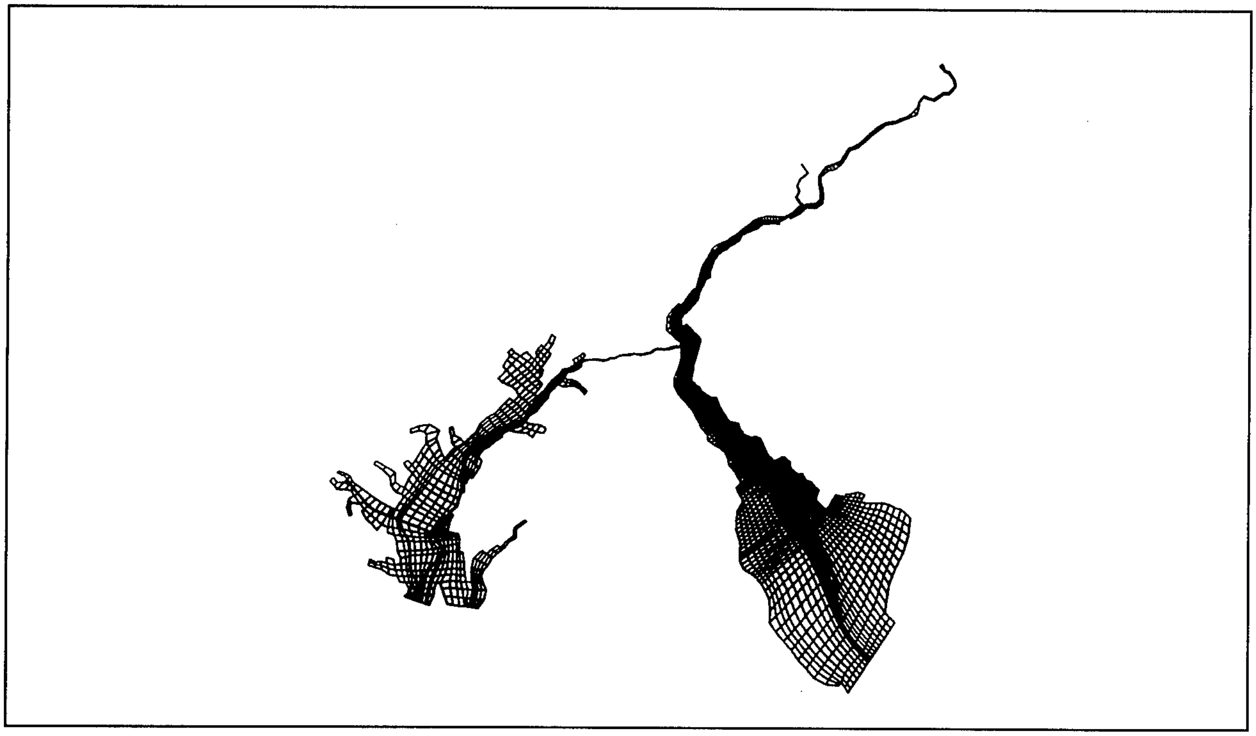

Figure 2. Boundary-fitted platform grid of Delaware Bay

Results from Model Verification

Data collected during October 1992 were employed in the model application that provided the results presented in Figure 3. The October 1992 period represented a relatively low inflow (about the half of the average annual freshwater inflow) event of fairly short duration. The initial salinity and temperature field were constructed from observed data and held constant for the first 5 days of the simulation. The comparison of computed and observed near bottom salinity at station R2.0B (see Figure 1) is presented in Figure 3. Although few data exist for comparison, the model appears to reproduce the observed salinity field well.

To gain confidence in the numerical model's ability to predict changes in salinity during prolonged extreme drought periods, the June - November 1965 low flow period (about 20 % of the average annual inflow) has been simulated. Such conditions result in the movement of salinity up river to Philadelphia, Pennsylvania and beyond. Interior data for comparison with the computed results were limited, with only salinity data available at a few stations. Figure 4 shows a comparison of computed and observed chlorinity at the Benjamin-Franklin Bridge, Philadelphia, Pennsylvania. The chlorinity labeled observed was actually constructed from conductivity measurements using the following expressions (Thatcher and Harleman 1978):

Table 1
Conversion of Specific Conductance to Chlorinity

Specific Conductance Ranges (K)	Equation
0 - 249.6	$Cl=8.092*10^{-4}(K)^{1.7687}$
249.6 - 525.7	$Cl=3.236*10^{-5}(K)^{2.3518}$
525.7 - 5,477	$Cl=2.686*10^{-2}(K)^{1.2789}$

where K is mhos at 25 degrees Celsius and Chlorinity (Cl) is in ppm. Salinity computed by the model is converted to chlorinity by

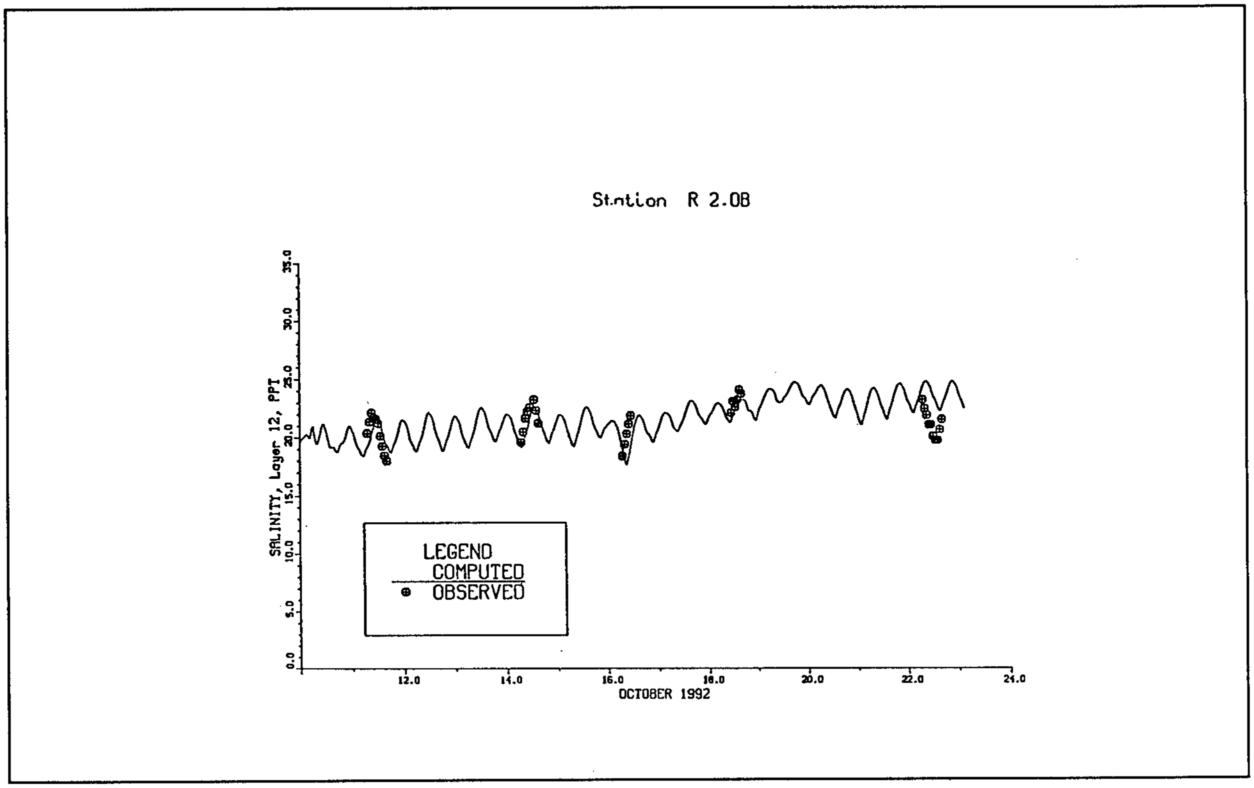

Figure 3. Near-bottom salinity at Station R2.0B

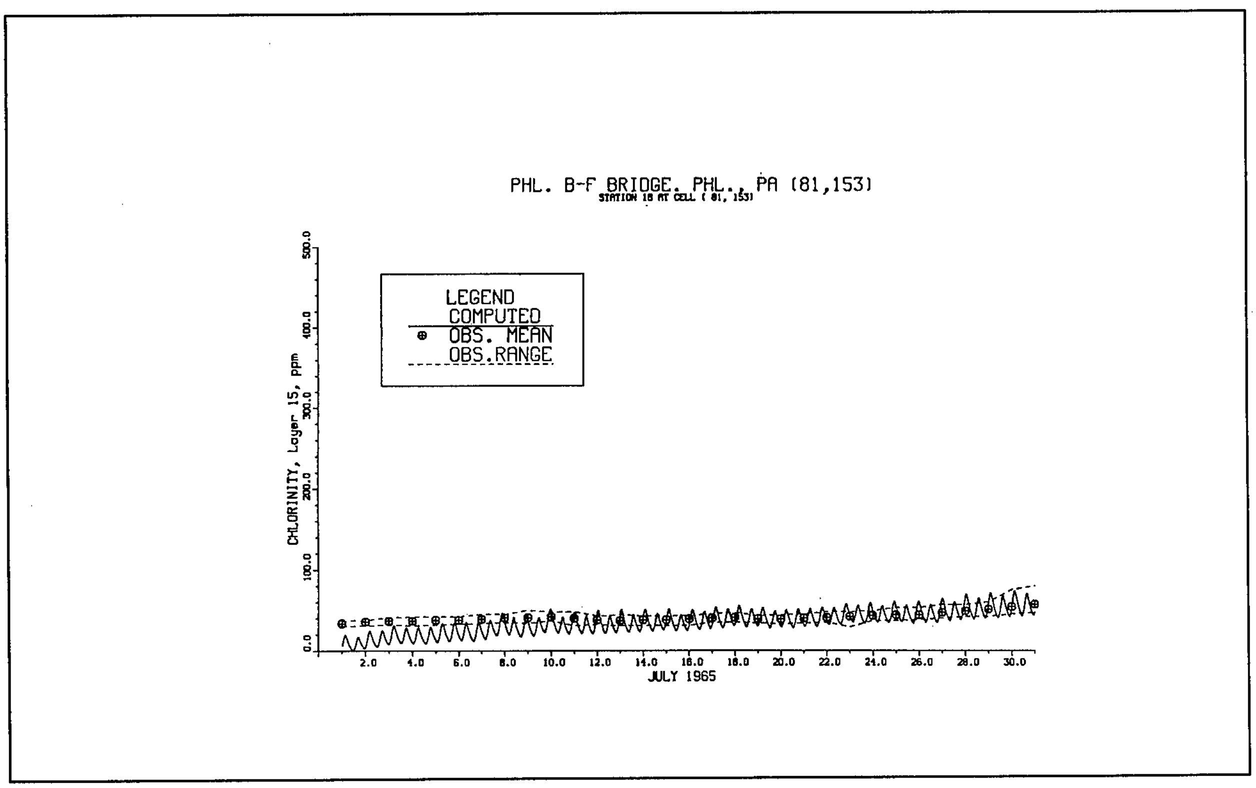

Figure 4. Near-surface chlorinity at Benjamin-Franklin bridge, Philadelphia

$$S = 0.03 + 1.805\ Cl \tag{1}$$

This comparison illustrates the ability of the numerical model to accurately compute salinity conditions in Delaware Bay - C&D Canal - Upper Chesapeake Bay system during prolonged drought periods.

Results from Channel Deepening Tests

To determine the impact of channel deepening on salinity intrusion in Delaware Bay, the verified model was run for regulated freshwater inflow conditions to a drought of record during June-November 1965 with and without channel deepening. The impact of channel deepening at the tidal time scale was determined by comparing time series plots of water surface elevation and velocity and salinity at multiple depths with and without channel deepening. Since salinity rather than chlorinity is computed in the 3D model, to provide chlorinity plots at the upper river locations, the computed salinities were converted to chlorinity by using Equation 1. Figure 5 is a time series plot for October 1965 that shows the impact of channel deepening on the chlorinity at River Mile 98.

Summary

A 3D numerical model of the Delaware Bay - C&D Canal - Upper Chesapeake Bay system has been developed to aid in assessing the impact of deepening the navigation channel in Delaware Bay from 12.20 m to 13.72 m. Model verification centered around reproducing the flow event of October 1992. Results from model runs with a 13.72 m channel were compared with results from the existing 12.20 m channel runs. Time series plots of chlorinity at River Mile 98 were compared to assess the impact of channel deepening.

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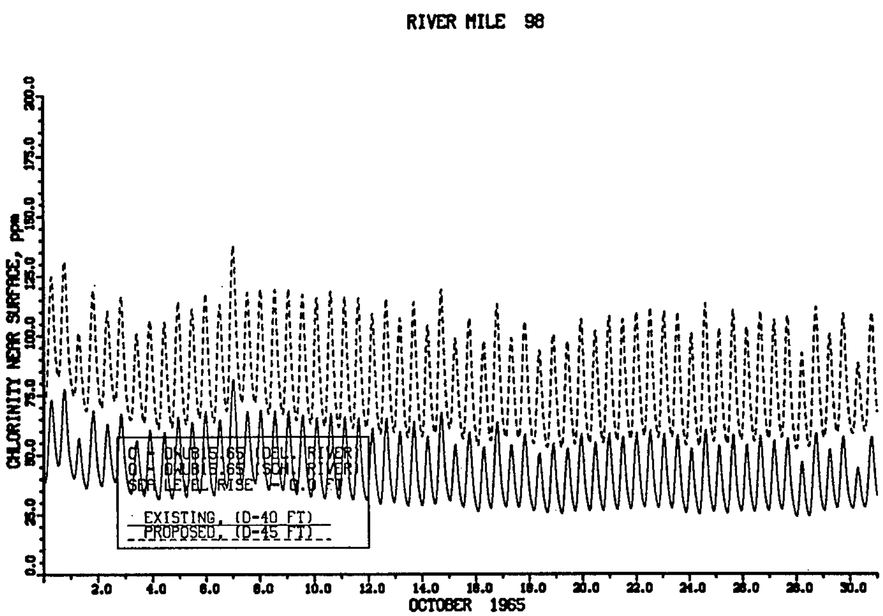

Figure 5. Impact of channel deepening on salinity at RM 98

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Effects of In-Water Disposal of Dredged Material on Fishes in Lower Granite Reservoir, Snake River

by
David H. Bennett,[1] *Teri Barila,*[2] *and Chris Pinney*[2]

Introduction

Sediment deposition in Lower Granite Reservoir, Idaho-Washington, has concerned managers over the ability of the levee system on the Snake and Clearwater rivers to protect the cities of Lewiston, Idaho, and Clarkston, Washington, from flood waters. Lower Granite Reservoir as part of the Lower Granite Project provides electrical power generation, flood control, navigation, and recreation. Estimates of sediment deposition by U.S. Army Corps of Engineer personnel have indicated that nearly 1,760,800 m3 annually enter the confluence of the Snake and Clearwater rivers at the upstream end of Lower Granite Reservoir.

Although several alternatives were evaluated to alleviate the accumulation of sediment, dredging and in-water disposal were immediate solutions. Sediment dredging was initiated in 1986 with land disposal (Bennett and Shrier 1986). In 1988, 1989, and 1992 experimental in-water disposal of dredged material was made approximately 32.2 km downstream of the confluence of the Snake and Clearwater rivers in Lower Granite Reservoir to construct an island at river kilometer 193 (Rkm 193) and an underwater plateau at mid-depth (6-18 m) located at Rkm 192. Additional dredged material was deposited into the thalweg of the channel in 1992 to construct a deep water site.

Use of dredged material to enhance habitat for fish and wildlife is common (Landin and Smith 1987). Most uses have been for wildlife resources or coastal fish habitat (McKern and Iadanza 1987). However, Anderson (1985) reported that additional habitat for food production for wild chinook salmon *Oncorhynchus tshawytscha* was created in the Campbell River, British Columbia. Also, McConnell et al. (1978) reported that the Miller Sands site, Columbia River, Oregon, created by use of dredged material was used extensively for feeding by juvenile chinook salmon. Both the Miller Sands and Campbell River sites were both estuarine systems; estuarine systems typically receive a high natural deposition of sediment. As these studies provided little information on the effects of in-water disposal of dredged material on up-river ecosystems, we conducted a 5-year monitoring study on the fish and benthic communities in Lower Granite Reservoir.

Study Area

Lower Granite Reservoir is a 3,602 ha impoundment on the lower Snake River (Bennett and Dresser, this volume). The Snake River originates in southwestern Wyoming, flows west through a series of reservoirs in southern Idaho, cuts north to Lewiston, Idaho, and Clarkston, Washington, and flows an additional 224 km to

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the confluence with the Columbia River in central Washington. The lower 224 km of the lower Snake River flows through four reservoirs with Lower Granite being the uppermost.

Eleven sampling stations were used to monitor fish abundance from 1988 through 1993 (Bennett and Dresser, this volume). Habitat at stations D1, D2, D4, and D7 was created by in-water disposal of dredged material; stations R3, R5, R6, R8, R9, R10, and R11 were classified as reference stations.

Methods

To assess abundance of juvenile and adult fishes, reduce sampling gear bias and make representative collections, we sampled the fish community in Lower Granite Reservoir by gill netting, beach seining, and electrofishing from 1989 through 1993. Pelagic abundance of fishes was assessed by gill netting and littoral abundance was assessed by gill netting, beach seining, and electrofishing.

Eight horizontal, experimental multifilament gill nets 68.6 m x 1.8 m consisting of three graded panels with bar measurements of 3.8, 4.4 and 5.1 cm (Webb et al. 1987) were generally fished monthly from April through November, although in recent years Endangered Species Act (ESA) permit limitations have precluded summer sampling. Gill nets were set perpendicular to the shoreline and fished on the bottom for approximately 3 hours of daylight and 3 hours of dark for a total of 6 hours and checked every 2 to 3 hours to avoid destructive sampling to salmonids and other fish species.

A 30.5 m x 2.4-m beach seine with a 2.4-m3 bag constructed of 0.64-cm mesh was used during the day from April through October to sample fish along the shoreline at shallow stations D1, D2, R3, R5, R9, R10, and R11. Three standardized beach seine hauls were made by setting the seine parallel and approximately 15.2 m from shore with attachment lines and then drawing the seine perpendicular toward the shoreline (sample area 1,524 m2).

Nighttime electrofishing was conducted by paralleling the shoreline, as close as possible, at shallow stations D1, D2, R3, R5, R9, R10, and R11. Electrofishing was conducted during April through November except when restricted by ESA provisions. Electrofishing effort generally consisted of two to three periods of 5 minutes using a constant output of 400 volts at 3-5 amps at each station.

All fish collected by the various gear types were identified to species, measured to total length (mm), and released. Adult salmonids were released immediately without being removed from the water.

Results and Discussion

We collected from 14,000 to 20,000 fishes annually from 1989 through 1993 comprising 28 genera and species (Table 1). Of these, four species were anadromous fishes while the remaining 24 species/genera were resident; only nine of which were native species.

Considerable variation in catch/efforts were determined among years and within both the disposal and reference sites (Table 1). Higher catch/efforts were observed in most of the native fishes with the exception of the introduced smallmouth bass *Micropterus dolomieu*.

Comparisons of catch/effort indicated a few differences among stations, years, and gear types. Overall, catch/efforts for juvenile chinook salmon were lowest at disposal stations D1 and D2 and highest at reference station R5. Catch/efforts by beach seining at reference station R5 and disposal station D2 were significantly ($P<0.05$) different while those for electrofishing were not. Although differences in abundance of juvenile steelhead *O. mykiss* based on catch/effort by beach seining were found among year and stations, numbers of steelhead collected was largely related to water year. Higher flows result in more rapid movements through Lower Granite Reservoir and lower catch/efforts. Catch/efforts for northern squaw fish, the most significant salmonid smolt

Table 1
Mean Catch/Efforts of Species Sampled at Disposal and Reference Sites by Beach Seining, Electrofishing, and Gill Netting From 1989 Through 1993 in Lower Granite Reservoir

	Disposal Sites					Reference Sites				
	1989	1990	1991	1992	1993	1989	1990	1991	1992	1993
White sturgeon	0.001	0	0.001	0.004	0	0	0.001	0	0.004	0.005
American shad	0	0	0	0.003	0	0	0.001	0	0.001	0
Sockeye salmon	0	0.006	0.012	0	0	0.003	0.006	0.020	0	0
Chinook salmon	1.128	0.268	0.695	0.544	0.120	3.461	1.090	1.072	0.975	0.368
Rainbow trout	0.171	0.105	0.261	0.274	0.096	0.361	0.329	0.823	0.826	0.114
Mountain whitefish	0.024	0.157	0.188	0.023	0.326	0.002	0.053	0.008	0.009	0.051
Chiselmouth	0.107	0.035	0.127	0.082	0.074	0.383	0.237	0.398	0.175	0.185
Common carp	0.015	0.013	0.006	0.009	0.012	0.087	0.135	0.020	0.033	0.020
Peamouth	0.016	0.012	0.063	0.053	0.033	0.024	0.007	0.014	0.020	0.010
Northern squawfish	0.261	0.217	0.390	0.100	0.097	0.312	0.369	0.639	0.226	0.101
Speckled dace	0	0	0	0	0	0	0.001	0	0	0
Redside shiner	0.024	0.040	0.006	0.005	0.006	0.014	0.038	0.009	0.009	0.001
Bridgelip sucker	0.024	0.018	0.059	0.004	0.023	0.059	0.063	0.074	0.029	0.037
Largescale sucker	1.044	0.932	1.446	0.817	0.815	2.774	1.562	1.582	1.070	1.074
Yellow bullhead	0	0.002	0.009	0.001	0	0	0.005	0.014	0.010	0.001
Brown bullhead	0.008	0.057	0.021	0.013	0.015	0.015	0.030	0.013	0.018	0.016
Black bullhead	0	0	0	0	0	0.002	0	0.004	0.002	0
Channel catfish	0.029	0.008	0.003	0.005	0.033	0.017	0.015	0.003	0.032	0.010
Tadpole madtom	0	0.026	0	0	0	0	0.111	0	0	0.001
Lepomis spp.	0.019	0.061	0.571	1.672	0.182	0.282	0.091	1.879	3.729	0.095
Pumpkinseed	0.004	0.012	0.082	0.182	0.078	0.277	0.109	0.266	0.437	0.072
Bluegill	0	0.099	0.010	0.002	0.043	0.007	4.070	0.017	0.259	0.021
Smallmouth bass	0.531	1.504	0.666	0.665	0.307	2.510	2.411	1.495	1.509	0.551
Pomoxis spp.	0	0.026	0.006	0.056	0	0.001	0.044	0.030	0.017	0.004
White crappie	0.062	0.086	0.017	0.008	0.022	0.283	0.135	0.056	0.012	0.019
Black crappie	0.024	0.242	0.027	0.005	0.093	0.050	0.396	0.052	0.063	0.064
Yellow perch	0.006	0	0.001	0.026	0.032	0.036	0.005	0.014	0.050	0.047
Sculpin	0	0.002	0.006	0	0	0.006	0.001	0.003	0	0.003

predators in the Columbia basin (Vigg et al. 1991), at disposal stations were not significantly different from several reference stations. Significant year-to-year variation in catch/efforts was found among stations. Catch/efforts of smallmouth bass, the second most significant predator on salmonid smolts in Lower Granite Reservoir (Curet 1994), generally were lower at disposal stations than at reference stations.

Several concerns expressed at the inception of the dredging and in-water disposal project have been allayed by our monitoring. A principal concern was over creating habitat highly favorable for predators, such as smallmouth bass and northern squawfish. Our data does not indicate habitat created by in-water disposal contributed to changes in predator abundance.

Another concern of dredging and disposal was that the habitat created could become too favorable and result in residualization of salmonid smolts. Comparisons of catch/effort by beach seining and electrofishing have indicated smolts do not concentrate in the newly constructed habitats and any residualization in the reservoir should be attributed to other factors (i.e., low flows as with juvenile steelhead in 1992) not related to the in-water disposal.

Use of dredged material has potential to create more shallow water habitat in Lower Granite Reservoir. Providing more shallow water habitat could increase fish species richness and increase available rearing habitat for subyearling chinook. However, if managers have concerns for increased species richness, dredged material can be disposed in mid to deep waters with no apparent ill effects as these disposal stations were represented by fewer species and low overall abundance.

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Larval Fish Abundance Associated With In-Water Disposal of Dredged Material in Lower Granite Reservoir, Idaho-Washington

by

David H. Bennett[1] *and Thomas J. Dresser, Jr.*[1]

Introduction

Sediment deposition of about 1,760,800 m3 annually enter the confluence of the Snake and Clearwater rivers in Lower Granite Reservoir, Idaho-Washington, interfering with navigation and arousing concern over the ability of the levee system to protect the cities of Lewiston, Idaho, and Clarkston, Washington. Dredging and experimental in-water disposal were immediate short-term solutions and were conducted over a 2-year period (1988-1989). An underwater bench and island were constructed at mid-depth (6.1-18.2 m) and additional disposal at a deep water (> 18.2 m) site. Dredged material was considered potentially beneficial in Lower Granite Reservoir as shallow water habitat constitutes about 8% of the surface area in Lower Granite Reservoir. Shallow water habitat provides foraging opportunities and short-term rearing for downstream migrating salmonid fishes (Bennett and Shrier 1986) and spawning and rearing habitat for resident game fishes (Bennett et al. 1993a,b). We compared the abundance of larval fishes at in-water disposal sites with those of reference sites in Lower Granite Reservoir. Effects of the underwater bench and deep water disposal on larval fishes are not evaluated in this paper.

Study Area

Lower Granite Reservoir, located in southeastern Washington, is the uppermost impoundment on the lower Snake River. Lower Granite Reservoir has a surface area of 3,602 ha with an average depth of 16.6 m and a maximum depth of 42.1 m (Curet 1994). Reservoir elevation is maintained between 240.5 and 242.1 m above mean sea level. Shoreline substrata in Lower Granite vary from rip-rap (45.9%), mud-sand beaches to basalt cliffs (32%) with adfluvial fans. Sand-talus (8.0%) and sand (9.8%) cover the remaining areas. *Potomogeton crispus* is the predominate submerged aquatic vegetation found in shallow water (< 3.3 m), although *P. filiformis* also has been reported (Bennett et al. 1993a,b).

Five sampling stations were sampled in Lower Granite Reservoir to assess larval fish abundance (Figure 1). Three stations were associated with habitat constructed from in-water disposal events (D1 and D2), while the remaining two were shallow water reference stations (R5 and R11).

Methods

Larval fish sampling was conducted at biweekly intervals from June through mid-September, 1989 through 1993 in Lower Granite Reservoir using paired (0.5-m) plankton nets and a hand-drawn beam trawl (LaBolle et al. 1985). Plankton nets were towed approximately 1.6 m/s at night whereas the beam trawl was used during the day. Three plankton net hauls were made at each station at the surface and

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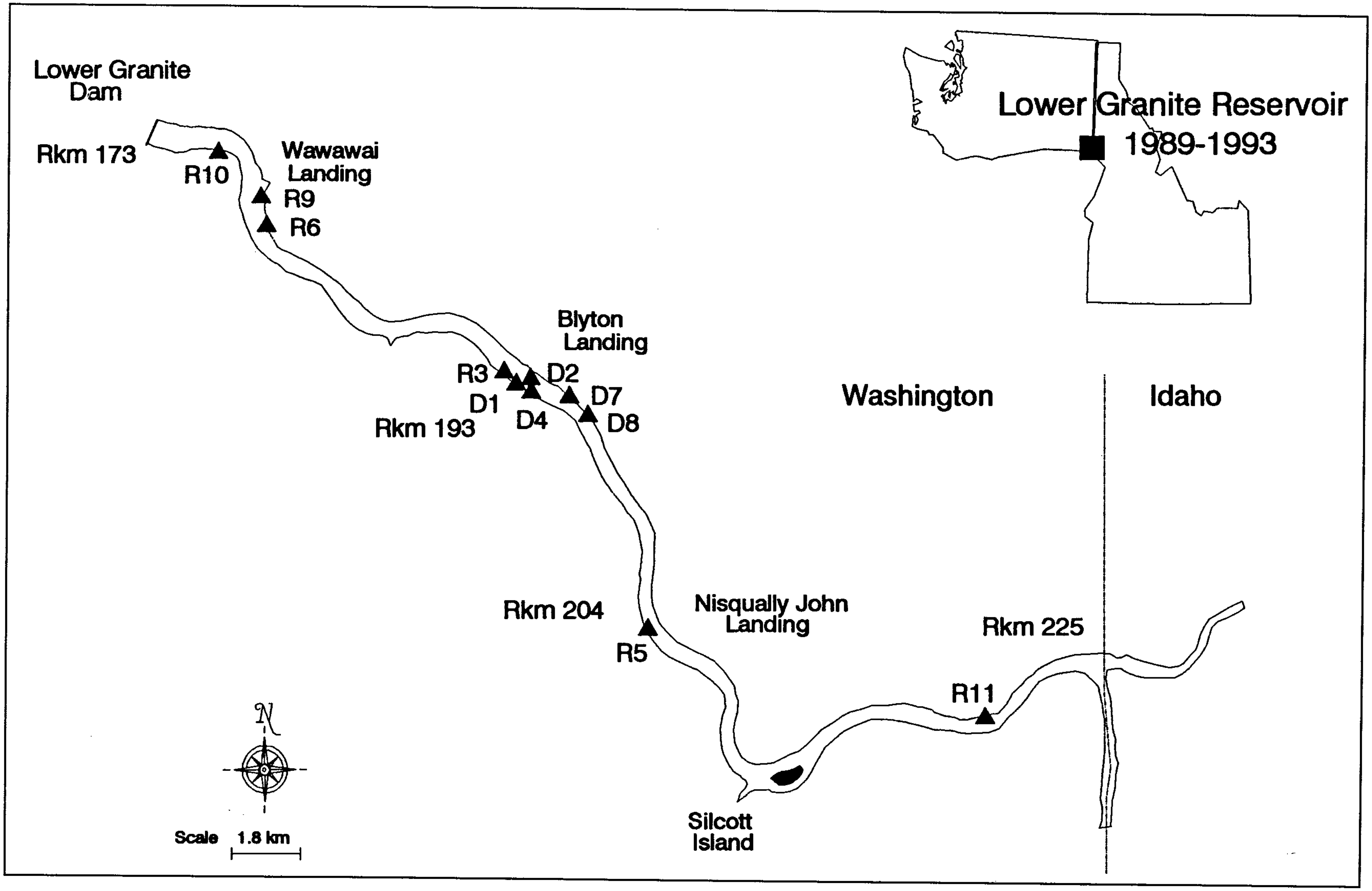

Figure 1. Map of sites sampled for larval fish abundance by paired (0.5-m) plankton nets and a hand-drawn beam and for fish abundance by beach seining, electrofishing, and gill netting for 1989 through 1993 in Lower Granite Reservoir, Idaho-Washington. Rkm indicates river kilometer

1.0 m depth for 3 minutes each night we sampled. The beam trawl was pulled three times each along the shoreline in shallow (< 1.0 m) and deeper (> 1.0 m) water by two people over a standard distance of 15.0 m. All samples were preserved in 10% buffered formalin solution for later identification. The degree of larval development regulated the taxonomic level to which larvae were identified. Estimates of larval fish density were determined by a quadrant sampling scheme for both plankton nets and handbeam trawl samples (Scheafer et al. 1986).

Results and Discussion

A total of 47,334 larval fishes was collected by 1,620 larval plankton net and 1,200 handbeam trawl samples between 1989 and 1993 (Table 1). Handbeam trawl samples taken accounted for 83.3% (n=39,436) of the total number of larval fishes collected in Lower Granite Reservoir. Highest numbers of larval fishes were collected at shallow water reference stations R5 and R11 and accounted for 88.6% (n=41,953) of the total number sampled. Shallow water disposal stations (D1 and D2) accounted for 9.9% (n=4,763) of the total number of larval fishes sampled in Lower Granite Reservoir. Few statistical differences in abundance of larval fishes were found between disposal and reference stations from 1989 through 1993, although larval fish abundance was generally higher at shallow water reference stations (R5 and R11) than at the experimental disposal stations.

Seven larval fish families were represented in the handbeam trawl and plankton net samples in Lower Granite Reservoir (Table 1). Catostomids accounted for 46.3% (n=21,903) of the total number of larval fishes collected followed by Cyprinids (34.8%; n=16,490) and Centrarchid fishes (18.4%; n=7,841). Highest estimates of Catostomids (no./ 10,000 m3) occurred during 1991 (mid-July) and peaked at approximately 1.5 million larval fishes whereas estimates for other years were substantially lower. Few station within year comparisons of abundance (no./10,000 m3) for *Catostomus* spp.

Table 1
Number of Larval Fishes Sampled by Paired (0.5-m) Plankton Nets and a Hand-Drawn Beam Trawl During June Through September From 1989 to 1993 in Lower Granite Reservoir, Idaho-Washington

	Year					
Family/Species	**1989**	**1990**	**1991**	**1992**	**1993**	**Total**
Clupeidae	10	1	16	15	132	174
Cyprinidae	513	470	1,902	2,044	3,306	8,235
Catostomidae	838	319	14,748	117	5,881	21,903
Ictaluridae	9	10	20	1	4	44
Centrarchidae	1,073	1,004	894	4,546	324	7,841
Percidae	0	0	4	17	2	23
Northern squawfish	213	235	6,514	1,248	45	8,255
Smallmouth bass	190	14	93	488	74	859
Total	2,846	2,053	24,191	8,476	9,768	47,334

were significant (P > 0.05) between disposal and reference stations. Northern squawfish *Ptychocheilus oregonensis* accounted for over half (51.4%) of the Cyprinids collected and the majority were collected at shallow water reference station R11. Abundances at station R11 were significantly different from other stations (D1, D2, and R5) during 2 (1991 and 1992) of the 5 years sampled. Highest abundances of larval northern squawfish generally occurred at reference station R11. Estimates of larval squawfish abundance after 1991 have remained low and may be related to high mortality of adult squawfish associated with the Sport Reward Program on adult northern squawfish conducted in the upstream sections of Lower Granite Reservoir. Other Cyprinids collected in Lower Granite Reservoir, but lower in abundance, included peamouth *Mylocheilus caurinus*, redside shiner *Richardsonius balteatus*, and chiselmouth *Acrocheilus alutaceus*. Centrarchids were represented by pumpkinseed *Lepomis gibbosus*, black crappie *Pomoxis nigromaculatus*, white crappie *P. annularis*, and smallmouth bass *Micropterus dolomieu*. Smallmouth bass accounted for 9.9% of the total number of Centrarchids identified during 1989 through 1993. Highest abundances of larval squawfish and larval fishes in general were at station R11, a shallow water station near the upstream part of the reservoir. Habitat at this site is similar to that at shallow water disposal station D1; however, low numbers of larval fishes were collected at the disposal stations. Substrata at both stations are sandy (< 2.0 mm), gentle sloping (<7.0%), and relatively shallow (< 0.9 m). Low abundances of larval squawfish at disposal stations (D1 and D2) may be related to interspecific associations with other fishes, or location in the reservoir. Abundances of larval, juvenile, and adult smallmouth bass were higher at disposal stations (D1 and D2) than shallow water reference station R11 (Bennett et al. 1991, 1993a, 1993b, 1995a, and 1995b).

Several habitat characteristics varied among years including water level fluctuations, water temperatures, and peak inflows (Kcfs) were associated with changes in larval fish abundance. High catches of *Catostomus* spp. and northern squawfish during 1991 coincided with lower mean water level fluctuations. Water fluctuations between 1 May and 13 August 1991 ranged from 0.0 m to approximately 0.23 m, whereas fluctuations during other years over the same period ranged from 0.04 m to 1.2 m. High catches for other larval fish families (Cyprinids) did not coincide with lower mean water level fluctuations suggesting other factors or combination of factors may regulate abundance. High larval catches of Centrarchids appear to be regulated by both water temperatures and water level fluctuations. During 1989, 1990, and 1992 high abundances of Centrarchids (occurring early August) coincided with higher water temperatures (> 25 °C) and variable water level fluctuations.

Our data suggests that several factors either acting independently or cumulatively affect larval fish abundance in Lower Granite Reservoir more than in-water disposal of dredged material. Having suitable habitat at a specific location in the reservoir may also be as important as other environmental factors in affecting larval fish abundance.

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Trends in Resident Fish Abundance Associated With Use of Dredged Material for Fish Habitat Enhancement

by

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Introduction

We evaluated trends in fish abundance associated with in-water disposal of dredged material in Lower Granite Reservoir, Idaho-Washington. Sediment deposition in the upstream end of Lower Granite Reservoir results in approximately 600,000 m^3 of sediments deposited at the confluence of the Snake and Clearwater rivers. As a result, flood control and navigation is maintained through periodic dredging. Disposal of dredged material has presented a problem in Lower Granite Reservoir due to limited availability of upland sites as a result of steep, talus cliffs surrounding the impoundment. Consequently, in-water disposal was proposed as an alternative and an experimental monitoring program began in 1985 to assess the value of using dredge material for fish habitat enhancement.

We sampled fish assemblages before (1985) and after (1993) construction of a 0.37 ha dredge disposal island to assess local changes in community structure. In addition, we monitored resident fish abundance for 5 years (1989-1993) near the disposal island and compared these patterns to fish abundance at other shallow water (reference) stations in the reservoir. Results were examined for evidence of negative impacts associated with use of dredged material as a habitat enhancement technique.

Study Area

Lower Granite Reservoir is a 3,602 ha impoundment on the Snake River, ID and WA. The reservoir is 51 km long with an average width of 643 m. Water depth averages 17 m and ranges from shallow, sandy shorelines < 1 m to a maximum depth of 42 m. Shallow water habitat (< 6 m) comprises less than 8% of the total surface area and occurs primarily in the upper end of the reservoir near the confluence of the Snake and Clearwater rivers (Bennett et al. 1990).

Methods

Disposal Island: Before versus After

We used data from electrofishing (no./5 min) and gillnet (no./hour) collections at river kilometer 192.4 to assess fish community composition before and after construction of a disposal island. We collected fishes along the shoreline using pulsed DC current from boat-mounted electrofishing equipment. Five to 15 minute electrofishing passes were usually made at weekly intervals from April-June, 1985 and 1993. Horizontal multifilament gillnets (68.6 m x 1.2 m with three graded panels 3.1, 4.1 and 5.1 cm) were used to sample deep water areas (> 3 m) from April-June 1985 and 1993.

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Fish Monitoring

In addition to examining fish communities before and after construction of a disposal island, we also monitored patterns in feeding guild abundance at six stations in Lower Granite Reservoir from 1989-1993. Four reference stations were selected to represent typical, shallow water habitat (< 6 m) found throughout the reservoir. Two disposal stations were selected immediately adjacent to the disposal island. We used electrofishing equipment to collect fishes at weekly intervals each spring (April - June, 1989-1993) along shallow shorelines of each station. We classified resident fishes into trophic feeding guilds based on published life history information (Scott and Crossman 1990) and followed the trophic classification scheme of Karr (1981).

Results and Discussion

Disposal Island: Before versus After

Ranked abundance of fishes collected along shallow shorelines (< 3 m) was similar 3 years before and 5 years after construction of the disposal island (Table 1). However, the fish assemblage associated with deep water habitat (> 3 m) changed significantly after construction of the disposal island. Of 13 common resident fishes in Lower Granite Reservoir, only six species were represented in gillnet catches prior to construction of the disposal island whereas after construction of the island, eleven species were represented in gillnet catches (Table 1). Construction of the disposal island effectively reduced maximum local water depth at the sampling area from 18 to about 8 m and may be

Table 1
Comparison of Ranked Fish Abundance From Electrofishing (Shallow < 3 m) and Gillnet (Deep > 3 m) Collections Before (1985) and After (1993) Construction of Shallow Water Disposal Island Using Dredged Material. Values in Parentheses Represent Proportion (%) of Catch

Species	Shallow		Deep	
	Before	After	Before	After
Chiselmouth	11 (15)	10 (6)	12 (16.9)	10 (6.9)
Carp	3 (0)	3 (0)	13 (57.8)	3 (1)
Bridgelip sucker	9.5 (12)	9 (4)	4 (0)	1.5 (0)
Largescale sucker	13 (30)	13 (51)	9 (4.8)	13 (61.7)
Brown bullhead	3 (0)	3 (0)	4 (0)	8 (3.2)
Channel catfish	3 (0)	3 (0)	8 (3.6)	12 (8.8)
Pumpkinseed	6 (0.3)	7.5 (2.1)	4 (0)	7 (1.8)
Peamouth	3 (0)	3 (0)	4 (0)	9 (7.8)
Smallmouth bass	12 (23.4)	12 (27.2)	4 (0)	9 (4.6)
White crappie	7 (1)	7.5 (2.1)	4 (0)	4.5 (1.4)
Yellow perch	3 (0)	3 (0)	4 (0)	6 (1.5)
Redside shiner	9.5 (12)	6 (1.1)	10 (7.2)	1.5 (0)
Northern squawfish	8 (6.5)	11 (6.5)	11 (9.6)	4.5 (1.4)

responsible for the observed shifts in assemblage structure (Schlosser 1982; Power et al. 1985; Chipps et al. 1994). Several warmwater gamefishes, including brown bullhead *Ameiurus nebulosus*, pumpkinseed *Lepomis gibbosus*, smallmouth bass *Micropterus dolomieu*, white crappie *Poxomis annularis* and yellow perch *Perca flavescens* were absent in collections made prior to construction of the disposal island. After construction of the disposal island, and subsequent reduction in maximum local water depth (to 8 m), all these species were present in gillnet collections implying an increase in species richness.

Fish Monitoring

Trends in feeding guild abundance were similar (repeated measures ANOVA; $P = 0.18$) at disposal and reference stations from 1989-1993 (Figure 1). We observed significant year to year variation ($P < 0.001$) in abundance of individual feeding guilds and postulate that reservoir-wide effects such as inflows, temperatures, and water level management may be more influential in regulating year-to-year variation in fish abundance than localized habitat characteristics. Results of our study suggest that construction of shallow water habitat using dredged material has potential for increasing habitat complexity in Lower Granite Reservoir. Islands created from dredged material reduce local water depth and can provide important rearing areas for several resident species, particularly warmwater game fishes.

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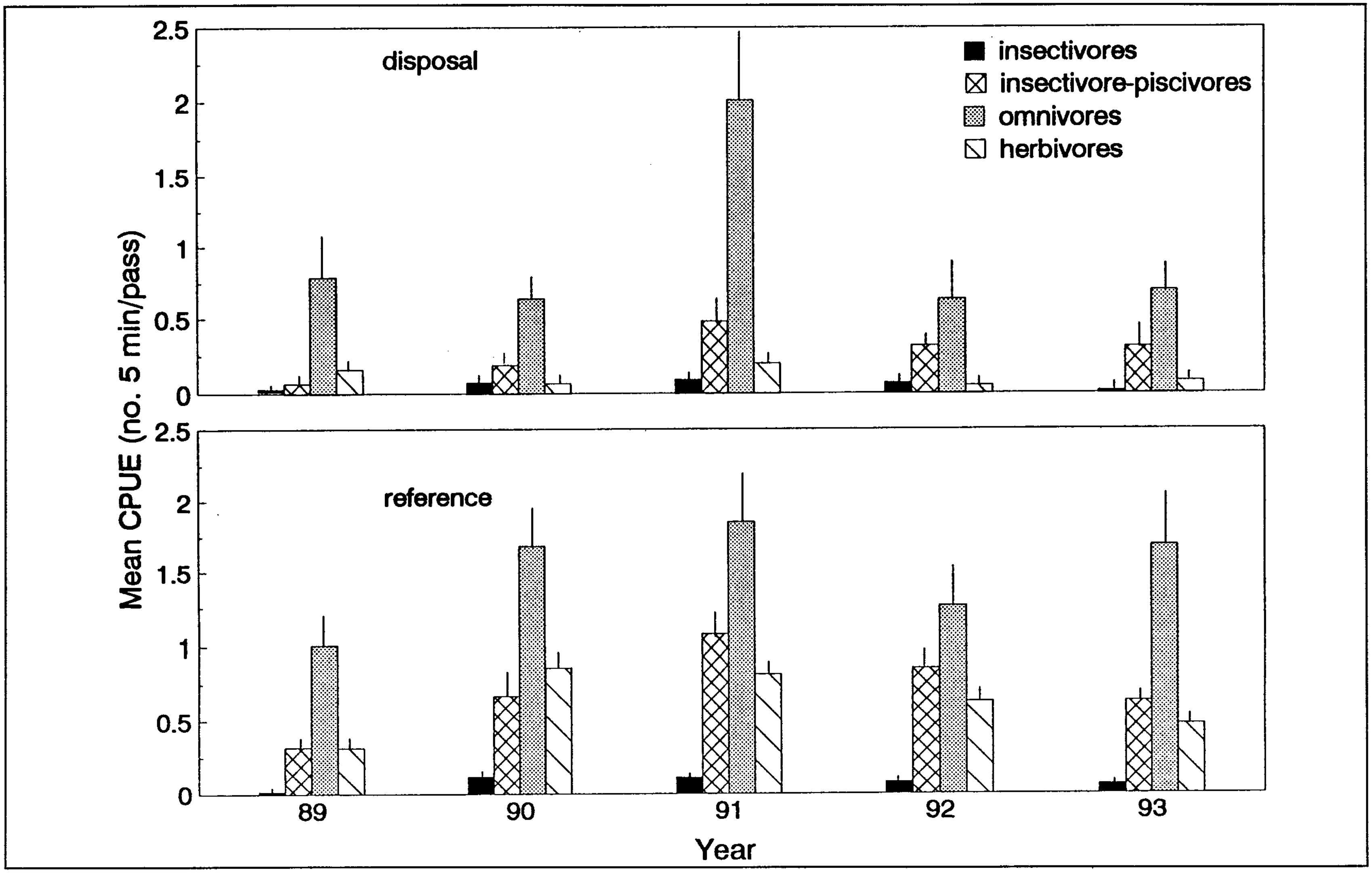

Figure 1. Trends in feeding quild abundance for fishes collected at disposal (top) and reference (bottom) stations in Lower Granite Reservoir, 1989-1993. Vertical bars represent ± 1 standard error of mean catch rates

Dredged Material Disposal and State Water Quality Standards

by
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Introduction

The Federal Water Pollution Control Act of 1972 (FWPCA) established, among other things, procedural requirements for determining compliance with applicable state water quality standards. Section 401(a)(6) of the FWPCA exempted Federal agencies from the state water quality certification process. The FWPCA was amended and renamed the Clean Water Act (CWA) in 1977. In those amendments, the Congress deleted the Federal agency water quality compliance exemption under Section 401(a)(6) and established a navigation protection provision for the Secretary of the Army in Section 404(t).

Following the 1977 amendments to the CWA, the Corps of Engineers (CE) published guidance requiring compliance with state water quality standards and instructed CE district offices on the appropriate course of action when a state and the CE could not reach an agreement or the state's requirements for compliance with Section 401 were determined excessive by the CE. In 1988, the CE codified into the Federal Register the essential components of that guidance.

Pursuant to the revision of 40 CFR 230 (the implementing regulation for the CWA) in 1975, the CE published a guidance manual (CE 1976) to assist CE Districts in determining compliance with state water quality standards in waters regulated by the CWA, although it was not yet required that the states certify that the standards had been met. This guidance consisted of the application of a numerical model with input variables being the numeric water quality standard, the numeric background concentration of the contaminant of concern, the concentration of the contaminant in a filtered elutriate, and the allowable mixing zone. The model has undergone major refinement over the years and is included in the 1994 draft revision of the manual (EPA/CE 1994).

Similarly, the ocean dredged material disposal guidance manual, first issued in 1977 (EPA/CE 1977) and revised in 1991 (EPA/CE 1991), uses the same model and approach to determine if the disposal of dredged material in the ocean meets Federal water quality criteria. The CWA and ocean disposal guidance manuals establish a Federal precedent dating back to 1976 with regard to the procedures to be followed in meeting either state water quality standards or Federal water quality criteria. Those manuals and the procedures established a process, still used today, to determine compliance with numeric state water quality standards.

Discussion

Early in 1978, the CE began to notice that some states were establishing non-numeric water quality standards. These non-numeric standards currently include goals, policies, designated uses, anti-degradation clauses, and other narrative standards, such as those dealing with odor,

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aesthetics, debris, etc. Obviously, it is not possible to use the numeric modelling approach to determine compliance with these non-numeric standards. Further, as these water quality standards were developed by the states, it became apparent that many of them were not always relevant to maintenance or enforcement of water quality standards. This led, in 1988, to the revision of the CE regulations for maintenance dredging and disposal activities at 33 CFR 209.145, as the new 33 CFR 335-338. This established, at 33 CFR 337.2, procedures for determining compliance with numeric state water quality standards.

Most recently, it has become evident that non-numeric standards have proliferated. Many states have now established a variety of non-numeric standards to address a number of state water quality issues. Because such standards are inherently subjective, they cannot be dealt with in a consistent and objective manner. Rather, they must be considered on a case-by-case basis. These non-numeric standards place the Federal project manager or a permit applicant in the position of having no quantitative standards to determine whether a project would be capable of water quality certification by a state under Section 401 of the CWA.

Non-numeric narrative criteria could be rendered quantitative by successful application of the numeric evaluation procedures of the 404(b)(1) guidelines or ocean discharge criteria. For example, where the state has a contaminant reduction "goal" for a specific chemical and a water quality standard for that chemical, the CE would be able to determine compliance using the numeric evaluation procedures. In another example, where the state has established an "aesthetic" standard and has uniformly and consistently applied measurable performance criteria, the CE would be able to use those performance criteria to comply with the non-numeric standard. However, where the state has established a narrative standard that applies on a categorical basis, such as a prohibition or policy against the discharge of dredged material in certain classes of waters, compliance cannot be determined by using numeric evaluation procedures.

The CE establishes a baseline or Federal standard for CE navigation projects as per 33 CFR 336.1(c)(1) and 335.7. The Federal standard requires that the least costly environmentally acceptable dredged material disposal alternative be selected using the evaluation procedures of the 404(b)(1) guidelines or ocean discharge criteria. The 404(b)(1) guidelines require compliance with applicable state water quality standards. Thus, compliance with the guidelines requires compliance with applicable numeric state water quality standards. The 404(b)(1) guidelines and ocean discharge criteria do not consider narrative water quality standards in the compliance process.

Several CE Districts have questioned the role of these non-numeric standards in establishing the Federal standard. Further, because of its technical expertise concerning the disposal of dredged material and its responsibility as the permitting agency for dredged material disposal under the CWA, CE personnel are often asked to guide and assist permit applicants with regard to meeting state water quality standards. Such guidance and assistance must be consistent with that applicable to Federal projects. Compliance with numeric state water quality standards has been straightforward until recently and has led to few, if any, problems or conflicts between the state over Federal projects, or permit applicants. Such is not the case for narrative standards.

An additional problem in complying with state water quality standards results from the standards of different states being applied to the same body of water. For example, four states (IL, IN, MI, and WI) exercise jurisdiction over southern Lake Michigan. All have different standards for this geographically small area. Hence, a given dredged material proposed for disposal in the same receiving environment is subject to significantly different water quality requirements. These requirements restrict the flexibility of disposal options and may lead to increased costs without appreciable environmental benefits.

The CWA requires that the CE consider mixing in determining compliance with state water quality standards. Unlike ocean dredged material disposal where the allowable mixing is specified by regulation (40 CFR 227.29), the individual states determine allowable mixing under the CWA. Some states allow no mixing, some specify allowable mixing, and others determine mixing on a case-by-case basis. As noted above in the case of southern Lake Michigan, adjacent disposal sites in different state jurisdictions may have vastly different water quality standards and allowable mixing. This makes neither environmental nor regulatory sense as dredged material, especially in dispersive sites, does not recognize state boundaries. There are instances where dredging takes place in one state and disposal in another. This has resulted in the project being "held hostage" over meeting standards by the state where disposal is proposed.

By withdrawing the water quality compliance exemption in 1977, the intent of the Congress was clearly to establish the right of the states to regulate water quality through the establishment and enforcement of water quality standards. In exercising this right, at least three states (NC, RI, and WI) have promulgated a blanket prohibition of open-water disposal of dredged material. This prohibition is administrative, as there is no technical justification for it. In the current atmosphere of downsizing the Federal government and the transfer of Federal programs to the states, it is likely that increased regulation of dredged material by the states will continue.

Conclusion and Recommendations

It is the policy of the CE that the CE will fully comply with State numeric water quality standards established under the authority of Sections 301, 302, 303, 306 and 307 of the CWA, where the evaluation procedures of the 404(b)(1) guidelines or ocean discharge criteria can be used to determine compliance.
In the case of narrative State water quality standards where there are no measurable compliance criteria, the CE should consider these on a case-by-case basis.

Regulatory guidance for this consideration is given at 33 CFR 337.1. Specifically, if a state imposes conditions or requirements which exceed those needed to meet the Federal standard and which have no technical basis, the CE may request that the state or local sponsor fund any additional costs. For permit applicants under these circumstances, the CE may deny the permit without prejudice and note that denial is based solely on the failure to meet non-numeric water quality standards.

The only long-term solution to the problem created by non-numeric water quality standards, allowable mixing, and absolute prohibitions of dredged material disposal would appear to be partnering with the states to reach accord. There is no reason why the processes and procedures in the manuals should not be adequate for this. However, the regulatory role of the states is clear and, if for non-technical reasons a state chooses to deny water quality certification under Section 401 of the CWA or otherwise prohibit disposal, the only recourse is 33 CFR 337.1.

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Technical Considerations for Sediment Quality Criteria

by
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Introduction

The U.S. Environmental Protection Agency (EPA) has recently proposed sediment quality criteria (SQC) under Section 304(a) of the Clean Water Act. Although SQC have not been implemented, the August 30, 1994 Federal Register notice for the contaminated sediments strategy recommends and encourages States to adopt SQC as water quality standards. This will result in the "de facto" imposition of SQC (U.S. Environmental Protection Agency 1994) as pass/fail criteria for dredged material because state water quality standards are by definition, pass/fail.

Three specific situations or reasons exist for evaluating sediments. The first is the necessity for disposal of navigation channel dredged sediments. For navigation channel sediments the only question to be answered is what unacceptable adverse effects, if any, will dredged material have in a particular disposal environment. EPA regulations at 40 CFR Parts 220-228 and 40 CFR Part 230 provide regulatory guidance on aquatic disposal of dredged material. The second reason to evaluate sediments is to determine what effects on aquatic ecosystems sediments may have if left undisturbed or if removed for environmental purposes. If sediments are determined to be exerting unacceptable environmental effects, consideration may then be given to some type of remediation, which may or may not include removal. If sediments are to be removed, the potential effects that these sediments will exert at the disposal site must be considered. The third reason, recently advanced by the EPA, is for source control of contaminants. Determination of locations where sediments, as sinks for contaminants introduced into the aquatic environment, are exerting unacceptable environmental impacts could lead to identification of the source of the contamination.

Sediment quality criteria and their potential uses have been discussed and critiqued in detail elsewhere (Chapman 1995; Iannuzzi et al. 1995; Lee and Jones-Lee 1995; Landrum 1995; Gaudet et al. 1995; Brannon et al. 1990). However, the use of SQC in a pass/fail mode and the potential impact on dredging and dredged material disposal raise many questions about the appropriateness of using SQC in such a manner. This paper explores the assumptions and uncertainties inherent in the SQC rationale and focuses on the impacts these will have if the SQC are used as pass/fail standards.

Single Chemical Criteria in Sediments With Mixed Contaminants

Sediments that the Corps of Engineers (CE) must dredge to keep ports, harbors, and waterways open to navigation are generally subject to multiple sources of contamination. Sediments containing a single contaminant are rare. The likelihood that sediments will contain contaminants for which criteria do not exist is high. For example, SQC are based upon only a few hundred water quality criteria (WQC) when approximately 65,000 chemicals are known today and approximately 1,000 new ones are developed each year (Lee and Jones-Lee 1995).

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Therefore, the chances are slim that a relatively few chemical specific criteria can protect the environment in multi-chemical sediments. Chemical contaminants that can cause environmental problems will go unregulated because of a lack of numeric concentration criteria (Lee and Jones-Lee 1995). The number of criteria available will always be limited (Landrum 1995), but the environmental safety of chemicals for which no criteria have been established is never guaranteed. The environmental effects of sediment contaminants for which SQC do not exist will always be an issue, even if the sediment is below all existing numeric SQC. This may result in toxicity testing requirements even if all applicable SQC are passed.

Single chemical criteria also fail to address the additive, antagonistic, or synergistic effects on toxicity of multiple contaminants in sediments. This may result in unexpected environmental effects or lack thereof, depending upon the interactions of the mixed contaminant suite. The only way to presently evaluate such situations is with whole sediment, effects based testing. The presence of multiple contaminants will always raise questions about the conclusions reached as a function of SQC numbers. If a sediment passes, questions will remain regarding the toxic effects of multiple contaminants or the effects of additional compounds in the sediment for which criteria do not exist. This will probably result in additional testing, even after the cost of obtaining the information for SQC evaluations. The application of single compound SQC to a sediment containing multiple contaminants will be conclusive when the sediment fails to meet the criterion. Passing will tend to raise further questions about the potential toxicity of the sediment that can only be answered by effects based testing. This can only add cost and complexity without necessarily increasing environmental protection.

SQC Versus Effects Based Testing

The SQC are claimed to be more protective of the environment than effects based testing. This assertion results from basing SQC numbers on chronic water quality values and the fact that present effects based testing measure mortality. Purportedly, SQC use more protective numbers and therefore provide more protection than is possible with acute bioassays. However, in the effects based sediment bioassays, test species are exposed in the laboratory to sediments proposed for disposal and the response of the organisms is evaluated in relation to reference organism responses in terms of a specified biological endpoint such as mortality, bioaccumulation, growth, reproduction, etc. The precise chemical composition of sediments need not be known to use this approach as it is a "whole sediment" test. Chemicals that may go undetected analytically or for which SQC do not exist are covered in the bioassay approach. Potential biological effects are, therefore, appropriately estimated because dredged sediments are managed on a "whole sediment" basis.

The advantage and logic of the bioassay approach is that the effect of sediment contaminant interactions such as synergism, additivity, and antagonism are integrated by the response of the organisms. In addition, this approach is consistent with the regulatory requirements and methods used to develop water quality criteria (effects-based bioassays), and thus render the approach and rationale technically acceptable and defensible (Chapman 1989). The sediment-specific nature of the bioassays also addresses many of the limitations of SQC such as inability to account for the effects of multiple contaminants and the varying bioavailability of sediment contaminants that will be explored in more detail later in this discussion.

The advantages of the sediment bioassay approach include ease of use, relative cost effectiveness, flexibility, widespread acceptance, and compliance with regulatory requirements. Sediment bioassays were adopted for use under EPA regulations specifically because this approach addresses considerations set forth in the cited regulations for dredged material (Engler 1988). Acute toxicity sediment bioassays are an effects-based approach that takes into account the fact that sediment is a complex mixture of substances whose potentially interactive effects cannot be predicted through chemical analyses. Bioassay methods can also be used to evaluate the effects that sediments exert in situ as well as to identify suspected sources of sediment pollution.

Most criticism of the present regulatory approach to evaluating sediment for dredging and disposal centers on the acute toxicity bioassay endpoint, which is usually mortality. However, active development of existing bioassays to include sublethal effects is ongoing. The approach has also been criticized because it does not identify the causative agents of observed effects and therefore, cannot be used in establishing limits for contaminants in discharged materials.

Contaminant and Organic Carbon Source

Recent research raises many questions about the universal applicability of the equilibrium partitioning SQC approach for predicting sediment pore water concentrations. Both the form of the contaminant and the organic matter have been shown to affect partitioning in ways that are not easily predictable from sediment organic carbon and contaminant concentrations. For example, a growing body of evidence shows that a large fraction of the total polyaromatic hydrocarbon (PAH) concentration in sediments is unavailable for partitioning into pore water due to their inclusion in combustion-derived particles (McGroddy and Farrington 1995; McGroddy, Farrington, and Gschwend 1995). For example, McGroddy and Farrington (1995) reported that only 0.2%-5% of sediment phenanthrene concentrations appeared to be readily available to partition into porewater. Sediment pyrene was more variable, but the amount available for partitioning into porewater ranged from 5% to 70% (McGroddy and Farrington 1995). These results indicate that predicted pore water concentrations of these PAH compounds would be greatly overestimated because sediment analysis includes forms of the contaminant that are unavailable for partitioning. This limitation on PAH bioavailability was demonstrated by McFarland and Ferguson (1994) who found that the biota/sediment accumulation factors (BSAF) of PAH compounds in shellfish exposed to contaminated sediments were about twenty-fold lower than the BSAFs reported for chlorinated compounds. These findings demonstrate that organic carbon normalization cannot accurately predict pore water PAH concentrations in sediments containing combustion derived PAHs when used to calculate SQC. Use of SQC in a pass/fail mode in such sediments will greatly overestimate the impacts of sediment PAHs on biota.

Equilibrium partitioning relationships developed for PAHs and sediment organic carbon were based on adsorption studies where all the contaminant added was available for partitioning. However, Burford, Hawthorne, and Miller (1993) found none of the spiking procedures they investigated were able to accurately represent the native analytes, which were normally bound much more strongly by the environmental matrices than the spiked analytes. Even under the conditions in which these partitioning relationships were developed, divergence occurred. Brannon et al. (1993, 1995) reported marked deviation of measured K_{OC} values from values predicted by empirical relationships for PCBs and fluoranthene. Puschel and Calmano (1995) reported that even for sorption, individual PAH compounds behaved differently, displaying behavior that could not be explained by either organic carbon normalization or the use of a reference compound. Measured and estimated values of K_{DOC} for fluoranthene in pore water also diverged.

The nature of sediment organic carbon can also affect partitioning of nonpolar organic contaminants. Evidence suggests that the aromaticity of organic carbon affects partitioning (Grathwohl, 1990; Garbarini and Lion, 1986; Gauthier 1987), with K_{OC} values increasing as the aromaticity of sediment organic matter increases. However, Davis (1993) did not observe a linear relationship between K_{DOC} and the fraction of aromatic carbon. This result implies that partitioning of nonpolar organic contaminants to sediments may be dependent on more than the nature of organic carbon. Kile et al. (1995) supported Davis's findings when they reported that the K_{OC} values for carbon tetrachloride and 1,2-dichlorobenzene were relatively invariant for either bed sediments or soils, with the values on bed sediments averaging about twice those on soils.

Rebhun et al. (1992) investigated partitioning in a system consisting of 10 percent clay, 90 percent sand, and various concentrations of adsorbed humic acid. Until the concentration of humic acid reached approximately 0.5 percent a linear relationship existed between the partitioning coefficient and the fraction of humic material. This result is in keeping with the kinetics studies of McFarland et al. (1996) who partitioned PCB-52 in a fish/water/sediment system involving sediments of differing organic carbon content. In those studies the constancy of the partitioning relationship between sediment and organism became unstable below 1.0 percent organic carbon and failed entirely at 0.33 percent. However, the technical documentation supporting SQC (DiToro et al. 1991) claims that SQC are valid at organic carbon concentrations as low as 0.2 percent in sediments.

Many other factors also affect K_{OC} and K_{DOC} in sediments. These include pH and ionic strength (Schlautman and Morgan 1993; Means 1995), surface reactions (McGinley et al. 1993), and competitive sorption behavior with mineral-bound humics (Murphy et al. 1990). For example, Means (1995) showed that sorption of pyrene increased as salinity increased, resulting in lower solution concentrations and K_{OC}s that were a factor of two higher in saline than in freshwater.

Use of Water Quality Criteria SQC

The use of chronic water quality criteria as the basis of SQC numbers has been subject to criticism by the CE and others (Lee and Jones-Lee 1995; Iannuzzi et al. 1995; Chapman 1995). Misgivings on the use of chronic water quality values as the limit for pore water contaminant concentrations has focused on the uncertainty inherent in chronic values, the limited toxicological database for benthic organisms, the limited availability of chronic water quality values, and the lack of direct chronic tests to demonstrate that the criteria are appropriate under chronic conditions. All the criticisms are valid and add to the uncertainty inherent in the use of SQC as pass/fail numbers. These results indicate that too little is understood about equilibrium partitioning to rely upon it for pass/fail values. Little certainty exists that site specific considerations can be ignored without compromising the accuracy of conclusions about potential environmental impacts.

Uncertainty in SQC

As the previous discussion has revealed, a high degree of uncertainty exists in the many factors used to derive SQC. Details of the uncertainty inherent in SQC are not discussed in this paper because of space limitations. However, the EPA has acknowledged uncertainty in measurement of octanol/water partition coefficients (K_{OW}) which are used to derive K_{OC} (Environmental Protection Agency 1993) for SQC. This is insufficient because K_{OW} is just one of the many sources of uncertainty inherent in SQC. In public forums, an uncertainty factor of two has been acknowledged for SQC (DiToro 1995). Such uncertainty, which is probably much too low, is still too much to use SQC on a pass/fail basis.

Conclusions

Use of SQC in a pass/fail mode is an approach that has not generated enthusiasm in either the scientific or regulatory community. The general consensus is that SQC be used as screening tools and not as pass/fail criteria (Iannuzzi et al. 1995: Chapman 1995; Landrum 1995; Brannon 1995). However, site specific effects-based evaluations are preferred for regulatory decision making(Iannuzzi et al. 1995; Lee and Lee-Jones 1995; Brannon et al. 1990). Chapman (1995) recommended using numerical criteria as screening tools, but also provided warnings that we would be wise to heed. He considered numerical criteria almost certain to be misused, no matter how many caveats are provided as to their proper use because of the hunger that exists for numerical guidance. He concluded that "We must recognize this and ensure that numbers cannot be used in any way, shape or form for definitive stand-alone decision-making." Use of SQC as other than clearly defined screening tools will lead to abuse that will cost this nation dearly without a corresponding increase in environmental protection.

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Wilmington Harbor Ocean Bar, Channel Deepening Project Wilmington Offshore Fisheries Enhancement Structure

by
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Introduction

Beneficial uses of dredged material refers to the concept that a byproduct of navigation dredging can be used in a way that is economically and environmentally acceptable and accrues natural resource benefits to society. There are over 30 Federal laws and Presidential Executive orders applicable to beneficial use activities (EM 1110-2-5026). It is the policy of the U.S. Army Corps of Engineers to use dredged material beneficially within existing authority and funding (33 CFR Part 337.9). However, innovative, beneficial uses of dredged material may be difficult to achieve for several reasons including, costs and the constraints of cost sharing partners, technical constraints, and existing resource interests. This paper will discuss the approaches taken to overcome these obstacles to beneficial uses of dredged material during construction of a Federal navigation project. The dredging of substantial quantities of rock from the Wilmington Harbor Ocean Bar Channel in southeastern North Carolina provided a unique opportunity to use dredged material beneficially.

Wilmington Harbor Ocean Bar Channel Project

The Wilmington Harbor Ocean Bar Channel Deepening Project involves the removal of approximately 1.6 million cubic yards of material to achieve authorized project dimensions not achieved previously due to rock obstructions and survey inaccuracies. About 1.0 million cubic yards of the material to be dredged is rock, while the remainder is a mixture of sand, silt, clay, and shell fragments. Samples of the rock indicate four different rock types; all of them are fossiliferous limestones. Dredging is being accomplished by a hydraulic pipeline dredge with a rock cutterhead. The rock cutterhead breaks and grinds the rock into pieces that can be lifted hydraulically into a scow moored alongside the dredge. The resulting dredged material is predominantly golf ball to softball sized rock pieces mixed with sands and smaller pieces. Some rock pieces are as big as volleyballs. The scow transports the dredged material to the disposal location. Some cleanup and maintenance dredging is accomplished by a hopper dredge.

Conventional Disposal Plan

The available dredged material disposal plan was to place all materials from the ocean bar channel deepening project within the Wilmington ocean dredged material disposal site (ODMDS) (Figure 1). The rock material was to be segregated into one quadrant of the ODMDS. The maintenance material was to be placed in another portion of the ODMDS.

Two concerns surfaced during review of the proposed ODMDS disposal plan. The first was a concern that Wilmington ODMDS as a disposal area "resource" would be affected by

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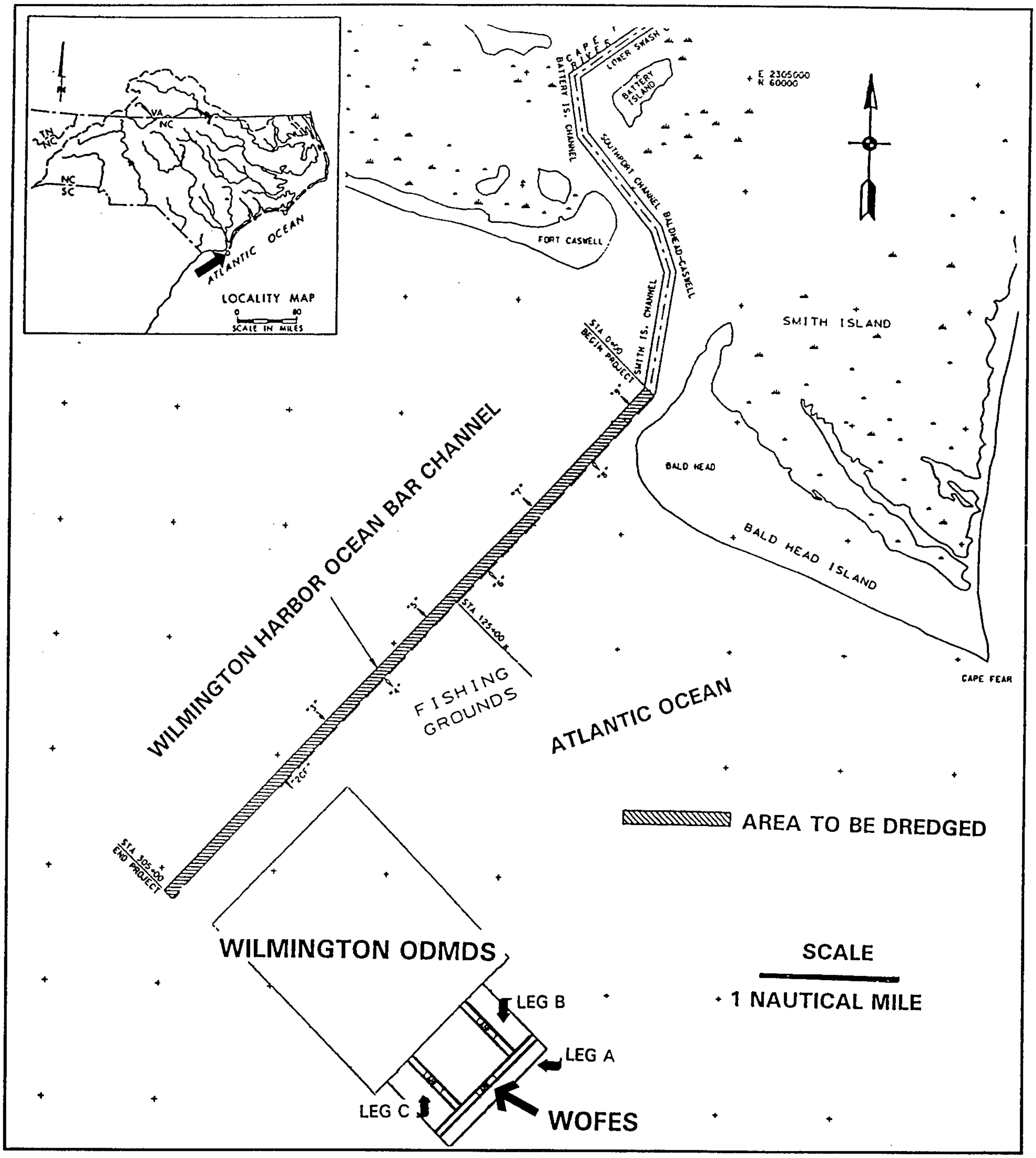

Figure 1. *Wilmington Harbor ocean bar channel deepening project*

expending of capacity. Projected use of the Wilmington ODMDS for the next five years from existing and foreseeable navigation projects is approximately 21 million cubic yards. Site capacity was already an issue of concern and the placement of large quantities of non-dispersive rock in the site heightened those concerns further. The second was the concern that placing substantial amounts of rock in the ODMDS would promote the development of hard-substrate habitat within the site. Either of these effects could have adverse long-term consequences. Colonization of the hard-substrate by marine organisms could create a resource that is not common to the area. Conservation of the newly created hard-substrate resource could restrict all or part of the ODMDS from use. Similar restrictions caused by habitat changes have occurred off Tampa, Florida (EPA 1993).

Beneficial Use Disposal Plan

The concerns for problematic creation of hard-bottom habitat led to the question, could the rock dredged from the navigation channel be purposely and beneficially used to make hard-bottom habitat? Two beneficial use alternatives were considered: augment an existing artificial reef or create a new reef or fisheries enhancing structure. The existing reef alternative involved a reef in the North Carolina Artificial Reef Program (NCARP). NCAR420 is approximately 2.9 nautical miles west of Wilmington Harbor Channel Buoy 4. This site contains barges, a bridge span, and concrete pipe. However, concerns by NCARP administrators for fine-grained materials which may be incidentally dredged and placed along with the rock adversely affecting existing reef life caused that plan to be dropped from further consideration. The new structure plan involved construction of a bathymetric anomaly to provide habitat and attract fish. It is believed that the rock dredged material would provide excellent marine habitat material because of its durability and stability, rugosity, the habitat complexity it would provide, and its availability.

Wilmington Offshore Fisheries Enhancement Structure (WOFES)

The plan developed was to use rock containing dredged materials from the Wilmington Harbor Ocean Bar Channel Deepening Project to create an offshore bathymetric feature as shown on Figure 1, hereafter referred to as the WOFES. The existing sand substrate and flat bathymetry would be replaced in the WOFES area by a significantly different rock-substrate and large scale, high vertical relief and "bumpy" topography. The WOFES site is in an area immediately to the east of the Wilmington ODMDS. Dredged materials used for the WOFES are to be new work materials to the extent practicable. Dredged material that are solely channel maintenance materials are not to be used for the WOFES. Dredged materials are to be placed within specified lanes to create a linear mound with up to three "legs" designated as A, B, and C. The targeted minimum average mound crest is 12 feet above the bottom. The maximum crest will be that which maintains a minimum vertical clearance of 25 feet at mean lower low water (m.l.l.w.) or a crest elevation of about 15 to 21 feet above the bottom. Charted depths at this location range from 40 to 46 feet below m.l.l.w.

The WOFES is designed with attributes and features that would provide habitat and attract fish. Physical factors believed to optimize fisheries utilization include material used, shape, orientation, vertical relief, side slopes, and total size. Rock and a mixture of rock and finer grained materials should provide habitat complexity and structure stability. Dr. Doug Clark of the U.S. Army Corps of Engineers, Waterways Experiment Station (WES), Environmental Laboratory, proposed a "U" shaped design to create a harboring effect from currents, allowing fish a "lee" area to occupy. A three legged "U" shape maximizes this effect and is planned. However, a two legged "L" and the single linear shapes also offer similar function but less than the "U" shape should the "U" not be achieved because of the physical properties of the dredged materials, placement factors, and site conditions. Leg A, the first leg to be built, is designed to be perpendicular to dominate current direction. The vertical relief of the WOFES is designed to rise above the seabed as much as possible. Vertical height or relief is constrained by a minimum required vertical clearance of 25 feet m.l.l.w. for navigation purposes. Providing for maximum side slopes is a design feature. The designation of deposition lanes which are as narrow as practicable will help provide both maximum vertical relief and side slope steepness. With respect to total size, fish are attracted to bathymetric anomalies. Accordingly, the WOFES design is to construct as big a feature as possible with the rock material available.

Factors Shaping the Beneficial Uses Plan

The idea of using rock dredged from the Wilmington Harbor Ocean Bar Channel to intentionally and beneficially modify marine habitat was shaped and defined by several factors. These factors included policy issues, costs and constraints imposed by cost sharing partners, technical issues, and existing resource interests. These factors are not independent of each other and can make beneficial uses of dredged material difficult to achieve.

Policy

As discussed previously, it is the policy of the Corps of Engineers to use dredged material beneficially within existing authority and funding. Accordingly, organizational support for the proposed activity was present. Locally, there was significant pressure to get on with the work of deepening the ocean bar channel. Initiatives perceived as potentially delaying the work could not be supported. The WOFES planning did not delay project and therefore was supported.

Other important policies included those contained in the National Artificial Reef Plan (Stone 1985) and the North Carolina Artificial Reef Master Plan (NCDMF 1988). The use of dredged material to build a structure for enhancing marine habitat and fishery resources would be in essence creating an "artificial reef." The National and North Carolina Artificial Reef Plans provide standards for siting, construction, monitoring, and management of the structure. The WOFES design was made consistent with national and State artificial reef standards and policies to the maximum extent practical. With regard to ocean dumping policies, EPA, Region IV supported the WOFES as a management option to use of the Wilmington ODMDS. EPA determined that the proposed WOFES placement was not transportation for ocean disposal of dredged material, a Section 103, Marine Protection, Research, and Sanctuaries Act of 1972 (MPRSA) action and the dredged material was acceptable for reef construction material.

Costs and Constraints Imposed by Cost Sharing Partners

Costs can be substantial obstacles to beneficial uses. There was a significant concern that costs would prevent the WOFES from being implemented even if it was a good idea. The Corps and their cost sharing partners such as States and ports authorities must maximize the value of every dredging dollar within a framework of environmental goals. The resource values added by beneficial uses of dredged materials, while of value, may not be considered in the same context as the prime objective of maintenance of navigable channels. There is a need to specifically identify beneficial use costs to facilitate decisions regarding those alternatives.

The concern for cost notably affected the WOFES plan in two ways. First, the WOFES was made as simple as possible. Technical aspects were designed with the idea that costs had to be kept to a minimum or else the project would not be achievable. For example, the WOFES was sited as near as possible to the conventional disposal alternative. Additional operational requirements for the WOFES placement were kept to a minimum. Secondly, the WOFES was included in the competitive bidding process. The disposal of dredged material at the WOFES was established as an option in the contract specification bidding schedule. WOFES would be bid separately but as a part of a total project contract. Bidders were required to quote on performing the work using the base of the disposal within the Wilmington ODMDS and then provide a separate bid on the WOFES option. The Government evaluated the bids by adding the total price for the options and the total requirement. Evaluation of the option did not obligate the Government to exercise the option. The excavation of rock and other sediments was expected to be the largest bid item.

The awarded contract price for the WOFES was $14,000. The up-front knowledge of these costs and their amount helped the Wilmington Harbor project's cost sharing partners be supportive of the WOFES.

Technical Constraints and Issues

Several technical issues developed during project planning and coordination and addressing these issues was fundamental to the development of the WOFES. In summary, the issues raised regarding the WOFES were as follows:

a. Stability of the proposed rock structure, movement of the rock materials to nearby trawling areas could adversely affect fishing.

b. Fisheries resource potential of the proposed WOFES.

c. Construction monitoring and supervision, implementation of the proposed plan.

d. Pre- and post-construction biological and physical monitoring.

In addressing these, the Wilmington District used a multi disciplined team. The District components of the team provided a local knowledge base and WES's, Coastal Engineering Research Center and Environmental Laboratory, provided coastal engineering and fisheries biology expertise.

Stability analyses suggest that the WOFES will be stable in the project area wave climate during a typical year (Pollock 1994). The highest waves during a typical year may cause the crest to flatten and material dislodged from the WOFES crest would be expected to migrate to the base, decreasing the height and increasing the base width. Stability then increases with greater water depth and the rock material is not expected to be moved from the project area. Thus, traditional trawling (commercial fishing) bottoms nearby and other existing uses of the marine environment would not be adversely affected. Concerning fisheries potential, the WOFES was designed with attributes and features that would provide habitat and attract fish. The existing sand-substrate and flat-topography will be replaced in the project area by a significantly different rock and rock-mixture (dredged material) substrate and large scale, higher relief, bumpy bottom. Because of the substrate and topography change, a change in biological communities of the area can be expected to occur (Clark 1994). The rugosity or crevices of the rock, dredged material, substrate will support epifauna and infauna that are different from existing sand-bottom benthic assemblages. These changes may shift the forage base of the project area and, as a result, change the higher trophic level structure. The relief or vertical aspects of the WOFES will alter current patterns in the area. Hydrodynamic lifts or current shadows down current of the structure may affect the distribution of plankton and the feeding on those organisms by subsequent trophic levels. As existing fisheries resource utilization, such as the traditional trawling areas are not expected to be adversely affected and marine habitat diversity and complexity will likely be increased, the impacts to fisheries resources are not considered adverse. However, the fisheries resource potential of dredged material structures have not been adequately documented (Clark et al., 1988). The WOFES will likely have a beneficial effect on recreational fishing. The location is close enough to the Cape Fear River Inlet and shore to provide recreational access. Fish species that will likely utilize the WOFES are desirable recreational catch. Construction management and post-construction are issues of commitment and trust as much as they are technical issues. Environmental enhancement has become a stated contractual goal of both the dredging contractor and the Corps of Engineers. Post-construction monitoring including appropriate biological and physical monitoring became a District commitment.

Existing Resource Interest

The WOFES site is in the Atlantic Ocean more than three nautical miles offshore near the Wilmington Harbor Ocean Bar Channel and the Cape Fear River Inlet. The area is important to several groups including recreational fishermen, commercial fishermen, and shipping interest. For example, traditional trawling grounds, principally for shrimp, occur near the project area. Loosing trawlable bottoms is a big issue for the shrimp fishermen. They see all the artificial reefs, the ocean disposal area, and other things as threats to their livelihood. The influences of the Cape Fear River estuary cause this area to be very important to both the local commercial and recreational fishing industry. The ocean bar channel leads to the river channel and the Port of Wilmington. A successful beneficial use project must consider existing resources of the area and the users of those resources, minimize adverse impacts to those resources, and do as much as possible to make those users view the project from a positive viewpoint.

An effort was made to first understand the concerns of the existing resource users then address them through a thorough public involvement process. The beneficial use of rock dredged from the Wilmington Harbor Ocean Bar Channel was discussed during several meetings with resource interests. For example, the WOFES plan was discussed during the Wilmington District's Interagency FY94 Dredging Coordination Meeting. This annual meeting brings together Federal, State, and local agencies and groups that have an interest in the District's dredging program. In addition, the WOFES was discussed with agencies and individuals with specific interest in the WOFES and the area planned for its construction. With the help of the local Sea Grant Advisory Agent, the opinions of local commercial and recreational fishermen were obtained. These persons supplied firsthand knowledge of the area, such as trawling areas that was invaluable. The public participation continued through the National Environmental Policy Act (NEPA) process. Without question, the public involvement shaped and improved the WOFES project and simplified its environmental approval.

WOFES Construction To Date

The *R.S. Weeks*, a hydraulic cutterhead dredge began work to deepen the Wilmington Harbor project on October 1, 1994. The rock cutterhead dredge discharge is directed to a "T" shaped discharge manifold on either side of the dredge where a dump scow, usually a 4,000 cubic yard scow, is moored. The dredge filled the scow, overflowing them until an economic load was obtained. The scow was then taken to the WOFES area and dumped. The tug captain has been provided centerline coordinates for each WOFES leg and is generally given a 2,000-foot section of the leg as a target. The tug captain is required to remain steady on course with the long axis of each lane during the discharge of material. The dump locations are recorded for each load and made a part of the contractors daily reports to the U.S. Army Corps of Engineers. Periodic WOFES surveys were performed by the contractor and the Corps to assess the WOFES development and direct the placement of additional material. As of January 31, 1996, 377 scow loads have been placed in the WOFES, Leg A, area. Additionally, 68 hopper dredge, *Padre Island*, loads have been placed in the WOFES, Leg B, area. The hopper dredge was used to clean up material that the cutterhead had fractured but not removed. It is estimated that the work is between 60 and 75 percent complete. Project completion is expected in the summer of 1996.

The dredging has proven to be challenging for several reasons including weather, the exposed open ocean and inlet location, and the hardness of the material to be dredged. During most producing days only one or two scow loads were placed on the WOFES. Equipment or weather delays resulted in long periods when no loads were placed.

The results of the placement of Wilmington Harbor channel deepening dredged materials at the WOFES site is illustrated in Figures 2-5. Figure 2 is an April 1994, preconstruction, survey of the WOFES site. The proposed WOFES plan, Legs A, B, and C are shown. Figure 3 is a January 11, 1996, survey of the WOFES site. Also shown on Figure 3 are the reported "begin dump" locations for all the scow loads dumps to date. A comparison of the pre- and during construction hydrographic surveys was made using surface modeling techniques. Figure 4 is a contouring of the differences between the pre- and during construction surveys (surfaces). Figure 5 is a cross-section through the WOFES at the specified location shown on Figure 3.

These data indicate that a bathymetric feature is being built at the WOFES site. Rock has been placed along specified legs. The targeted crest elevation of 12 feet above the bottom has been achieved along portions of Leg A. Side slopes of between 1V:20H and 1V:15H are common. Most of the dredged material is located within 300 feet on either side of the centerline of the WOFES leg (Leg A). However, an apron footprint extending 500 feet or more from both sides of the centerline is seen.

Summary and Conclusions

The concerns that rock materials dredged during the Wilmington Harbor Ocean Bar Channel Deepening Project and disposed of in an ODMDS may encourage colonization hard-bottom organisms and significantly reduce site capacity led to a plan to intentionally use of rock dredged material to create hard-bottom marine habitat elsewhere. While the idea of beneficial uses receives support in terms of Corps policy its application requires substantial coordination, planning, and good fortune. Significant factors that shaped the WOFES were costs and constraints imposed by cost sharing partners, technical issues, and existing resource interest.

The WOFES is being carried out as planned. The WOFES was designed with attributes and features that would provide habitat and attract fish. Factors in the design to optimize fisheries utilization include material used, shape, orientation, vertical relief, side slopes, and total size. The features seen in place, to date, appear to fulfill these design elements.

As dredging of large quantities of rock is likely not common in the project area, beneficial use of those materials should be considered when those materials are encountered. Based on the WOFES experiences, to date, a fisheries enhancement structure constructed with rock dredged material is an acceptable alternative if costs, technical issues, and impacts to existing resources are carefully considered. Monitoring of the response of marine organisms to the WOFES structure remains to be accomplished and should provide useful information applicable to similar dredged material uses in the future.

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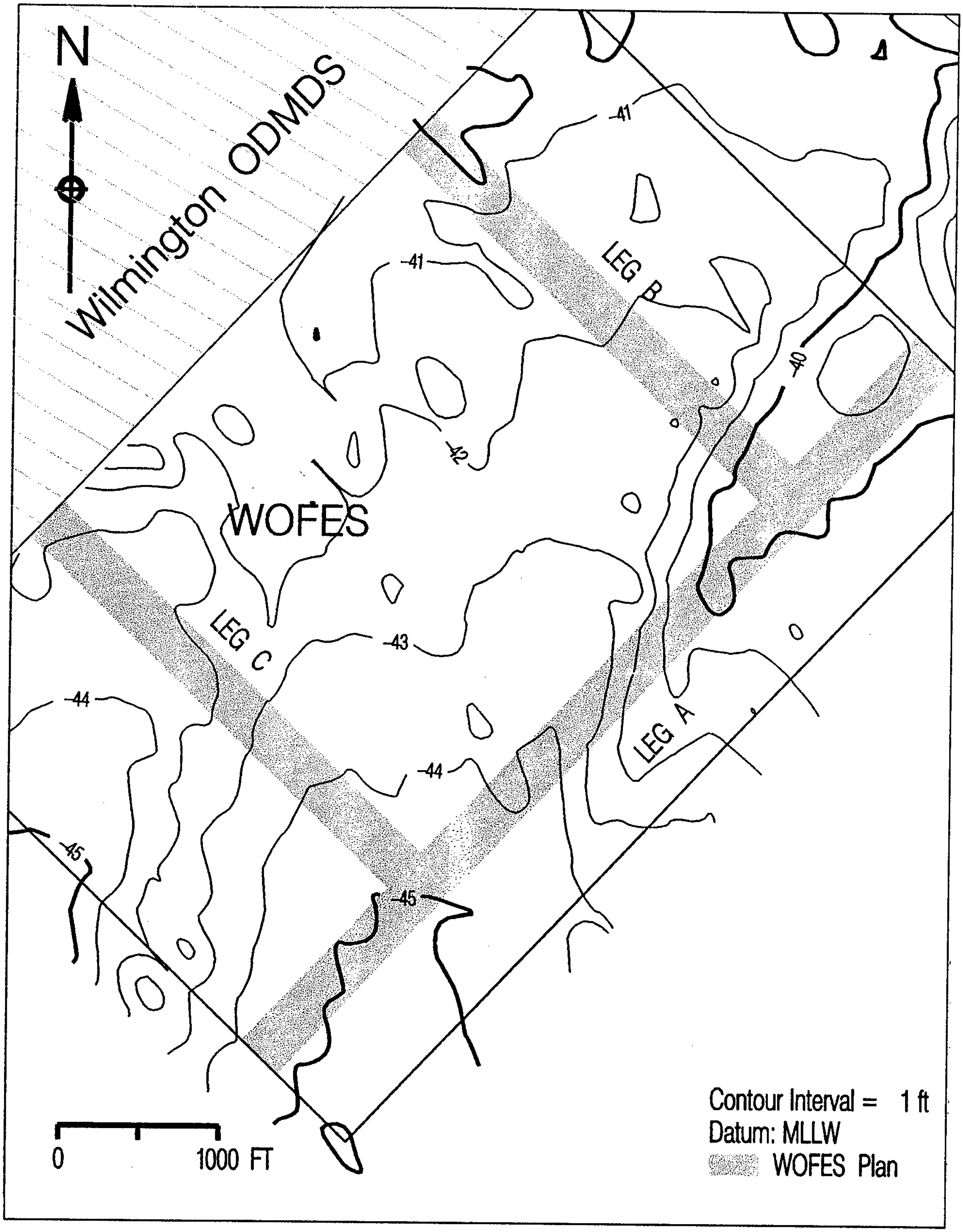

Figure 2. WOFES area pre-construction survey, April 1994. Proposed WOFES plan shown

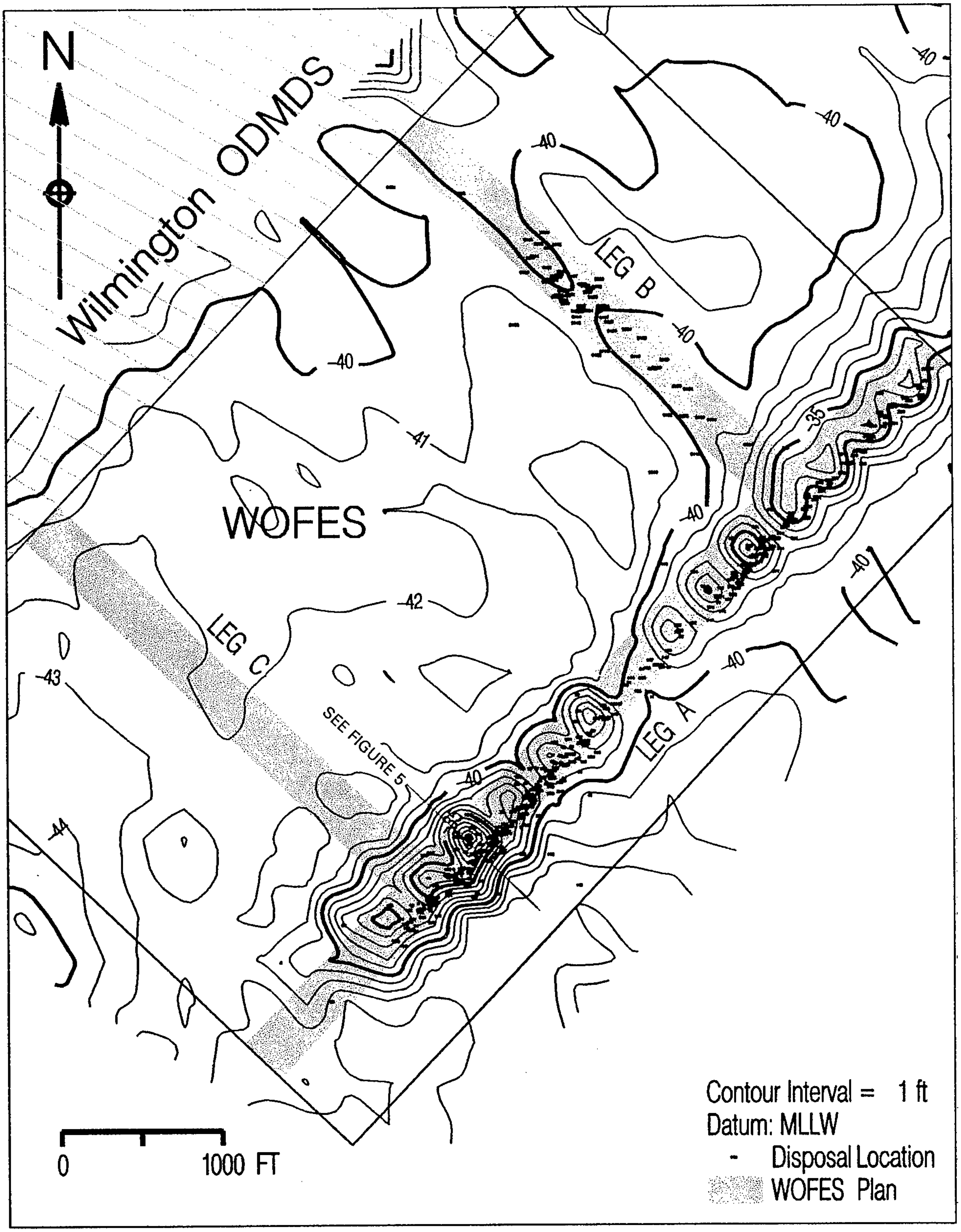

Figure 3. WOFES area during-construction survey, January 1996. Proposed WOFES plan shown. Begin dump locations shown

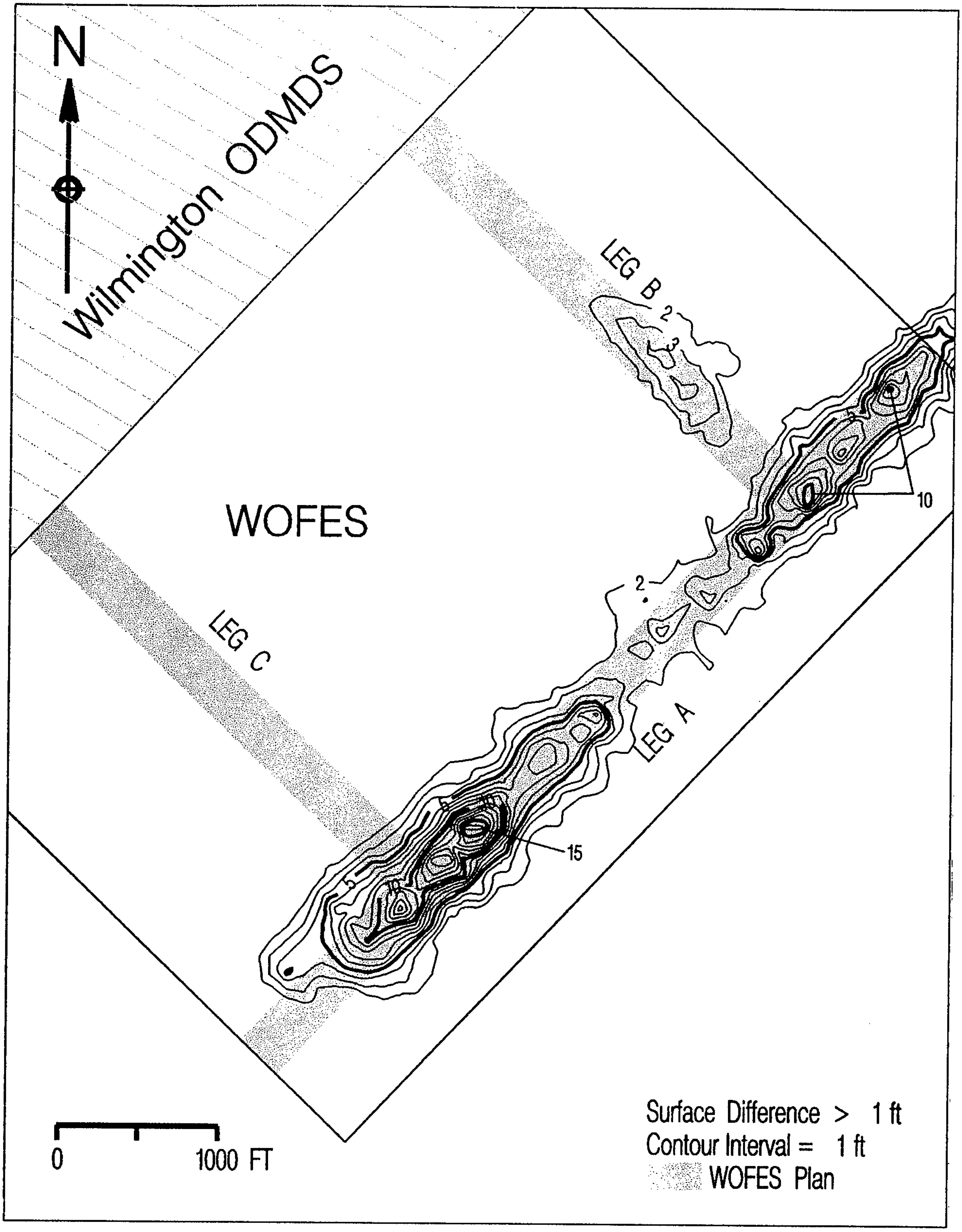

Figure 4. WOFES area contours of differences greater than 1 foot between surfaces created from April 1994 and January 1996 surveys

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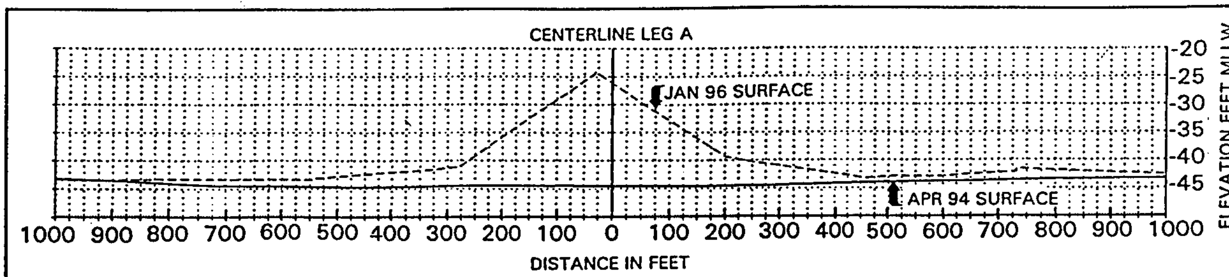

Figure 5. WOFES cross section comparing April 1994 and January 1996 surfaces. See Figure 3 for location.

Puget Sound Dredged Disposal Analysis Program: An Interagency Approach to Sediment Management

by

Stephanie Stirling,[1] *Therese Littleton,*[1] *David Kendall,*[1] *Justine Barton,*[2] *John Malek,*[2] *Rachel Friedman-Thomas,*[3] *Tom Gries,*[3] *and Ted Benson*[4]

The management of clean dredged material and open-water disposal in Puget Sound is accomplished through the Puget Sound Dredged Disposal Analysis program (PSDDA). This program is a cooperative venture of the four agencies with jurisdiction over dredging and disposal (Corps of Engineers (COE), Environmental Protection Agency (EPA), Washington Department of Natural Resources (DNR), and Washington Department of Ecology (Ecology)). The PSDDA program grew out of a common need and emphasizes consensus decision-making and shared resources to accomplish common goals.

In 1984 several events took place which focused attention on the environmental health of Puget Sound. A number of activities that had been seen as "business as usual," including the disposal of dredged material, were called into question. Out of these events grew the political support for the creation of a new agency, the Puget Sound Water Quality Authority. In addition, concern over the potential presence of chemicals in dredged material, and their effects on fish and other marine life, led to the closure of two major disposal sites within Puget Sound.

These closures, as well as the general concern for the health of the Sound, led to a comprehensive interagency study on future dredging activity and the need for unconfined open-water disposal sites. The four agencies with jurisdiction launched a Puget Sound-wide effort to identify locations for open-water disposal, define the evaluation procedures for material to be placed there, and develop site management plans. The study was completed in two overlapping phases, each lasting 3 ½ years (4 ½ years total), at a total cost of $4.5 million. The process was an open one, and involved input from all stakeholders, including tribes, ports, environmental groups, state and federal agencies and research scientists. These studies resulted in two Environmental Impact Statements. These documents, and the technical appendices that accompany them, form the basis of the PSDDA program (DSSTA, 1988, 1989; EPTA, 1988; FEIS, 1988, 1989; MPR, 1988, 1989, MPTA, 1988).

The program identified eight open-water disposal sites, five of which are non-dispersive and three dispersive. (At non-dispersive sites, environmental conditions are such that the material will stay on-site; at dispersive sites, stronger currents dissipate the material.) The PSDDA management plan emphasizes cooperative implementation and management of the open-water disposal sites. The Corps works cooperatively with DNR to conduct periodic chemical, biological and physical monitoring of the nondispersive sites. The Corps is responsible for conducting physical monitoring at the three dispersive sites.

Agency responsibility and authority are spelled out in the PSDDA management plan:

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[1] U.S. Army Engineers District, Seattle; Seattle, WA.
[2] U.S. Environmental Protection Agency, Region 10.
[3] Washington Department of Ecology, Olympia, WA.
[4] Washington Department of Natural Resources.

Table 1
General Sampling for Full Characterization and DMMU Volume Analysis Guidelines Based on Area Rank

Area Concern Rank	Surface Sediment (< 4 ft) Analysis Requirements	Surface Sediment (< 4 ft) Sampling Requirements	Subsurface Sediment (> 4 ft) Analysis Requirements	Subsurface Sediment (> 4 ft) Sampling Requirements
Low	48,000	6	72,000	9
Low-moderate	32,000	4	48,000	6
Moderate	16,000	4	24,000	6
High	4,000	1	12,000	3

Corps:
- Lead PSDDA agency
- Section 10/404 permits
- Permit compliance and enforcement
- Dredging and disposal site inspections (with DNR and Ecology)
- Initial project data quality assurance
- Dredged material data management - Dredged Analysis Information System
- Physical monitoring of disposal sites
- Disposal site identification (with EPA)
- Biennial PSDDA report (testing and site use)

Ecology:
- Section 401 Water Quality certification
- Coastal zone management consistency determination
- Coordinated state response
- State sediment management data - SEDQUAL
- Dredging site inspections (with Corps and DNR)
- Biennial PSDDA Report (Management Plan Assessment)

DNR:
- Disposal Site use permits and site use records
- Collection of disposal site user fees
- Disposal site inspections (with the Corps)
- Chemical and biological monitoring of the disposal sites
- PSDDA biennial report (monitoring of disposal sites)
- Shoreline permits for disposal sites

EPA:
- Section 10/404 reviews
- Disposal site identification (with the Corps)
- Refinement of assessment (testing) methods

Each of the PSDDA agencies participates directly in evaluation of permits, reviews of disposal site monitoring data, and decisions regarding program modifications through the annual review process.

Program Overview

Procedures for the evaluation of material proposed for dredging were developed during the PSDDA study. Urban embayments in Puget Sound were evaluated based on existing data and known sources of contamination, and these areas were given one of four ranks (low, low-moderate, moderate and high). In addition, some types of facilities, such as marinas, were given a specific ranking. The site ranking determines the amount of testing required for the sampling effort.

Applicants may opt to do a partial characterization (PC) survey prior to doing a full characterization in order to reevaluate the area rank (and reduce the amount of testing required). This is a cost-effective measure for the applicant if 1) they believe the area is cleaner than the ranking would indicate and 2) the project is of a sufficient volume that the lower rank would justify the cost of the PC. Full characterization ranks and minimum analysis requirements are reported in Table 1.

Project volumes are divided into dredged material management units (DMMU), that is, the smallest volume of dredged material for which a separate disposal decision can be made. Each DMMU must be capable of being independently dredged.

Chemical testing is conducted on 57 chemicals which have been identified to be of concern in Puget Sound. Testing for additional analytes may be required if there is reason to believe they may be present, e.g., tributyltin near marinas or areas of boat maintenance activities. "Tiered" chemical and biological testing is required. Screening levels and maximum levels have been established for each chemical of concern. If any chemicals exceed the screening level, biological testing is required. If two or more analytes exceed the maximum level, the material is generally considered unsuitable for open-water disposal. The current bioassay suite consists of the amphipod ten-day acute test, the *Neanthes* 20-day biomass test, and the sediment larval (echinoderm/bivalve) bioassay. The Microtox® bioassay, which has been a part of the suite, is currently under evaluation, and the use of Microtox has been suspended while the PSDDA agencies evaluate the relative merits of the solid phase versus the saline extract test.

Program Implementation

Both the development of PSDDA and its implementation have emphasized cooperative management and decision-making. The Corps of Engineers' Dredged Material Management Office (DMMO) acts as a one-stop office for coordination with dredging proponents. In other parts of the country applicants must deal with each agency separately. While agencies have not relinquished any of their mandated authorities and responsibilities, the consensus approach has eliminated duplication of review for both the applicant and the agencies.

Successful implementation involves close interagency coordination. On a project-specific basis, the agencies review all existing information and sign a suitability determination memorandum (SDM). The suitability determination provides the mechanism for a technical review of the sediment characterization results for each project, and for consensus agency judgements regarding sediment suitability for unconfined open-water disposal.

Permit Process

PSDDA sampling and testing are undertaken as part of the requirements for a permit under Section 404 of the Clean Water Act. Each applicant follows a series of steps:

a. Submit application for permit, receive permit number.

b. Prepare a sampling and analysis plan (SAP) for characterization of proposed dredged material.

c. Receive approval of SAP from PSDDA agencies.

d. Perform sampling and chemical/ biological analysis.

e. Submit testing results to PSDDA agencies.

f. Receive suitability determination for open-water disposal from PSDDA agencies.

g. Complete application details required to issue public notice.

h. Issue public notice, undergo 30-day public comment period.

i. Public interest review and permit decision.

These steps are coordinated with the four PSDDA agencies through the Corps' DMMO. Data submitted upon the completion of testing is given to each of the agencies for concurrent review. Additional coordination of data also occurs with Indian tribes, when their usual and accustomed fishing places occur in the vicinity of the dredging and/or disposal activity. PSDDA agency review of the data normally occurs within two to three weeks. If the data are acceptable for decision-making, the DMMO prepares an SDM, which documents the PSDDA agency consensus on the suitability of the material for unconfined open-water disposal. The SDM is coordinated with each agency, and is signed by each agency representative. Federal navigation projects are subject to the same guidelines and evaluation procedures and must meet the same suitability guidelines as permitted dredging projects prior to any disposal action.

The Dredged Analysis Information System (DAIS) was developed by the Corps for the PSDDA program as a part of program implementation. This is a relational database designed to track all aspects of data associated with projects tested under PSDDA guidelines. The chemical testing modules include all sample information as well as relevant quality assurance/quality control (QA/QC) data. The bioassay module includes test data, water quality measurements and reference toxicant data. Automated quality assurance routines provide checks on accuracy and precision, and quality control coverage for chemical analyses. For biological analyses the QA routines cover test conditions, sensitivity and validity. The administrative tracking system within DAIS allows the management of project milestones, dredged material volumes, points of contact and other valuable information for project managers. Extensive reporting capabilities have been built into DAIS. In addition, it is completely compatible with a geographic information system, and is being used for site-specific spatial analysis of chemical data, as well as in broader investigations.

Annual Review Meeting—Public Responsiveness

Each year the PSDDA agencies conduct an Annual Review Meeting (ARM), moderated by the Corps, and hosted by one of the four agencies. The ARM provides a forum for airing program concerns and problems, receiving public input, clarifying and proposing changes to the PSDDA Management Plan, discussing disposal site management, and discussing the status of any ongoing agency actions. Proposed revisions to the program can be relatively minor, such as a new reporting requirement, or they can be major, such as the requirement to conduct a new bioassay. Papers covering these issues are mailed one month prior to the annual review meeting to all interested parties, including tribes, environmental groups, ports, dredgers, and other agencies. Both oral and written comments and questions are encouraged. Last year the agencies combined this review with the that of the state's Sediment Management Standards, to provide a more comprehensive discussion of all sediment management programs.

Report Preparation—Public Accountability

The PSDDA program prepares a biennial report documenting all aspects of the program. Included are project testing overview summaries, testing cost evaluations, PSDDA program/regulatory processing evaluations, documentation and summaries of disposal site monitoring, program protocol clarifications and refinements, and updates to the PSDDA management plan.

Site Management and Monitoring

All four PSDDA agencies cooperatively manage the disposal sites and participate in data review and management decisions affecting the sites. Initial baseline monitoring studies were conducted at each of the five nondispersive sites: Bellingham Bay, Port Gardner, Elliott Bay, Commencement Bay, and Anderson/Ketron. Each site has its own monitoring schedule, determined primarily by the amount of material deposited at the site.

DNR generally takes the lead on disposal site monitoring. It is responsible for contracting for chemical and biological monitoring services. Major funding for the contract is generated by site use revenues collected by DNR. In cases where special studies are required, funding can come from the other PSDDA agencies. The Corps accomplishes physical monitoring at all eight disposal sites in lieu of paying a site use fee to DNR for its unsponsored federal maintenance projects. The PSDDA agencies review the request for proposals, contractor bids, and other documents related to the monitoring effort. Monitoring reports are prepared by the contractor who conducts the chemical and biological testing. This information is reviewed by the four PSDDA agencies to determine if management standards have been met, whether additional studies are needed, or whether adjustments should be made to site use conditions, site standards, or other program elements.

Monitoring at a PSDDA site is not triggered by a specific disposal event, but rather is triggered by the cumulative volume disposed at a site. Only dredged material which meets the PSDDA guidelines is allowed at the site (no guns, no bridges). No attempt is made to isolate physical types of material (sand from clay).

DNR is the land management agency for the State of Washington, and holds the shoreline permit for each of the PSDDA disposal sites. These permits are for five year periods, and all were renewed within the past two years. As the holder of the permit, DNR is potentially liable for any damages resulting from a disposal at a PSDDA site. The Corps and DNR were jointly named in a lawsuit in 1992. All four agencies worked cooperatively to address litigation issues, and the lawsuit was ultimately resolved in the agencies' favor.

Monitoring Data: Non-Dispersive Sites

The chemical, biological and physical monitoring conducted at the PSDDA non-dispersive sites is designed to answer specific questions to determine if the site management objectives outlined in the management plan have been achieved. The three questions are:

1. Does the dredged material stay on-site?

2. Is the biological effects condition (minor adverse effects) for nondispersive sites exceeded due to dredged material disposal?

3. Are unacceptable adverse effects occurring to biological resources immediately off-site due to dredged material disposal?

Six post-disposal site monitoring events have taken place to date, covering four of the five non-dispersive sites. These are 1990/1992 monitoring of the Elliott Bay site, 1990/1994 monitoring of the Port Gardner site, 1993 monitoring of the Bellingham Bay site, and the 1995 monitoring of the Commencement Bay site. The results of these monitoring events have confirmed that the site management objectives have been met.

If site management objectives are not met, the PSDDA agencies meet to consider a range of management options to remedy the problem, which could range from modification of evaluation procedures, to additional data collection, or possibly site closure.

Monitoring Data: Dispersive Sites

At the three dispersive sites, monitoring requirements are limited to bathymetry for verification that dispersion of the material is taking place. Guidelines used to evaluate material proposed for dispersive site disposal are more restrictive than those used to evaluate material for nondispersive sites. The only monitoring question for dispersive sites is:

1. Is the dredged material mounding on-site, or is it being effectively dispersed?

Post-disposal bathymetry monitoring of the Rosario Strait site in 1991 and 1994 indicated that the site is highly dispersive, with no discernible mounding of material compared to initial baseline bathymetry.

Five Year Review

Projects

The number of projects evaluated under the PSDDA program is shown in Figure 1, and has averaged eleven per year, with a range from 5 to 17 projects each year. The number of suitability determinations has averaged 10 per year. The difference reflects, in part, the completion of several partial characterizations during DY93. In addition, there are a few projects where, based on the results of the PSDDA testing, the applicant withdrew the project or opted for a different disposal option. In some of these cases, the data were not submitted to the PSDDA agencies, and a suitability determination was not made.

Between 1990 and 1995, the average time required for PSDDA evaluation was 176 days (Figure 2). This included the time for agency review of sampling and analysis plans (average 15 days), sampling and testing by the applicant (average 145 days) and data review/suitability determination by the agencies (average 16 days). The agency review times are well within the agency established guidelines of three weeks.

Chemical and Biological Testing

Figure 3 shows the dredged material management units undergoing chemical and biological testing since the PSDDA program was initiated. There was a large increase in the number of DMMUs tested in DY92, even though the number of projects remained relatively constant. This is the result of several large volume projects, as well as more projects being located in high and moderate-ranked areas. The number of DMMUs requiring the biological testing tier (those that had one or more exceedances of PSDDA screening levels) varied considerably: from a high of 72 percent in DY92 to a low of 19 percent in DY93. Approximately fifty-nine percent of DMMUs undergoing chemical testing have required bioassay testing during the program's implementation.

Site Use and Capacity

As part of the disposal site technical evaluation under the PSDDA study, site capacities were estimated for each site. Boundaries of the sites were drawn so that each will retain adequate capacity for fifteen years, given the projected use. Figure 4 depicts the total volume of material undergoing PSDDA characterization, and the volume found suitable each dredging year. In general, most of the material (96%) tested was found suitable for unconfined open-water disposal. Over the five years of PSDDA implementation, approximately 2,378,200 cubic yards have been placed at the PSDDA open water sites, averaging 518,279 cubic yards per year. A total of 244,000 cubic yards of material has been found unsuitable for unconfined open-water disposal during the past five years. Five year summaries of site use show that site capacity appears to be sufficient for at least fifty years (Table 2).

Cost Data

One aspect of the PSDDA program that is not found in many other regulatory programs is a commitment to monitor program costs. By tracking these costs, the agencies can give new applicants an estimate of the cost of conducting testing for a proposed project. In addition, it allows the agencies to keep an eye on the cost implications of any proposed program changes. Higher unit costs tend to be associated with smaller projects, which lack the ability to spread

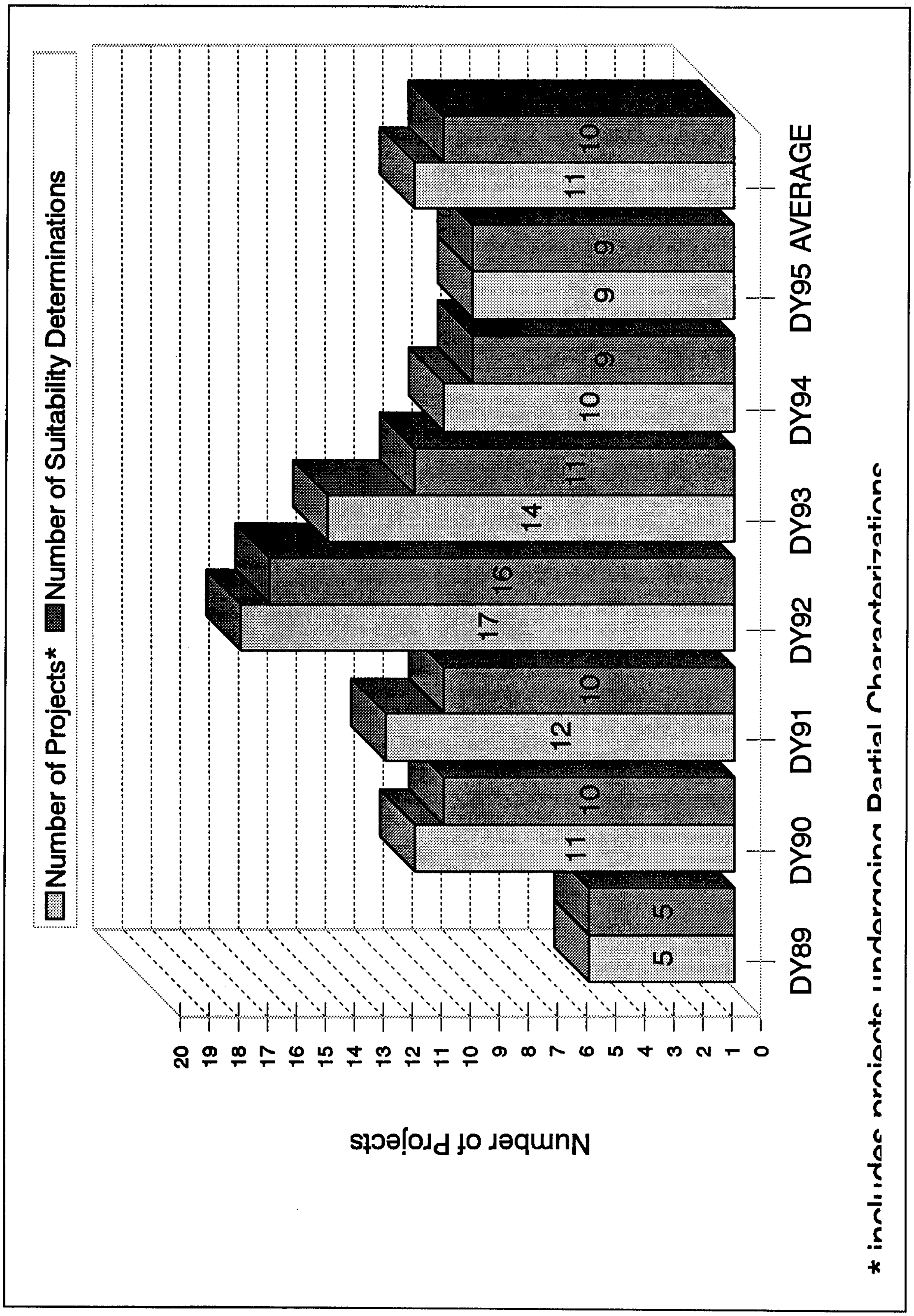

Figure 1. Number of projects evaluated under PSDDA

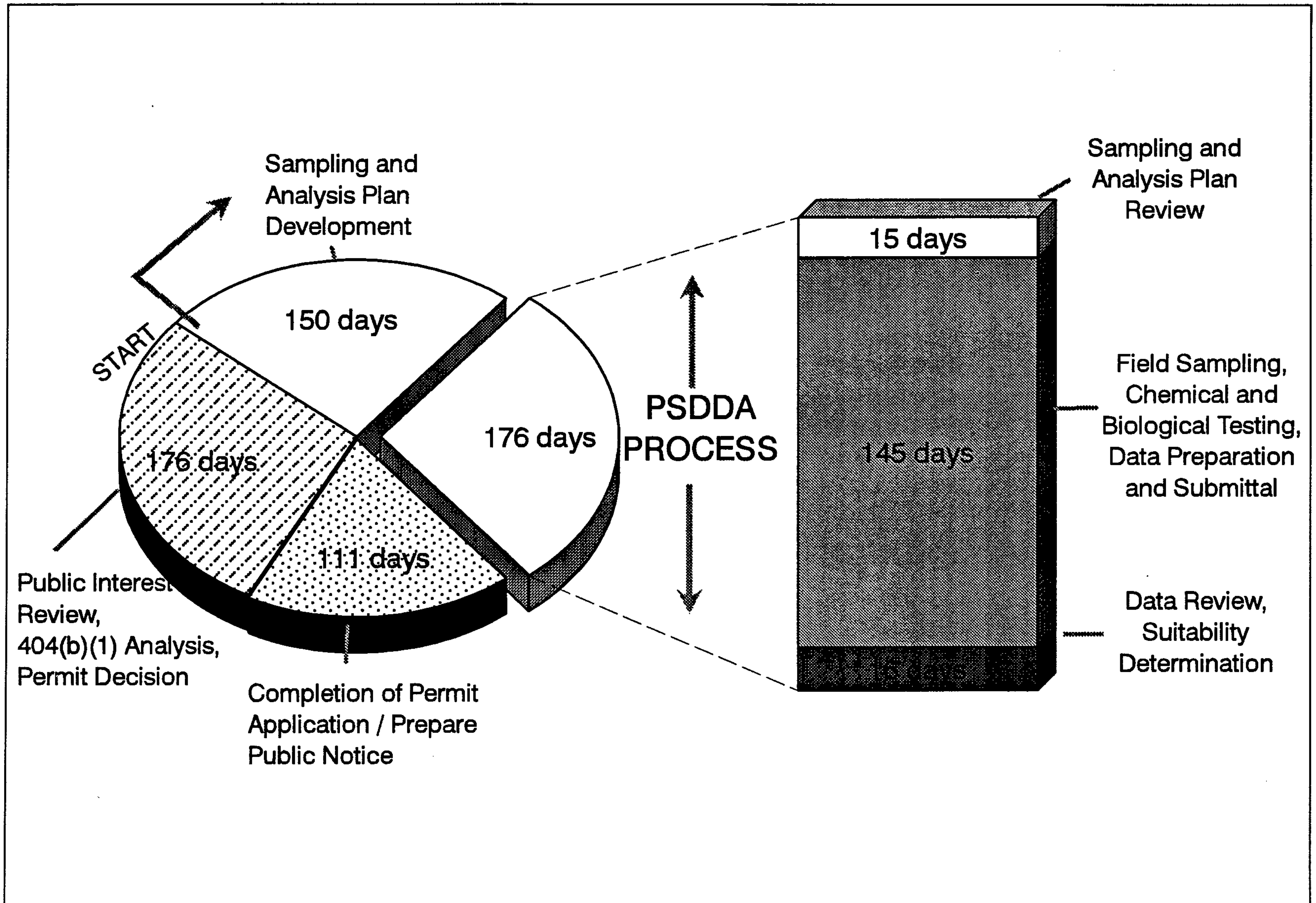

Figure 2. PSDDA processing time retrospective summary (1990-1995)

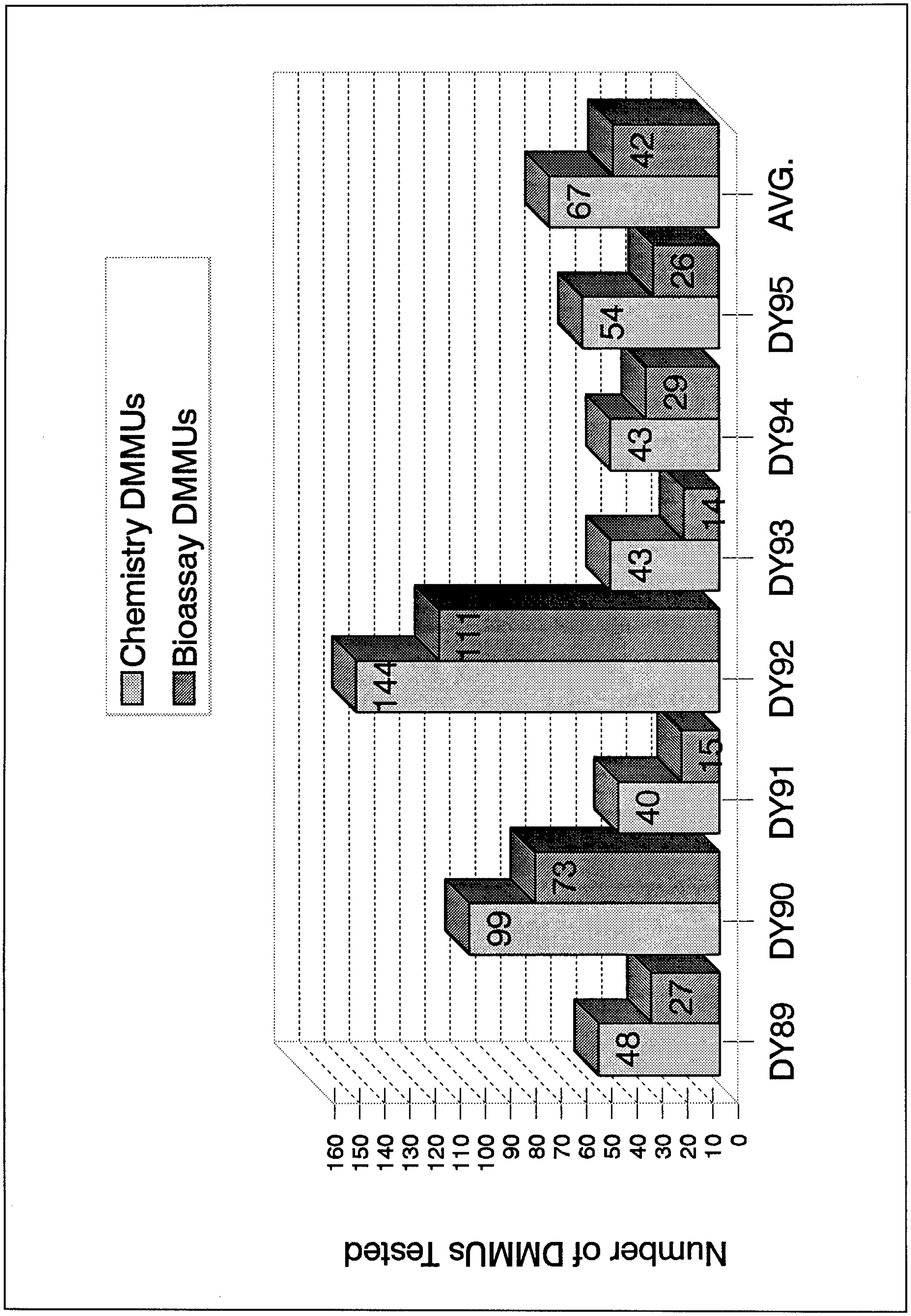

Figure 3. Number of chemistry and bioassay DMMUs tested

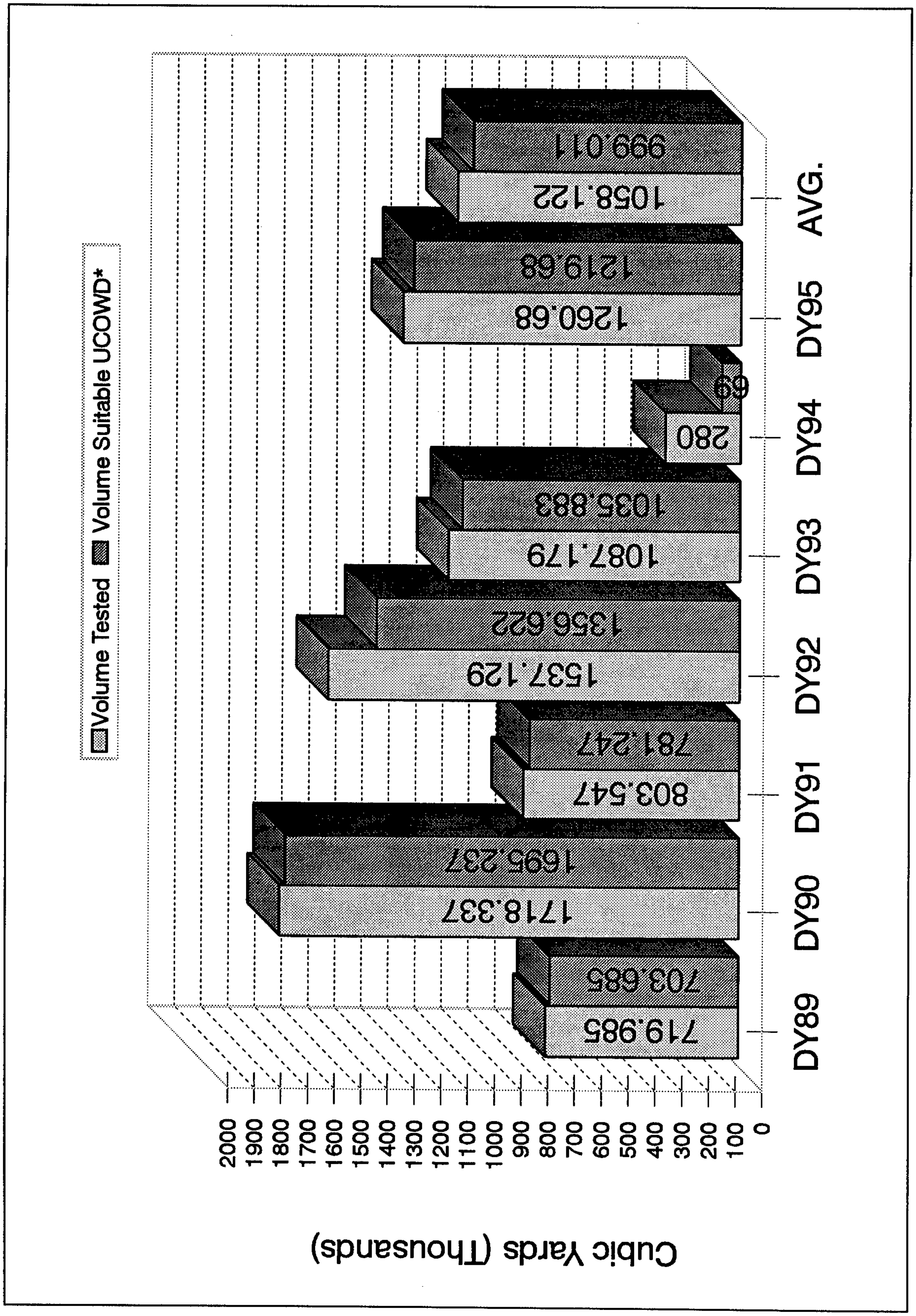

Figure 4. Testing outcome summary

Table 2
Five Year Site Use Summary

Nondispersive Disposal Site	Cumulative Volumes (CY)	Average Volume (CY/YR)	15-Year Predictions MDR Phase I/II (CY)	Percent of 15-Year Prediction	Estimated Time to Exceed Site Capacity[1] Years
Port Gardner (1989-1993)	1,118,835	223,767	8,243,000	13.6	40
Elliott Bay (1989-1993)	389,065	77,813	10,525,000	3.7	>50
Bellingham Bay (1990-1993)	32,883	8,221	1,181,500	2.8	>50
Commencement Bay (1989-1993)	17,548	3,510	3,929,000	0.45	>50
Anderson/Ketro Island (1990-1993)	10,197	2,549	785,000	1.3	>50
Subtotals	1,568,528	315,860	24,763,500	6.3	N/A
Dispersive Disposal Site	**Cumulative Volumes (CY)**	**Average Volume Per Year (CY/YR)**	**15-Year Predictions MPR Phase I/II (CY)**	**Percent of 15-Year Prediction**	**Estimated Time to Exceed Site Capacity[2] (Years)**
Rosario Strait (1990-1993)	787,030	196,758	1,801,000	43.7	N/A
Port Townsend (1990-1993)	22,642	5,661	687,000	3.3	N/A
Port Angeles (1990-1993)	0	0	285,000	0	N/A
Subtotals	809,672	202,419	2,773,000	29.2	N/A
Grand Totals	2,378,200	518,279	27,536,500	8.6	N/A

[1] Site capacity estimated in Phase II disposal site selection technical appendix for nondispersive sites is approximately 9,000,000 cubic yards.
[2] Actual site capacity for dispersive sites is not limited, assuming complete dispersal of dredged material off site.

the QA and sampling costs over several DMMU. Over the past five years, sampling and testing costs have averaged $0.57 per cubic yard (Figure 5) and have remained fairly uniform. Due to more biological testing than usual, and higher ranking of projects, the average cost for DY92 year was $0.78 a cubic yard. There was also a sharp increase in costs for DY94/95 (to $0.93), due to a number of small projects in higher-ranked areas.

The Future

Due to resource limitations, the PSDDA program has focused on the management of unconfined open-water disposal of clean dredged material. It is becoming evident that both agencies and project proponents are increasingly involved in complex projects that include the potential dredging and disposal of contaminated

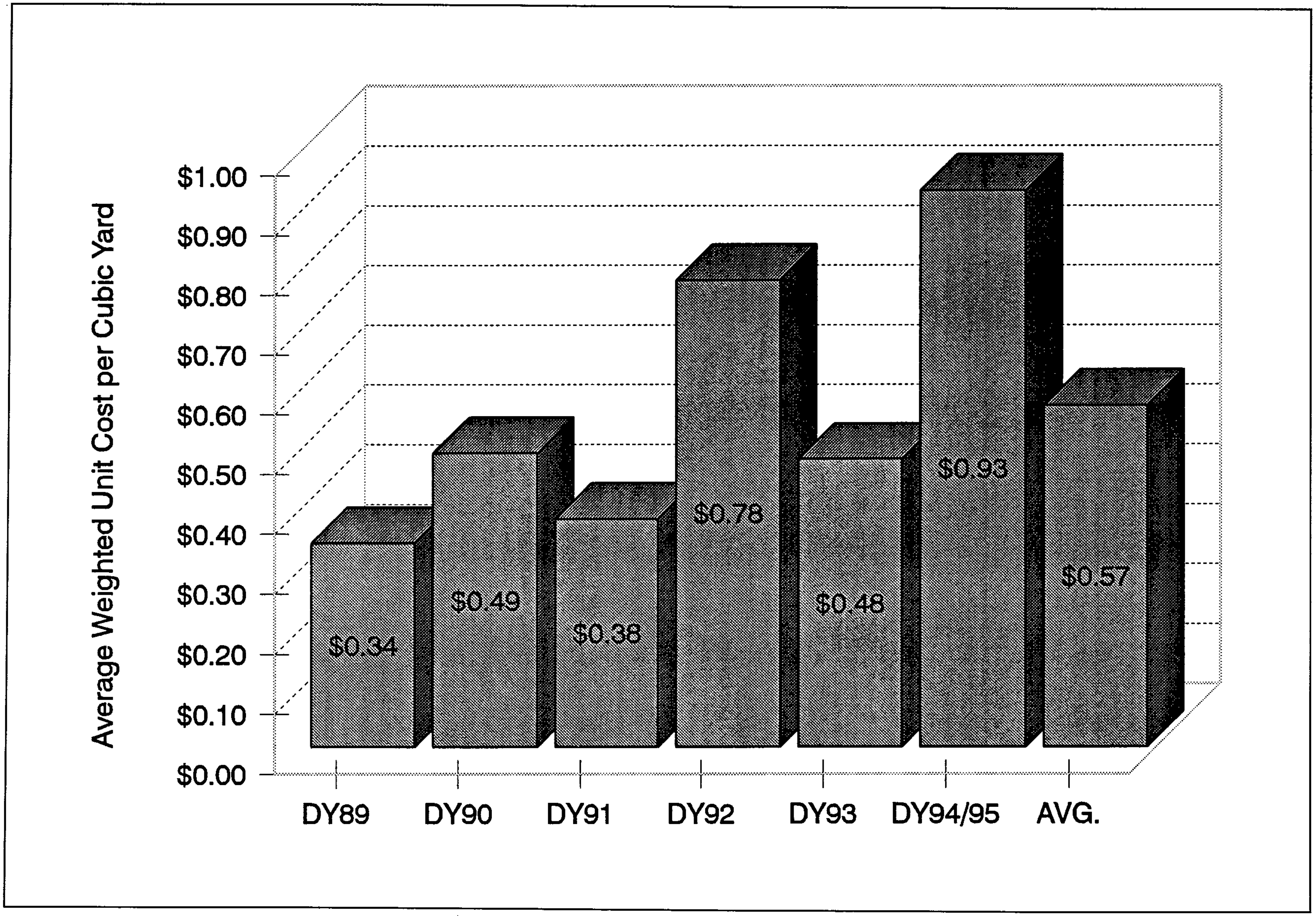

Figure 5. Average sampling and testing costs

sediment. No single agency has complete and unilateral authority to cover all contaminated sediment issues. In addition, no one agency has the resources to tackle these issues.

In recognition of these facts, the PSDDA agency directors (with the addition of the Puget Sound Water Quality Authority) signed an interagency/intergovernmental agreement (IAG) in May 1994. The IAG formalizes interagency cooperation and coordination to establish regional priorities, jointly develop technical and policy responses, and implement solutions for the management of clean and contaminated sediments. To be successful in crafting viable solutions to these problems, the initial joint actions require the active participation of all stakeholders, including state and federal regulatory agencies, resource agencies, environmental groups, Indian tribes, industry and the public.

The initial actions identified by the agency directors included the following: initiation of a sediment cleanup strategy, preparation of an action plan for multi-user confined disposal sites, and the establishment of interagency/intergovernmental policies for the beneficial use of dredged material. Workgroups have been formed to accomplish these goals. The sediment cleanup workgroup (convened in July 1994) has focused on development of a range of approaches to facilitate the cleanup of contaminated sediments in the aquatic environment. The group reported the results of its efforts to agency directors on December 20, 1994.

The beneficial uses workgroup began meeting in January 1995, and is focusing its efforts on the identification of existing agency policy and regulations regarding beneficial uses. The group will recommend ways to encourage the beneficial use of dredged material in a coordinated and environmentally sound manner.

The action plan for multi-user confined disposal sites received a substantial boost through a Congressional appropriation to the Corps to conduct a reconnaissance study on multi-user sites. This effort (coupled with additional funds from the Department of Ecology) complements existing efforts and identifies the steps necessary for the development of one or more multi-user confined disposal sites in the Puget Sound region. The reconnaissance study was completed in 1995, and scoping is underway for the completion of an EIS for the project.

Conclusion

Five years of implementation have confirmed the success of the PSDDA program. The cooperative management approach has provided predictability to the regulated dredging community. The program has provided technical feedback to the resource agencies and the public on the environmental acceptability of the evaluation procedures and management plans. The program is now evolving to deal with complex dredging projects, with hopes that the same spirit of cooperation can speed the cleanup of contaminated sites in Puget Sound.

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Evaluation of Polychlorinated Biphenyl and Polycyclic Aromatic Hydrocarbon Concentrations in Two Great Lakes Dredged Material Disposal Facilities

by
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Introduction

Several Great Lakes CDFs are now nearing or exceeding design capacity. Of the 26 Federally-funded CDFs built under PL 91-611 as of June 1992, six had been filled to capacity and all but two are scheduled to be filled by the year 2006 (U.S. General Accounting Office 1992). Concern has been expressed about the potential for release of contaminants from these sites and about closure requirements. Two CDFs near design capacity were selected for evaluation of long-term losses from a closed disposal facility. Results of the analysis will be used in determining the need to cap the sites, recommend future uses, and assist in the design of the next generation of CDFs. This paper will focus only on concentrations of PCBs and PAHs within the CDFs and theorize about factors influencing their distribution.

The Jones Island CDF at Milwaukee, Wisconsin and the Renard Island CDF at Green Bay, Wisconsin were selected for study. Both CDFs were constructed on native sediments with no liner, which makes it difficult to differentiate native sediment from dredged material at the bottom of the CDF. Both CDFs were filled mechanically, which means that the oldest material is located proximal to the off-loading site and the most recently deposited material is farther away. At the Jones Island CDF the dredged material was trucked and barged to the CDF. Therefore, filling has occurred more in a south to north trend. Similarly, the Renard Island site was filled by lifting material over the north dike wall. As the material emerged from the water, it was graded toward the center of the CDF. Therefore, the dredged material deposited earliest is near the dike walls and the most recently deposited material is near the center west of the weir.

The Renard Island CDF is an island located approximately one quarter mile from the mouth of the Fox River in Green Bay. The sixty-acre site was constructed in 1979 with a design capacity of approximately 1,200,000 cubic yards, and is considered 99 percent filled. The Federal Navigation Channel extends from just below the DePere Dam in the Fox River at the city of Green Bay, approximately eleven miles out into Green Bay. Sediments in the navigation channel contain relatively low concentrations of PAHs; however this site has historically contained high concentrations of PCBs (U.S. Army).

The 44-acre Jones Island CDF at Milwaukee, Wisconsin was completed in 1975 and is located in the south outer harbor. The design capacity of the Jones Island facility is 1,600,000 cubic yards and it is expected that the site will be filled to capacity by 1999. The Jones Island CDF is a nearshore facility, having one side abutting the shoreline. The CDF is located in a

[1] U.S. Army Engineer District, Detroit; Detroit, MI.
[2] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.
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heavily industrialized area and a petroleum handling operation is located close by. The CDF serves the Milwaukee Harbor, which has been one of the most contaminated sites operated and maintained by the Detroit District. Concentrations of PAHs in sediments from the Milwaukee Harbor navigation channel have been as high as 160 $\mu g/g$ (U.S. Army Engineer District, Detroit 1989). Samples from above and beside the navigation channel have exceeded 500 $\mu g/g$ of PAHs (Li et al. 1995). Coal tar, coal tar air emissions, gasoline engine exhaust tar, and highway runoff were the major contributors to Milwaukee Harbor's PAH burden (Li et al. 1995). PCB concentrations are elevated also, with typical concentrations in the inner harbor exceeding 10 $\mu g/g$ in the mid-1980s (U.S. Army Engineer District, Detroit 1989).

Materials and Methods

Borings were collected at six locations on the Renard Island CDF (Figure 1) and six locations on the Jones Island CDF (Figure 2) during August and October 1993 under a work order developed by the Detroit District. At each station split-spoon samples were collected to a depth of approximately twenty-five feet. Cores were split into five discrete samples, with "A" samples reflecting the top 1.5 feet of dredged material, down to "E" samples representing the bottom 7.5 to 9 feet of the core. As elevations varied along the CDF and depths to the original lakebed varied, all cores were sectioned at the same depths. Due to the presence of soft material at the Jones Island CDF, the site was unable to support a drilling rig at two stations and these samples were collected using a tripod-mounted rig which could not reach the bottom of the CDF.

Analysis for PCB congeners was conducted using the Canadian Standard (CLB-1). The 51 congeners present in CLB-1 include several of those that are considered most toxicologically significant (Clarke et al. 1989). This was felt to be the most appropriate in light of current

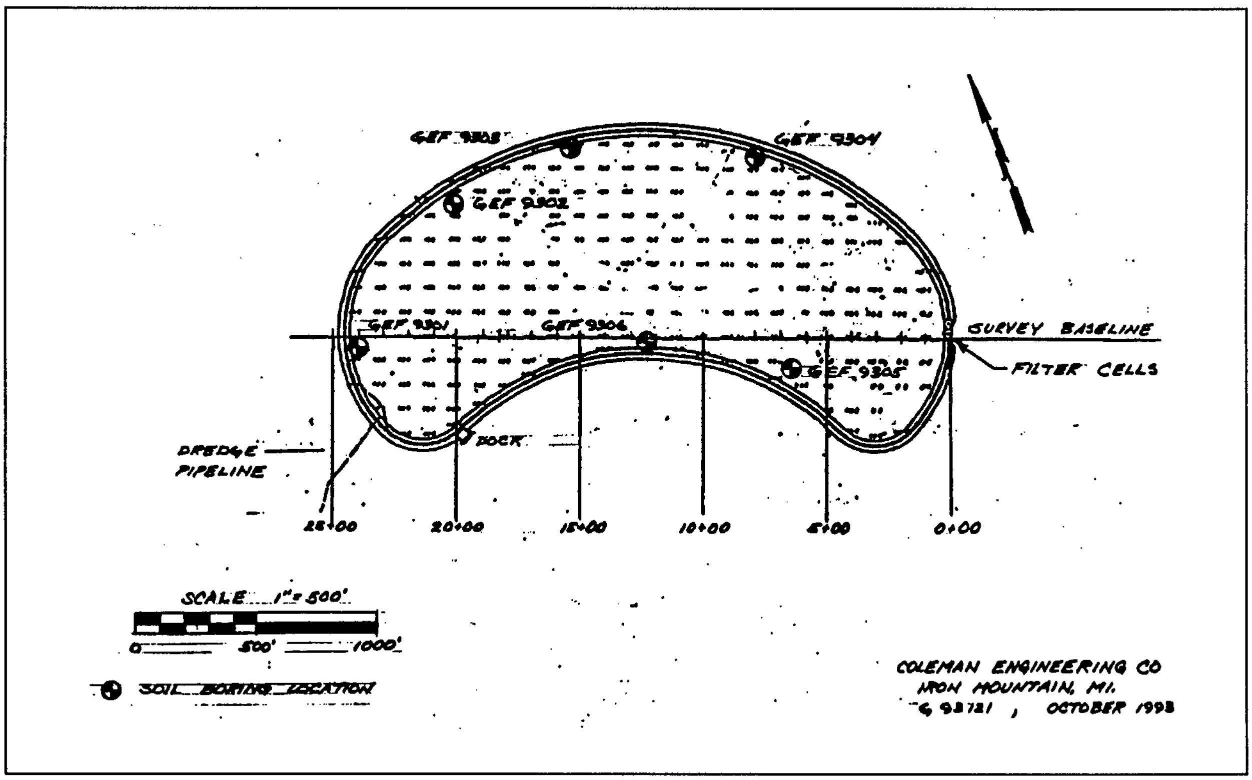

Figure 1. Sampling locations at the Renard Island Confined Disposal Facility

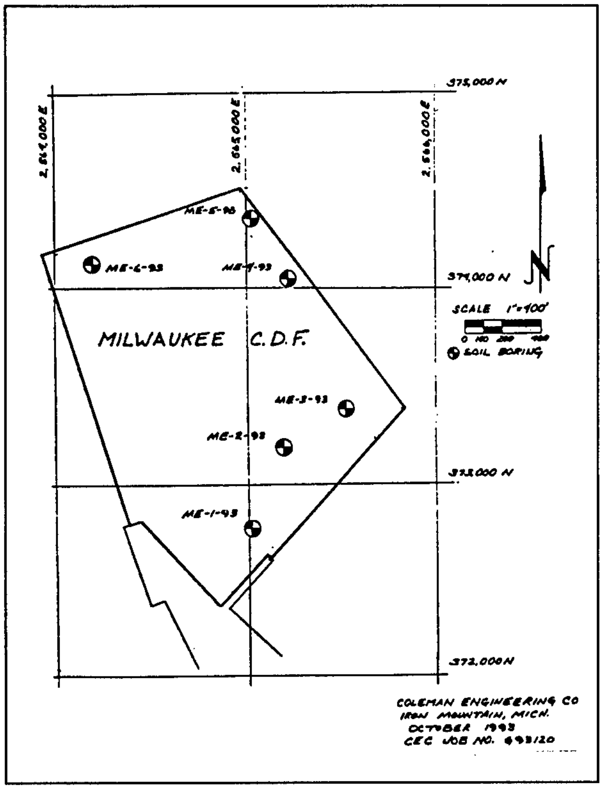

Figure 2. Sampling locations at the Jones Island Confined Disposal Facility

analytical capabilities, as the study was directed at estimating potential environmental impacts of releases from the CDFs.

Results and Discussion

PCB Concentrations - Jones Island CDF

Concentrations of PCB congeners in the surface of the CDF are very low in comparison to the concentrations in sediments of Milwaukee Harbor. The sum of detectable CLB-1 congeners in the "A" samples in the Jones Island CDF was less than 1 μg/g.

Concentrations of several PCB congeners increase from the surface to the bottom of the CDF (Figure 3). Increases are most substantial in congeners measured with the lowest degree of chlorination under the CLB-1 Standard. Congeners 18, 31, 40, 44, 49 and 52 show substantial (3-7X) increases in the "D" and "E" sections of the cores at the Jones Island CDF. Highly chlorinated PCBs were generally nondetectable or showed little change in concentration. The pentachlorobiphenyls show approximately a 25 percent increase while the hexachlorobiphenyls are unchanged. The tetrachlorobiphenyl congener 77 doubles in concentration near the bottom of the CDF. This congener is one of the non-ortho coplanar configurations and has been identified as one of the most toxicologically significant PCB congeners (Clarke et al. 1989).

PCB Concentrations - Renard Island CDF

PCB concentrations at the surface of the Renard Island CDF also appear low in comparison to the high amounts of PCBs present in the harbor sediments. The sum of detectable CLB-1 congeners only reaches to about 1 μg/g. Similar to the Jones Island site, several cores show pockets of much higher PCB concentrations with depth, but a comparison of the distribution of congeners does not show a substantial change in congener patterns. A plot of mean concentration versus depth shows some increases with depth. Congeners 18 and 31 show an approximate doubling in the "E" section, with congener 31 showing a second peak in the "C" section at depth in the Renard Island CDF. As with the Jones Island CDF the more heavily chlorinated congeners generally are not detectable or remain at low concentrations through the CDF core.

PAH Concentrations - Jones Island CDF

PAH concentrations at the surface of the Jones Island CDF were found to be comparable to concentrations in the harbor. The average sum of detectable PAHs in the surface samples was 40 μg/g (Figure 4a). A survey of the entire navigation channel during 1993 found an average of 20 μg/g (n = 22) (U.S. Army Engineer District, Detroit 1993). However, sampling as recent as 1989 found average PAH concentrations of 92 μg/g (n = 11) (U.S. Army Engineer

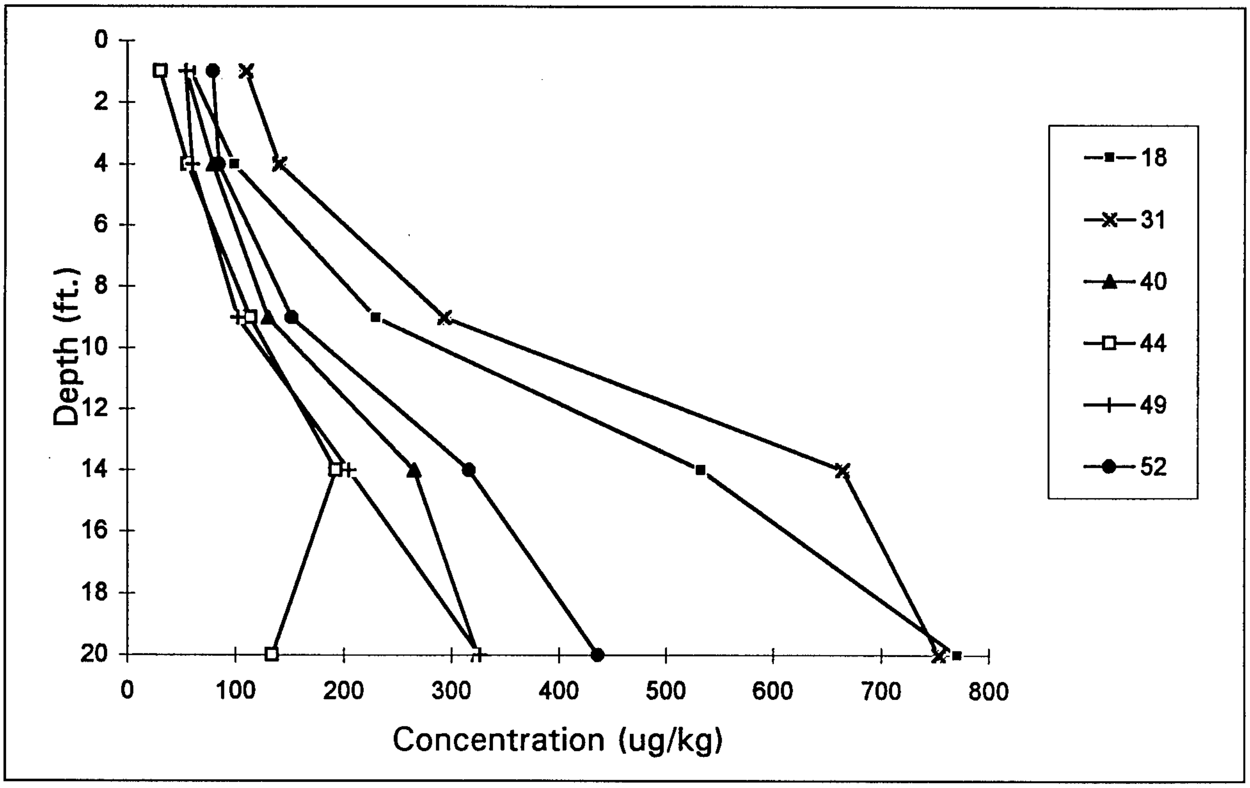

Figure 3. Average PCB congener concentrations Milwaukee CDF

District, Detroit 1993). Samples from the 1.5 to 6 foot depth contours ("B" section) appear comparable in PAH concentrations to the surface samples. The PAH distribution in these samples appears to be dominated by the 4- and 5-ring compounds such as fluoranthene, benzo(b)fluoranthene, benzo(a)pyrene, and benzo(k)fluoranthene. The 2- and 3-ring PAHs such as acenaphthene, acenaphthylene, anthracene, and naphthalene are not detectable or are present at low concentrations in the upper six feet of the CDF.

The PAH component in CDF cores below the 6-foot depth level demonstrated substantial increases with depth. Approximately half the priority pollutant PAHs show substantial increases in concentration as compared to surface samples, with none showing a decrease. The compound showing the strongest increase is anthracene, which typically increased 50 μg/g or greater in the "C" and "D" sections. Acenaphthene also increased substantially, such that these two simple PAHs comprise 40 to 60 percent of the total PAHs (Figure 4b). One exception to this trend is Station MVF9301. Although this core does demonstrate substantial increases in total PAH concentrations in the "C" and "D" sections, there is little change in the relative proportions of PAHs from the top of the core to the bottom.

As indicated above, the District's 1989 sampling, and surveys conducted by others, show high concentrations of some PAHs in the harbor sediments. However, a comparison of the relative proportions of the various PAH components in the sediment and the CDF does not correspond with those seen at depth in the CDF. In the Detroit District's 1989 harbor survey the highest concentration reported for anthracene was 7.3 μg/g whereas concentrations of greater than 200 μg/g were reported at the bottom of the Jones Island CDF. Interestingly, anthracene, which is in very high concentration in the "D" sections, is virtually undetectable in "E" sections.

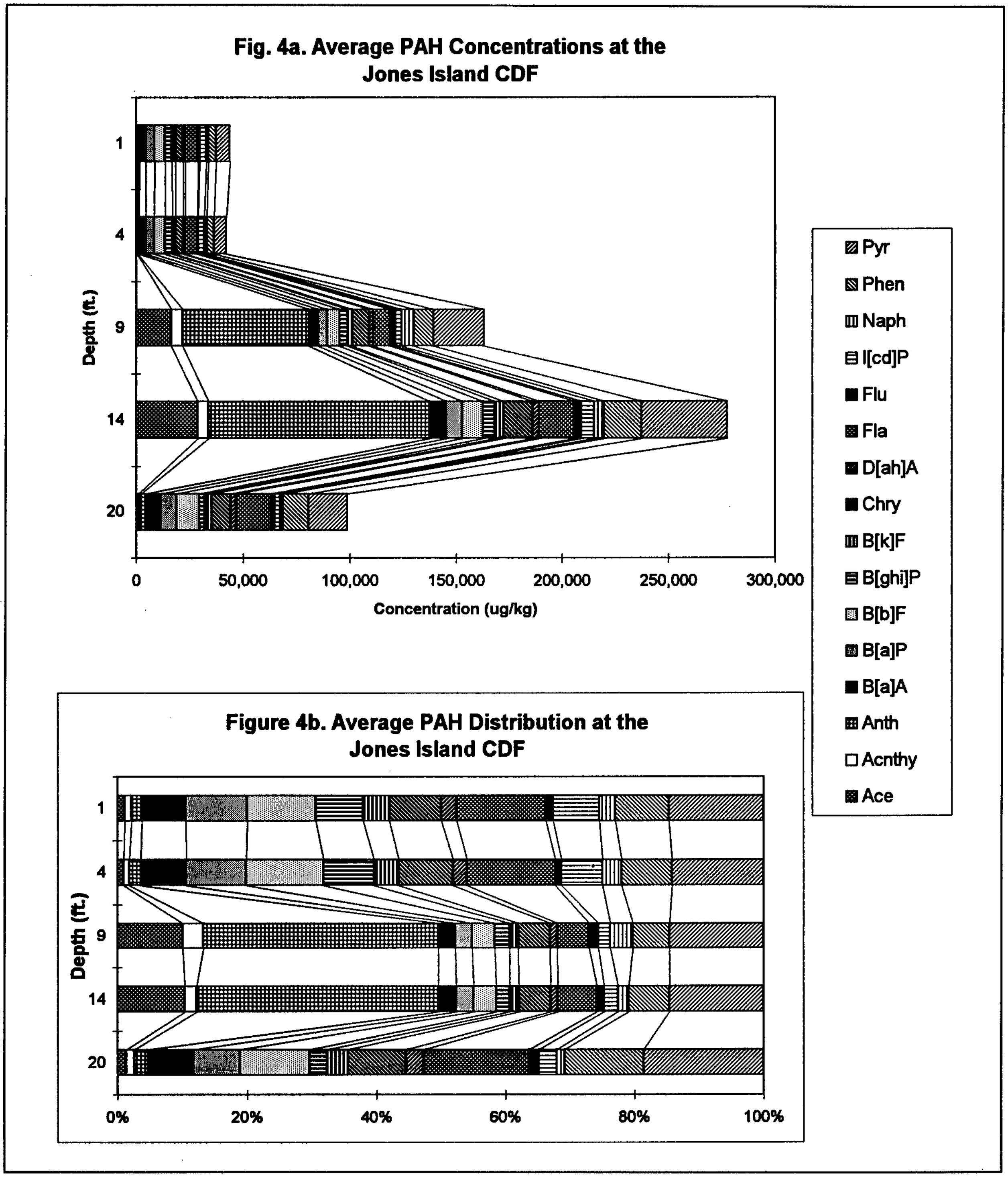

Figure 4. Average PAH concentrations at the Jones Island CDF

PAH Concentrations - Renard Island CDF

As with the Jones Island CDF, the Renard Island CDF also showed substantial increases in the concentrations of certain PAHs over the depth of the CDF. PAH concentrations in the 1988 survey found total PAH concentrations of less than 1 $\mu g/g$; however, concentrations as high as 38 $\mu g/g$ were reported at depth in the CDF. As with the Jones Island site, the increase in concentration is mainly due to acenaphthene and anthracene (Figure 5). The acenaphthene/acenaphthylene/anthracene component comprises 60 to 80 percent of the total PAH at depth in the Renard Island CDF. Although PAH concentrations decline near the bottom of this CDF, the PAHs are still present at significant quantities in the deepest sections of the cores.

Factors Influencing Contaminant Concentrations at Depth in the CDF

As both CDFs were constructed in-water and captured water was not removed, initial placement of dredged material was directly into standing water. Under these conditions dredged material pores filled with water restrict the diffusion of gases and oxygen is consumed faster than it can be replaced, causing reducing conditions. In the early stages of filling, anaerobic dredged material is placed below the saturated zone and likely remains in the original anaerobic condition. Although redox conditions were not measured as a part of this study, the drilling operations at the Jones Island CDF were delayed after encountering methane pockets.

PCBs

Studies by Abramowicz et al. (1993) detected anaerobic dechlorination of PCBs at the relatively low PCB concentrations (5 - 10 $\mu g/g$) seen in dredged material. Rhee et al. (1993) evaluated the reductive dechlorination of Aroclor 1254 by microorganisms eluted from Hudson River sediments and found microbes were able to remove chlorine from the PCB molecule. Attack was seen to occur in the meta- and para-positions, resulting in the accumulation of ortho-substituted congeners. Under anaerobic conditions, microorganisms were able to reductively dechlorinate A-1254 to a mixture composed largely of di-, tri-, and tetrachlorinated congeners.

The increase in concentration of congeners 18 and 31 near the bottom of both CDFs is consistent with these findings. The only dichlorobiphenyl in the CLB-1 Standard (para-substituted congener 15) is not detected. The congeners that are seen to increase with depth most strongly (18 and 31) are both ortho-substituted trichlorobiphenyls. This does not demonstrate, however, that reductive dechlorination is occurring in the CDF. A review of the concentrations of the more highly chlorinated congeners does not demonstrate a corresponding decline in these compounds. In fact, these compounds are generally not detectable or present at very low concentrations even in recently deposited sediments which have been placed above the saturated zone. Furthermore, analysis of Milwaukee Harbor sediments by Ni et al. (1992) shows little change in congener distributions with depth in harbor sediments, some up to 40 years old. The Rhee et al. (1993) study found 30 percent of the total chlorines removed over the first five months, however only 8 percent additional had been removed in the remaining 19 months of the study. Therefore, any reductive dechlorination which takes place likely occurs before sediments are placed in the CDF.

PAHs

The finding of elevated concentrations of anthracene at depth in both CDFs was surprising as anthracene was generally present at very low concentrations in Milwaukee Harbor, and at non-detectable levels in the Fox River and Green Bay, in sediment surveys conducted in the late 1980s. Milwaukee harbor sediments placed within the CDF during the early stages of

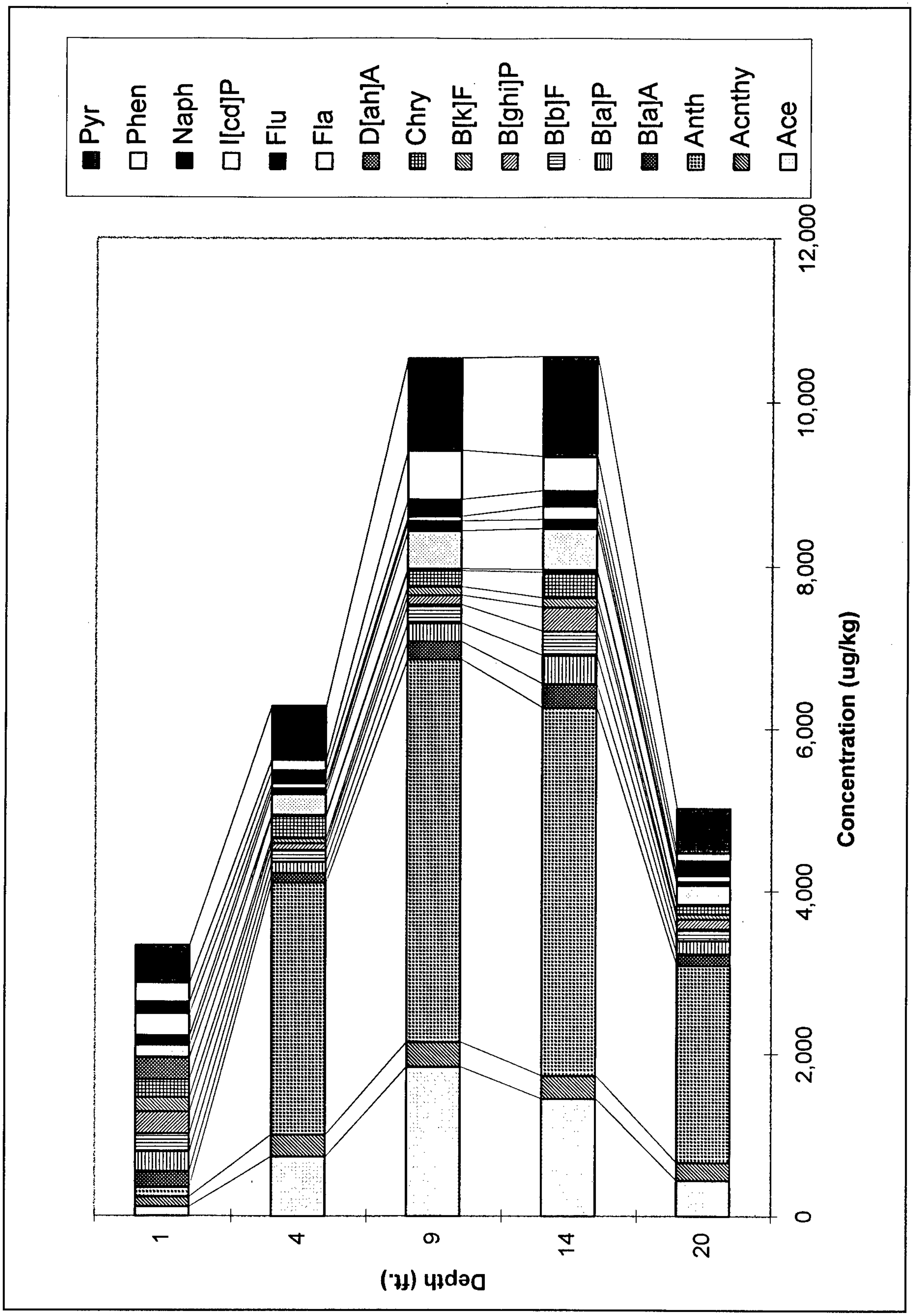

Figure 5. Average concentrations of PAHs in the Renard Island CDF

filling (late 1970s) contained much higher concentrations of PAHs than are seen in contemporary sediments (Gin, 1992). Research by Li et al. (1995) correlated the highest PAH concentrations in Milwaukee Harbor to a coking facility just upstream of the navigation channel. Despite extensive sampling, none of those cores exceeded approximately 10 μg/g anthracene.

Enzminger and Ahlert (1987) point out that coal tar contains an estimated 10,000 compounds, including 2-ring compounds (such as naphthalene) up to compounds with more than 20 rings. Approximately 50 percent of the PAHs in coal tar pitch contain more than seven rings. The entire PAH component of sediments is seldom quantified. However it seems possible to hypothesize that these complex compounds decompose to 2- and 3-ring compounds during the first phase of degradation. The generation of simple PAHs through the decomposition of complex PAHs under aerobic and/or denitrifying conditions would possibly explain other anomalous data associated with these samples. Sediment samples collected from Milwaukee Harbor in 1993 and allowed to air dry displayed a doubling in priority pollutant PAH concentrations over the first five to ten days (C. L. Price, WES, unpublished data).

Other authors have reported similar findings in sediments containing high amounts of coal tar. Murphy et al. (1994) studied biodegradation of organic contaminants under denitrifying conditions in the Dofasco Boatslip in Hamilton Harbor and suggested the production of naphthalene during initial degradation. During the first phase of treatment naphthalene concentrations in sediments of the Boatslip increased 196 percent, which they suggest was due to a breakdown of these complex PAHs into naphthalene. A significant increase in anthracene concentrations was not noted however. Palermo et al. (1987) evaluated the concentrations of PAHs in Indiana Harbor sediment in the original highly reduced state and after six months of "aging" (placement outdoors in the shade). Total priority pollutant PAH concentrations in the aged sediment declined by an order of magnitude, mainly due to the loss of naphthalene, however anthracene concentrations increased by 17 percent.

If these PAHs were subjected to reducing conditions, it is unlikely that they would undergo further degradation. Although there has been increasing interest concerning anaerobic decomposition of PAHs, the phenomenon has yet to be documented (Cerniglia 1992). Mihelcic and Luthy (1988) were unable to detect significant degradation of naphthalene and acenaphthene under anaerobic conditions over periods of 50 and 70 days, respectively.

Factors Influencing Contaminant Concentrations at the Surface of the CDF

As CDFs continue to fill, the height of the dredged material rises above the water level. Placement of dredged material in upland conditions allows drainage of excess porewater and creates aerobic conditions within the dredged material. Dredged material exposure likely enhances the rates of volatilization and photolytic processes. Biodegradation of 2-ring and 3-ring PAHs in soils and sediments has been documented under varying conditions. Herbes and Schwall (1978) found turnover times to be very rapid for naphthalene (7 hours) and anthracene (400 hours) in freshwater sediments from contaminated areas. Four- and five-ring PAHs were transformed, but rates were three orders of magnitude lower.

PCBs which have undergone reductive dechlorination are rapidly degraded in aerobic environments. Bedard et al. (1987) isolated a bacterial strain which reduced A-1242 concentrations by 81 percent and A-1254 by 35 percent over a two-day period (initial PCB concentration 10 μg/g). This study was conducted in Hudson River sediments which had undergone extensive anaerobic dechlorination such that 62 to 73 percent of the PCBs were present as mono- and dichlorobiphenyls (Harkness et al. 1993). Mineralization is likely a more important loss pathway for PCBs than volatilization after the first few days of CDF placement. Fairbanks et al. (1987) evaluated mineralization of PCBs in sludge-amended soils and found that, at a PCB concentration of 5 μg/g, mineralization was seen to be the major loss pathway.

Conclusions

Placement of dredged material in CDFs has been occurring for more than 25 years and provides a relatively inexpensive alternative for handling contaminated sediments. The finding of high concentrations of PAHs and low molecular weight PCBs at depth in the CDFs has some implications in the long-term management of these sites. As these compounds are relatively mobile, the long-term potential for migration of these compounds from the CDF needs to be assessed. The low molecular weight ortho-substituted congeners are less toxic but more mobile than the original Aroclor congeners. Anthracene (and potentially benzo[a]pyrene) is highly phototoxic to fish and benthic organisms (Landrum et al. 1987). Well monitoring presently being conducted will allow modeling of leaching rates through CDF dikes. Closure assessments will need to consider this migration pathway.

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Natural Attenuation of Explosives in Groundwater

by
Judith C. Pennington[1] *and Ted Ruff*[2]

Background and Rationale

Clean up of groundwater contaminated with explosives can be prohibitively expensive. Achieving concentrations that are acceptable to regulatory limits when concentrations reach low levels is often especially difficult. Natural attenuation may be an attractive alternative to available remediation technologies at sites that meet well-defined selection criteria, acceptable risk levels, and that satisfy specific regulatory concerns. Since natural attenuation is a relatively "non-invasive" technique, it has potential for application where structures, e.g., buildings and highways, prevent access to the surface, or where disturbances of the surface are undesirable, e.g., wetlands and endangered species habitat. Research had demonstrated that certain explosives are naturally immobilized in soil systems under specific environmental conditions. Other explosives are degraded by microbial and/or abiotic processes. Either of these sets of processes may provide evidence of attenuation in the subsurface. However, the hydrologic and geologic dynamics of the site groundwater together with the location and risk associated with potential contaminant receptors may also provide support for selecting natural attenuation as a remediation alternative. If the site is isolated making receptors far removed, and/or if the groundwater flow is extremely slow, risk to receptors may be deemed acceptable.

Natural attenuation is not a "no action" remediation alternative. Selection of natural attenuation requires a thorough understanding of relevant site hydrology, geology, and the contaminant plume. Natural attenuation also requires long-term site monitoring to assure that receptors are being protected and that no significant changes have occurred in anticipated contaminant migration toward receptors. The objectives of this study are to provide guidance to site personnel for selection of natural attenuation as a remediation alternative, for monitoring the site for prolonged protection and regulatory compliance, and for eventual site closure.

Natural attenuation has been defined as "the reduction of contaminant concentration to environmentally benign levels through natural processes" (ADL 1995). Active processes include, but are not limited to, advection, dispersion, diffusion, volatilization, biotic degradation (mineralization and transformation), sorption/desorption, ion exchange, complexation, abiotic transformation, and plant and animal uptake. This definition excludes any intervention to enhance natural processes.

Studies of natural attenuation for petroleum hydrocarbons by the Air Force in cooperation with EPA have contributed significantly to maturation of the technology (Downey and Hall 1994, Miller 1993a, Miller 1993b, Miller 1994, Stauffer 1993, Stauffer et al. 1994, Weber 1993, Wiedemeier et al. 1994). Natural attenuation of BTEX has also been investigated at several sites: Sleeping Bear Dunes, MI (Korreck 1993, Borden 1993); Sampson County, NC (Borden 1993); Rocky Point, NC (Borden 1993). At least three states have developed guidelines for permitting natural attenuation, Wisconsin (Barden 1993, Giesfeldt 1993a, Giesfeldt

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[2] U.S. Army Environmental Center, Aberdeen Proving Grounds, MD.

1993b), Florida (Larson 1993, Department of Environmental Regulations, State of Florida 1990) and Michigan (Shauver 1993).

Technical Description

Site Selection and Characterization

An ideal candidate site for demonstrating natural attenuation is one in which the groundwater has (1) sufficient residence time to allow natural attenuation processes to occur, (2) limited or no risk of contamination of local receptors, e.g., drinking water wells, streams, wetlands, and (3) a receptive local/regional regulatory community. We selected the Louisiana Army Ammunition Plant at Bossier City, Louisiana as our initial demonstration site for these and several other reasons. The source of contamination, several liquid waste storage lagoons, was removed six years ago. Absence of an active source of contamination will simplify the interpretation of the groundwater data and the plume migration. The site has been extensively characterized in terms of geology, hydrology and contamination in soils and groundwater. The area of interest already has over 40 groundwater monitoring wells; therefore, cost for installation of wells will be minimal. Contaminant data for many of these wells is available for each year since remediation of the lagoons. Region VI of the EPA has been receptive to considering natural attenuation as a remediation alternative for the site.

Although this demonstration site has received extensive characterization, we hope to develop procedures for site characterization that will minimize the cost of implementing site characterization for future users. We anticipate a need for good geology, hydrology and contaminant data that focuses on the objectives of natural attenuation. An objective of the study is to define an efficient, cost-effective sampling protocol.

To justify selection of natural attenuation, the site must be sufficiently characterized to support the following performance criterion: The rate of contaminant immobilization/degradation is sufficient to protect the nearest possible receptor over a "reasonable" period of time. The definition of "reasonable" will usually be dictated by results of local risk assessments and regulatory concerns.

Monitoring

A site monitoring plan consisting of monthly sampling of 30 existing wells has been initiated. Monthly sampling will occur for the first six months of the project only. After that, sampling will occur quarterly for the remainder of a two year period. The monthly sampling will allow us to refine sampling protocols, identify sources of variability and determine the degree of variability in the site. Parameters that will be monitored initially include the following: explosives and their transformation products, potential electron acceptors (iron, manganese, sulfate, nitrate/nitrite), dissolved oxygen, conductivity, pH, total dissolved solids, and water level. As research on the potential for use of stable isotopes of carbon and nitrogen and on biomarkers progresses, additional assays may be added.

Modeling

A three-dimensional, multicomponent transport code is under development to describe and predict the subsurface fate of explosives. While code preparation is being conducted on a Unix workstation platform; eventual migration to other platforms (personal computers to multiprocessor systems) is anticipated. Long-range plans include bringing a mature version of this code into the DoD Groundwater Modeling System (GMS), a standard user interface with which to visualize simulation input/output. Available GMS tools will be used to visualize hydrogeology, monitoring well data, and model predictions.

Guidance Document

A user-friendly guidance document will be prepared for use by facilities personnel and others who must consider natural attenuation as a remediation alternative. The document will include pertinent site characterization guidance, monitoring protocols, selection of compliance criteria, and guidance on site closure.

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Air Sparging/Soil Vapor Extraction Remediation, Landfill 4, Fort Lewis, Washington

by
William Goss[1]

Introduction

The air sparing process pumps fresh air, under pressure, into the water table aquifer. As the pressurized air travels through the contaminated groundwater it forms minute bubbles. These bubbles carry volatile organic compounds such as trichlorethylene upwards through the water table into the vadose zone. Air sparging can be thought of as an in-situ air stripper. The volatile organic compounds are then removed from the vadose zone by soil vapor extraction, which imparts a vacuum to the vadose zone. The air that is removed by soil vapor extraction is processed through granular activated carbon units to remove the volatile organic compounds. The treated air is discharged to the atmosphere.

At the request of the Environmental and Natural Resources Division, Public Works Department, Fort Lewis, an air sparging and soil vapor extraction pilot study was conducted at Landfill 4 in December, 1994. The pilot study was successful in removing contamination in the groundwater beneath Landfill 4. A remediation contract is due for award in the latter part of 1996.

Fort Lewis is located at the southern end of Pudget Sound, approximately 12 miles southwest of Tacoma, Washington, and 18 miles northeast of Olympia, Washington. Landfill 4, shown on Figure 1, occupies approximately 52 acres at surface elevations between 225 and 245 feet above mean sea level. No landfill records exist and it is surmised the waste materials consist of the type of refuse typical of unregulated municipal solid waste landfills. The Springs are located within the study area at the east end of Sequalitchew Lake. Sequalitchew Springs are the primary water supply for Ft. Lewis.

Nature and Extent of Contamination

Volatile organic compound (VOC) contamination in the Vashon Aquifer is primarily trichloroethene (TCE) and its degradation products, dichloroethene (DCE) and vinyl chloride (VC). Within and proximal to Landfill 4 (LF4), TCE concentrations in groundwater exceeded the maximum contaminant level (MCL) for drinking water (5 parts per billion (ppb)). The highest TCE concentration measured was 330 ppb. Detection of TCE was limited to the upper part of the Vashon Aquifer. DCE and VC were also detected in the Vashon Aquifer with maximum values of 12 ppb and 7.8 ppb, respectively. Tetrachloroethene (PCE) and TCE were commonly available degreasing solvents and historically were used at the Fort. DCE and VC are present at LF4 as breakdown products of PCE and TCE.

TCE and/or PCE disposed of outside the boundaries of LF4 would likely still exist as TCE, unless and anaerobic (oxygen-deficient) environment developed at the point of disposal. The TCE could partition as follows: onto soil particles; into underlying groundwater; into

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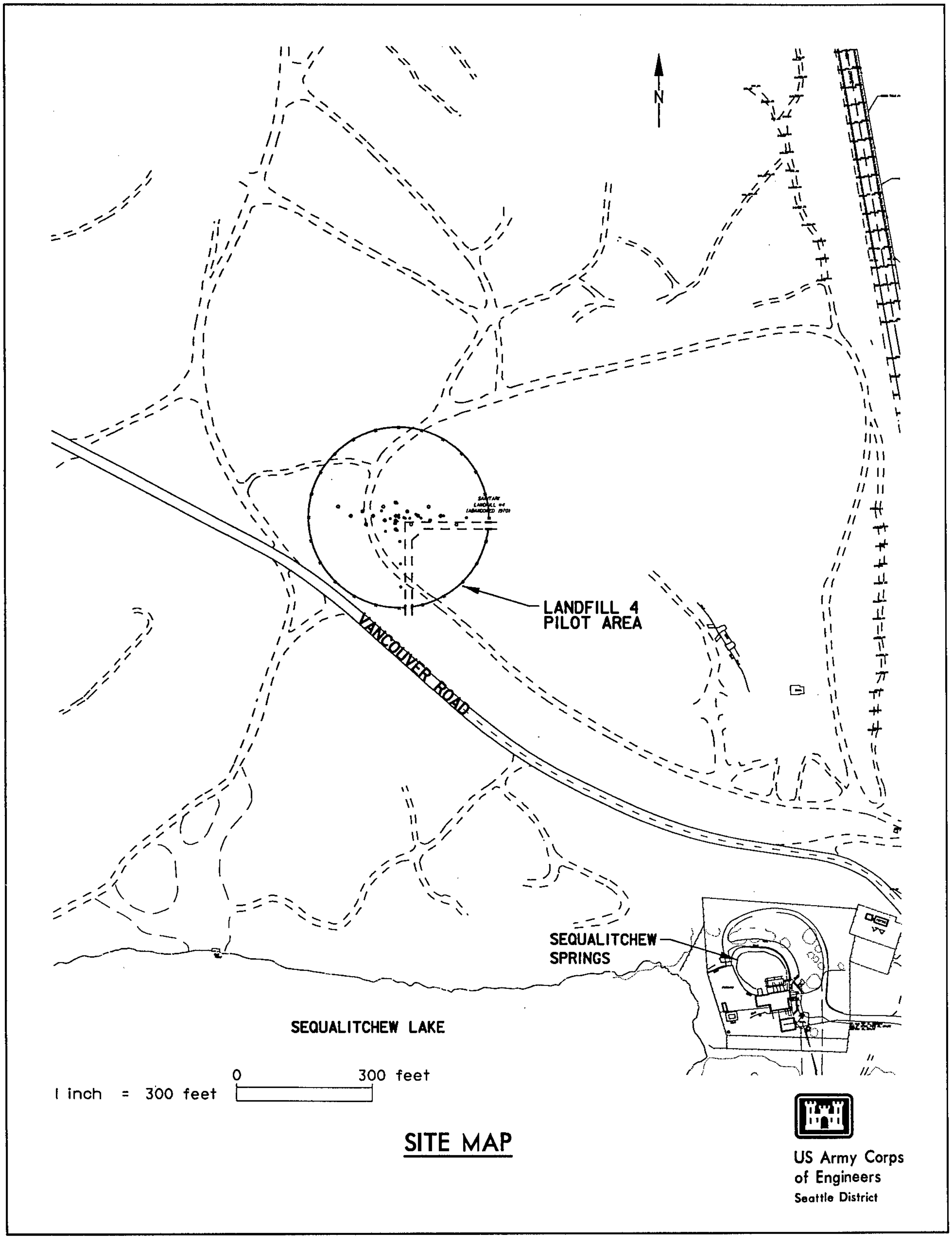
N
LANDFILL 4
PILOT AREA
VANCOUVER ROAD
SEQUALITCHEW
SPRINGS
SEQUALITCHEW LAKE
0
300 feet
1 inch = 300 feet
SITE MAP
US Army Corps
of Engineers
Seattle District

Figure 1.

infiltrating rainwater; and into the ambient air. The coarse soils at the LF4 site typically would not be amenable to the retention of TCE. However, if the TCE were co-disposed with oil and grease, the less mobile oils and grease would act to retain part of the TCE in the coarse soils. Areas where TCE has partitioned into unsaturated soil are suspected groundwater contamination sources.

Seasonal groundwater fluctuations of 3 to 4 feet were observed in the LF4 monitoring wells. TCE concentrations in groundwater are lower in the summer months corresponding with a lowering of the water table. This is attributed to the presence of contamination in a zone between the seasonal high and low water tables. The steepest hydrogeological gradient towards Sequalitchew Springs would be in the summer when the pumping rate at Sequalitchew Springs is at its peak. However, groundwater flow towards Sequalitchew Springs would have the least impact during the summer months because TCE concentrations are at their lowest. The retardation factor for TCE in the upper aquifer indicates that TCE in the groundwater will move at a velocity approximately one-third as fast as the groundwater, or about 1 foot per day to the south.

Pilot Study

A pilot study was undertaken to investigate the applicability of air sparging and soil vapor extraction (AS/SVE) as remediation technologies to clean groundwater at LF4. The objective of the pilot study was to install and operate a small scale AS/SVE system over a short duration and assess the potential to meet required cleanup goals within a reasonable time frame, such as 2 to 5 years. Maximum contaminant levels of 5 ppb for TCE and 1 ppb for VC are the regulatory limits prescribed for cleaning groundwater at this project. The main phases of work related to the pilot study include; pilot test design, well installation, cover installation, and the actual running of the pilot system. Types of wells installed for the pilot study include: soil vapor extraction (SVE) wells, monitoring wells (MW), vadose zone piezometers (VZP), air sparge (AS), and passive injection (PI) wells. The pilot program was run as a series of five 8-hour tests and a continuous 72-hour test.

Remediation and monitoring activities during the pilot study were conducted within the Vashon Formation, the uppermost water table aquifer beneath Ft. Lewis. Water bearing materials vary from sandy gravel to sandy silty gravel with clay. Overall, the materials are coarse with high porosity and permeability. Zones of lower porosity and permeability were encountered, but only in isolated lenses.

Soil vapor was sampled from the SVE effluent stream on an hourly basis. The samples were analyzed at an on-site mobile lab. The four analytes targeted in the soil gas analyses were; TCE, VC, DCE and PCE.

The TCE mass removal rate in soil-gas during the 72-hour test of the pilot study shows an overall decrease (Figure 2). The mass removal rate was calculated from the SVE flow rate and TCE concentrations. These were initially greater than 200 ppb (volume), but decreased and leveled off at approximately 60 ppb at the end of 72 hours. During the five 8-hour SVE tests an overall decrease in TCE concentrations was also noted.

Sparging produced no increase in the concentration of TCE from the SVE effluent stream. It is suspected that the addition of clean air into the sparge well diluted the air present in soil pore spaces thereby reducing TCE values. The air injection rate was less than 50 percent of the extraction rate (45 cfm vs 110 cfm). The contribution of TCE to the effluent stream from sparging is not discernible from soil gas data, but TCE concentrations in groundwater show a decline in TCE levels (Figure 3). Groundwater samples collected prior to and at the end of each sparging event (six rounds of groundwater samples) were analyzed for VOCs.

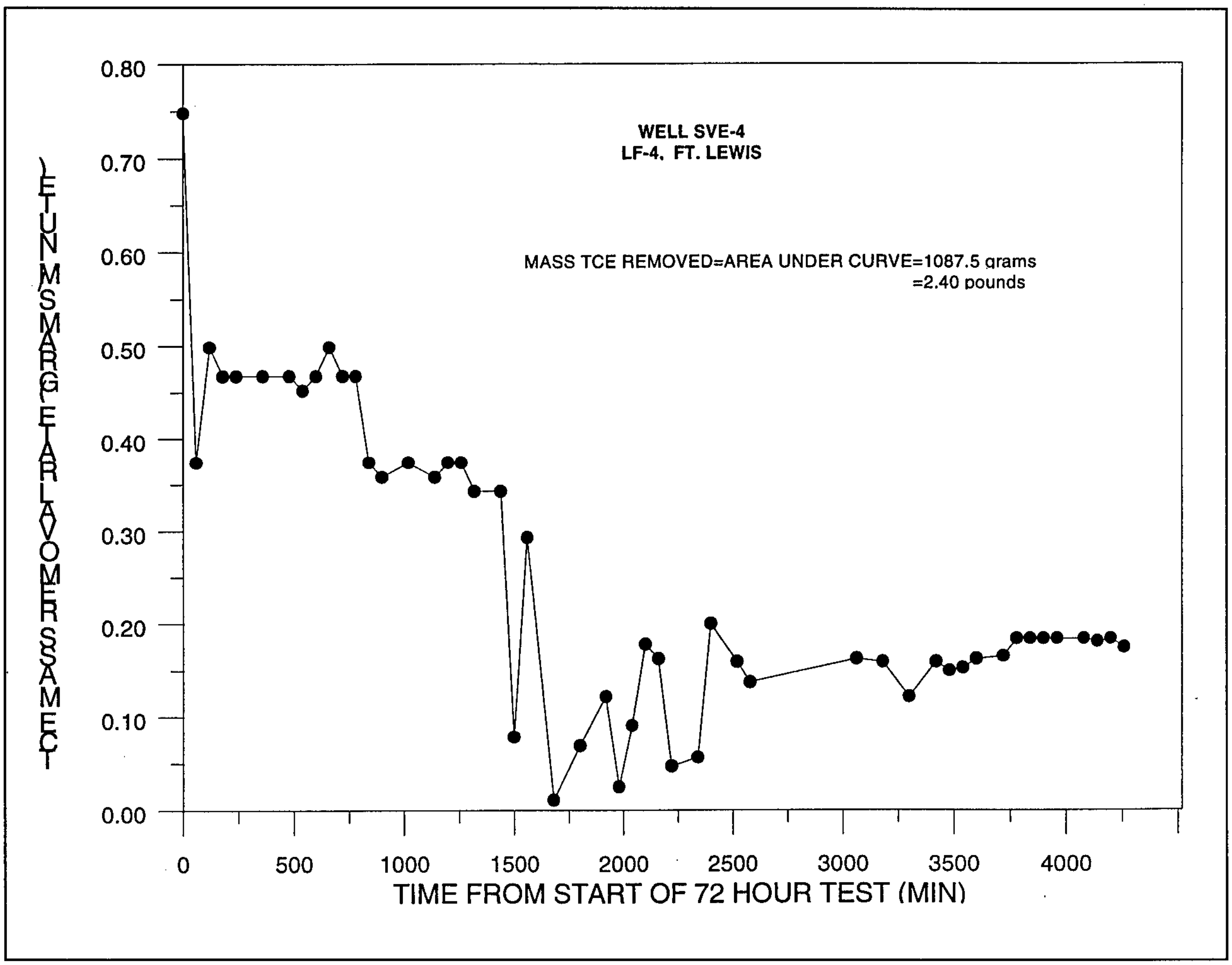

Figure 2.

During the course of the pilot study four separate sparge events were conducted. Each of the first three events were 6 to 7 hours in duration. Theses events occurred on 7 to 9 December, 1994. The fourth event took place during the final 48 hours of the 72-hour continuous test between 13 and 15 December. Concentrations in TCE for monitoring well MW8A are shown on Figure 3. TCE decreased in the well during the second and fourth events, and showed an increase during the first and third events. The decreases in TCE concentrations are related to sparging and the increases may be attributable to introduction of new source material to groundwater such as infiltrating precipitation. TCE concentrations fluctuated up and down during the pilot study, but the net effect was a decrease in TCE concentration.

Treatment of the vadose zone by SVE should reduce the amount of contamination entering the groundwater. Biodegradation of TCE is an anaerobic process. The processes of AS and SVE will inhibit anaerobic activity by imparting aerobic conditions to the subsurface. However, the introduction of additional oxygen into the subsurface may accelerate the biodegradation of non-chlorinated hydrocarbons such as oil and grease. TCE is sorbed to these compounds and released slowly. Accelerating the breakdown of oil and grease compounds may increase the desorption rate of TCE from oil and grease compounds. This desorbed TCE will be removed from the vadose zone by SVE.

During the air sparging and soil vapor extraction processes, dissolved oxygen levels

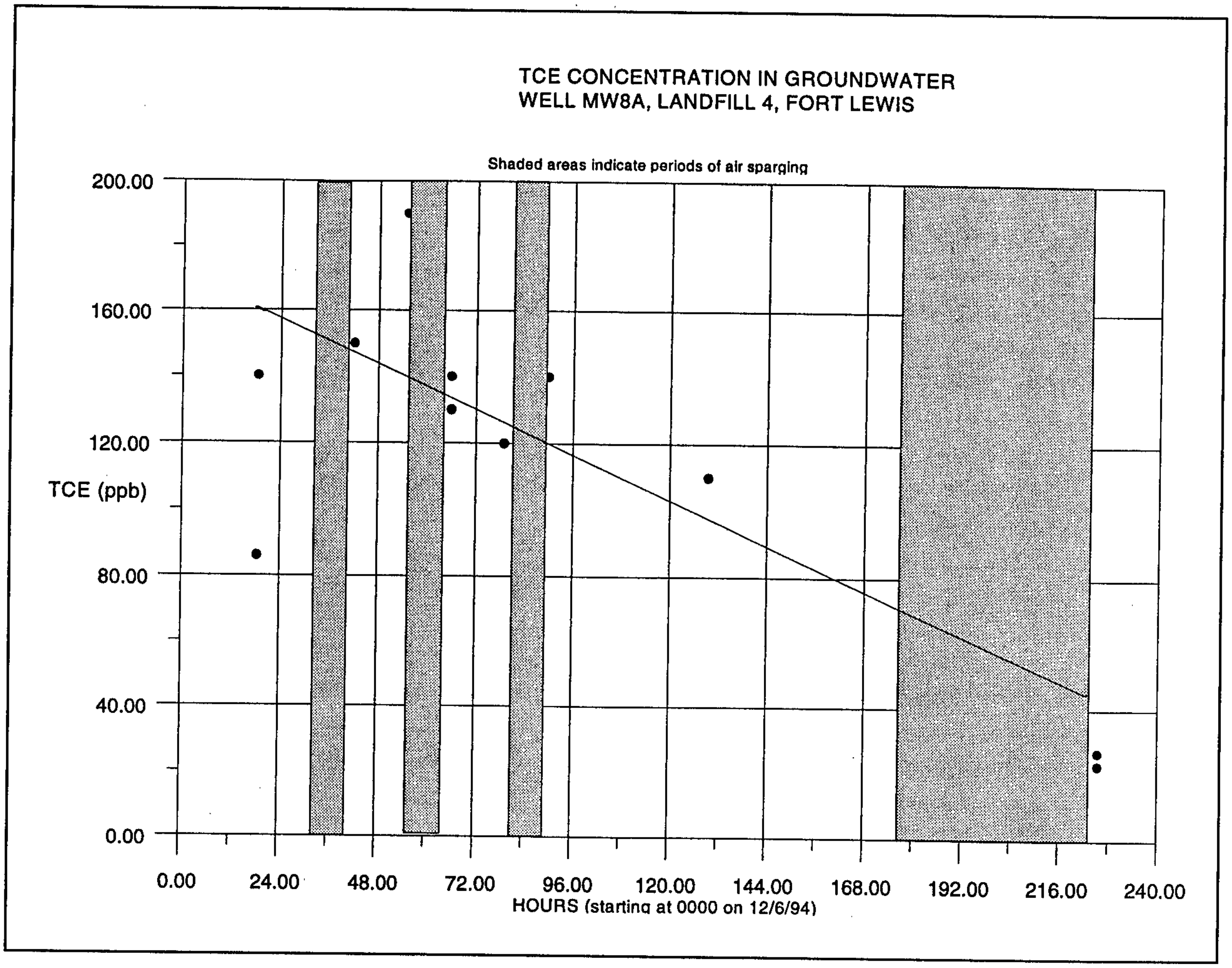

Figure 3.

were obtained from two dissolved oxygen sensor wells and one existing monitoring well. The levels were used to determine the radius of influence of air sparging. Eleven vadose zone piezometers installed in the pilot area were used to measure the vacuum induced by the soil vapor extraction and to determine the radius of influence of the SVE system.

The concentration and extent of TCE in groundwater and the vadose zone have not been fully determined. Without this information the mass balance calculation of contamination to be removed cannot be compared with the TCE extraction rate obtained during the pilot study.

Summary

The use of AS/SVE technology should result in a short-term decrease in volatile contaminants found in groundwater that meets or exceeds the cleanup level for TCE of 5 ppb as stated in the Record of Decision (ROD). Estimates to clean-up the highest concentrations of TCE observed in groundwater within the 3-year goal stated in the ROD are reasonable as long as no new source of contamination is introduced.

Removal of TCE from groundwater and the source area (the vadose zone) utilizing air sparging and soil vapor extraction is feasible as shown by the pilot study. Approximately 2.4 pounds of TCE were removed during the 72-hour portion of the pilot study (Figure 2). Results from the pilot test and from a preliminary 3-dimensional model indicate that an extraction rate of approximately 110 cubic feet per minute (cfm) will likely capture all volatilized contaminants within approximately 200 feet of the SVE well. The pilot test also shows that the radius of influence of the air sparging wells is about 25-30 feet, indicating that multiple air sparge wells will be required to remediate the site.

Currently preparations for plans and specifications for a remediation system are underway. The system will consist of 5 AS wells, 6 SVE wells, 4 PI wells, compressors, vacuum pumps, associated piping, valving, appurtenances and carbon treatment systems. It is anticipated the system will run for a period of 3 years with concurrent monitoring of groundwater and soil gas. Vapor effluent will be treated with carbon prior to atmospheric discharge. After 3 years of AS/SVE remediation an evaluation period will follow to determine if further remedial action is required.

Remediation of Explosives-Contaminated Groundwater at Umatilla Army Depot Superfund Site, Oregon

by
Michael M. Easterly[1] *and Richard E. Smith*[1]

Background

Umatilla Depot Activity (UMDA) is a 20,000-acre Army ordnance facility in northeastern Oregon, about 180 miles east of Portland, Oregon, approximately 3 miles south of the Columbia River. The Corps of Engineers, Seattle District is currently performing Remedial Design and Remedial Actions (RD/RA) for UMDA at six operational units. UMDA is commanded by Lt. Col. Marie Baldo, and the UMDA Environmental Coordinator is Mark Daugherty.

The Army originally purchased the UMDA property in 1940 and used the installation as a storage facility for both conventional and chemical weapons. Beginning in the 1950's, UMDA operated an explosives washout plant where munitions were opened and explosives removed with hot water. The wash water was disposed of in two nearby unlined lagoons. The Explosives Washout Lagoons received a total of about 85 million gallons of wash water over a period of about 12 years. Although wash-water sludge was removed regularly during operation, dissolved explosive compounds migrated into the soil and ground water about 47 feet below the Lagoons.

UMDA became an Installation Restoration Program site in the late 1970's; various investigations were conducted through the 1980's, which showed explosives contamination in both soil and ground water at the Explosives Washout Lagoons site. The Lagoons were placed on the National Priorities List in 1987, and in 1989 a Federal Facilities Agreement was signed between the Army, Oregon Department of Environmental Quality, and EPA Region X.

A remedial investigation and feasibility study (RI/FS) of the entire UMDA installation was initiated in 1990 by the Army Environmental Commission (AEC) and their contractor Dames and Moore Inc. Approximately 40 monitoring wells were installed, and pumping tests were performed at some of the wells in conjunction with a UV oxidation ground water treatability test.

The Feasibility Study Report was completed in 1994 by Arthur D. Little Inc. A. D. Little used USGS's MOC ground-water transport code to examine pump-and-treat alternatives, and explored various water treatment methods. Locations for three extraction wells and one infiltration field were identified, and the preferred alternative consisted of a 10-year remediation using an activated carbon water treatment plant. It was also determined that the contaminated soil under the Lagoons could be remediated by recharging the treated water into the Lagoons after the surface soils were excavated.

The Record of Decision (ROD) for the Washout Lagoons Ground-Water Operable Unit was signed in 1994, calling for extraction of the ground water over an estimated 10- to 30-year period, water treatment by granular activated

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carbon (GAC), in-situ flushing of soil beneath the Lagoons with treated ground water for an estimated period of one year, and reinfiltration outside the RDX plume of treated water that is not recharged to the Lagoons. The Remedial Action (RA) contract was awarded to ICF Kaiser Engineers Inc. in September 1995, and construction of the ground-water treatment facilities is scheduled to begin in March 1996.

The upper 10-15 feet of contaminated soil under the Washout Lagoons were removed in September 1994, and are currently undergoing bioremediation by composting.

Site Conditions

Hydrogeology

Umatilla Depot is located in the Deschutes-Umatilla Plateau. Surface soils consist primarily of Quaternary Lake Missoula flood deposits overlying Miocene basalt flows. The Explosives Washout Plant is located on the eastern edge of Coyote Coulee, a 0.25-mile-wide, 50-foot-deep linear depression formed by the last flood episode. Explosives washout water was discharged via a trough from the plant to the Lagoons at the bottom of the slope.

The alluvium consists of 90 to 150 feet of sand and gravel with occasional silt beds and lenses. The thickness of the silt layers varies from a few inches to several feet, and their frequency increases west of the Lagoons. The upper, unconfined aquifer, known locally as the Ordnance aquifer, occupies the lower 30 feet of the alluvium. A 5 to 10-foot-thick silt layer generally occurs at the aquifer base immediately above basalt bedrock. The top of the basalt is generally weathered and permeable to a depth of about 10 feet. The thickness of the alluvium and the unconfined aquifer increase to about 180 and 70 feet respectively at the south boundary of UMDA. The aquifer thins to the north and pinches out about 2400 feet north of the Washout Lagoons against a fine-grained flood deposit known as the Northern Aquifer.

The ground-water level at the site varies about 4 feet seasonally with an average elevation of about 497 feet (NGVD), probably in response to irrigation pumping outside the UMDA boundaries. Average gradients across the contaminant plumes, determined from year-round well data, vary between extremes of 0.00015 ft/ft NW in late spring, to 0.00020 ft/ft SE in late summer. The average gradient is 0.00005 ft/ft ESE. The average ground-water velocity in the Lagoon vicinity is estimated at 180 feet/year ESE. Average annual precipitation at UMDA is 9.01 inches/year, but infiltration is expected to be only 0.5 inches/year. Well data from outside the Depot indicates that the average water level in the Ordnance Aquifer dropped 16 feet from 1965 to 1973 as agricultural irrigation increased, and then rose 11 feet from 1977 to 1984 when irrigation canals were installed from the Columbia River. No data for these variations exists at the Washout Lagoons site.

Contamination

Soil and ground water under the Lagoons are contaminated with nine explosive compounds. A 4500-foot long by 5500-foot wide hourglass-shaped RDX plume stretches downgradient from the Lagoons, with a maximum concentration of 3000 $\mu g/L$ (see Figure 1). A smaller, 1100-foot-diameter TNT plume is located close to the Lagoons, with a maximum concentration of about 6000 $\mu g/L$. HMX, Tetryl, 2,4-DNT, 2,6-DNT, TNB, DNB, and NB all exist in lesser amounts with plume sizes and locations similar to the TNT plume. The ground water also contains nitrate over approximately the same area as the RDX plume with levels as high as 40 mg/L. Monitoring wells show that contamination diminishes rapidly with depth in the silt and weathered bedrock beneath the Ordnance Aquifer, and has not penetrated into the deeper basalt aquifers.

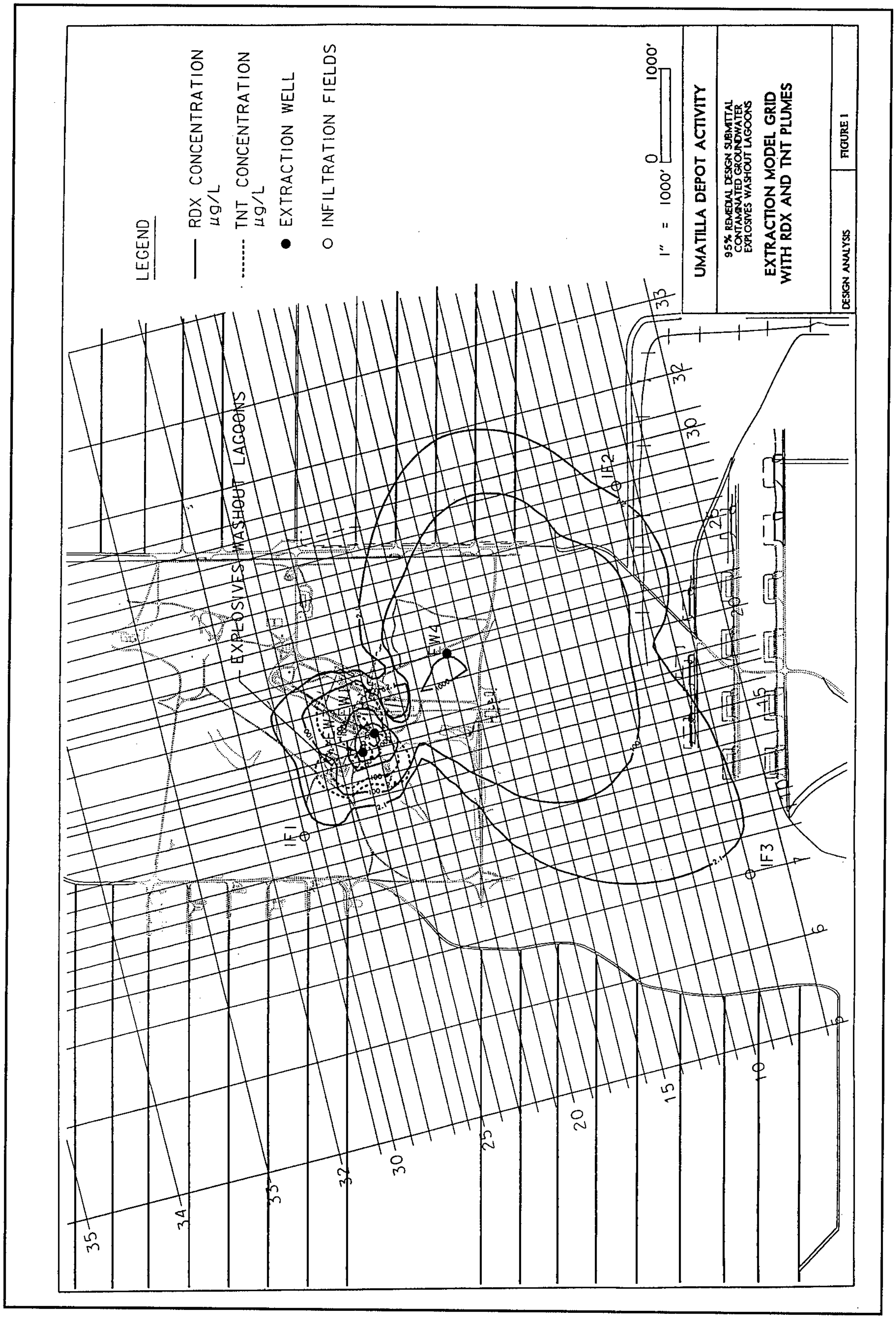
LEGEND
RDX CONCENTRATION µg/L
TNT CONCENTRATION µg/L
EXTRACTION WELL
INFILTRATION FIELDS
1" = 1000'
0
1000'
UMATILLA DEPOT ACTIVITY
95% REMEDIAL DESIGN SUBMITTAL
CONTAMINATED GROUNDWATER
EXPLOSIVES WASHOUT LAGOONS
EXTRACTION MODEL GRID
WITH RDX AND TNT PLUMES
DESIGN ANALYSIS
FIGURE 1
EXPLOSIVES WASHOUT LAGOONS
EW4
IF1
IF2
IF3

Figure 1.

Remedial Design Field Investigation

Seattle District began work on the Remedial Design in January 1994. The project team agreed that more subsurface information was needed before the design could proceed, so a field investigation was performed during summer 1994. Two extraction wells were installed and tested at locations recommended in the FS, and twelve monitoring wells were installed to provide additional data for pumping tests. Ground water discharged during the extraction well pumping tests was passed through a 20,000-lb granulated activated carbon (GAC) treatment plant. In addition, infiltration tests were performed in the Washout Lagoons excavation and at anticipated infiltration gallery sites.

Analyses of pumping test data showed horizontal hydraulic conductivities in the range of 1000 to 6000 feet per day, specific yield values from 0.22 to 0.35, and ratios of horizontal to vertical hydraulic conductivity from 30 to 300.

Remedial Design Modeling

Two contaminant transport models were used for the remedial design: a 3-dimensional, unconfined, saturated zone model which simulated the contaminant plume, extraction wells, and infiltration fields; and a 3-dimensional variably saturated model simulating the flushing of Lagoon soils above the static water level. Both models utilized the finite-volume flow and transport code PORFLOW (ACRI 1993). The code was selected because of its robust capabilities, including multispecies transport and a wide variety of source and boundary options. Initially only RDX and TNT were simulated in the models because the other contaminants of concern (COC) rank between RDX and TNT in mobility (K_d value), and they all exist in lower concentrations and smaller plumes than either RDX or TNT. Nitrate contamination became an issue during regulatory reviews of the Remedial Design Report, and nitrate transport was finally added to both models.

Extraction Model

The extraction model utilizes a 38-row by 38-column by 10-layer, variably-spaced grid oriented orthogonal with the axes of anisotropy determined from pumping tests (see Figure 1). The horizontal node spacing is 200 feet in the central area; vertical node spacing is 10 feet.

Calibration runs were performed until the model successfully reproduced Northern Aquifer water level contours presented in the RI report, average gradient and water level elevations in the Ordnance Aquifer, and approximate RDX and TNT plume sizes and concentrations shown by RI sampling.

Various pumping and recharge configurations were used in the simulations, and pumping rates were adjusted to cause contaminant concentrations to decrease below the remediation goals within 10-year and 30-year cleanup times. The first simulations used the configuration recommended in the FS Report: 10-year cleanup with a single primary infiltration site 700 feet northwest of the Washout Lagoons with 200 gpm initially recharged into the Lagoons. The duration of the Lagoon recharge was determined by the flushing model, and well discharges were adjusted to remove the plumes in 10 years. Simulations performed with only one infiltration field had difficulty retrieving the southern portions of the RDX plume, and very high pumping rates were required to meet the 10-year goal; therefore a second set of simulations was performed using 3 infiltration fields. The final simulation utilized 3 extraction wells and 3 infiltration fields, with flow rates adjusted to produce a 30-year cleanup.

The sensitivity analysis showed the model to be slightly to moderately sensitive to input parameters, with distribution coefficient (K_d) and hydraulic conductivity variations causing the greatest effect in the results.

Flushing Model

The flushing model provided Lagoon infiltration rates and durations, as well as leachate concentrations to be used as input for the extraction model. The K_d values reported in the RI were determined from batch desorption tests; therefore TNT was simulated with a K_d equal to 5 times the RI value to allow for differences between sorption and desorption isotherms. The flushing model utilized a 23-row by 23-column variably-spaced grid with a 20-foot horizontal node spacing in the central area. Vertical node spacing ranged from 1 to 10 feet. Grid orientation was the same as for the extraction model, and as many nodes as possible coincided with extraction model nodes to facilitate data transfer and possible future changes. The discharge of treated water into the Lagoons was simulated as a areal flow source distributed over 36 nodes within the Lagoon excavation. Contaminated soil was simulated as six flow-driven solubility-limited contaminant source zones.

Calibration consisted of adjusting empirical soil-moisture coefficients until the model reproduced the same retention characteristic demonstrated by field measurements.

Three Lagoon infiltration rates were simulated: 200, 400 and 600 gpm. Contaminant concentration vs. time was monitored at seven nodes near the phreatic surface. In all cases, the concentration of TNT with K_d = 5 ml/g took the longest to drop below the remediation goal. The TNT concentration of the leachate was therefore used to determine cleanup times.

The sensitivity analysis showed only slight to moderate sensitivity to all inputs except infiltration rate, which had a major effect on cleanup time.

Results

The extraction model showed that the FS preferred alternative with three extraction wells and one infiltration field would not efficiently retrieve all of the RDX plume. A more economical cleanup could be performed using three infiltration fields surrounding the contaminant plumes. The remediation could be performed in 10 years with a total extraction rate of 4000 gpm from five wells, or a 30-year cleanup could be accomplished with a 1300-gpm extraction rate from only three wells.

Both the field infiltration tests and the flushing model showed that several thousand gpm of treated water could be infiltrated in the Lagoons, however an infiltration rate greater than 400 gpm would spread the existing contaminant plumes. The flushing model showed that a 400-gpm infiltration rate could remediate the Lagoon soils in about two years.

Issues, Philosophy, and Lessons Learned

Partnership Between Designers, Owner, and Regulators

The project has benefited from close communications and good relations between team members, including owner and regulator representatives. So far, an equitable balance has been maintained between the customer's economic requirements and the regulatory agencies' continuing desire for more data and analysis.

Importance of the Remedial Design Field Investigation

The consensus of the project team regarding the lack of design data in the FS was well-founded. The proposed design in the FS was based on a 2-D transport model with a uniform aquifer thickness, along with underestimated specific yield and hydraulic conductivity values from slug tests and pumping tests of small-diameter wells. The FS gave a total design discharge of 333 gpm for a 10-year remediation; RD studies based on a 3-D model with data from extraction well pumping tests predicted a 3900 gpm total extraction requirement for the 10-year cleanup.

Practical Approach to Modeling

Modeling studies have a reputation for being expensive and time-consuming, and requiring specialized modeling experts with high-powered computers. The Washout Lagoons modeling study was performed in two months, entirely on desktop PC's, and by two geologists who had also supervised the field investigation. An aggressive project schedule and austere budget were both met, and technical continuity was maintained through the design process.

During the extraction model calibration phase, an attempt was made to recreate seasonal gradients and alleged historical water levels. This effort was finally abandoned due to time constraints and the lack of definitive historical data. Instead, initial concentrations were mapped onto the extraction model grid from field sampling data, and the remediation simulations were run using present hydrogeologic conditions. Regulatory reviewers supported this approach, taking the position that operational adjustments during the remediation were inevitable, thus the modeling effort was probably nearing the point of diminishing returns.

Regulatory Focus

Since the ROD defines the project as a pump-and-treat ground-water cleanup, hydrogeology, transport modeling and well design might be expected to be central issues; however, reviewers tended to focus on other, more peripheral areas during the design process.

Lagoon soil flushing: The most heavily-contaminated Lagoon soils had already been remediated under a previous action, and the ROD stipulated that recharge in the Lagoons was merely a beneficial use of treated ground water. EPA continued independent studies after the RD was completed, however, and requested that the Army revisit the soil flushing analysis.

Treatment plant carbon: GAC usage and disposal was a prime area of interest for reviewers. EPA proposed to add instrumentation to the carbon vessels, and requested additional sampling and data recording at the treatment plant by the RA contractor.

Nitrate contamination: Although explosives were the only COC's listed in the ROD, the State of Oregon was concerned about the possible spread of nitrate by the discharge of treated water in the infiltration fields. The Corps included nitrate to the extraction model and eventually added monitoring wells to the RA contract to detect nitrate migration.

Chemistry: Sampling protocol, analytical methods, and data management were ongoing and consuming issues throughout the design reviews, and promise to be a major preoccupation throughout the project life.

Pump-and-Treat as "Conventional" Technology

Although the extraction, treatment and recharge of contaminated water is now regarded as a traditional practice, the technology continues to evolve. Better analytical methods are being developed, equipment and instruments are becoming more sophisticated, and the practitioners are still learning.

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Cheers to NORM

by
Cheryl G. Peyton[1]

Introduction

Someone yells NORM and we all look, expecting a stocky, out-of-work accountant to darken the doorway and hasten his way to the familiar stool on the corner of the bar. Other than the fact that the beer nuts may be served in fiesta ware that is slightly radioactive, there is very little connection between that image and the subject of this paper. NORM, in fact, is Naturally Occurring Radioactive Materials, and is a subject that is receiving much attention lately.

NORM is a subgroup of Natural and Accelerator-produced Radioactive Materials (NARM). Obviously, Accelerator-produced Radioactive Materials (ARM) is the other subgroup. While extensive regulation exists for certain types of radiation related activities, very little currently exists for NARM. As early as the 1970's, various agencies began recognizing the need for regulation on the federal level,[2] but politics interfered and to date, very little has been done. The Environmental Protection Agency (EPA) has a few guidelines with promises for more. However, the bulk of the regulation is on the state level.

Radiation is all around us - in the marble in our hallways and tables, the potassium in our food, the cosmic radiation from the sun and outer space, so on and so forth. It is a natural part of our lives and has been since the beginning of time. The average person in the U.S. receives approximately 360 mrem/yr from background and other radiation sources.[3]

Sources of NORM include cosmic radiation (penetrating radiation from the sun and outer space that bombards the earth's atmosphere), internal radiation (natural radionuclides found in the food we eat, the water we drink and the air we breath), and terrestrial radiation (radiation from natural nuclides in the earth's crust, mostly radium-226 (^{226}Ra), uranium-238 (^{238}U), thorium-232 (^{232}Th), and potassium-40 (^{40}K)).[4] Regulated NORM comes from the terrestrial radiation group and is generally associated with the oil and gas production, fertilizer production, and military instrumentation applications, among others. The focus of this discussion will be NORM associated with the oil and gas industry.

NORM in the Oil and Gas Industry

Several isotopes are the primary sources of NORM associated with the oil and gas industry. These are primarily ^{226}Ra, a daughter product of ^{238}U, ^{228}Ra, a daughter product of ^{232}Th, and radon-222(^{222}Rn), a daughter product of ^{226}Ra. Uranium and thorium are found in conjunction with the sandstones, shales, and other rocks

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1 U.S. Army Engineer District, New Orleans; New Orleans, LA.
2 Francis X. Cameron, "The Odyssey of the Good Ship NORM: The Search for a Regulatory Safe Harbor," *NORM/NARM: Regulation and Risk Assessment, Proceedings of the 29th Midyear Topical Meeting*, Health Physics Society, 1996, p. 14.
3 *Norm Survey and Control Certification,* (Baton Rouge, LA: Radiation Technical Services, 1994), p. 36.
4 Ibid.

which host oil and gas. The act of drilling a well brings these radioactive isotopes to the surface in a petroleum emulsion containing oil, gas, water, and sediment and in the drill tailings, although, the level is usually low. Radium from the tailings can become dissolved in waters at the surface (e.g., leachate and runoff in contact with the tailings). The produced liquids from any production activity may contain dissolved radium, the primary type of NORM in the oil and gas industry.

If the well produces sufficiently and is placed into production, more NORM may be brought to the surface in the produced liquids. Obviously, the longer a well produces, the more NORM can be brought to the surface. The radium, like many other minerals, tends to remain dissolved until something causes it to precipitate out. Thus, contamination will follow the course of the contaminated liquids.

The produced liquids, drilling fluids, and completion fluids are usually contained in pits or lagoons. However, sometimes containment is not provided, allowing the fluids to run overland or the fluids may breach containment. Thus, NORM contamination from a well can travel far from the source, perhaps even miles, if conditions exist for it to do so. Such may be the case, for example, if the contaminated liquids enter a surface water pathway, such as a ditch, stream, or channel.

When a pit or lagoon is used, it may be operational for a very long period of time, sometimes as long as 30 years or more. The sludges, and thus, the radiation levels in the pit become more and more concentrated over time as water continually evaporates or otherwise leaves the pit. Often, pits are closed by merely filling and capping, leaving no visible sign at the surface of the previous activity and associated danger. Likewise, the scales which build up on the inside of transport pipes may also contain high levels of radiation. When the pipes are reamed out for maintenance, the dust generated is a source of air contamination, and the disposed scales are a potential source of soil and groundwater contamination. One study has shown, however, that the NORM from pipe scale has a very low solubility internally, due most likely to the fact that it is bound in the same matrix that includes barium and radium sulfate.[1]

The Toxicology of NORM

Types of Radiation

Radiation damage occurs when the radiation interacts with the atoms that make up the cells of our bodies. Three of the primary types of ionizing radiation are alpha, beta, and gamma, each with different properties and means of interacting with other atoms.

Gamma radiation is an energy wave, having no mass or charge. Remembering that atoms are made up of electrons (negative charge) orbiting a nucleus (positive charge) with a lot of empty space in between, the only way gamma radiation can influence an atom is to directly hit one of the electrons, knocking it out of orbit. Since atoms comprise so much empty space, it is unlikely that gamma rays will collide with the electrons. As such, gamma rays tend to go a long way without interacting with their environment. Or, putting it another way, denser mediums are better at attenuating gamma rays. About an inch and a quarter of lead, or 6 inches of concrete, or about 60 feet of air would be required to attenuate 90 percent of the gamma rays normally associated with NORM.

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1 Otto G. Raabe, "Studies of the Solubility of Naturally-Occurring Radionuclides in Petroleum Pipe Scale," *NORM/NARM: Regulation and Risk Assessment, Proceedings of the 29th Midyear Topical Meeting*, Health Physics Society, 1996, p. 121.

Beta radiation is particulate in nature. It is a relatively small particle with the mass of one electron, and may be either positively or negatively charged. Because it has a charge, beta radiation can interact with other atoms not only by direct collision, but also by attraction or repulsion of charge. Thus, a beta particle only has to pass near an electron in order to influence it. As a result, beta particles tend to interact more with the environment than gamma rays. Only an eighth inch thickness of aluminum is required to stop most beta particles. The beta radiation normally associated with NORM will only travel about three to six feet in air.

Alpha radiation is also particulate in nature. It is a much larger particle than beta radiation, though. It consists of two protons and two neutrons. Note that this is a helium nucleus. In fact, an alpha particle neutralizes when it gains two electrons and becomes a helium atom. The alpha particle, being very large and having a +2 charge, interacts readily with other atoms, both by direct collision and by attraction or repulsion. A piece of paper or the dead layer of skin on our bodies is enough to stop alpha particles. They will only travel a few centimeters in air.[1]

Each radionuclide consistently emits its own signature combination of alpha, beta, and/or gamma radiation. The radionuclides normally associated with NORM almost always emit gamma radiation in combination with either alpha or beta radiation. This becomes important in surveying for the detection of NORM. Radium 226, one of the more common concerns associated with the oil and gas industry emits both gamma and alpha radiation, predominately alpha.

Interaction in the Body

Because radium is a mineral and is similar in nature to calcium, if it enters the body, the body will treat it as though it were calcium, concentrating it in the bones. The effective half life of a nuclide is a measure of the actual half life of the nuclide inside the body based upon a combination of the half life of the radionuclide and the removal rate of the nuclide through normal body functions. The amount of the nuclide that the body takes up out of the contaminating material is dependent upon the matrix of the material (how soluble is the radionuclide in the matrix?), the abundance of the nuclide in the material, and the contamination pathway. The effective half life of ^{226}Ra is 43.8 years.[2] Given this length of time, it is likely that a contaminated individual would retain a significant amount of the radiation for the remainder of his/her life. Since alpha radiation is a very large particle, it has the ability to do tremendous damage to the internal tissues of the contaminated individual. The chronic effect of alpha radiation exposure is considered to be cancer.

In the oil and gas industry, radium is usually a soil or water contaminant. The most likely contamination pathways are inhalation (breathing contaminated dusts, say at a construction site) and ingestion (eating with unwashed, dusty hands or drinking contaminated water). Good hygiene and dust control and/or breathing protection are the best methods of controlling contamination at a construction site. Below is a table outlining exposure limits:

Table 1
Exposure Limits

Body Part	REMs/quarter
Whole body; head and trunk; blood-forming organs; lens of eyes or gonads	1-1/4
Hands and forearms; feet and ankles	18-3/4
Skin of the whole body	7-1/2

1 *Norm Survey and Control Certification*, (Baton Rouge, LA: Radiation Technical Services, 1994), p. 16.
2 Ibid., p. 20.

Surveying for NORM

There are two types of surveys for NORM contaminated or potentially contaminated equipment and lands. The first is the confirmatory survey to determine if the potential for NORM exists in an area. The confirmatory survey is a screening level effort.

The second is the release survey. This is a comprehensive survey performed prior to the release of equipment or land for unrestricted use. For a release survey, the entire area to be released must be surveyed and sampled.

The regulations are based upon alpha radiation emissions in the units of picocuries/gram (pCi/g). However, these emissions can only be determined through laboratory analysis. Screening level analysis can be done in the field for gamma radiation which is measured in microroentgens per hour (uR/hr) using a scintillation detector. Thus, since ^{226}Ra is both an alpha and a gamma emitter, surveys are conducted for gamma radiation to find "hot spots." Samples are then taken at the "hot spots" for laboratory analysis of alpha radiation. What is considered a "hot spot" depends upon whether equipment or soils are being measured and what state you are in. Figure 1 shows a generalized map of gamma radiation contamination in the U.S.

Regulations for NORM

History

Beginning in the early 1970s, agencies, organizations, and task forces studying the issue began to call for the regulation of NORM by the federal government. Early recommendations were for the Nuclear Regulatory Commission (NRC) to expand it's jurisdiction to this area. The NRC consistently declined, however, suggesting that such regulation more appropriately belonged to the EPA or the states. Regulations from EPA have trickled with very little specifically addressing the issue. The EPA does address NORM, however, under the drinking water standards (criteria: 5 picocuries/liter (pCi/L) of ^{226}Ra and ^{228}Ra, 15 pCi/L of gross alpha beta activity including ^{226}Ra but excluding uranium and radon), in the proposed "Federal Radiation Protection Guidance for Exposure of the General Public," and in the Clean air Act's National Emission Standards for Hazardous Air Pollutants (NESHAPs) (40 CFR 61).[1] Further regulation of NORM by the EPA is in the works, but may be slow in coming.

One of the organizations originally calling for federal regulation was the Conference of Radiation Control Program Directors (CRCPD), made up of directors of the state radiation control programs. As the federal government's reluctance to develop regulations became apparat, the CRCPD resolved to assist the states in taking the lead by developing model state regulations. The bulk of the regulation is currently occurring on the state level. Eight states now have regulations specifically addressing discreet NORM, with several others in the process of developing regulations or guidance.[2]

Louisiana Regulations

Louisiana regulations are divided between equipment and soils. Equipment with gamma readings of more than 25 uR/hr are regulated. If a company is not yet registered with the state, equipment with alpha emissions of 5 to 30 pCi/g may be exempt from regulation. However, if the company has equipment or materials

[1] F. L. Galpin and V. C. Rogers, "NARM Classifications: Their Ethical and Regulatory Implications," *NORM/NARM: Regulation and Risk Assessment, Proceedings of the 29th Midyear Topical Meeting*, Health Physics Society, 1996, p. 5.
[2] Ibid., p. 6.

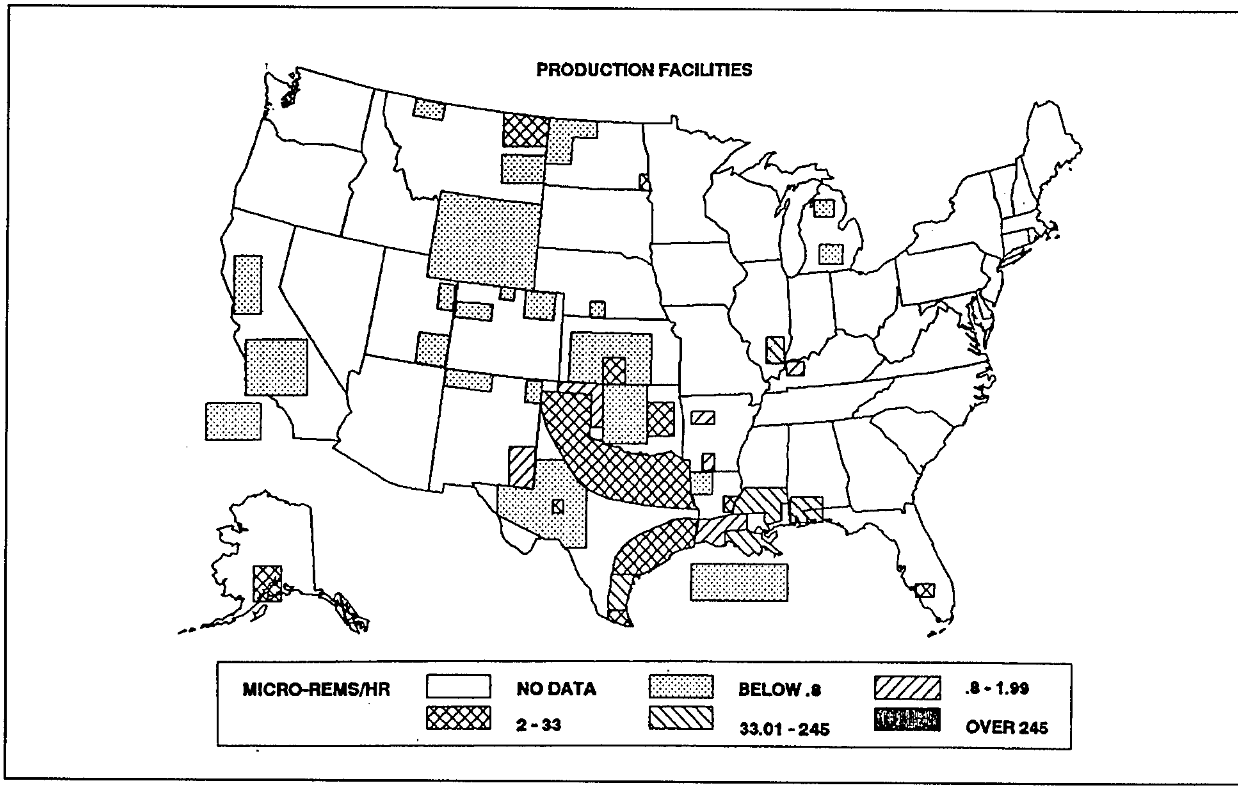

Figure 1. American Petroleum Institute National Survey on NORM - 1989 Production Facilities (Values shown are the median of readings less background[1])

registering levels of more than 30 pCi/g, they must become licensed. Once registered, any equipment or materials registering more than 5 pCi/g must be licensed. For soils, any area exhibiting gamma radiation levels more than twice background are suspect. Background levels for Louisiana are approximately 4-10 uR/hr.[2] Soil samples may then be taken for alpha emission analysis. Readings are averaged over a 100 square meter area. The alpha radiation criteria for soils is 5 pCi/g above background in the first 15 cm layer and 15 pCi/g in any subsequent 15 cm layer.

Liabilities

Of a far greater concern than the actual regulations are the liabilities established by the courts and the insurance industry. Unfortunately, the liabilities are determined, not necessarily by the actual risk based on a scientific evaluation, but rather by the public's perception of risk. Peter MacDowell, in his paper *The Insurance Industry's Proactive Response for Total Radioactive Liability Avoidance,* states:

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[1] American Petroleum Institute, "Bulletin on Management of Naturally Occurring Radioactive Materials (NORM) in Oil and Gas Production," (BUL E2), (Washington, DC: American Petroleum Institute, 1992), p. 10.

[2] *Norm Survey and Control Certification,* (Baton Rouge, LA: Radiation Technical Services, 1994), p. 83.

Further evidence of the American public's unconscious fear of radiation can be found in the legal proceedings of what is today considered "the Law" on the issue of personal injury from radiation exposure: the *Silkwood* trial (*Silkwood vs. Kerr McGee Corp.* (1979, WD Okla) 485 F Supp 566, 5 Fed Rules Evid Serv 765, 10 ELR 20708). A summation of Dr. David S. Gooden's excellent work on the review of Silkwood in *Radiation Injuries - Ionizing Radiation*, (*Radiation Injuries-Ionizing Radiation*, David S. Gooden, M.S., PhD., and J.D., Proof of Facts, American Jurisprudence, 3d Series, Lawyers cooperative Publishing, 1991) evidences the extent of pervasiveness and distrust of federal regulatory RPG's (Radiation Protection Guidelines) in existence in 1979, even by the judge. The transcript shows that the court's instructions to the jury specifically noted that the jurors were not to be bound by federal regulations in bringing in their verdict. Rather, they were to individually decide what constitutes reasonable care on the contamination trial issue, then weigh the defendant's conduct against their own perceived standard.[1]

MacDowell later quotes Roland Fletcher, past chairperson of the CRCPD: "Fact is fact, but perception is reality."[2]

Conclusion

NORM is an environmental contaminant that is receiving much attention lately. The primary radionuclides of concern in the oil and gas industry are ^{226}Ra, ^{228}Ra, and ^{222}Rn. These isotopes from deep within the earth are brought to the surface during oil and gas operations. During the drilling and refining processes, these low level radionuclides can become concentrated so that the level of radioactivity becomes quite high. The health risks due to radiation are dependent upon the type of radionuclide, its concentration, it's solubility in the matrix, and the exposure pathway, among other things. NORM is currently regulated under EPA's drinking water standards, in the Clean Air Act's NESHAPs, and a smattering of other federal regulations. Most of the current regulation is on the state level, with more and more states rapidly developing regulations following CRCPD's release of model state regulations. Further regulation on the federal level is needed. In the absence of regulations, and sometimes in spite of them, actions regarding NORM are often based on insurance liabilities and court decisions.

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Trends of PCBs, PCDDs and PCDFs in Federal Navigation Channel Sediments, Saginaw River, Michigan

by
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Introduction

Saginaw Bay is located on the west shore of Lake Huron, approximately 90 miles north-northwest of Detroit, Michigan (Figure 1). Saginaw Bay is a shallow, inland projection of the western shore of Lake Huron, which forms the thumb of the Lower Peninsula of Michigan. The Saginaw River basin is the largest basin of the total Saginaw Bay watershed (72 percent). The Saginaw River is at the head of the 51-mile-long bay and drains into the southern portion of Saginaw Bay forming a commercial harbor of considerable importance to the industries of the area. The flow on the Saginaw River originates from four major tributaries - the Cass, Flint, Shiawassee and Tittabawassee Rivers (MDNR 1988).

The mouth of the Saginaw River is the site of a Federal Commercial Navigation Project that consists of a 36 mile long Federal navigation channel which extends from 14 miles in Saginaw Bay, Lake Huron, to a point on the Saginaw River, 22 miles upstream of its mouth. The navigation channel is 200 feet wide in the river and 350 feet wide in the bay. Channel depth varies from 16.5 feet at the upstream end to

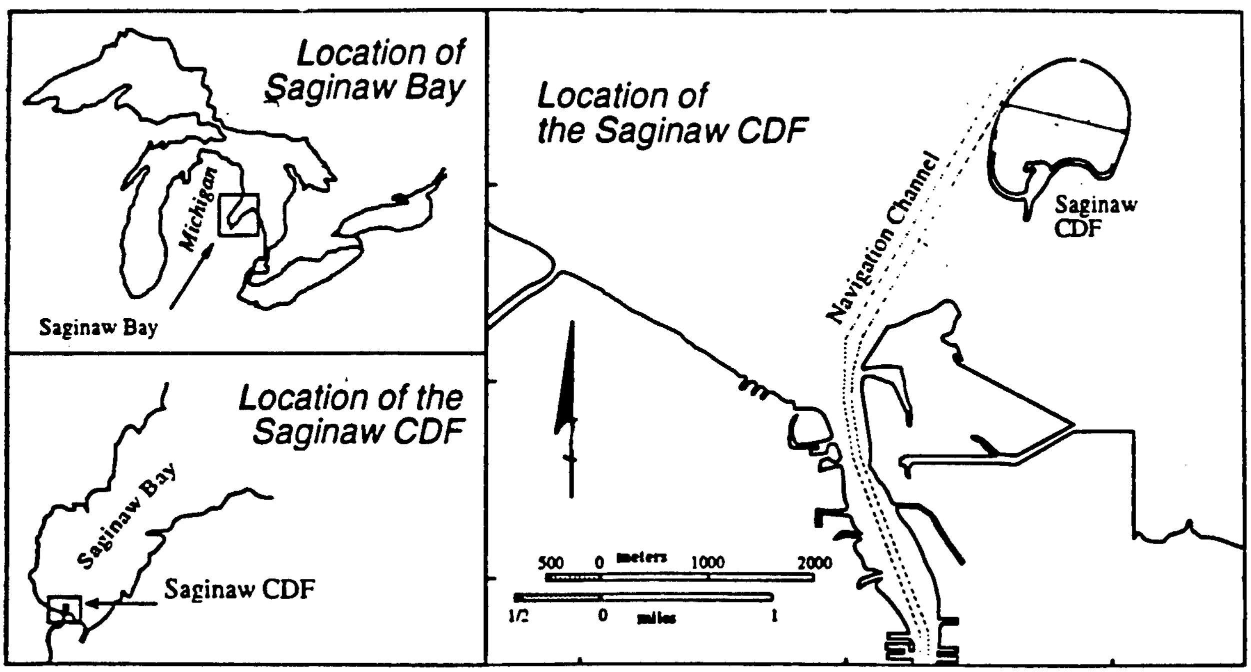

Figure 1. Location of Saginaw Bay, and the Saginaw CDF

[1] U.S. Army Engineer District, Detroit; Detroit; MI.

27 feet in the Bay. Several turning basins are included in the river channel. The Saginaw River Project is maintained by the Detroit District Corps of Engineers (COE). Average annual maintenance dredging for the Saginaw River and Saginaw Bay is 300,000 cubic yards.

Contamination

General

Water quality in the Saginaw Bay and River is degraded from wastewater, industrial discharges, agricultural runoff, and contaminated sediments. Agricultural runoff contributes to turbidity, siltation, and nutrient buildup in the bay which causes algae blooms, reducing water clarity and contributing to oxygen depletion. The Saginaw River and Saginaw Bay have been defined as one of 42 Great Lakes Areas of Concern by the International Joint Commission because degraded water quality conditions impair certain uses for which these waters are designated (MDNR 1988). Reductions in point and non-point source discharges have resulted in water quality improvements.

The Saginaw Bay and River area contains abundant fish and wildlife habitat, including refuges and game areas, over 40,000 acres of marsh along the bay shore, extensive beds of aquatic vegetation in shallow areas of the bay, and several wetland areas along the river. This habitat supports a variety of fish and waterfowl species. The lowlands around the bay and river also support many small mammals. This abundance of habitat provides many opportunities for hunting and sportfishing. Fish consumption advisories are in effect for certain species, particularly bottom dwelling species. Fish consumption is impaired because of organic chemicals such as PCBs, PCDDs, PCDFs, dieldrin, mirex and toxaphene (MDNR 1988).

Sediments throughout the harbor have been contaminated with PCBs, heavy metals, and dioxin as a result of discharges from industries that operated along the river and non-point source discharges. Non-point sources of contaminants include: combined sewer overflows, urban and agricultural runoff, and atmospheric deposition. PCBs in the Saginaw River originated principally from ten point source dischargers (Table 1) which were identified by the Michigan Department of Natural Resources in 1971 (COE 1978).

Historical PCB Contamination

The Detroit District COE conducted extensive sampling and analysis of sediments in the Saginaw Federal Navigation Channel in 1976,[1] to discover the extent of PCB contamination. Of the 48 samples analyzed, only four stations were found to contain levels of PCBs in excess of the U.S. Environmental Protection Agency

Table 1
Ten Significant Point Source Discharges in the Saginaw River Basin

Point Source Discharges	
Bay City WWTP Effluent	Dow Corning (Midland) to Lingle Drain
Buick Division (Flint) to Flint River	Flint WWTP to Flint River
Chevrolet (Bay City)	Grey Iron Foundry (Saginaw) to Saginaw River
Chevrolet Engine (Van Slyke) to Carmen Creek	Nodular Iron Foundry (Saginaw) to Saginaw River
Chevrolet WWTP (Flint) to Flint River	Saginaw WWTP Effluent to Saginaw River

1 COE data from U.S. Army Engineer District, Detroit.

(EPA) heavily polluted sediment criteria of 10 mg/kg dry weight used at that time. Stations with heavily contaminated sediments were located in the Saginaw River rather than Saginaw Bay and all were opposite to, or downstream of, known former PCB point source dischargers. The source of the PCBs was primarily from PCB-containing hydraulic fluids. The highest level of PCB found was 22.9 mg/kg located approximately 15.5 miles upstream of the river mouth. River sediments varied in density and particle size gradation tending towards a silty clay.

As a result of the sampling, two sections of the Saginaw River were classified as heavily contaminated with PCBs (Figure 2). The section furthest upstream stretched from 14.25 to 15.7 miles from the mouth of the river, referred to as the Upper Contaminated Region. The downstream heavily contaminated section stretched from 2.2 to 4.25 miles from the river mouth, referred to as the Lower Contaminated Region.

Testing results during the 1980's showed the same areas of the Saginaw River were still highly contaminated with PCBs. The highest level of PCB found was 27 mg/kg located approximately 3.5 miles upstream in the Lower Contaminated Region. The trend of PCB concentrations over the years are shown on the insets in Figure 2. Although significant dredging had taken place in 1978 to remove the most contaminated material, levels of PCBs remained consistent in the river until the 1990's.

A flood event occurred in the region in September 1986. More than 12 inches of rain fell on Midland and Saginaw, Michigan, within a 36 hour period on September 9-11, 1986 (MDNR 1988). As a result of this rain event, extensive flooding occurred in the whole area. The Tittabawassee River rose from 8 feet to 33 feet. Flow in the rivers in this area were greater than the 100-year record. It is not known if the high flows that occurred in the Saginaw River affected surficial sediment concentrations by depositing additional sediment, exposing the most contaminated layers, or eroding the materials into Saginaw Bay.

Confined Disposal Facility

In 1978 a Confined Disposal Facility (CDF) for Saginaw River dredged material was constructed by the Detroit District COE, under Section 123, Public Law 91-611 (Figure 1). The CDF is located in Saginaw Bay, 1/4 mile east of the Saginaw River's mouth on the previous site of two small islands which were created by former dredging. The dikes of the facility are constructed of a graded mixture of prepared limestone protected with riprap and are designed to contain solids while allowing water to permeate through them. This filtering capability initially minimizes the amount of weir overflow from the facility thus reducing the release of any contaminants. The dikes eventually become impermeable as a result of the accumulation of dredged material.

The Upper and Lower Contaminated Regions were dredged in 1978 using hopper dredges with disposal into the newly constructed Saginaw CDF. The disposed material was then covered with sediments dredged from the less contaminated shoals located at the mouth of the Saginaw River. An estimated total of 390,000 cubic yards of heavily contaminated material was removed in all. To reduce the release of heavily contaminated dredged material during dredging, hopper overflow was restricted during removal of sediments from the heavily contaminated areas. Approximately 2,000,000 cubic yards of critically shoaled sediments from the mouth of the Saginaw River were disposed into the Saginaw CDF on top of the higher PCB contaminated material.

Current PCB Contamination

Recent test results from 1992 and 1993 have shown considerably lower levels of PCBs overall in shoal material. In 1992, extensive sampling took place in shoaled areas of the river

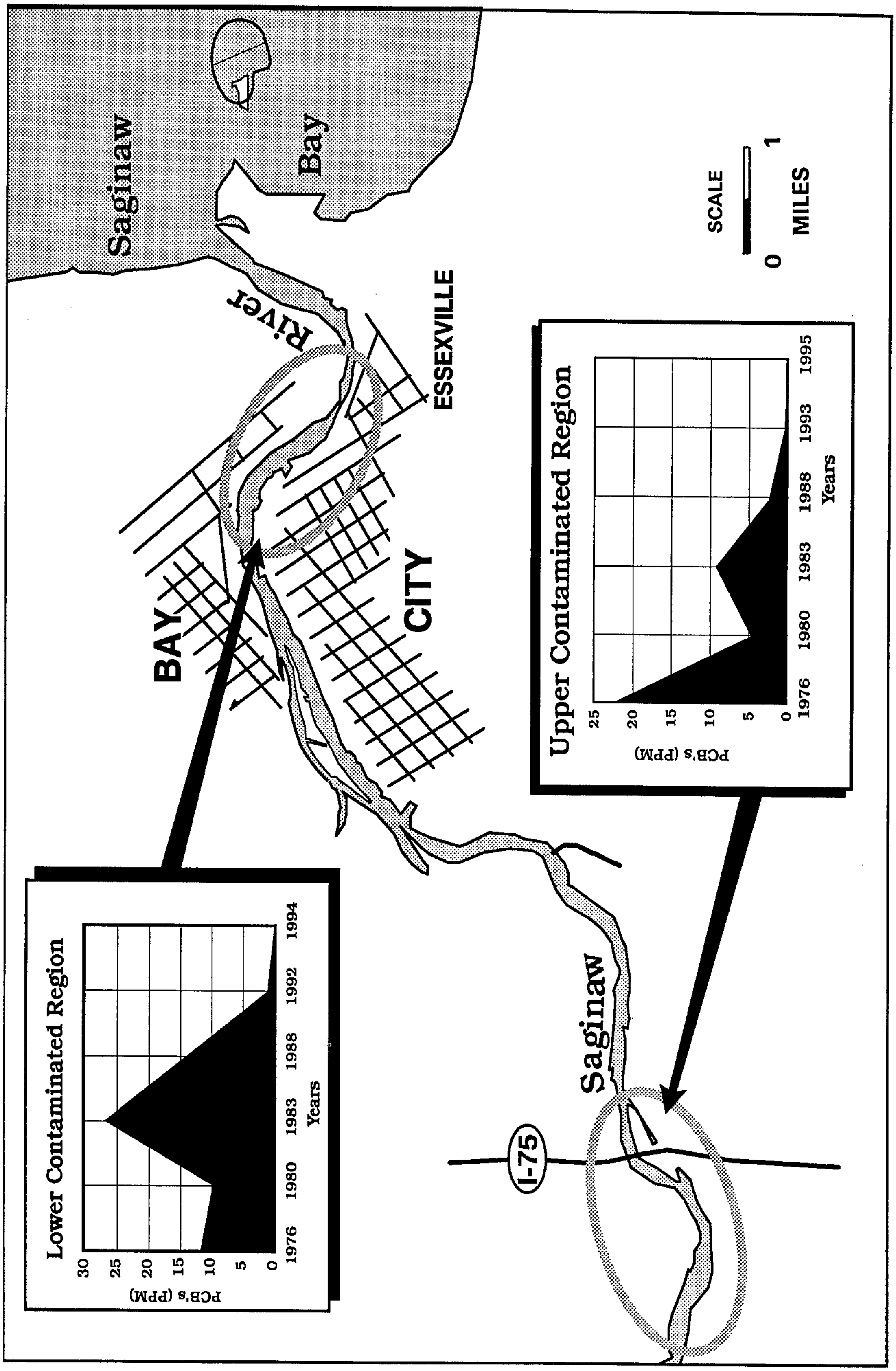

Figure 2. Trends of PCB Levels in the Saginaw River, MI

(34 stations). With this effort, no samples were obtained in the Upper Contaminated Region of the Saginaw River; five samples were obtained in the Lower Contaminated Region. The PCB levels in this Lower Region ranged from 0.32 ppm to 1.0 ppm. In 1993, sampling of the river was conducted (34 stations) beginning about four miles upstream of the river mouth to the upstream end of the Federal project. Only one sample was obtained from the Upper Contaminated Region which showed a level of 0.16 ppm. This sampling did not include the Lower Contaminated Region of the harbor.

Sampling that took place in 1994 was to characterize the lower portion of the Saginaw River project (i.e., lower 4 miles of river and the entire bay). Of these samples, two were obtained in the Lower Contaminated Region. The PCB levels were non-detectable in both samples with rather high detection limits (<0.132 ppm).

In 1995, a limited number of samples were obtained, composited and used for leaching tests. Of these composites, two samples were obtained in the Upper Contaminated Region and were composited with a third sample from a historically low PCB area. No samples were obtained in 1995 in the Lower Contaminated Region. All PCB results in the 1995 sampling were below detection limits (detection limits of 0.05 ppm or less).

As shown above, there were a limited number of samples obtained in the last few years from the two contaminated areas of the Saginaw River. These limited results indicate low or non-detectable levels of PCBs in these areas. An explanation for the recent low levels is unknown. However, the decline could be attributed to the increased flows of the 1986 Flood. This event may have moved contaminated surficial sediments into the Federal navigation channel. Material may have been subsequently removed by dredging with placement into the CDF.

Test results of sediment (since 1987), outside the navigation channel, adjacent to the Lower Contaminated Region, indicate areas of high PCB contamination (up to 297 ppm) still exist in the Saginaw River. Although recent testing suggest declining PCB levels in the Federal navigation channel, evidence still warrants concerns over PCB contamination. Therefore, more detailed sampling of shoaling in these contaminated areas is required before conclusions can be made.

Past and Present PCDD and PCDF Contamination

Analysis for dioxin in channel sediments was initiated in 1983 after a proposal was made to dispose of dredged material in the Crow Island State Game Area adjacent to the river. Some dredged material was to be used to create islands to enhance wildlife habitat. The U.S. Fish and Wildlife Service raised concerns particularly for PCDD levels and PCDF levels in the sediment, since they are persistent pollutants in the aquatic environment and are considered toxic. Dioxins have been shown to bioaccumulate and are known to transfer through the foodchains of fish and wildlife resources (FWS 1984).

The levels of 2,3,7,8-tetrachlorinated dibenzo-*p*-dioxins toxic equivalents (dioxin) in the sediments from the navigation channel have varied from an average low of 38 ppt in 1983 to an average high of 199 ppt in 1993. Data for dioxin levels from various sources are identified in Table 2. This summary includes results from sediment data in the Federal navigation channel, background lake sediments, remote lake sediments, and various upland soils.

Available dioxin data from EPA Region 5 show levels in a few Great Lake states (upland soil and lake/river sediments in Wisconsin, Ohio and Indiana) are mostly non-detectable. There were several areas with low levels of dioxin

Table 2
Summary of Dioxin Levels from Various Media

Media	2,3,7,8-PCDD Toxicity Equivalency, Average	Reference
Saginaw River sediments - 1994	65 ppt	COE 1994
Saginaw River sediments - 1993	199 ppt	COE 1993
Saginaw River sediments - 1990	83 ppt	COE 1990
Saginaw River sediments - 1984	110 ppt	COE 1984
Saginaw River sediments - 1983	38 ppt	COE 1983
Saginaw River sediments - 1990	697 ppt	EPA 1993
Lake Huron sediment - 1986	9 ppt	Czuczwa & Hites 1986
Lake Huron (SE) - lake	25 ppt	Czuczwa et. al. 1984
Zilwaukee Upland Site	278 ppt	COE 1995
Detroit River EPA Data	33 ppt	EPA 1995[1]
Dow Chemical Company plant property	.1 ppm	Smith 1978
Lake Michigan	32 ppt	Czuczwa & Hites 1986
Lake Erie	0.5 ppt	Czuczwa & Hites 1986
Lake Ontario	<10 ppt	Czuczwa & Hites 1986
Lake Siskiwit (Isle Royale)	10 ppt	Czuczwa et. al. 1984
Wisconsin, Indiana, & Ohio EPA Data	25 ppt	EPA 1995[1]

[1] U.S. Environmental Protection Agency, Region V, Chicago, Illinois.

(single digit ppt), only a few areas with dioxin levels up to 100 ppt, and two areas with levels over 100 ppt. These higher dioxin results were located in Wisconsin and were 358 and 638 ppt dioxin toxic equivalents. The only data available from EPA in Michigan is from the Detroit River, of which twenty-nine sites were analyzed for dioxin. The dioxin toxic equivalents were <30 ppt for most of these stations except for one area where sediments were obtained in Monguagon Creek in an industrial area in Riverview, Michigan (across the Detroit River from the north end of Grosse Ile). The levels in five samples from this area were all over 50 ppt; one sample was 210 ppt dioxin toxic equivalents.

Levels of dioxin in Lake Huron are approximately 9 ppt dioxin toxic equivalents (Czuczwa & Hites 1986). Various other levels have been seen in other Great Lakes: Lake Michigan - 32 ppt (Czuczwa & Hites 1986); Lake Erie - 0.5 ppt (Czuczwa & Hites 1986); Lake Ontario - <10 ppt (Czuczwa & Hites 1986); Lake Siskiwit - 10 ppt (Czuczwa et.al. 1984). Testing of an upland site in Zilwaukee, Michigan, showed dioxin toxic equivalents at an average level of approximately 278 ppt. The source of this material at the Zilwaukee site was from previous dredging from a local user.

Results of sediment testing of the material in the Federal navigation channel of the Saginaw River have shown varied levels of dioxin toxic equivalents. However, these levels are above sediment levels in Lake Huron. The levels of dioxin in the dredged material appears to have increased over the years. The levels of dioxin are significantly higher in material outside of the Federal navigation channel, and this is a potential source of dioxin contamination. Therefore, dioxin levels in dredged material should continue to be monitored.

Conclusions

The Saginaw River is the site of an important commercial navigation channel as well as valuable wildlife habitat and a recreational corridor. Maintenance of the Federal Navigation Channel at Saginaw requires the removal of some 300,000 cubic yards of dredged material annually. Historically, material removed from the channel has been placed in a CDF. The character of the material, proposed disposal methods, associated disposal costs and environmental soundness all play a role in selected disposal sites.

Due to the industrial nature of the area, sediments in the Saginaw River Federal Navigation Channel are degraded. Several contaminants have been highlighted, and their levels have been monitored over the years, specifically PCBs and dioxins. There has been an interest in the levels of both of these classes of contaminants due to their persistence and toxicity.

Sampling and analysis efforts for both PCBs and dioxin has been intermittent. Existing limited data, when viewed over time, gives an idea of the trends in these contaminant levels. Recent analyses suggests declining PCB levels in the Federal channel, however, evidence still warrants future concerns over PCB contamination. The levels of dioxin in the dredged material appears to have increased over the years. The levels of dioxin are significantly higher in material outside of the Federal navigation channel, and this is a potential source of dioxin contamination. Development and implementation of a comprehensive, cyclic sampling regime with dedicated stations would provide a more accurate picture of specific contaminant trends.

Channel dredged material at Saginaw is dynamic, being influenced by natural processes, storm events, and ship passage. However, based on historical data, trends in character can be seen. In the search for dredged material disposal solutions, the changing character of dredged material should be considered. Because space in the existing Saginaw CDF is becoming a limiting factor, disposal options include evaluation of open water disposal, upland placement, and raising the dikes on the existing Saginaw CDF.

Periodic removal of material from the Saginaw River, as in maintenance dredging, and greater control of watershed contaminant sources over time should result in an overall reduction of sediment contaminant levels in the channel. Indeed, this effect has been observed with the channel sediments at Saginaw Bay and River with respect to PCBs.

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Modified Solid-Phase Bioassay for Geotextile Confined Contaminated Sediments

by
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Abstract

The Miami River maintenance project involves dredging 5.5 miles of channel from the river mouth to the end of the Miami River Federal Project. The volume of dredged material to be removed is estimated at 800,000 cubic yards. The failure of bioassays in 1992, failure of a bioassay at one station in 1988, and the known presence of pollutants such as arsenic, cadmium, copper, chromium, lead, mercury, silver, and zinc have indicated the unsuitability of sediments from the Miami River for ocean disposal and There are no environmentally acceptable upland sites available. The modified bioassay was used with Miami River sediment to test the efficacy of the geotextile containers in preventing the escape of the contaminated material. Results of a comparison of the modified test procedures with traditional bioassay and bioaccumulation procedures indicate the geotextile containers did reduce release of toxic materials and bioavailability of metals under most circumstances and that the modified procedure is a valid means of testing toxicity of geotextile confined contaminated sediments.

Introduction

Disposal of contaminated dredged material is a major problem in many areas of the United States. The presence of pollutants in channel and harbor sediments significantly increases the problems associated with dredged material disposal. In some cases it may be impossible to maintain harbors simply because there is no environmentally acceptable means of disposing of contaminated dredged material.

Open water disposal of dredged material in an Ocean Dredged Material Disposal Site (ODMDS), near shore site or inland waterway is often the most cost effective means of disposal and in some cases is the only practical alternative available. However, the environmental impact of placing contaminated material in these sites is significant. In an effort to address the need to dispose of contaminated materials in a cost effective and environmentally acceptable manner the U.S. Army Corps of Engineers Waterways Experimental Station has done extensive work on the use of clean material to cap contaminated material in open water sites. Capping appears to offer a means of preventing migration of contaminants from the disposal site, but is not entirely free of problems. In general the problems associated with capping contaminated sediments can be divided into two categories, water column impacts from materials released during the disposal operation and movement of material away from the disposal mound after deposition. In the former case, fine material is striped out of the descending sediment as it passes through the water column resulting in the dispersal of contaminants into the environment. In the latter case, as the material impacts the bottom it spreads out into a layer that can be

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quite extensive. The volume of capping material needed to adequately cover the site is greatly increased by the spread of the contaminated material from the disposal site. Placing contaminated sediments in geotextile containers has the potential of eliminating both the problem of contaminant stripping during disposal and the spread of disposed materials from the site. This process has in fact been used in a project at Marina Del Rey, Los Angeles, California, however the question remains; how do we know that the geotextiles are in fact preventing the release of contaminants to the environment? There are no standard tests available to evaluate the effectiveness of the geotextile barrier.

The modified bioassay procedure described in this paper was designed to quantify the ability of geotextiles to retain contaminants and restrict the bioavailablity of contaminant species in dredged materials placed in an open water disposal site. An attempt has been made to replicate the conditions that would exist if geotextile containers were filled, placed in an open water site and capped with one meter of clean sediment. Sediment from the Miami River was tested using both the USA/USACE 1991 "Green Book" bioassay procedures and a modified procedure using sediment confined in geotextile bags.

Modified Test Procedure

Geotextile Containers

The modified test procedure incorporates a geotextile bag consisting of an outer layer of woven 100-percent polyester (24-ounce/square yard) and an inner layer of nonwoven continuous-filament 100-percent polyester (16-ounce/square yard) material. Trial studies were performed on six different weights of nonwoven material to measure their ability to retain sediment and eluate with in the bag when under pressure. The nonwoven material selected produced an eluate with the lowest total suspended solids. The material was provided by Nicolon Corporation and assembled into bags (sewn on three sides) at their Crestview Florida plant. The day before each test initiation the bags were filled with sediment and the final open side sealed by Nicolon Corporation employees with a portable sewing machine using two single seams per bag.

Solid Phase Bioassays

The day before test initiation, the bags were filled with 2.5 liters of Miami River sediment and placed in a labeled aquarium. The open ends of the filled bags were sewn shut. Each bag was then placed in a labeled stainless steel pan (19" × 11" × 6") and covered with a perforated acrylic sheet (18" × 10" × 0.25"). Six 10 cm diameter cylinders were placed vertically on each acrylic sheet. Plastic drop cloths were used to cover the cylinders and protect the bags from dirt and contamination. Wooden planks (18" × 10" × 2") were placed directly on the plastic cloths to provide a stable base for the placement of the weights. Five 30 pound weights were balanced on each bag setup, resulting in 150 pounds applied to each bag. The pressure was maintained on the bags for 1 hour. After pressing, each bag was transferred to a 5-gallon aquarium, with any eluate poured directly on each bag. The bags were covered with approximately 2.5 liters (2 cm) of clean Miami Harbor sediment and then covered with 8 liters of natural seawater (*Leptocheirus* - 20ppt, *Mysidopsis* - 25ppt). Five test chambers were placed in each aquarium, with 2 cm of clean Miami Harbor sediment covering the bottom of each test chamber. The test chambers consisted of 10 cm diameter PVC cylinders with eight 3 mm openings for water flow. The bottom and sides of each container were covered with 300 um Nytex polyester screen secured with silicon sealant and polyester ties. For the remainder of the test the 1991 Green Book bioassay procedures were followed.

Bioaccumulation

The geotextile materials and assembly procedures used in the modified bioaccumulation procedures were essentially identical with the bioassay procedures with the exception that the *Macoma* tests were run using a 3.5 liter bag compressed with weights totaling 210 pounds. After assembly and pressing the bags were transferred to aquariums and covered with approximately 2 cm of clean Miami Harbor sediment. The remainder of the test was run using the 1991 Green Book procedures.

Methodology and Procedures

Test Protocols

To provide a basis for comparison of geotextile confinement to unçonfined disposal it was necessary to first determine the toxicity of the test sediment using chemical analysis, acute toxicity and bioaccumulation techniques as provided in the 1991 Green Book. The data from these tests were compared with data from the modified acute toxicity and bioaccumulation tests.

Previous Evaluations

Miami River environmental evaluations were carried out in 1987, 1988, and 1992 by the Jacksonville District of the U.S. Army Corps of Engineers (USACE, Jacksonville, 1988 and 92) for the purposes of off-shore disposal and water quality certification. Samples were collected at locations along the length of the project including one station in the Seybolt Canal. The results of the testing indicated high levels of heavy metals, PAH's, oil and grease, and PCB's in one or more bulk sediment samples for each test series but much lower levels of heavy metals and no synthetic organics in the elutriates. Solid phase bioassay results varied but generally indicated the material was not suitable for ocean disposal.

In addition to the Corps evaluations, the Florida Departmnet of Environmental Regulation (FDER, 1985) carried out water quality studies that included 12 sample stations located in the Miami River or its tributaries, the Seybold Canal and the Tamiami Canal, and involved chemical analysis of water, elutriate and sediment. Bulk sediment analysis showed elevated values for arsenic, cadmium, copper, chromium, lead, mercury, silver, and zinc at some or all river stations. Detectable levels of Aldrin, Chlordane, DDT and Aroclor 1254 were also found at some or all river stations. No phenols were detected, but PAH's were found at all river stations. Elutriate analyses were done for metals only and indicated that lead, mercury and silver might exceed state water quality standards at some river stations.

Sample Plan Design

Sample sites were selected to test or retest areas of specific concern that were thought to be contaminated (Figure 1). A composited area reference sample (consisting of two subsamples taken at two stations located near the Miami ODMDS) was collected to characterize the ODMDS. The reference station samples were composited and tested as a single sample. Location of the reference station was 1 mile north of the NE corner of the ODMDS at 25° 48'00"N and 80° 02'02"W (Figure 2).

A sample of clean Miami Harbor sediment was also collected (Figure 2). Sufficient clean sediment was collected for physical and chemical testing as well as provide cover material for the modified sediment bioassay and bioaccumulation studies and one additional sediment bioassay using clean sediment with the modified procedure.

Sampling and Test Procedures

Samples were taken at the edge of the navigation channel using a stainless steel or other non-contaminating box core, grab, or core sampler as required. Samples were taken from the benthic surface to slightly below the project dredge depth.

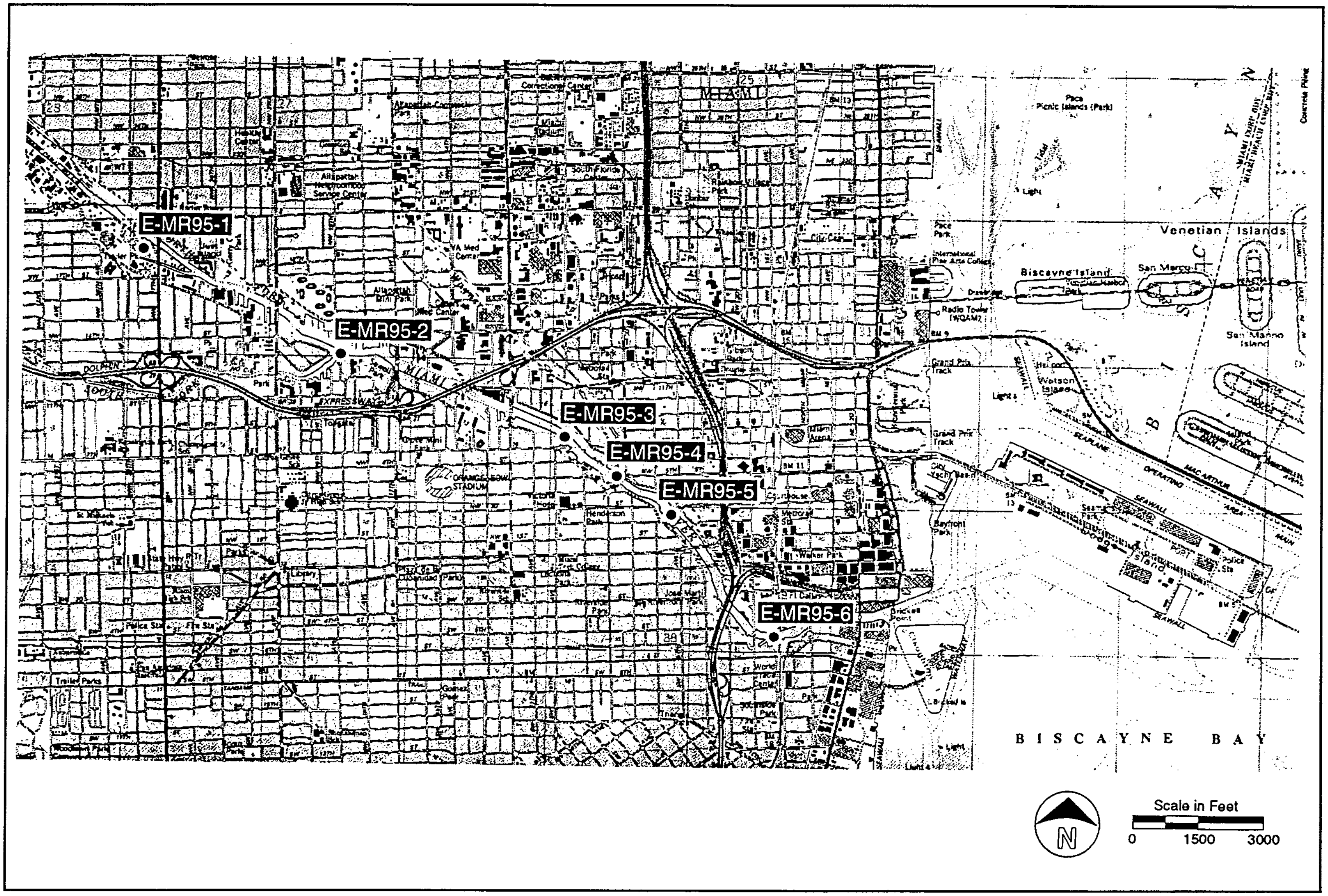

Figure 1. Sample station locations at the Miami River—Sampled July 26-31, 1995

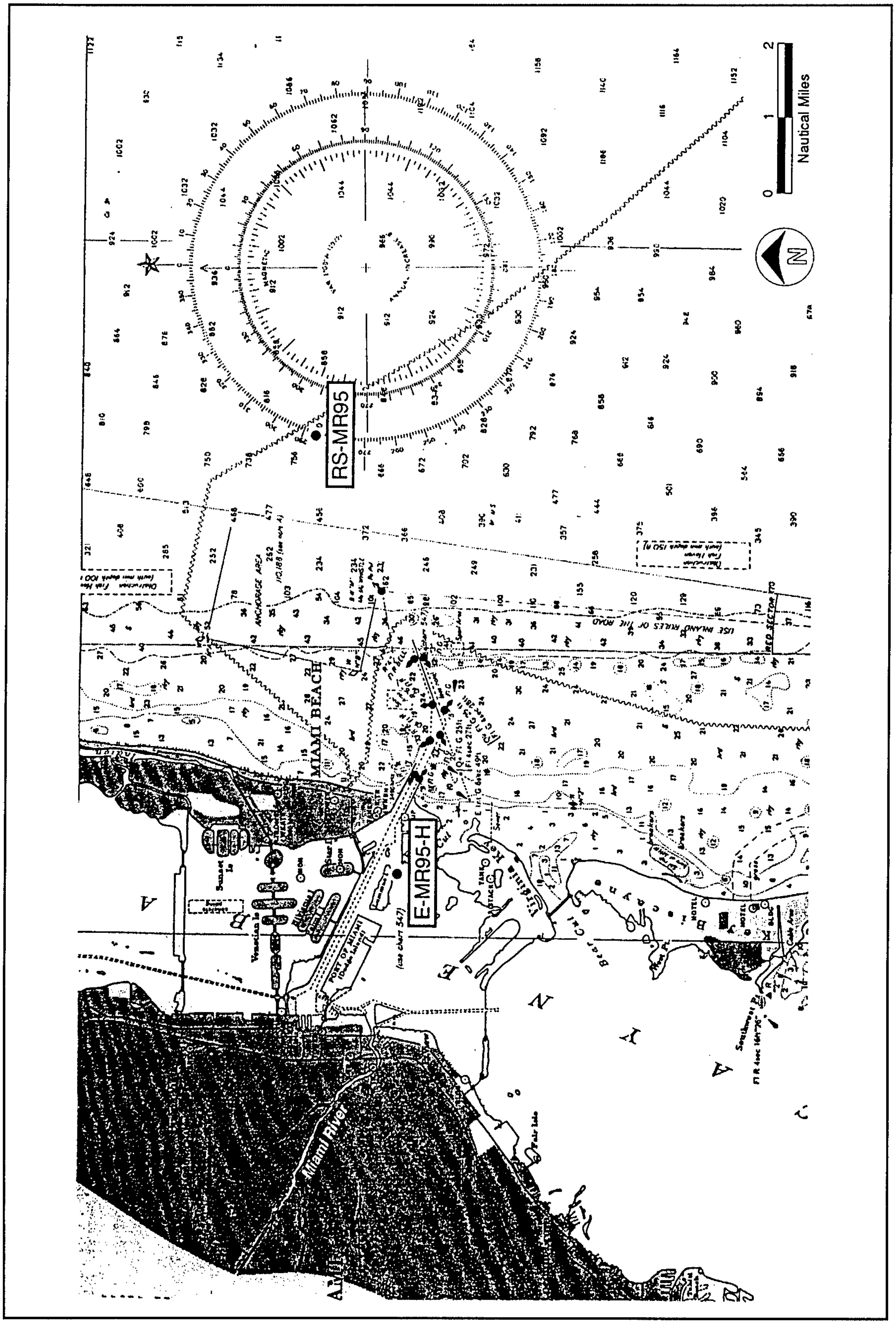

Figure 2. Miami River reference station and Miami Harbor Station—Sampled July 1995

Elutriate Bioassays

Standard Green Book elutriate bioassays were performed for each sample station and a control. No reference station samples were run. Tests were 96 hour (48 hours for zooplankton) static bioassays on concentrations of 10, 50, and 100 percent with five replicates of 10 animals each for each concentration. In addition a 100-percent dilution-water test was run for each species. The species *Lytechinus variegatus* (Sea Urchin Fertilization Test), *Mysidopsis bahia*, and *Meninida beryllina* were used.

Whole Sediment Bioassays

Sediment bioassays were run in two series, standard sediment bioassays and modified bioassays. The modified bioassay procedure was used to test the use of geotextiles for confinement of contaminants. Sediment bioassays were performed for each sample station (regular and modified procedures), a duplicate (regular procedure only), the reference station (regular procedure only), control sediment (regular procedure only) and the Miami Harbor sediment (modified procedure only). Tests were 10-day tests conducted in accordance with the 1991 Green Book and Methods for Assessing the Toxicity of Sediment Associated Contaminants with Estuarine and Marine Amphipods, EPA600/R-94/025, with five replicates of 20 animals each for each station. The tests were conducted using the species *Leptocheirus plumulosus* and *Mysidopsis bahia*.

"Bag Effect" Bioassays

In addition to the 1991Green Book and modified bioassay tests a series of tests using just control sediment placed over bag material was run to determine if there existed a "bag effect". Tests were conducted in beakers and aquariums. In the beaker tests 200 ml of sediment were added to each test chamber (2 liter beaker). One replicate of each species was run for each of the following conditions: 1) control sediment only; 2) control sediment covering simulated bag (two outer woven layers and two inner gray layers); 3) control sediment covering two woven outer layers; and 4) control sediment covering two gray inner layers. The aquarium tests were run using test chambers identical to those used for the whole sediment bioassays placed in 5 gallon aquariums under the same conditions as for the beaker tests. After test setup, test procedures were in accordance with the 1991 Green Book.

Bioaccumulation Studies

Bioaccumulation studies were also performed using both a regular and a modified procedure. Bioaccumulation studies were performed for six samples (regular and modified procedures), the reference sample (regular procedure only), control sediment (regular procedure only), and the Miami Harbor sediment (regular and modified procedures). Chemical analysis of tissue was performed only for those contaminants of concern identified during chemical analysis of sediment. The species *Nereis virens* and *Macoma nasuta* were used for bioaccumulation studies.

QA/QC

One sample station, selected at random, was run in duplicate for bioassays and bioaccumulation studies. All sample stations were run in duplicate for sediment chemistry. The control sediment used was analyzed for all sediment chemical parameters and biological parameters. A reference-toxicant test was conducted for each species used in elutriate bioassays, sediment bioassays and bioaccumulation studies.

A Miami ODMDS water sample was tested for the same metals and chemical compounds as the elutriate samples. Elutriate tests were performed using site-water obtained from the ODMDS site.

Results

Miami River sediments are silts and silty sands with a very high organic carbon content. High levels of heavy metals, oil and grease, and ammonia were found in bulk sediment samples throughout the project area (Table 1). All stations were essentially free of organic chemicals including pesticides, PCBs and PAHs except for one PAH (phenanthrene) detected at a level of 0.7 ppm at station E-MR95-3, and one pesticide compound (4,4'-DDT) detected at a level of 2.8 ppm at station E-MR95-6. Elutriates from stations E-MR95-1 and 6 were tested for heavy metals. Analytical results from elutriate samples showed metals below detection limits or at very low levels at both stations (Table 2). No organic chemicals were detected in the elutriate samples. The results of the bioassay, bioaccumulation, and "bag effects" tests are shown in the Tables 3 - 7.

Discussion and Conclusions

The results of the standard bulk sediment bioassays clearly indicate that Miami River sediments are not suitable for unconfined ocean disposal. Stations 1, 3, 4, 5, and 6 failed one or both of the solid phase bioassays.

The results of the *L. plumulosus* modified bioassay tests indicated a much lower mortality for *L. plumulosus* than found in the standard tests with surviorships ranging from 49 to 86 percent. The surviorship in the standard bioassays ranged from 0 to 76 percent. Comparison of the *L. plumulosus* modified and standard bioassay using the paired samples t-test indicates that overall the geotextile containment did in fact reduce mortality by an average of 38.5 percent ($a < 0.01$).

The results of the *M. bahia* modified bioassays were ambiguous. The tests run using the geotextile containers had generally higher mortalities than the standard bioassays. However comparison of the *M. bahia* modified and standard bioassay using the paired samples t-test indicates that overall there was no significant difference between the modified and standard test results ($a = 0.01$). A possible explanation of the high mortalities in the modified *M. bahia* bioassays was the presence of significant amounts of ammonia in the sample. During the bioassay period un-ionized ammonia in the overlying water ranged as high as 1.7 ppm. Total ammonia was found as high as 48 ppm. Miller, et al. found *M. bahia* to be sensitive to unionized ammonia at levels as low as 0.08 ppm. The EPA and USACE have issued guidance for reducing ammonia concentrations in benthic toxicity tests (EPA/USACE 1994) and recommend that total ammonia be no higher than 20 ppm at a pH of 7.5 (unionized ammonia = 0.3 ppm). A geotextile barrier would have little if any effect on the movement of ammonia from the sediment to the overlying water and would therefor not prevent ammonia toxicity in the test animals.

The tests of the geotextile materials used for the containers showed a slight toxic effect in *M. bahia* associated with the woven outer fabric. The cause of this effect has not been determined and additional studies are needed to clarify what, if any, toxicity is actually caused by the geotextile materials.

Bioaccumulation tests were run for cadmium, chromium, copper, lead, and mercury using the test species *Nereis virens* and *Macoma nasuta*. *N. virens* tissue concentrations were in general lower than *M. nasuta* tissue concentrations. The modified *N. virens* bioaccumulation test showed no significant reduction in bioaccumulation of any of the metals when compared to the standard test ($a = 0.01$). However the *N. virens* tissue concentrations from the Miami River tests were similar, and often lower than those from the control, reference and Miami Harbor sediments tests indicating that *N. virens* is not bioaccumulating metals from the Miami River sediments. *M. nasuta* tissue concentrations for cadmium and chromium in both the standard and modified tests were in line with expected values for marine bi-valves (Ratkowsky et al. 1974, Kopfler and Mayer, 1967, and Korte et al. 1983) Tissue concentrations of copper in *M. nasuta* were elevated in both the modified and standard tests. However this is not unexpected as copper concentrations in the Miami River sediments were extremely high (138 ppm to 1,540 ppm). The modified *M. nasuta* bioaccumulation test showed an average reduction in copper bioaccumulation of 45 percent when compared to the standard test for *M. nasuta*. The difference was statistically significant at the

Table 1
Results of Chemical Analysis for Sediments Collected From The Miami River. All Data Reported in ppm (μg/g) Dry Weight Basis

Station ID	Ammonia as N	Total Org. Carbon	Oil and Grease	Aluminum	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Silver	Zinc
Control	<1.0	750.0	<30	250.0	1.3	<0.1	2.3	1.2	0.6	<0.05	1.2	<0.1	<0.1
Reference	6.0	180000.0	<50	563.0	1.3	<0.1	10.2	2.4	2.0	<0.05	11.6	<0.1	3.1
Harbor A	6.6	78000.0	85.0	903.0	3.5	0.1	5.8	1.2	3.8	<0.05	5.8	0.2	2.2
Harbor B	5.9	87000.0	<30	919.0	3.8	<0.1	5.6	0.5	3.6	<0.05	5.2	1.4	1.7
E-MR95-1-A	33.0	67000.0	75.0	5900.0	8.4	1.4	48.7	349.0	307.0	1.0	22.1	1.3	361.0
E-MR95-1-B	32.9	76000.0	<40	5880.0	7.4	1.4	55.4	558.0	368.0	1.0	22.6	1.4	387.0
E-MR95-2-A	37.0	120000.0	<50	5290.0	8.7	0.8	40.6	173.0	185.0	1.2	17.0	1.4	206.0
E-MR95-2-B	35.1	120000.0	92.0	4910.0	8.6	0.8	35.8	144.0	194.0	1.2	17.2	1.4	182.0
E-MR95-3-A	147.0	140000.0	<40	7220.0	7.9	3.6	44.0	138.0	205.0	4.6	21.6	5.6	269.0
E-MR95-3-B	151.0	150000.0	<40	6920.0	7.5	3.7	43.2	160.0	209.0	4.9	21.3	5.2	270.0
E-MR95-4-A	50.6	190000.0	<40	5540.0	8.0	2.6	51.6	246.0	495.0	5.6	23.1	7.6	402.0
E-MR95-4-B	50.9	130000.0	160.0	5710.0	6.8	2.7	53.2	217.0	311.0	7.4	21.9	7.3	436.0
E-MR95-5-A	58.5	110000.0	84.0	6350.0	8.6	3.1	42.1	146.0	217.0	4.8	21.3	5.6	284.0
E-MR95-5-B	60.5	100000.0	147.0	6490.0	7.6	6.0	42.8	144.0	222.0	4.8	23.1	6.0	312.0
E-MR95-6-A	18.1	45000.0	<30	10600.0	7.9	0.6	49.7	829.0	301.0	7.0	36.0	1.6	363.0
E-MR95-6-B	17.8	53000.0	102.0	4440.0	10.0	1.1	46.2	1540.0	372.0	8.1	28.5	2.5	575.0

Table 2
Results of Chemical Analysis for Elutriates of Sediments Collected From The Miami River. All Data Reported in ppb (micrograms per liter, $\mu g/\ell$)

Station ID	Aluminium	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Silver	Zinc
Reference	14.2	<0.8	<0.5	0.4	<0.3	1.7	<0.1	1.0	<0.2	2.2
E-MR95-1	15.2	3.8	<0.5	0.9	0.5	<0.2	<0.1	<0.2	<0.2	<0.2
E-MR95-1	19.9	2.7	<0.5	1.4	<0.3	<0.2	<0.1	<0.2	0.2	<0.2
E-MR95-6	71.8	3.0	<0.5	1.1	0.6	1.0	<0.1	<0.2	<0.2	8.2
E-MR95-6	23.9	3.7	<0.5	1.1	0.8	1.6	<0.1	0.7	0.8	2.8

95-percent confidence level but was not significant at the 99-percent confidence level. Bioaccumulation of lead and mercury in *M. nasuta* was significantly lower (75 and 50-percent lower respectively, a = 0.01) in the modified test than in the standard bioaccumulation test.

Conclusions: Data indicate that the modified bioassay and bioaccumulation test procedures do serve to quantify the efficacy of the geotextile barrier in preventing the release of contaminates. The use of the modified test can provide a means of determining the suitability of geotextile confined sediment for open water disposal should this become an available option. However the results of this study raise several questions. The *M. bahia* bioassay results did not show any reduction in mortality between the standard and modified tests. This may have been due to high ammonia levels. However this cannot be stated with certainty until additional tests have been run.

The toxic effects exhibited by the woven material are difficult to explain. This may be a result of the way the test was run rather than a real toxicity impact. The "bag effect" test series was not extensive enough to say with certainty that the effect is real. Further studies will be necessary to determine the significance of these results.

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Table 3
Surviorship of *Mysidopsis bahia* Tested in Sediment Bioassays for The Miami River

Station ID	Dauphin is Control	Reference	Harbor	E-MR95-1	E-MR95-2	E-MR95-2D	E-MR95-3	E-MR95-4	E-MR95-5	E-MR95-6
Standard	90%	90%	NR	89%	90%	91%	75%	68%	85%	62%
Modified	29%	NR	88%	43%	88%	57%	48%	71%	48%	74%

Table 4
Surviorship of *Leptocheirus plumulosus* Tested in Sediment Bioassays for The Miami River

Station ID	Dauphin is Control	*L. plumulosus* Control	Reference	Harbor	E-MR95-1	E-MR95-2	E-MR95-2D	E-MR95-3	E-MR95-4	E-MR95-5	E-MR95-6
Standard	96%	96%	79%	NR	44%	76%	75%	12%	13%	22%	0%
Modified	20%	0%	NR	80%	80%	86%	85%	76%	55%	81%	49%

Table 5
Surviorship of *Leptocheirus plumulosus* and *Mysidopsis bahia* Tested in Clean Harbor Sediment Covering Geotextiles (1 rep each)

Sample	*L. plumulosus*		*M. bahia* 1st Test		*M. bahia* Retest	
	Beaker	Aquarium	Beaker	Aquarium	Beaker	Aquarium
Control (D. I.)	100%	80%	90%	50%	100%	35%
Control + Bag	45%	0%	10%	60%	0%	0%
Control + White	5%	0%	85%	85%	85%	50%
Control + Grey	95%	75%	80%	90%	60%	40%

Table 6
Tissue Heavy Metal Concentrations in *Nereis virens* Exposed to Miami River Sediment. All Data Reported in ppm (μg/g) Dry Weight Basis

Station ID	Cadmium		Chromium		Copper		Lead		Mercury	
	Standard	Modified	Standard	Modified	Standard	Modified	Standard	Modified	Standard	Modified
Control	0.40	0.44	0.80	0.60	11.90	9.00	1.10	0.90	<0.05	<0.05
Reference	0.43	NR	2.90	NR	53.00	NR	1.60	NR	<0.05	<0.05
Harbor	NR	0.35	NR	1.50	NR	9.40	NR	0.80	NR	NR
E-MR95-1	0.22	0.19	1.80	1.00	9.00	8.40	1.80	1.10	<0.05	<0.05
E-MR95-2	0.66	0.39	1.30	2.40	14.90	6.20	4.10	1.00	<0.05	<0.05
E-MR95-2	0.22	0.20	3.50	1.50	27.60	8.40	2.20	1.20	<0.05	<0.05
E-MR95-3	1.39	0.33	4.20	0.90	103.00	6.30	12.10	0.60	<0.05	<0.05
E-MR95-4	0.74	0.32	11.30	1.00	63.70	6.20	155.00	0.60	<0.05	<0.05
E-MR95-5	0.68	0.37	1.60	1.80	39.50	6.10	6.80	0.80	<0.05	<0.05
E-MR95-6	0.58	0.40	2.30	1.40	27.30	7.30	6.30	0.70	<0.05	<0.05

Table 7
Tissue Heavy Metal Concentrations in *Macoma nasuta* Exposed to Miami River Sediment. All Data Reported in ppm (μg/g) Dry Weight Basis

Station ID	Cadmium		Chromium		Copper		Lead		Mercury	
	Standard	Modified	Standard	Modified	Standard	Modified	Standard	Modified	Standard	Modified
Control	0.58	0.53	2.40	2.50	35.00	34.80	3.00	3.50	0.18	0.13
Reference	0.38	NR	2.80	NR	33.90	NR	2.50	NR	0.15	NR
Harbor	NR	0.30	NR	7.80	NR	144.00	NR	2.80	NR	0.16
E-MR95-1	1.87	0.52	5.90	2.40	69.40	23.80	17.10	2.90	0.24	0.11
E-MR95-2	0.40	0.46	2.90	6.60	24.60	109.00	9.90	5.70	0.14	0.10
E-MR95-2	0.51	0.33	5.80	2.40	188.00	21.60	12.40	2.00	0.20	0.10
E-MR95-3	0.89	0.65	2.80	7.10	187.00	26.20	11.50	5.20	0.20	0.14
E-MR95-4	0.50	0.78	3.20	2.90	102.00	27.80	12.90	2.30	0.35	<0.05
E-MR95-5	0.46	0.32	5.70	2.50	101.00	6.10	9.00	2.60	0.32	0.11
E-MR95-6	0.36	0.41	2.90	2.10	41.80	7.30	22.10	2.90	0.17	0.12

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Development of a Corps of Engineers Comprehensive Watershed Water Quality Model

by
Patrick N. Deliman[1]

Water quality of receiving waters (e.g., CE reservoirs) depends upon landuse and other activities in the watershed. There is a growing awareness that water quality improvements in these receiving waters depends greatly upon developing and implementing Best Managements Practices (BMPs). Increasingly, the CE must interact with other agencies on landuse/ management practices in the watershed to reduce nonpoint source pollution. The primary tool in determining the effectiveness of proposed BMPs is the application of a comprehensive watershed water quality model. Currently, the CE is developing the capability for determining non-point source water quality (nutrients, solids, and chemicals) runoff through watershed modeling.

Several watershed water quality models are being reviewed to determine applicability to meet the modeling needs of the CE. These models include those based on the Stanford Watershed Model such as ARM, NPS, and HSPF and also CREAMS, GLEAMS, PRZM2, ARMSED and HEC1. The model which is most applicable to the CE modeling needs will be selected as the building block for the comprehensive Corps watershed water quality model. Perceived enhancements to the model included improved process descriptors and algorithms and interfacing of the model with other capabilities such as GIS, remote sensing, and water quality assessment methods.

The developed CE comprehensive watershed water quality model will be verified with existing data and documented with a users guide. CE offices needing to evaluate long term impacts of landuse changes in a watershed will have an evaluation tool. It is anticipated that the model will be documented in a way that minimizes the need for consultation during model application. Following the completion of this project the CE will have a state of the art watershed water quality modeling capability.

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Water Quality and Contaminant Modeling

by
Toni Schneider[1] *and Dorothy H. Tillman*[1]

The Water Quality and Contaminant Modeling Branch, Environmental Laboratory, Waterways Experiment Station (WES), conducts research, development, and special studies to predict, through mathematical and numerical (i.e., computer simulation) models, environmental quality of aquatic systems resulting from natural and man-induced influences. Typical activities include: develop, maintain, and apply water quality simulation models for surface water systems; evaluate, modify, develop, maintain, and apply contaminant (i.e., toxic substances) transport/fate models for surface water systems and groundwater resources; evaluate the effects of reservoir operations on in-pool and tailwater fisheries and aquatic habitat; and develop and apply ecosystem process-based models.

Some of the models the Corps of Engineers relies upon to assess and to answer critical environmental questions are CE-QUAL-R1, CE-QUAL-W2, CE-QUAL-RIV1, CE-QUAL-ICM, TWQM, PREWET, and RECOVERY. A description of each of these models will be presented during the poster session. Previous model applications will also be displayed for each model. These models are available through the WES Water Quality Bulletin Board (601-634-4216).

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A Rainfall Simulator/Lysimeter System for Predicting Surface-Runoff Water Quality

by
Richard A. Price[1], *John G. Skogerboe*[1] *and Charles R. Lee*[1]

Introduction

The CE dredges approximately 400 million cubic yards of sediment from waterways and harbors annually. A small percentage of this sediment contains elevated levels of contaminants such as heavy metals, polychlorinated biphenyls (PCBs), polyaromatic hydrocarbons (PAHs), and other contaminants. In most cases, confined upland disposal of contaminated dredged material is selected as an alternative to more environmentally sensitive methods of disposal. Movement of contaminants from the upland confined disposal facility (CDF) must be addressed to reduce environmental impacts to the surrounding ecosystem. Testing protocols for predicting these impacts were developed under the Environmental Impact Research Program and field verified under the Field Verification Program. The prediction of surface runoff water quality from CE upland disposal sites is one of the evaluations described by Francingues et al. (1985) for the management of dredged material. The interpretation of the test data has generally been described in the decision making framework of Peddicord et al. (1986) and Lee et al. (1991). The WES RSLS is an important tool developed to predict surface runoff contaminant concentrations from the upland disposal of contaminated dredged material and has been used extensively as such. The RSLS has also been used to quantify the effectiveness of vegetation to reduce soil erosion from CE construction sites. This paper will summarize the development of the RSLS, discuss the results of a study on upland disposal of dredged material and summarize some of the other uses of the RSLS.

The WES Rainfall Simulation Development

The WES RSLS incorporates a rotating disk type simulator and an aluminum bin lysimeter measuring 4.6 m by 1.2 m (Figure 1). The WES simulator is a modified version of a design by Morin et al. (1967, 1970). The design accurately duplicates the drop size distribution and terminal drop velocities of natural rainfall (Westerdahl and Skogerboe, 1982). The depth of the lysimeter can be adjusted from 0.15 m to 0.9 m in increments of 0.15 m and can vary in slope from 0 to 20 percent. The RSLS was also modified to be field portable (Figure 2), allowing in-situ rainfall simulation and field verification of laboratory tests.

The WES RSLS went through a series of testing and verification of results from laboratory and field studies. A study conducted on an overland flow wastewater treatment facility compared hydrographs of the RSLS to natural storm events and found the RSLS accurately simulated the effects of complex natural storm events (Westerdahl and Skogerboe 1982). A study conducted on a disturbed CE construction site compared the effects of biomass of various vegetation types on soil loss in both laboratory and in-situ rainfall simulations. The laboratory simulations on vegetated lysimeters accurately predicted actual soil loss from similarly

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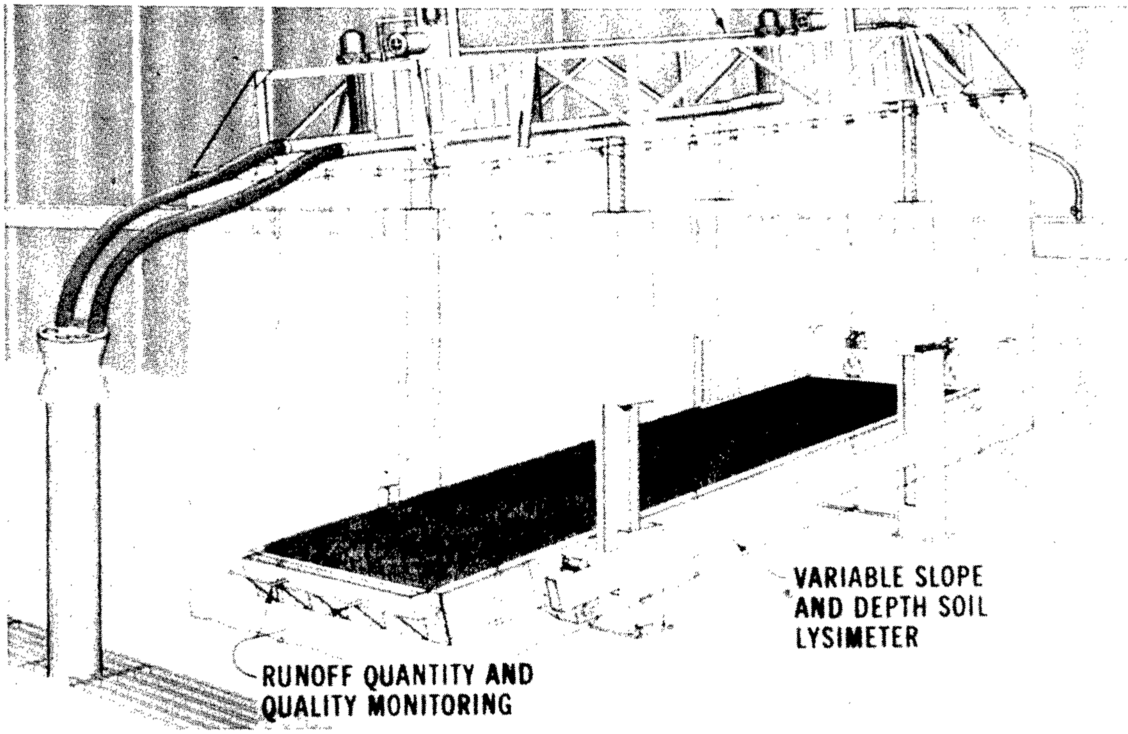

Figure 1. The WES RSLS in the laboratory

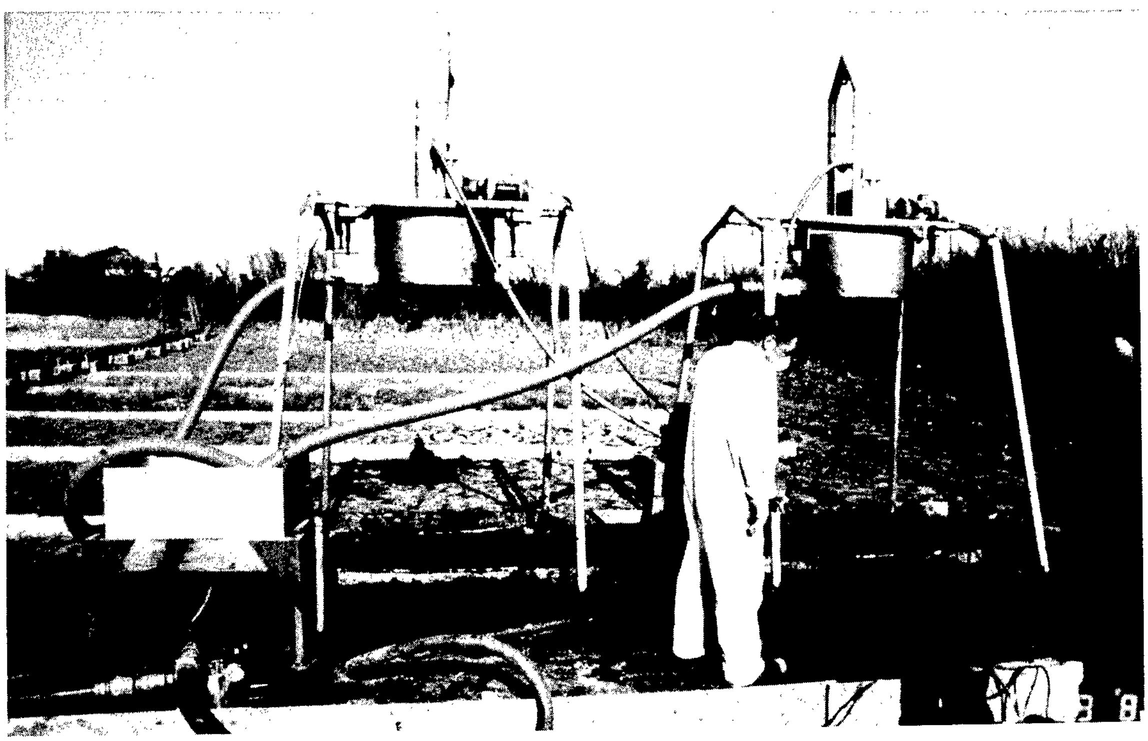

Figure 2. The field portable system on a upland disposal site

vegetated plots in the field (Lee and Skogerboe 1984). Another study was conducted to predict the surface runoff water quality from dredged material placed in an upland confined disposal facility (CDF) prior to its placement. Sediment from the dredging site was collected and placed in lysimeters at the WES and allowed to naturally dry and oxidize. Rainfall simulations were conducted on the wet, anaerobic dredged material and subsequently on the dry oxidized material. The CDF was eventually filled with dredged material and rainfall simulations were conducted in-situ on wet and on eventually dried dredged material using the portable RSLS. The laboratory simulation on wet and dried dredged material accurately predicted the surface runoff water quality from the field site (Skogerboe et al. 1987).

Evaluation of Upland Disposal Dredged Material

Methods and Materials

Sediment is normally collected from the proposed dredging site to sufficiently represent the dredging cut in question. The surface runoff test requires eleven 55-gal drums of sediment. The sediment is placed into the lysimeter, thoroughly mixed and the standing water removed. Three 30-min storm events at 5.08-cm/hr are conducted on the wet, anaerobic sediment on successive days (Figure 3). Runoff rates are measured every minute and runoff samples are collected for chemical analysis at 5, 15 and 25 min after runoff begins to occur. The runoff samples are composited and analyzed for filtered and unfiltered contaminants. The lysimeter is then placed outside the laboratory and covered with a transparent top to allow air drying for six months. The rainfall simulation and sampling procedures are then repeated on the dry, oxidized sediment (Figure 4). The results are then compared to appropriate State or Federal Water Quality Standards. Site design and runoff controls can be engineered such that surface water runoff from the upland disposal site will meet the applicable water quality standards.

Results of a Previous Study

The evaluation of surface runoff water quality from upland disposal of dredged material has been conducted for many locations including Black Rock Harbor, CT (Skogerboe et al.

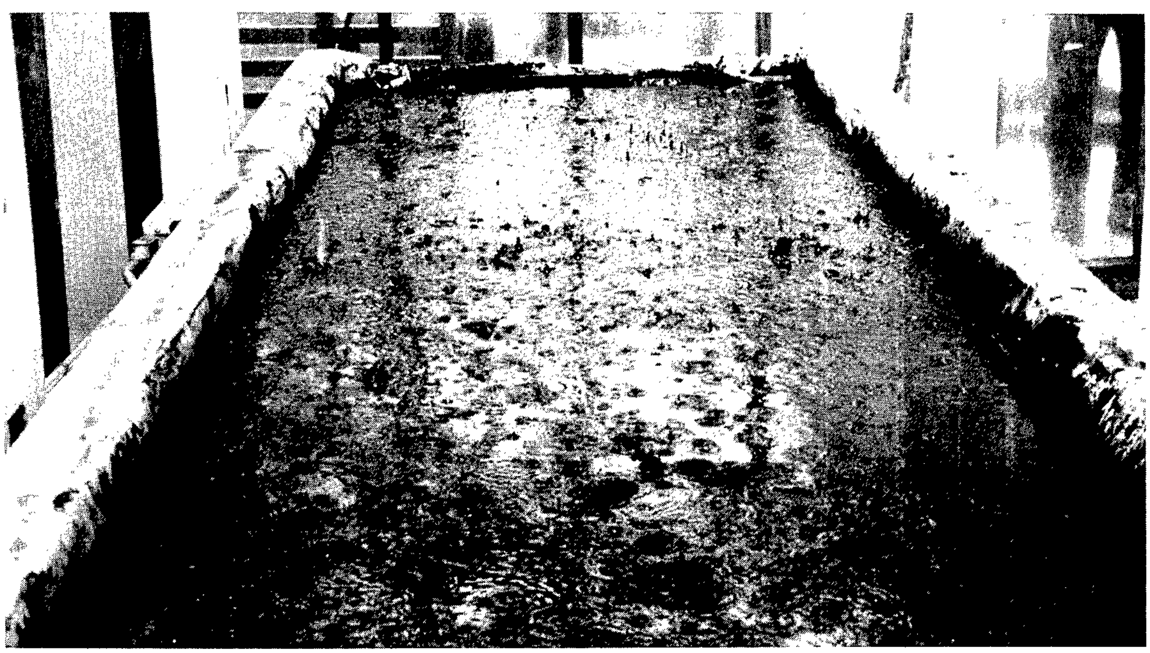

Figure 3. Rainfall simulation on the wet, unoxidized sediment

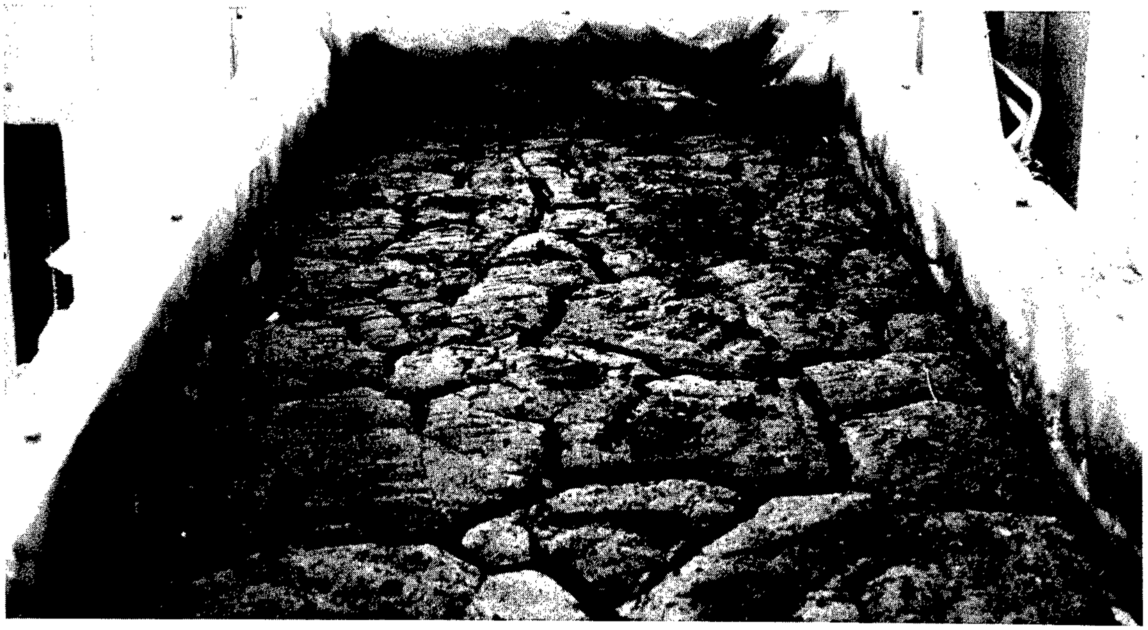

Figure 4. Six months later, the sediment is dry and oxidized

1987), Indiana Harbor, IN (Environmental Laboratory 1987), New Bedford Harbor, MA (Skogerboe et al. 1988), Oakland Harbor, CA (Lee et al. 1992, 1993) and others. A summary of results and interpretation from a study conducted on two Oakland Harbor sediments (Lee et al. 1993) is presented.

Table 1
Heavy Metal Concentrations in Sediment, μg/g Dry Weight (Lee et al. 1993)

Parameter	Oakland Inner	Oakland Outer
Silver	0.110	0.203
Arsenic	5.620	6.90
Cadmium	0.148	0.233
Chromium	381.3	364.0
Copper	24.53	31.07
Mercury	0.110	0.166
Nickel	65.2	84.03
Lead	14.15	18.67
Selenium	0.173	0.267
Zinc	61.10	84.80

Concentrations of heavy metals in sediment from the proposed dredging site are shown in Table 1. Although these levels are not significantly elevated, chromium and lead concentrations were higher than in soils from the Twichell Island disposal site evaluated with previous Oakland Harbor sediments (Lee et al. 1992). Results of surface runoff tests using the RSLS in the laboratory are provided in Tables 2, 3 and 4. Suspended solids and pH in runoff were reduced after drying while conductivity increased (Table 2). Heavy metal concentrations that exceeded water quality criteria or standards are shown in bold (Tables 3 and 4). While the sediment is wet and unoxidized, metals will be bound to eroded particulates in the runoff from an upland disposal site. However, soluble (filtered) arsenic and copper exceeded the Receiving Water Limitations requiring additional considerations such as the availability of a suitable mixing zone to dilute concentrations below the standard or treatment controls to reduce discharges.

Table 2
Physical Characteristics of Unfiltered Surface Runoff Water (Lee et al. 1993)

Parameter	Oakland Inner		Oakland Outer	
	Wet	Dry	Wet	Dry
Suspended solids, mg/L	4,447	1,686	4,610	1,749
pH	7.97	6.88	8.11	7.02
Conductivity mmhos/cm	0.99	2.41	1.55	2.97

In the dry oxidized material, arsenic, cadmium, copper, chromium, and zinc was found to be a potential problem and required removal of particulates from the runoff water. Only soluble arsenic was found to require additional considerations for a mixing zone or treatment controls. Under both wet and dry conditions, controlling particulate movement from the site will control the release of most contaminants from the site.

Table 3
Heavy Metals in Oakland Inner Laboratory Runoff (Lee et al. 1993)

Parameter	Wet Unfiltered	Wet Filtered	Dry Unfiltered	Dry Filtered
Silver	0.16	0.14	1.11	0.93
Arsenic	22.5[4]	12.6[4]	3.72[4]	2.08[4]
Cadmium	9.12[1,2,4]	0.99	2.69[4]	0.29
Chromium	78.0[1,2,4]	2.97	34.0[1,2,4]	1.30
Copper	162[1,2,3,4]	6.66[3]	84.4[1,2,3,4]	3.46
Mercury	0.0014	0.0006	0.0043	0.0019
Lead	46.0	0.71	22.0	0.34
Zinc	395[1,2,3,4]	21.7	206[1,2,3,4]	11.3

Table 4
Heavy Metals in Oakland Outer Laboratory Runoff (Lee et al. 1993)

Parameter	Wet Unfiltered	Wet Filtered	Dry Unfiltered	Dry Filtered
Silver	0.21	0.18	1.45	1.22
Arsenic	32.0[4]	17.9[4]	5.29[4]	2.96[4]
Cadmium	3.19[1,4]	0.35	0.94	0.10
Chromium	69.3[1,24]	2.64	30.2[1,2,4]	1.15
Copper	155[1,2,3,4]	6.33	80.2[1,2,3,4]	3.29
Mercury	0.0007	0.0003	0.0020	0.0009
Lead	65.2[1,4]	1.01	31.2	0.48
Zinc	294[1,2,3,4]	16.2	153[1,3,4]	8.41

[1] Mean Concentration was statistically greater than the assumed Effluent Limitation Criteria.
[2] Mean concentration was statistically greater than the assumed USEPA Fresh Water Acute Criteria.
[3] Mean concentration was statistically greater than the assumed USEPA Marine Water Criteria.
[4] Mean concentration was statistically greater than the assumed Receiving Water Limitation Criteria.

Simplified Laboratory Test

Evaluation of the long-term effects of placing a particular sediment in an upland environment requires eleven drums of sediment for the surface runoff test. Disposal expenses for the material upon completing the tests can be high depending on the contaminants present in the material. This expense must be passed on to the sponsor in the project cost. The WES began to develop a simplified laboratory test to address the concerns of surface runoff water quality from wetland or upland dredged material disposal sites. The simplified test was designed to require a fraction of the sediment needed for the RSLS test and could be performed by most analytical laboratories. The time required to conduct the test was also reduced significantly from 7 months to 4 weeks. The procedure incorporates oxidation of the dredged material with hydrogen peroxide to simulate the long-term effects of drying and a simple water/sediment dilution method to address various suspended solids concentrations expected under wet, anaerobic and dry, oxidized sediment conditions. The basic procedure has been previously described (Environmental Laboratory 1987).

The CE Districts will benefit by the simplified test in having a less expensive alternative to the RSLS test and will be able to conduct the test in-house or by a contract laboratory. The simplified test is not designed to replace the RSLS test but, to provide a less expensive screening mechanism to determine if the more expensive RSLS tests are required. However, the simplified test has not been fully correlated to a wide range of sediments and thus is not accepted at this time as an alternative method to address surface runoff water quality. Additional sediments need to be subjected to both the simplified test and the RSLS test to verify the accuracy of predictive capabilities.

Evaluation of Erosion Control Techniques

Techniques to control soil erosion from CE construction sites have also been evaluated using the RSLS. The amount and type of vegetation can have a drastic effect on the amount of soil eroded from a site as shown in Figure 5. Various species of vegetation can be grown in soil from the site in question and the lysimeter can be tilted to slopes up to 20 percent. Suspended solids in runoff from a simulated storm event can be compared to assess erosion control effectiveness of each species at various stages of development or under various environmental stresses such as drought and dormancy. Management strategies such as burning, fertilization and mowing can also be compared. The information derived from simulated storm events can determine minimum vegetation requirements and management practices necessary to meet particular water quality requirements.

Data Generation for Water Quality Modeling

Recently, military installations have come under fire for discharging elevated levels of suspended solids in rivers and streams leaving the installations. Many of these installations have land uses and activities unique to the military that current watershed and soil erosion models may not address. The RSLS can be valuable in generating surface runoff water data necessary for the modeling of watersheds and water quality of water resources. Land management and land use effects on soil erosion, nutrient and contaminant runoff, and runoff rates can be accurately determined using the RSLS in the laboratory or in-situ.

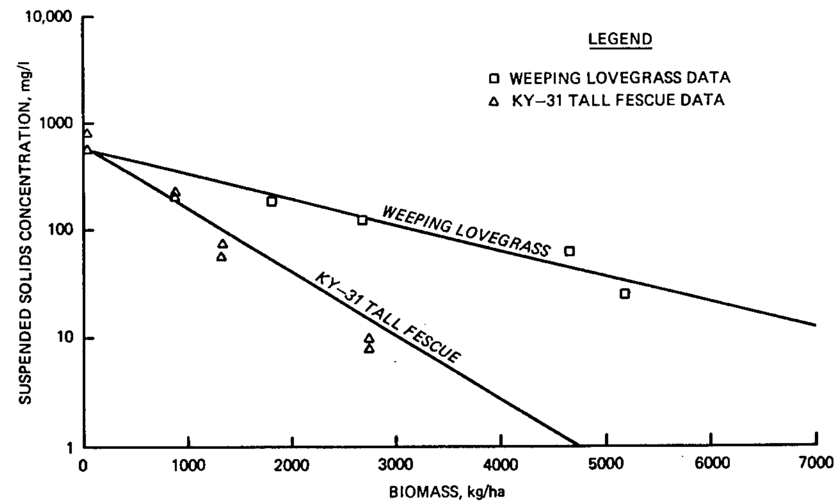

Figure 5. Suspended solids concentrations versus biomass from laboratory simulations (Lee and Skogerboe, 1984)

Summary

The WES RSLS has been very effective in predicting surface runoff water quality from the upland disposal of dredged material and quantifying contaminant runoff from existing sites. A simplified test to screen sediments for potential contaminant problems has been developed and is under further evaluation. Soil erosion as affected by various vegetation types and coverages has been quantified for various sites. The WES RSLS can be applied to a wide array of surface runoff problems and land management strategies to assist in the evaluation and elimination of environmental impacts to adjacent environments.

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Contaminants Database

by
Charles H. Lutz[1] *and Victor A. McFarland*[1]

Introduction

Many chemical contaminants released into the environment become associated with soils and sediments, are transported in surface waters, and unless degraded by natural processes may be bioaccumulated by exposed organisms. When the sediments of rivers and harbors are proposed for dredging, Public Law (Section 404 of the Clean Water Act, and Section 103 of the Marine Protection, Research, and Sanctuaries Act) requires that they be evaluated for ecological effects before disposal at open water sites. Standard testing procedures are provided in the implementation manuals for the Laws: the "Green Book" for Section 103 (USEPA/USACE 1991), and the "Inland Manual" for Section 404 (USEPA/USACE 1994). The manuals share a tiered approach to determine the suitability of dredged sediments for open water disposal.

Bioaccumulation is first addressed in Tier II in the manuals by a screening test, the "theoretical bioaccumulation potential" (TBP), that applies an AF to sediment chemistry data to estimate the potential for bioaccumulation of neutral organic chemicals such as polychlorinated biphenyls, polynuclear aromatic hydrocarbons, chlorinated pesticides, or dioxins/dibenzofurans. The AF expresses the ratio of the concentration of a neutral organic chemical in the organic carbon fraction of a sediment to the chemical concentration in the lipids of an exposed organism at equilibrium or after prolonged exposure. Knowing the concentration of chemical and the organic carbon content of a sediment, and the approximate lipid content of an organism of concern, a TBP calculation indicates the magnitude of bioaccumulation that can be expected.

The Green Book and the Inland Manual both use a default AF value of 4 to calculate TBP for all neutral organic chemicals, and for all combinations of sediment and biota. However, it has been demonstrated that different classes of organic chemicals, different organisms, and different exposure and measurement practices can result in substantial deviations from this default value (McFarland and Ferguson 1994, McFarland 1995), leading to inaccurate estimates. Conversely, the use of laboratory and field-generated AFs for specific chemicals and types of organisms and sediments in TBP calculations can be highly predictive (Ankley et al. 1992, Cook et al. 1993, McFarland and Ferguson 1994, McFarland 1995). The Contaminants Database has been created to compile and make readily accessible empirically measured AFs as an alternative to the default AF for use in TBP calculations or for other purposes. In addition, other data concerning contaminated sediments are available. Data may include location, chemical concentration in sediment and tissue, collection methods, and type and duration of exposure. Searches can be conducted by chemical, waterbody, organism, state and country. These data can be used to put into perspective levels of contamination encountered in a local project, or to identify previous evaluations that may have been done in a project area.

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Data Sources

The data entered in the Contaminants Database come from a variety of sources. Laboratory and field-generated AFs and contaminated sediment data are extracted from refereed journal articles, symposia and conference proceedings, government reports, and otherwise unpublished sources such as the reports of contractors. Only experimentally obtained AFs are entered into the database and complete citations are given allowing the user to go to the original source for quality assurance/quality control data and other information. When available, the following data are included with each record: sample location, sample type, common and scientific name of organism, chemical name, tissue and sediment chemical concentrations, sample identification, AF value including variance and type of variance, exposure type and duration, percent lipid, percent TOC, lipid analysis methodology, and bibliographic citation. When another convention is used to report the lipid/organic carbon-normalized concentration ratio in the source (for example, BSAF or PF), the ratio is reported as AF in the database record. If necessary, a conversion is made. For example, PF or APF (the reciprocal of AF) has been used in some reports.

Program Description

The Contaminants Database program is written in dBASE IV™, which allows large amounts of data to be manipulated. The program is designed to be a simple vehicle for searching data and it is not necessary to be familiar with dBASE IV™ to use the Contaminants Database program. The program is completely menu driven, requiring only that the user select the search parameters desired (from a selection list) and the type of output wanted for the report (screen, printer, or file). Once these inputs are made, the requested report is automatically generated.

Availability

The Contaminants Database program may be accessed on line via The Contaminants Bulletin Board System (BBS) by dialing 601-634-4380 using a microcomputer and modem (see Lutz and McFarland 1994 for complete instructions). Currently, modems up to 14400 baud are supported using data parameters of N,8,1. Set your terminal emulation to IBM-PC if available or ANSI-BBS if not. After logging in to the BBS, The Contaminants Database is accessed by selecting the *(D)oors* option from the main menu, then door 1. The program will load and present you with the opening menu. The Contaminants Database may also be installed on your own microcomputer in a stand-alone mode. The necessary files to do this may be downloaded from file area 4 on The Contaminants BBS. Four files must be downloaded to run The Contaminants Database on a microcomputer: RUN.EXE, CONTAMDB.EXE, CONTAMDO.EXE and AFINSTAL.BAT. These are the dBASE™ engine, data files, program files and the installation program. In addition, the file AFMAN.EXE contains the user manual and should also be downloaded. The CONTAMDO and CONTAMDB files will be updated as new data and changes are made to them. The file RUN.EXE is the same file used by the first version of The Contaminants Database and the Ocean Disposal Database. If you already have either of these programs on your microcomputer, you do not need to download another copy of that file. The copy you already have will work.

The on line version of The Contaminants Database accessed through the BBS is slightly different than the stand-alone version. The same data are accessible in either version, but the menu options available on the BBS version only allow the caller to view data or create reports for viewing or downloading. Adding, editing and printing data are not allowed on the

BBS. The stand-alone version contains the complete program with all the data addition/ editing options and with print capability. The Contaminants BBS is the distribution mechanism for the most current versions of data and program updates for The Contaminants Database.

A copy of The Contaminants Database program may also be obtained via electronic mail. Send a message to lutzc@ex1.wes.army.mil requesting the program. Some electronic mail providers do not pass files attached to an electronic mail message, causing corruption or total loss of the files. The Corps of Engineers x400 electronic mail system has no problem passing attached files.

History of the AF Database

The AF Database was originally designed to be accessed as a separate program via the Contaminants BBS, described in Lutz and McFarland (1994). Based on feedback from system users, the AF Database was combined with The Contaminants Database in 1995. This action eliminated duplication between the two databases and provided a uniform platform for both types of data, allowing many of the same modules and databases to be shared. The AF Database no longer exists as a discrete program, but AF-related data can be found in The Contaminants Database. The search options allows the user to choose between AF and contaminants data.

Installation

In order to run this program in a stand-alone mode, the following minimum hardware must be available. An 80286-compatible microcomputer, 1 MB of RAM, MS DOS 3.3, and a hard drive with at least 8MB of free disk space. A printer is recommended. The program will run with these minimum requirements but a faster processor and more RAM will accelerate program execution. The Contaminants Database program can be installed under Microsoft Windows™ 3.1, although it is a DOS-based program. Be aware that if you do not have enough low memory available, you will experience random lockups when running under Windows. An icon file (AF.ICO) is included with the program. The Contaminants Database has not been tested under WIN 95, but it should work with fewer problems than under Windows 3.1.

After the necessary files are downloaded, the program must be installed on a hard drive to work properly. Each of the first three files are compressed and must be uncompressed before they can be used. Installation is simple and the installation program performs the necessary uncompression chores. To install The Contaminants Database on your microcomputer, copy the four files (RUN.EXE, CONTAMDB.EXE, CONTAMDO.EXE, AFINSTAL.BAT) to a subdirectory on your hard drive (e.g., CONTAM), change to that directory and type AFINSTAL. After the installation program finishes, type DIOXIN to start the program. In order to conserve disk space, the original four files may be copied to floppy disks for storage and deleted from the hard drive. If you have a printer attached to your microcomputer, the first option you should choose after initially starting the program is to select a printer driver. The program installs a generic printer driver as the default. To change from the generic driver, select *Defaults, Printer*, and press F2 from the main menu. You will be presented with a list of printer manufacturers and models, allowing you to select the correct one. If you are not certain of the correct printer port to choose, most non-networked printers are installed on LPT1. In some instances you may have to quit The Contaminants Database program and restart it to invoke the newly selected printer driver. Directions on using the program may be found in Lutz and McFarland (1995) or the electronic user manual.

Using The Contaminants Database

From the opening menu, highlighting a menu item results in the appearance of a short text message at the bottom of the screen describing what that selection does. Pressing the enter key will activate the highlighted menu item. The program contains options for data entry or editing in addition to report generation, however, most users will be interested in getting data from the database as opposed to entering data into it. Selecting the *Reports* option allows you to extract data from the database. Under *Reports* there are two options, *Contaminants* and *AF*. To locate AF data, the current version of the program permits searching by three different qualifiers: *Chemical*, *Organism*, or *Chem./Org.* (chemical/organism combination). Highlight one of the search options using the arrow keys, and press the Enter key. A mouse may also be used if it has been installed for DOS programs. Selecting the *Chemical Reports* option will bring up a small window requesting you to enter the name of the chemical for which you are seeking AF data. Pressing F2 at this point will cause a list to pop up allowing you to select a chemical for which there are data in the database. It is recommended that you use this selection list instead of simply typing in a chemical name. The program must match chemical names exactly to find data, so any typing error will prevent data from being located. After selecting the chemical, press the enter key. The final task is to select the type of report required, *screen, printer* or *file*, and the report will be generated. Selecting *screen* or *printer* will send the report to that device. Selecting *file* will prompt you for a filename and an ASCII text file with that name and ending in *.TXT* will be created in the Contaminants Database directory. If the printout format does not look correct, you may not have selected the correct printer driver for the program to use (see Installation, above). Searching by *Organism* or *Chem./Org.* is accomplished in the same manner as searching by *Chemical*. You will find this a very easy way to query the database for AF values. The Contaminants data are searched in the same manner, but additional options are available to narrow searches.

Several key combinations used by the on-line BBS version of the software are different from the stand-alone version due to the failure of many communications programs to correctly send function and arrow keys. The required key combinations are clearly displayed on the appropriate BBS screens.

Current Database Status

The Contaminants Database currently contains over 2500 records, covering 32 discrete genera and several other organism groupings and 240 contaminants. The Contaminants Database/BBS is an on going project with new data being periodically added. Since AFs are used for predicting organic chemical concentrations, the data gathering effort has concentrated on those chemical groups. The database contains very little information on metals.

This project is sponsored by the Long Term Effects of Dredging Operations (LEDO) program from Headquarters, U.S. Army Corps of Engineers.

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TNT Transformation in Soils Under Controlled Eh/pH Conditions

by
Cynthia B. Price,[1] *James M. Brannon,*[1] *and Charolett A. Hayes*[2]

Introduction

The explosive 2,4,6-trinitrotoluene (TNT) is widely used in propellants and explosive charges. Manufacturing processes have been underway for several decades in the United States and have generated wastewaters containing as much as 100 mg/L TNT (Palazzo and Leggett 1986) with former disposal practices resulting in contamination of soil and groundwater at a number of active and inactive munitions sites. TNT and some of its transformation products have been shown to be toxic to fish and other aquatic organisms (Nay et al. 1974; Won et al. 1974).

There are relatively few data describing the processes that control the transformation of TNT in soils. Previous studies have shown that TNT undergoes transformation (McCormick, Feeherry, and Levinson 1976; Kaplan and Kaplan 1982; Pennington, and Patrick 1990; Pennington et al. 1992). Both abiotic and biotic transformations in soils have been reported (Ainsworth et al. 1993) as well as transformations under both oxidized and anaerobic conditions (McCormick, Feeherry, and Levinson 1976; Pennington and Patrick 1990).

The objective of this study was to determine the effects of different oxidation-reduction (redox) potentials and pH levels on TNT transformation in soil. Soil components responsible for TNT transformation were also investigated.

Materials and Methods

Eh/pH Incubations

The Sharkey clay soil used in this study is an agricultural surface soil from the Mississippi River floodplain. Organic matter (0.5 percent w/w) collected from the Atchafalaya Basin was added as an energy source for the oxygen-consuming microorganisms. Tests were conducted in 2,800 ml Fernbach flasks. Sufficient distilled deionized water and soil were added to produce an 18:1 water to solids ratio. The soil slurries were kept in suspension by magnetic stirring and were maintained at room temperature (~30.5 °C).

Control of redox potential and pH in the systems was maintained using the methods developed by Patrick, Williams, and Moraghan (1973) with some modifications (Brannon 1983) (Figure 1). The suspensions were allowed to incubate and stabilize for approximately 2 weeks, depending upon the Eh level, at the chosen Eh/pH levels before addition of TNT. Laboratory investigations consisted of four Eh levels, +500, +250, 0, and -150 mV. Tests at each Eh level were incubated with each of four pH levels, 5.0, 6.0, 7.0, and 8.0, carried out in duplicate. Following the initial incubation period 100 ug of TNT/g soil was added to each flask. Slurry samples were withdrawn at 1, 4, 9, and 14 days after addition of TNT and the samples were separated by centrifugation.

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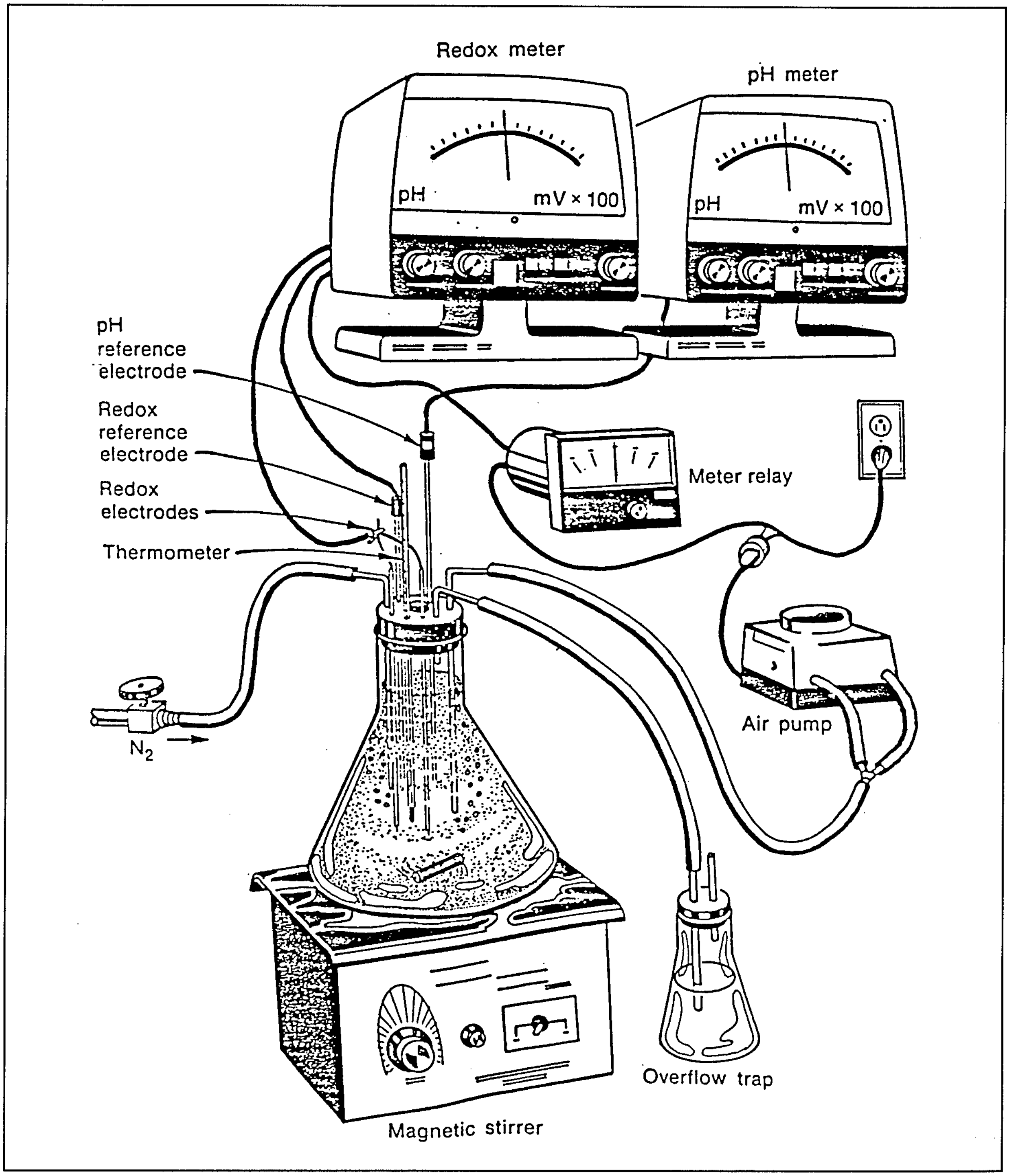

Figure 1. System used to control Eh and pH over time

Water samples were analyzed, at all 4 sample times, for TNT, 1,3,5-trinitrobenzene (TNB), 1,3-dinitrobenzene (DNB), 4-amino-2,6-dinitrotoluene (4A-DNT), 2-amino-4,6-dinitrotoluene (2A-DNT), 2,6-dinitrotoluene (2,6-DNT), 2,4-dinitrotoluene (2,4-DNT), 3,5-dinitroanaline (DNA), 2,6-diamino-4-nitrotoluene (2,6-DANT), and 2,4-diamino-6-nitrotoluene (2,4-DANT). Analyses were performed according to EPA SW-846 Method 8330 (EPA 1982). Soil samples were analyzed, only on day 14, for the above constituents and also total tetranitroazoxytoluenes (EPA 1990).

Mass Balance

Mass balance of TNT in the Eh/pH reactors was determined by using uniformly ring-labeled TNT ((^{14}C)TNT). Tests were conducted in duplicate at +250 and -150 mV at pH 7. Procedures for loading and incubating the slurries were identical to earlier testing. A bubble trap containing 1 N KOH was added to each of the reactors to trap CO_2. Traps were sampled on a daily basis. The waters and soils were sampled on day 14. Radiolabeled TNT and transformation products in the aqueous phase and KOH were determined on a Packard Tricarb 2500 TR Liquid Scintillation (LS) Counter. The soil was analyzed for radiolabeled products by complete combustion in a Model 307 Packard Sample Oxidizer.

Soil Component Effects

The ability of various soil components (clays and cations) and Fe^{+2} to facilitate transformation of TNT was investigated under anaerobic and aerobic conditions. Tests consisted of distilled deionized water containing sufficient iron oxide, manganese dioxide, aluminum hydroxide, montmorillonite, or kaolinite to produce a solution concentration of 100 mg/L of each component. Each test was spiked with 50 mg/L TNT and shaken for 24 or 72 hours, at which times samples were taken. Ferrous chloride was tested alone under anerobic conditions, and in the presence of the two clays.

Results and Discussion

Eh/pH Incubations

Redox potential and pH of the soil suspensions had a significant effect on TNT transformation and stability. TNT was not stable at any Eh/pH test combination as less than 8 percent of the added TNT remained in any treatment. Under highly reducing conditions (-150 mV), TNT was not present in any pH treatment. The small amount of TNT that was recovered was present in the soil while in the aqueous phase TNT was only found at 0 mV and pH 5 following the 14 day incubation period (Figure 2).

At pH 6, 7, and 8 essentially all of the added TNT disappeared from solution after 24 hours (Figure 3). These results indicate that TNT added to uncontaminated soils would not persist in solution under a wide range of redox potentials and pH. Similar results were also noted by Folsom et al. (1988), who reported that the percent recovery of TNT in soils tended to decrease as soil pH increased.

Transformation products found in solution were strongly related to the Eh of the slurries. Following one day of incubation, the transformation products 4A-DNT and 2A-DNT were present in solution at all Eh/pH combinations (Figure 4). Other products, 2,6-DANT and 2,4-DANT, were present only in the more reduced systems of 0 and -150 mV. No other products were detected. The data in this study show that rapid reduction of nitro groups to amino groups is favored under highly anaerobic conditions. Concentrations of 2,6-DANT and 2,4-DANT increased in the -150 mV treatment compared with the 0 mV treatment with a corresponding decrease in the monoamino compounds. These data suggest that the transformation of TNT is a stepwise reduction process progressing from amination of a single nitro group to two nitro groups. McCormick, Feeherry, and Levinson (1976) reported that the nitro groups on the TNT molecule are reduced in both aerobic and anaerobic systems, and depending upon the intensity of reduction, either one, two, or three of the nitro groups may be reduced to aminos.

Results of solution analyses indicate that as Eh decreased, reduction of nitro groups on TNT

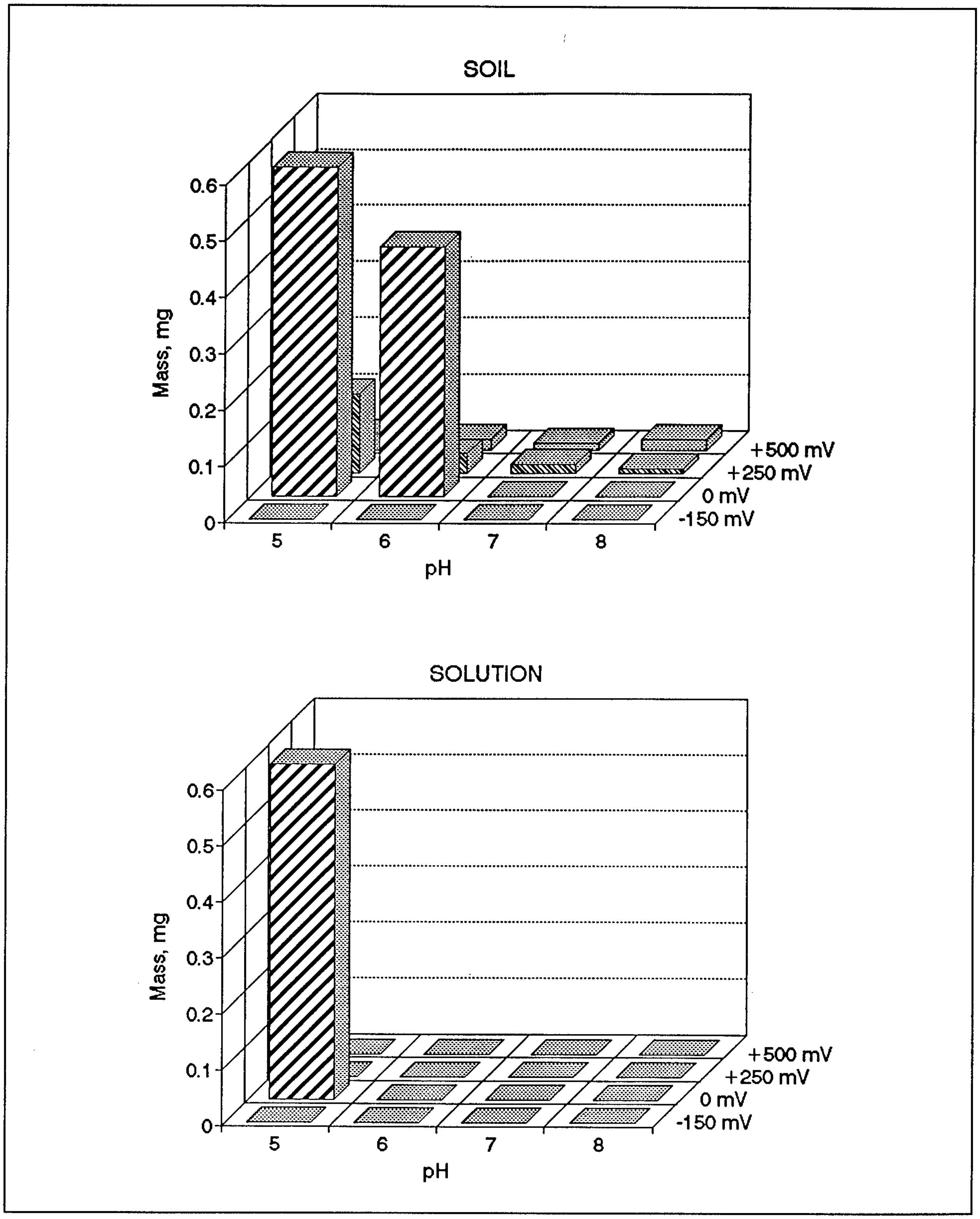

Figure 2. Mass of TNT remaining (aqueous and soil phases) 14 days after addition of 15 mg TNT a various Eh and pH levels

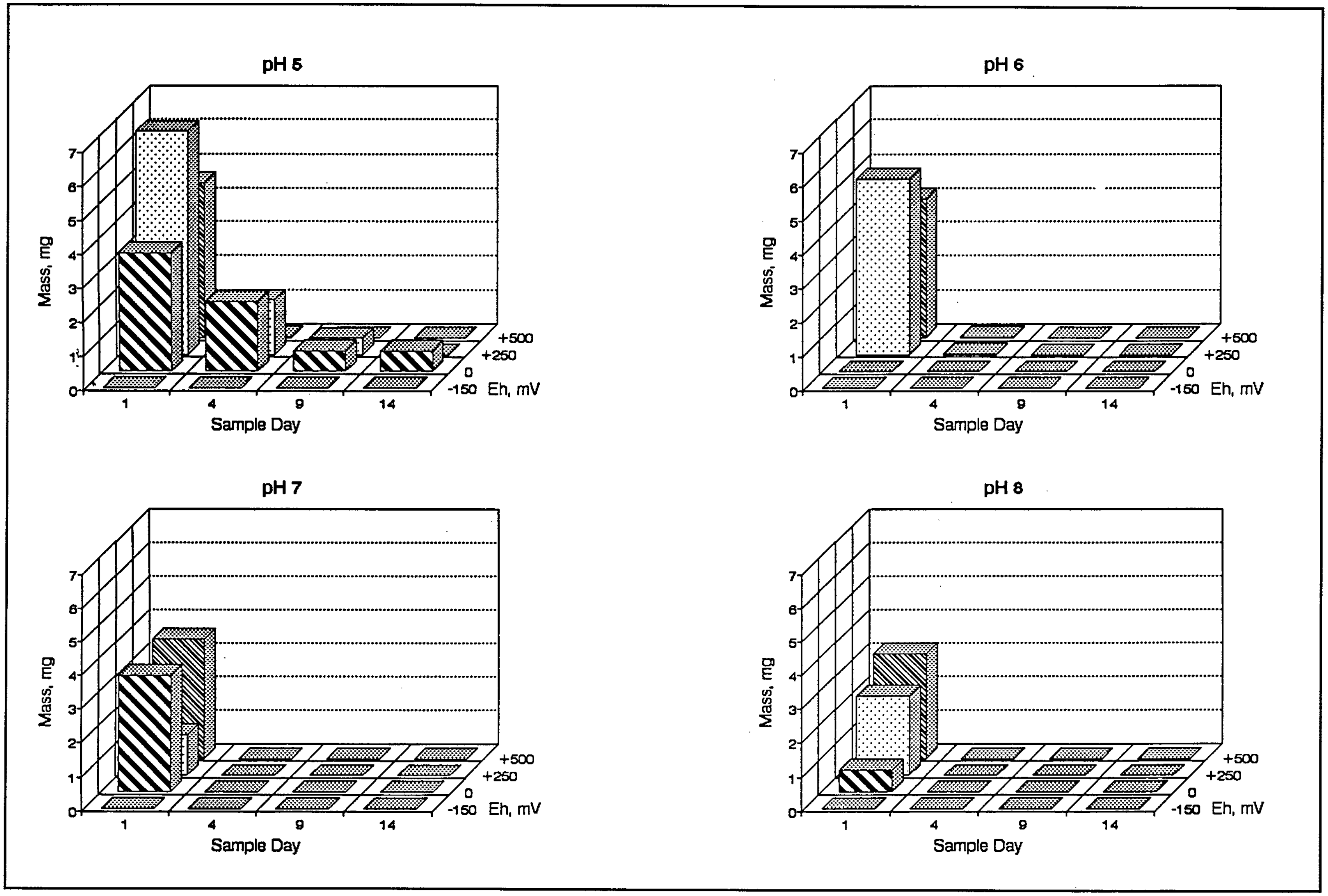

Figure 3. Aqueous mass of TNT over time at each pH level

Figure 4. Aqueous mass TNT and transformation products following 1 day of incubation

to amino groups progressed from one to two nitro groups. Soil analysis at 14 days confirmed this trend (Figure 5). In the -150 mV test, TNT was not present in the soils and transformation products were only present at pH 5. These results indicate that TNT in a highly reduced system would not persist as either the parent compound or its transformation products.

Mass Balance

Results of mass balance investigations showed that most of the radiolabeled carbon was present in the Eh/pH reactors. A small portion was released to the KOH traps as mineralized CO_2. In the +250 mV test 83 percent of the radiolabel was present in the soil, 31 percent in the water, and 2.7 percent in the KOH. The -150 mV tests revealed similar results. The soil contained 82 percent of the radiolabel, with 9.6 percent in the water and 0.09 percent in the KOH. These data support the conclusion that the low percent recovery in the earlier tests was a result of the degradation products being bound to the soil in unknown forms.

Soil Component Facilitated Transformation

TNT disappeared rapidly in the clay + Fe^{+2} treatments (Figure 6). Approximately 90 percent of the TNT was lost within 24 hours after addition. Tests containing Fe^{+2} or clays alone showed no significant effect on the disappearance of TNT. Transformation products were not detected in any of the treatments. These results are consistent with the findings of Heijman et al. (1995) wherein a mechanism for transformation of nitroaromatics was presented; Fe^{+2} sorbed to surfaces was shown to reduce a nitro group to an amino group. Following this reduction TNT can form insoluble and difficult to analyze azoxy compounds (Kaplan and Kaplan 1982). The disappearance of TNT without the detection of transformation products supports this conclusion.

The rapid disappearance of TNT under highly anaerobic conditions may be explained by the presence of Fe^{+2} sorbed to surfaces. Gotoh and Patrick (1974) showed that the reduction of Fe^{+3} to Fe^{+2} is a rapid and continuing process. The sorbed Fe^{+2} can reduce the TNT nitro groups to amino groups and Fe^{+3} which will be quickly reduced to Fe^{+2} again under highly reducing conditions and the cycle continues.

Conclusions

Lower pH levels promoted the stability of TNT, especially at pH 5 under oxidizing and moderately reducing conditions. Reduced soil conditions promoted rapid and complete disappearance of TNT. The soil component investigations showed that the presence of Fe^{+2} sorbed to surfaces may explain the rapid disappearance of the TNT under the highly anaerobic conditions of -150 mV. TNT persists at very low concentrations in soils only under moderately reducing and oxidized conditions at pH 6 or lower.

TNT in groundwater moving into areas of intense anaerobic conditions would not persist. It would be rapidly transformed into mono and diamino compounds, which also disappear from both solution and soil. Soil analysis showed the recovery of only a small portion of the added TNT, indicating that the degradation products of TNT were bound to the soil in unknown and unextractable forms. Radiolabel recovery investigations revealed that the labeled carbon was not mineralized, but remained associated primarily with the soil.

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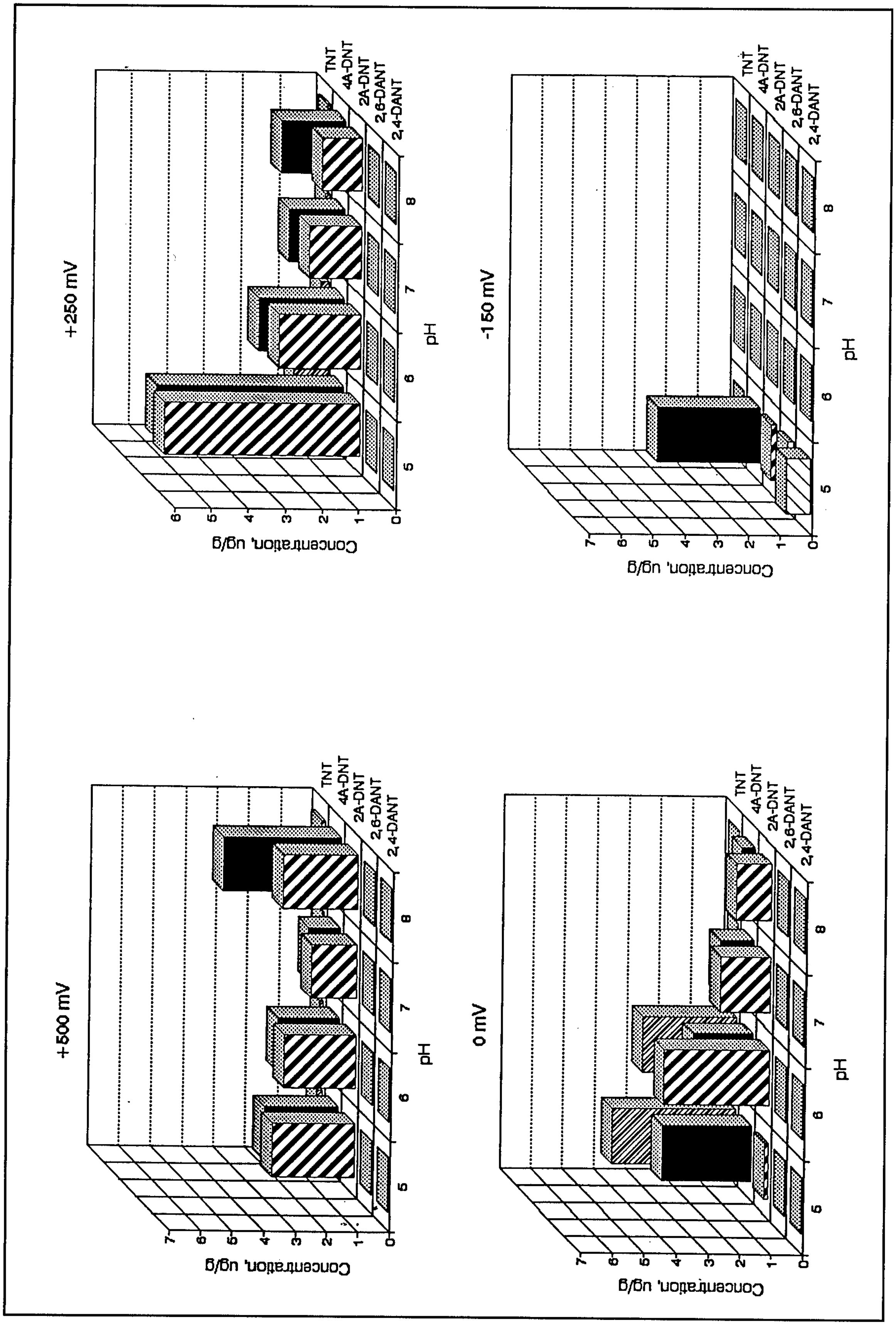

Figure 5. TNT and its transformation products in soil at each pH and Eh 14 days after addition of TNT

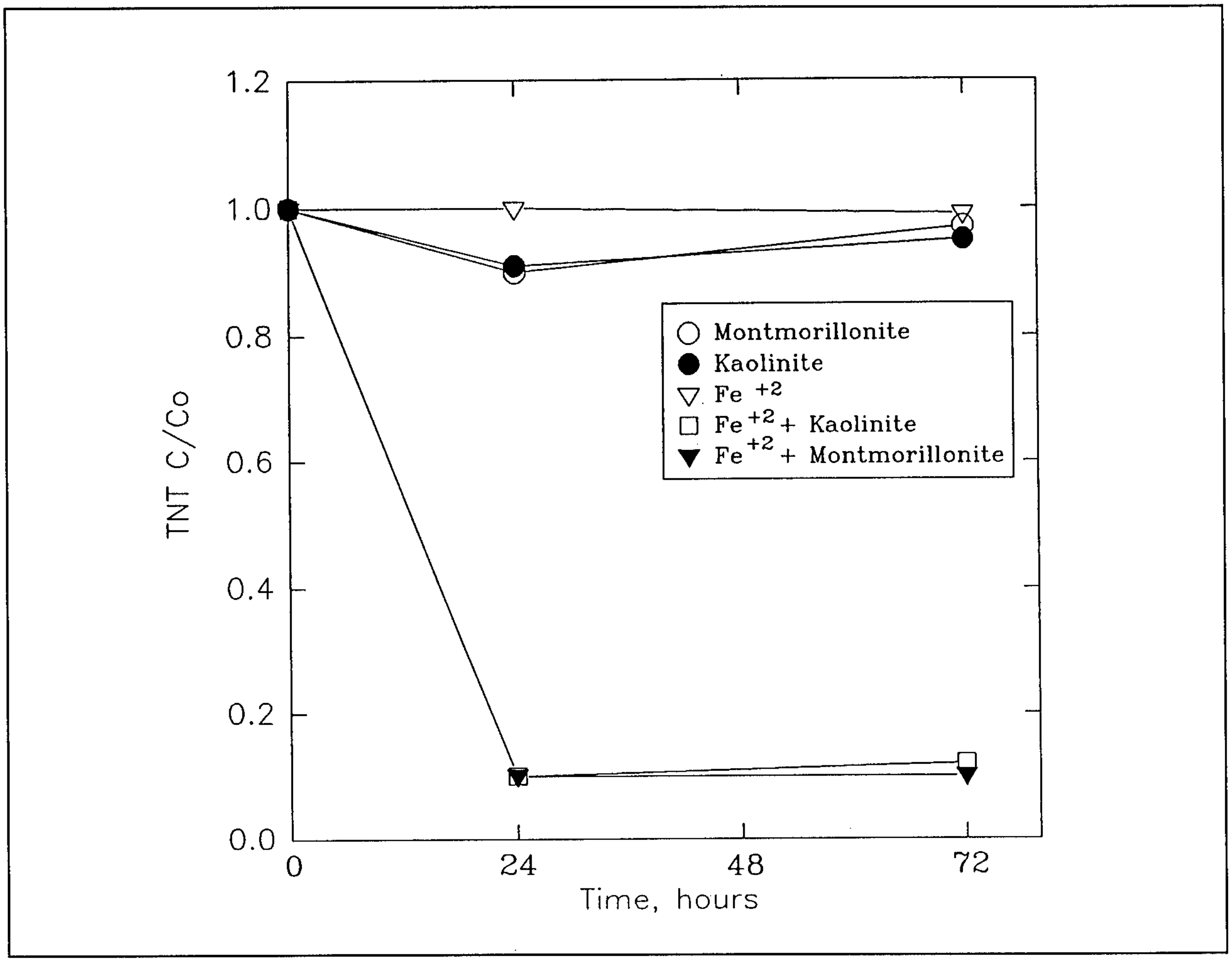

Figure 6. TNT in solution over time at each treatment versus the concentration of TNT at sample time over the initial concentration (C/Co)

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An Integrated, Coordinated Study of the Effects of Dredged Material on the Marine Environment— Mamala Bay Example

by
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Introduction

We initiated an interagency survey/monitoring effort in Mamala Bay, in the vicinity of the EPA-designated South Oahu Ocean Dredged Material Disposal Site (ODMDS), one of five ODMDS's in the State of Hawaii, in anticipation of and in support of the development of Site Management and Monitoring Plans required by the Water Resource Development Act of 1992 for all ODMDS's. Three federal agencies were involved in the study design and funding: U.S. Environmental Protection Agency, Region IX, the U.S. Geological Survey, Menlo Park, and the Corps of Engineers, Pacific Ocean Division. University of Hawaii research staff and private consultants also contributed to the success of the study.

Our primary intent is to define and document the impacts on the marine environment resulting from ocean disposal activities of dredged material. We designed the study to determine firstorder effects of dredged material on the environment. Ultimately, we plan to compare the impacts associated with dredged material disposal with impacts resulting from other anthropogenic inputs (see below).

We chose Mamala Bay off the South coast of the island of Oahu as our study site because we believe that the Bay has the greatest potential for exhibiting adverse impacts as a result of anthropogenic inputs. Mamala Bay has been used as a repository of various materials and wastes for over a century. The majority of material disposed consists of sediments dredged from Honolulu and Pearl Harbors, the most heavily industrialized harbors in the State. In addition, primary-treated wastewater effluent is being discharged into the Bay from two treatment facilities, and other types of material, notably, military debris, were disposed in the study area before ratification of the London Dumping Convention. Moreover, prior to promulgation of the Ocean Dumping Regulations (40 CFR 220-228) in 1977, no pre-disposal testing of dredged material was required.

Three relatively distinct disposal areas have been identified. We refer to them as the Old Pearl Harbor site, the Old Honolulu Harbor site, and the EPA-designated South Oahu site (Figure 1). Neither the Pearl nor Honolulu Harbor sites have been used since designation of the South Oahu site in 1980.

Study Design

The study was undertaken in three phases. Phase 1, conducted in 1993, deployed side-scan sonar and subbottom profilers to map and

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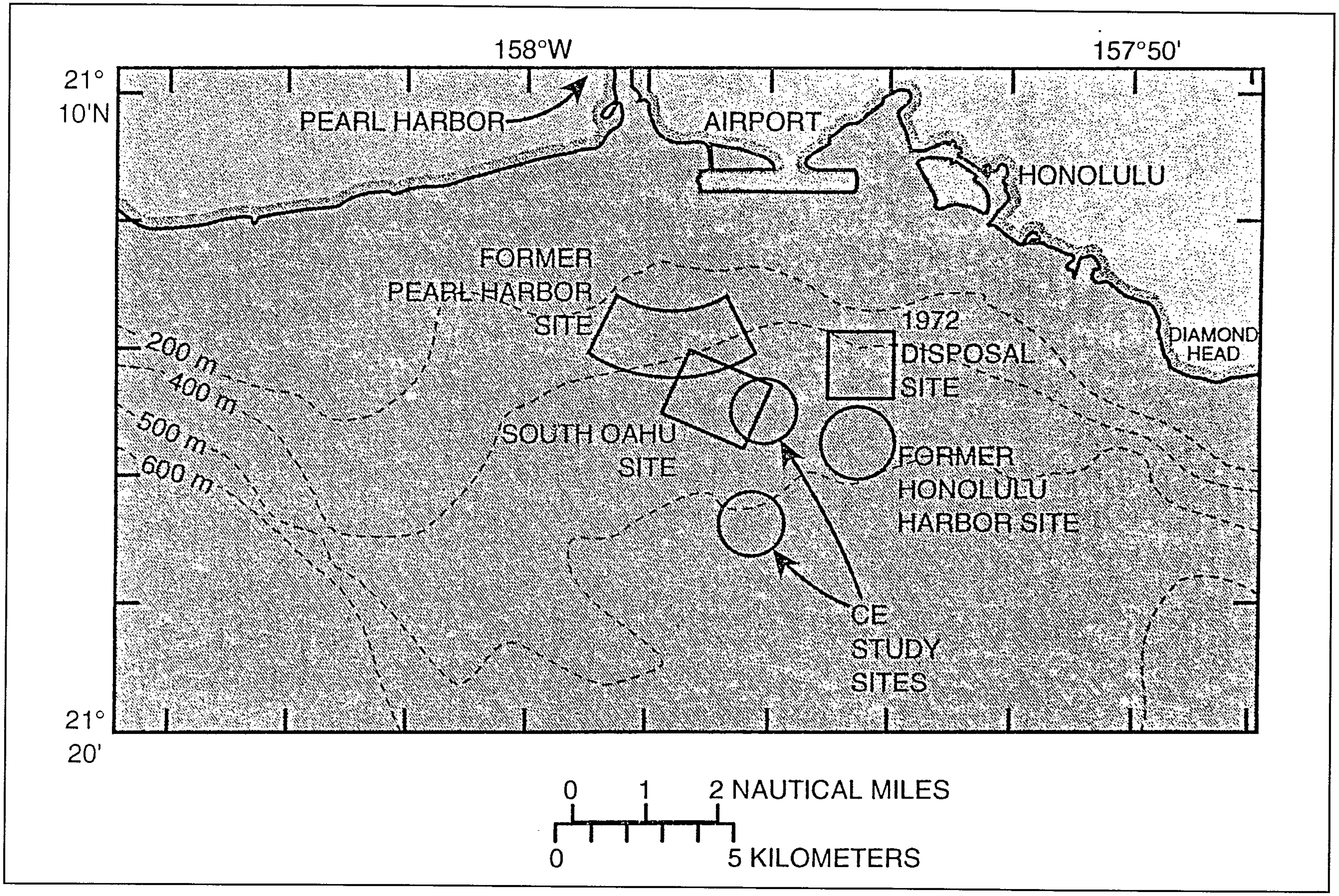

Figure 1. Map of study area showing existing and former disposal areas (two sites examined during the S. Oahu site designation process, and a one-time disposal site are also identified)

characterize the seafloor in the study area, which encompassed all three disposal areas as well as surrounding areas unaffected by disposal activities (Figure 2). The resulting data were used to design Phase 2 in 1994, an extensive study employing subbottom profiling, video and still bottom photography, seafloor sediment sampling. The latter was intended to ground-truth seismic data obtained in Phase 1 and to provide sediment for laboratory sampling, including bulk chemistry, X-radiography, and physical characteristics. 53 box cores recovered sediment from all three disposal areas, including samples of dredged material, and from the surrounding areas not impacted by disposal. Phase 3 in 1995 was designed primarily to evaluate biological impacts. Samples were retrieved from thirty-nine box cores for evaluation of benthic biota community structure (crustaeans, molluscs and polychaetes), including population density and taxonomy. In addition, composite samples from the three disposal areas and composites of three native sediment sites were obtained for laboratory bioaccumulation tests.

Results and Preliminary Interpretations

Prior to the initiation of our study, the commonly-held opinion was that most of the disposed dredged material never reached the seafloor. This opinion, which was stated in the Environmental Impact Statement required by the site designation, was based on the fine-grained nature of the dredged harbor sediments, the depth to the bottom (350 to 500 meters), the existence of mid-water and bottom currents, and modeling results. Most of the dredged material was believed to be dispersed in the water column and then deposited, as a more or less even blanket over a fairly large area. The fine-grained nature of the sediment and great depths could conceivably result in large volumes of material remaining in the water column for long periods of time. Furthermore, a thin, unconsolidated blanket of dredged material was believed to be more susceptible to resuspension and transport by bottom currents. These phenomenon evoke concerns regarding the potential effects of disposal of dredged material, including degradation of water quality and adverse impacts on benthic biota over a wide area.

Side-scan sonar data obtained during Phase 1 of our study indicate that dredged material produces a backscatter signal distinct from that exhibited by native sediment (i.e., that not impacted by disposal). As shown in Figure 3, dredged material appears as light-toned (high backscatter), in contrast to the darker (low backscatter) native sediment. The bright patches in the western portion of the study area were theorized to be drowned carbonate reefs. This theory was confirmed by bottom samples which recovered reef materials. In addition, distinct high-backscatter circular "mound" features on the ocean bottom are observed at the edges of the disposal areas, (particularly adjacent to the Old Honolulu area; see Figure 3) which coalesce in blankets near the centers of the disposal areas. Individual mounds cannot be distinguished in the central areas.

We tentatively interpret the mounds to be the results of single disposal events, strongly suggesting that dredged material descends rapidly and fairly coherently through the water to the seafloor. Evidence of "mud balls" of cohesive fine-grained sediment within cores of dredged material lends evidence to this hypothesis. Such a rapid descent argues strongly against significant adverse water quality impacts, particularly over a long time period, as a result of disposal of dredged material.

Side-scan sonar data, bottom photography, and visual descriptions of core samples indicate that most dredged material remains where it was originally disposed, and apparently little is resuspended and transported by bottom currents. Thus, dredged material deposited within the disposal site tends to stay within the site. Nevertheless, the presence of sand waves and other bedforms do suggest the existence of currents capable of moving sediment. Current directions are variable throughout the study area and additional work to better define current velocities and directions is proposed.

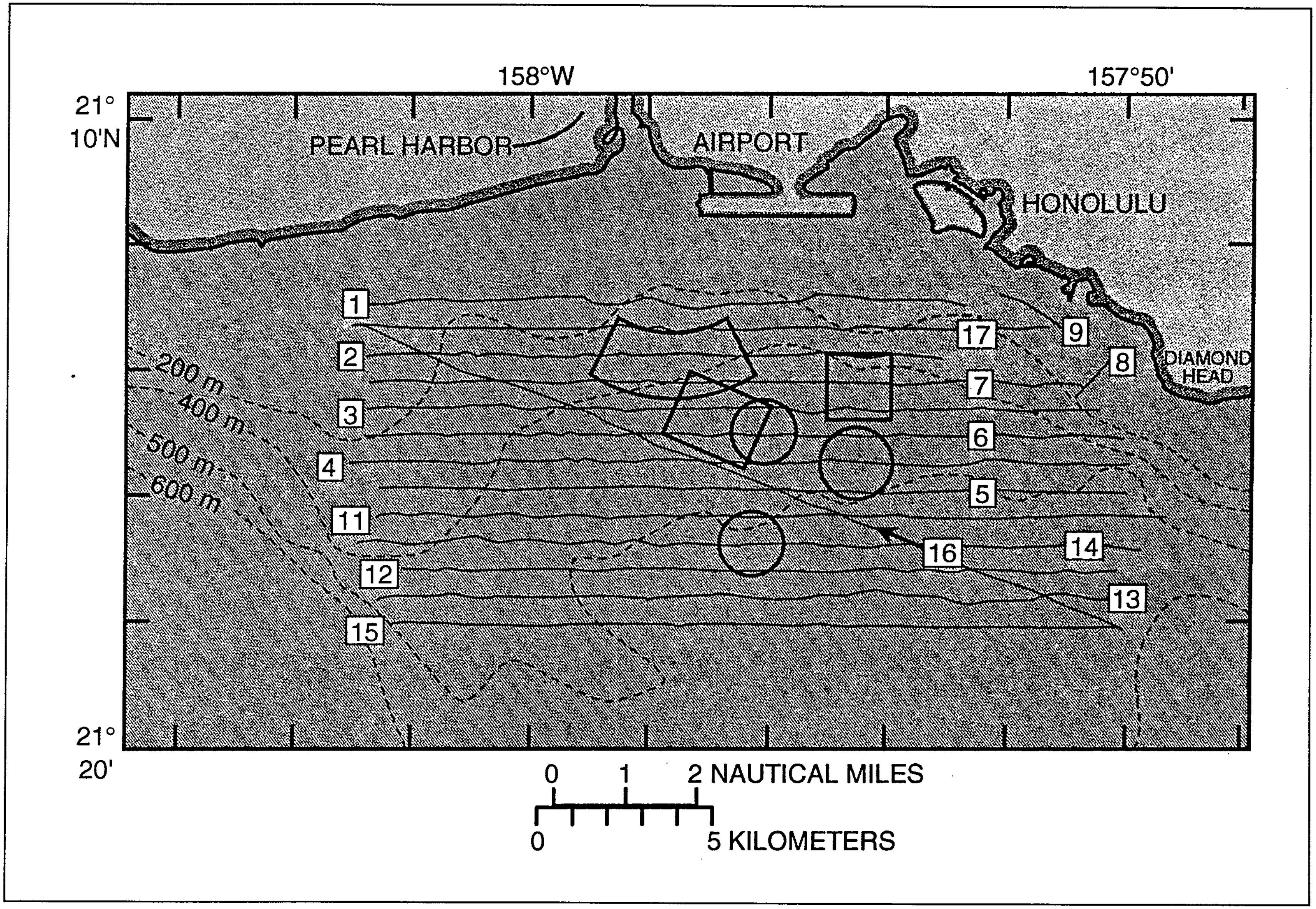

Figure 2. Ship-tracks showing locations of side-scan Sonar Data

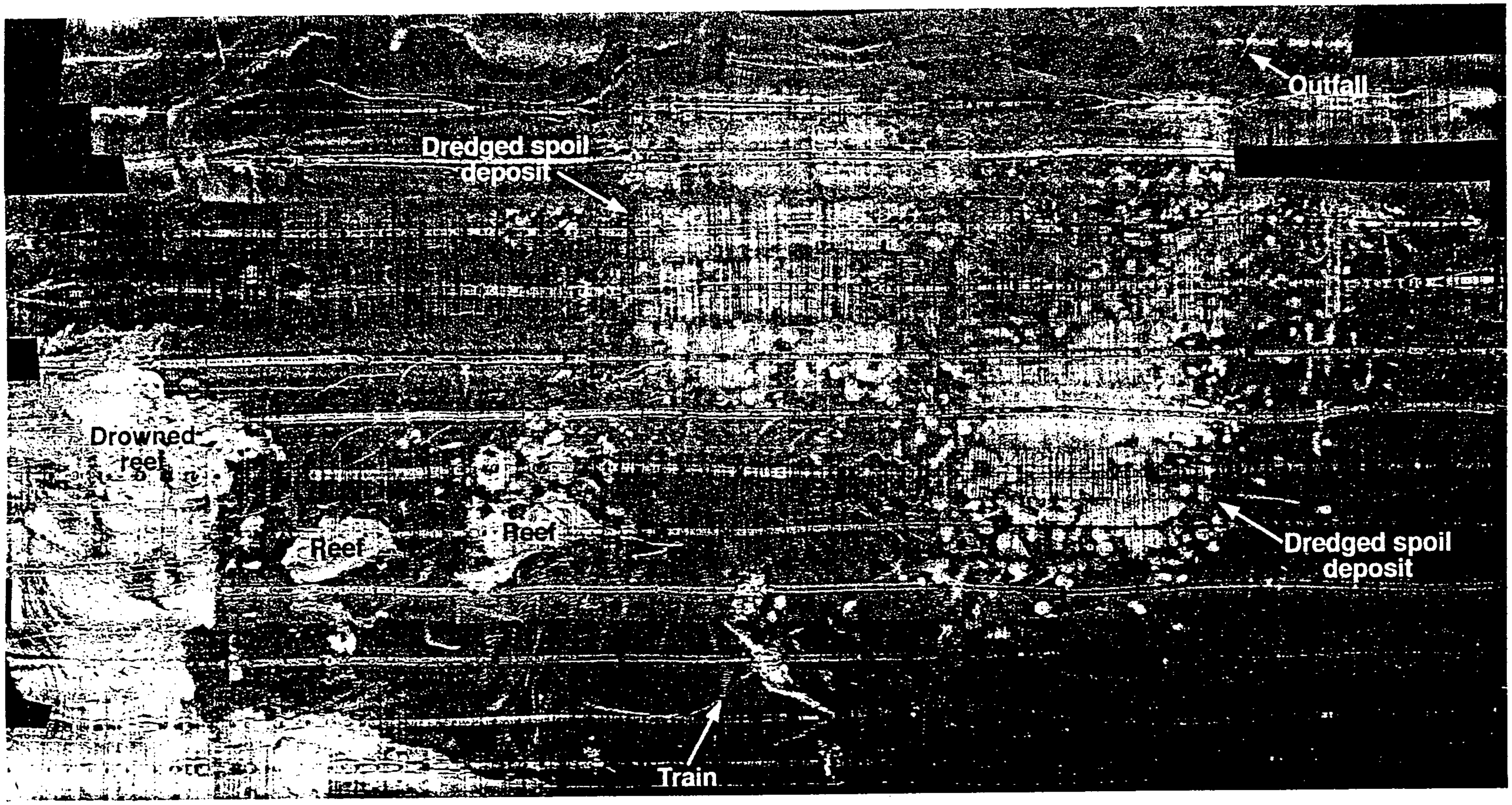

Figure 3. Side-scan Sonar Mosaic of study area in Mamala Bay

Dredged material is differentiated from native sediment both visually and in X-radiographs. As described in core logs, the dredged material is characterized as a fine-grained, cohesive grey clayey-silt to silty-clay, with cobble-sized material consisting of coral fragments, shell and man-made detritus, commonly admixed. Native sediment is tan to beige colored silty sand to sandy carbonate material. Burrows and other evidence of biota-sediment interaction is ubiquitous in the native sediment and was also observed within dredged material deposits. Dredged material, over 15 cm thick, was recovered in box cores within the disposal areas. In a few incidences, a layer of dredged material was overlain by coarser-grained, lighter-colored native sediment. The bases of these deposits were either sharply defined or bioturbated, as a result of benthic infaunal burrowing. Burrows within the dredged material deposits indicate that at least some organisms are living in the dredged material.

Benthic community structure analyses, undertaken in FY 1995, are intended to evaluate the variability in quantity and type of organisms in the native sediment and between the dredged material and native sediment. Although these analyses are not yet complete, preliminary results suggest that total abundances of benthic organisms in the upper three to five centimeters of the dredged material are greater than that in the upper three to five centimeters of native sediment. Similarly, the taxonomic diversity in the upper portions of the dredged material is greater than that in native sediment.

Final bioaccumulation test results are not yet available. However, we do have data on organism mortality in the 28 day flow-through tests. Two test organisms, Macoma natsuta (a mollusc) and Nepthys caecoides (a burrowing polychaete) exhibited high survival rates in dredged material obtained from the disposal areas (86 to 99 percent and 86 to 96 percent, respectively; J. Word, pers. comm., 1995).

Although preliminary, the biological data suggest that organisms do recolonize the dredged material and that toxicity of the material is not significant. We further speculate that high nutrient concentrations in the dredged material relative to the native sediment may attract organisms.

Bulk chemical analyses of the sediments have yielded inconclusive results. Nearly all analytes, chosen from the list of contaminants of concern in the region, exhibited low concentration values. Measured concentrations of pesticides, semi-volatile organics, PCB's, and polynuclear aromatic hydrocarbons (PAH's) were mostly below detection limits. Metals, however, exhibited higher concentrations in the native sediment relative to the dredged material in 1994 samples (arsenic, chromium and nickel; Figure 4), while other analytes (mercury; Figure 5) displayed relatively high concentrations (hot spots), and others (e.g., cadmium; Figure 5) exhibited both higher and lower concentrations in the dredged material relative to native sediment. Samples obtained during Phase 3 in 1995 show conflicting results. Analytes elevated in dredged material 1994 samples do not necessarily show the same pattern in the 1995 samples. Our chemical results further suggest that although elevated concentrations of certain components exist within the dredged material, these concentrations appear to be localized. Even samples collected at different times (Phase 2 in 1994 and Phase 3 in 1995) from the same stations exhibit different concentrations, and some vary greatly (Table 1).

Preliminary Conclusions and Further Study

Although additional data, particularly bioaccumulation test results and benthic community structure data, and rigorous statistical evaluations are needed, our tentative conclusions are that the disposal of dredged material in Mamala Bay has not resulted in significant adverse environmental effects to the water column or to the benthos. We expect that additional data and analyses will confirm this theory.

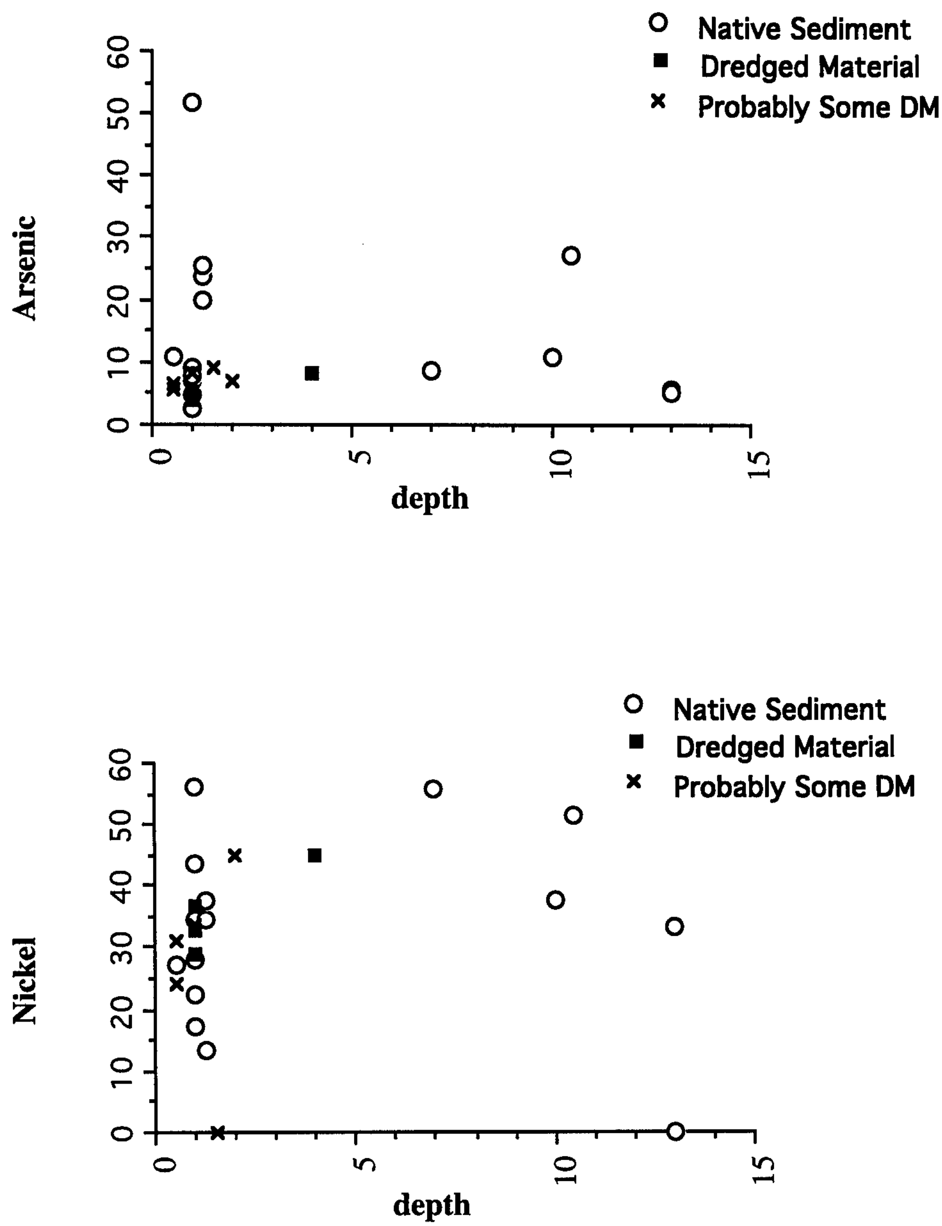

Figure 4. Bulk concentrations of arsenic and nickel in dredged material and native sediment from Mamala Bay (1994 data)

In addition, we plan to compare, semi-quantitatively, the effects of dredged material on the marine environment with the effects of other anthropogenic inputs in the study area. This will entail evaluation of the impacts associated with discharge of primary treated wastewater through two ocean outfalls and impacts related to the disposal of other anthropogenic material prior to the London Convention. We plan to work with, and derive information from, the

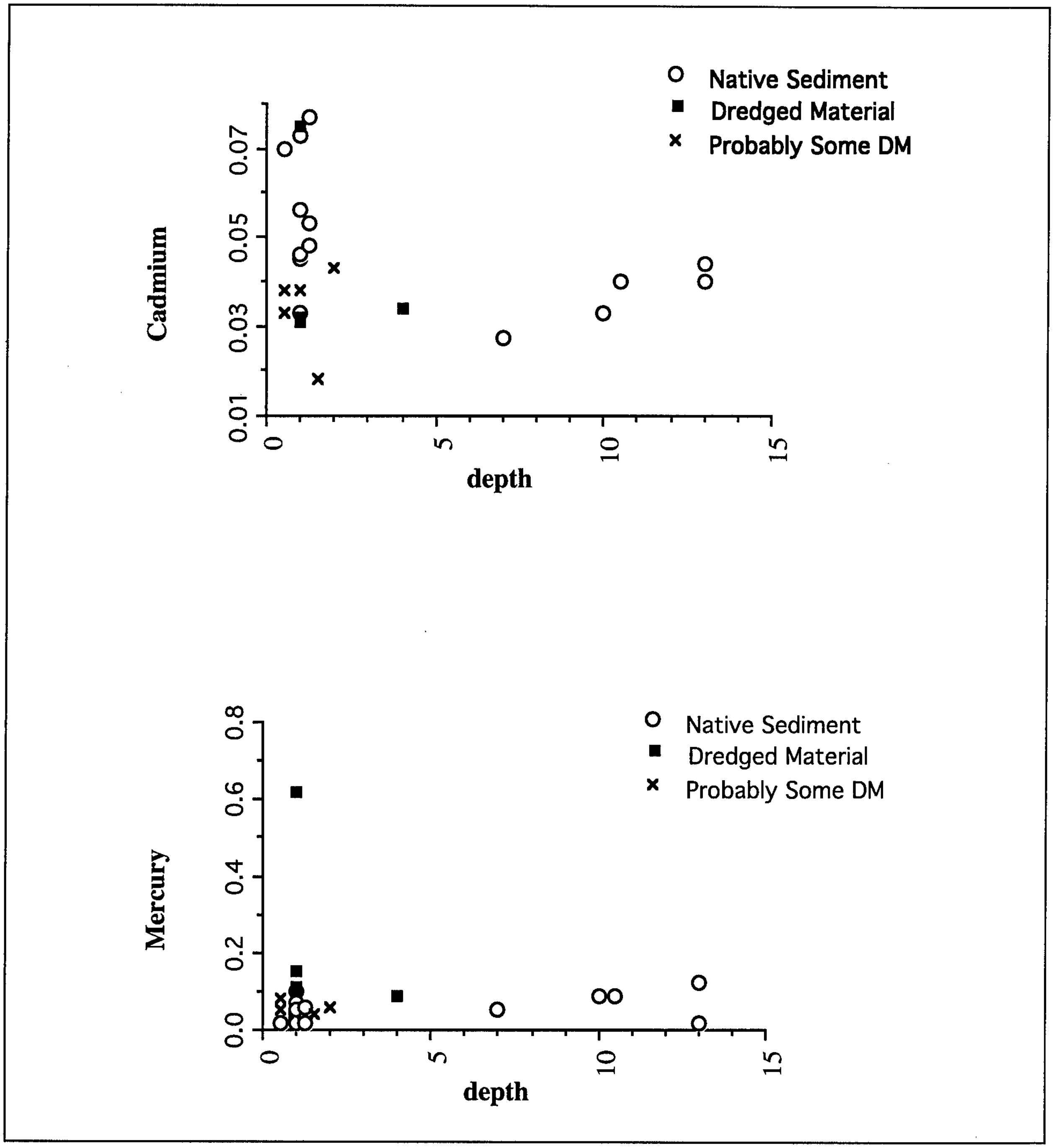

Figure 5. Bulk concentrations of cadmium and mercury in dredged material and native sediment from Mamala Bay (1994 data)

Mamala Bay Study Commission and the University of Hawaii Environmental Center,respectively, to complete these evaluations.

Furthermore, this project serves as an excellent example of interagency coordination efforts and has become a springboard for grant proposals to other agencies. We discovered that inter

Table 1
Comparison of Chemical Concentrations in Samples from the Same Station

Core	Arsenic	Cadmium	Chromium	Copper	DRO	Lead	Mercury	Nickel	TOC	% Solids
1(Phase 2)	4.6	0.031	57.1	2.7	42	8.3	0.11	28.8	0.0083	0.57
72(Phase 3)	4.5	0	50.1	22.8	17	8.2	0.11	28.9	0.81	0.55
25(Phase 2)	3.9	0.032	64.2	25	85	4.5	0.15	32.6	0.0099	0.43
89(Phase 3)	3	0	51.6	23.3	86	5.4	0.1	28.9	0.1	0.5
60(Phase 2)	23.6	0.053	69.9	18.7	12	15.8	0.06	34.4	0.0045	0.58
91(Phase 3)	11.1	0	45.8	14.9	14	13.9	0.051	22.3	0.0028	0.56
46(Phase 2)	19.7	0.048	33.5	4.3	3.9	3.2	0.03	13.3	0.0078	0.68
96(Phase 3)	10.8	0	30.2	6.5	0	1.8	0	16.3	0.019	0.62

Note: All concentrations are reported in mg/kg, except in the case of solids, which is reported as a percentage.
DRO: Diesel Range Organics
TOC: Total Organic Carbon.
Each pair of samples were obtained from approximately the same station (e.g., Cores 1 and 72 were obtained from the same location in the South Oahu ODMDS).

agency work provides at least two significant benefits:

- Cost benefits—use of other federal agencies can reduce costs by nearly 50 percent versus the sole use of private contractors; and

- Enhanced interaction and understanding among regulatory and resource agency staff.

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Water Quality Studies in America's Greatest Wetland

by
Richard Punnett[1]

General

To understand the landscape changes that have resulted in the urgent need to understand and improve water quality in the Everglades, a synopsis of hydrologic changes must first be considered. Therefore, an historical perspective has been provided. Following the perspective, a synopsis of water quality problems and of scope water quality studies was provided. In addition to the water quantity and quality problems, the Everglades has been reduced to only half the original nine million acres.

Historical Perspective

Pre-drainage (prior to 1881)

In the historic Everglades, water overflowed the southern banks of Lake Okeechobee during most wet seasons. The water, combined with local runoff, moved as slow, broad waves so expansive that they were nearly imperceptible. The waves filtered through the vegetation in a lingering journey through vast, wet prairies. The wave movement provided a dynamic storage of fresh, clean water which nourished the wetland through dry seasons. Sawgrass, with its ability to thrive in a low-nutrient environment, dominated much of the landscape. South of the sawgrass plains were extensive open-water prairies with periphyton-based communities. The fresh water not only supported the expansive marshes, abundant with diverse and beautiful creatures, but created rich, productive nurseries where it mixed with salt water. It was the abundance of fresh, low-nutrient water and warm climate that created the unique Everglades.

Wetland Drainage (1881 to 1928)

In 1847, U.S. Senator Westcott, from Florida, made the first proposal to drain the lower peninsula of Florida. The State of Florida contracted with Hamilton Disston in 1881 to begin construction of drainage canals. This work continued until Disston's death in 1896.

In 1906, the State of Florida contracted to have 225 miles of canal made through the Everglades. By 1927, a total of 440 miles of canals, 47 miles of levees, and 16 locks and dams were constructed. This enabled the first significant increase in farm settlement south of Lake Okeechobee.

In 1926 and 1928, hurricanes resulted in the death of more than 2,400 people and great financial loss. These disasters led to involvement of the Federal government.

Water Control Activities by the Corps of Engineers (1930 to 1992)

At the behest of the public, through the U.S. Congress, the Corps of Engineers (the Corps) began south Florida flood control activities in 1930 when a dike was placed on the southern rim of Lake Okeechobee to protect the growing agricultural communities.

In 1947, 100 inches of rain fell (over twice the normal amount) ending a severe drought. Then two hurricanes and a tropical depression

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[1] U.S. Army Engineer District, Jacksonville; Jacksonville, FL.

resulted in 90 percent coverage of water over the land from Orlando to the Keys. In 1948, the Corps developed a comprehensive report on flood control, drainage, water supply and related problems in south Florida. From that report, the U.S. Congress approved the Central and Southern Florida Flood Control Project. A congressional act of 1948, and subsequent acts in 1954, 1958, 1960, 1062, 1965, and 1968, expanded on earlier works to create 990 miles of levees, 978 miles of canals, 30 pump stations, 212 control and diversion structures, and 25 navigational locks.

By 1970, it was clear that significant reductions in wildlife, especially wading birds, had occurred. In 1970, an act provided for minimum flow to the Everglades National Park (ENP). This act was further enhanced in 1989 to restore natural hydrological conditions to the ENP.

Current Activities by the Corps of Engineers (1992 to 1996)

Because the activities of the Corps are directed by Congress, they have reflected the values of society. As such, the Corps has been, and continues to be, the executor of public will. However, the public will has shifted priorities on environmental values. Therefore, the activities of the Corps have shifted emphasis as well. The reasons for the priority shift has been postulated and presented in a later section titled: Levels of Environmental Awareness.

In 1992, the Water Resources Development Act directed the Corps to begin a reconnaissance-level "Restudy" of the Central and Southern Florida (C&SF) Project with ecosystem restoration as a priority. It was clear that over a decade of human activities had caused environmental damage that was beyond early predictions. The final report of the reconnaissance-level Restudy[1] (a three-volume, double-sided, 4-1/2 inch thick tome) defined the problems and opportunities for the C&SF Project. Additionally, the report identified several plans having potential improvements to the C&SF Project. However, many shortcomings in the understanding of the restoration actions were also identified.

In September of 1995, the Corps began a Feasibility Study for restoration of the Everglades including the improvement of existing project purposes. The $20 million, 4-year study is being cost-shared by the South Florida Water Management District. In the Plan of Study[2], many required studies were identified. However, the listing does not encompass the scope of water quality studies in south Florida.

Water Quality Issues

Although a popular premise to wetland restoration was "wet it and it will grow," the hydrology was not the only problem in the remaining Everglades. Severe water quality problems exist today. Years of upstream agricultural activities have resulted in huge nutrient loads into historically oligotrophic marshes. Catastrophic landscape and water quality changes have been documented. The atmospheric deposition of mercury and internal production of methylmercury have resulted in toxic levels in some animals and high concentrations in fish throughout the system. Fortunately, pesticide or herbicide pollution problems are decreasing.

Because of the changes in "hydropatterns" (in flow magnitudes, timing, and location) of the Everglades, it has been difficult to separate water quantity impacts from water quality

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1 Reconnaissance Report, Central and Southern Florida Project, Comprehensive Review Study, U.S. Army Corps of Engineers, Jacksonville District, November 1994.

2 Plan of Study, Comprehensive Review Study, U.S. Army Corps of Engineers, Jacksonville District, June 1995.

impacts. As a result, ongoing studies have to deal with both possibilities. For example, there has been a landscape shift from sawgrass to cattails, but it is difficult to pinpoint the exact cause of the change when both water level and nutrient changes are documented—perhaps it is a combination of both. Twenty-six animal species have been listed as endangered or threatened. There has been a 90 percent decline in the wading bird population alone over the last few decades.

Water Quality Studies

To account for all the ongoing studies related to understanding the role of nutrients in the current system would be unreasonable for this paper. However, a 100-page South Florida Ecosystem Restoration Plan[1] covers the water quality studies underway. To begin to describe the details of these water quality studies would make this paper to lengthy. The Restoration Plan discusses the various basin areas individually and provides a good overview of the goals, costs, and schedules. The Plan, which covers most projects, will cost almost $2 billion to complete.

The scope of the ongoing water quality studies can be gleaned from the main study purposes:

1. To establish a Florida State Standard for water quality (including nutrients and metals) which would assure no imbalance in native fauna or flora;

2. To be able to numerically model both chemical and water quantity interactions within the wetlands including vegetation and animal interactions of the region; and

3. To be able to predict the effects of untested alternative plans for restoration.

To accomplish these purposes, laboratory tests, mesocosym tests, and field tests and model development are underway. The magnitude of the program is supported by several agencies including the South Florida Water Management District, the Florida Department of Environmental Protection, the U.S. Geological Survey, the Environmental Protection Agency, the U.S. Department of Interior, the Corps, universities, private businesses (such as the Sugar Cane Growers Cooperative), public interest groups, and tribes. Thus, hundreds of millions of dollars are committed to the water quality effort alone.

Levels Of Environmental Awareness

To characterize the process of gaining the public support necessary to commit vast resources to improve and protect the environment, Levels of Environmental Awareness are herein proposed.

Level 1: Reduction in Animals

There has a small cross-section of the public that predicts dire consequences of environmental changes. They may have been correct; however, science was not sufficiently developed to support such predictions. Usually, the first significant clue that the environment has been harmed may be a noticeable reduction of wildlife. Since natural population fluctuations occur, this first sign takes years to document. Draining a "swamp" (something not as attractive as a "wetland") provides an example. After the initial drainage, there may still be an abundance of wildlife, but a shift in species eventually takes place. Certain individuals would be aware of the changes, however the general public would not be concerned. Hence, the First Level of Environmental Awareness can be

[1] South Florida Ecosystem Restoration Plan, South Florida Water Management District, West Palm Beach, Florida, November 1995.

characterized by meager public support ("trouble-makers") with no significant local, state, or federal governmental involvement.

Level 2: Decrease in Healthy Conditions

There seems to be a public sector that will recognize and accept actions that increase health risks as long as there are associated benefits. Living high smog areas, pouring pollutants in sewer systems, and using persistent pesticides on crops are examples. However, there also coexists a public sector that opposes these actions. Hence, the Second Level of Environmental Awareness can be characterized has having vocal public support with some local or state government involvement (making rules).

Level 3: Decrease in Liberties

At this level, a large public sector has been adversely impacted by environmental degradation and the provocative environmental regulations that followed. Restriction on the use of certain chemicals, restrictions on the freedom to clear-cut timber or drain the wetlands on private property, and restrictions on hunting and fishing as well as on consumption are examples of the provocative regulations. At this level, the environment has been seriously impacted, the causes have been clearly identified, and the restrictions make the problem well known. Hence, the Third Level of Environmental Awareness can be characterized by local, state, and sometimes federal government involvement (making and enforcing rules) with restricted freedom and financial losses incurred by some of the public.

Level 4: Loss of Revenues

This is the proverbial straw. At this level, the public is outraged over economic losses, both potential and realized. Key public officials perceive that the issue has campaign importance. A clear case of cause and imminent adverse financial consequence exists. For example, consider the economy of the Florida Keys (tourism based) which is dependant upon the health of Florida Bay and the reefs. If water quality conditions make sport fishing and diving undesirable, the economic base could collapse. As another example, if the mangrove zones (south of the Everglades) become less productive due to fresh water quality and quantity changes, a \$600 million-a-year fishing industry could wane. Hence, the Fourth Level of Environmental Awareness can be characterized by involvement of special interest groups, all levels of state and federal government with massive amounts of aid (money) to resolve the problem. Like a "Code Blue" situation in a hospital emergency room, there may be a frantic effort to resuscitate the ecosystem.

In Closing: Hope For The Future

The public sector (numbering in the millions), having reached the Fourth Level of Environmental Awareness, has motivated the state and federal agencies to "fix" the Everglades. It may seem ironic that the Corps would be directed to lead in the restoration of the Everglades since the Corps was also directed in years past to institute water control (including drainage) throughout the system. However, the Corps is the only federal agency which has the ability to carry-out the congressional mandates that require huge infrastructure changes. The public will has changed; and with that change, the mission of the Corps in Florida has changed.

When considering the water quality in America's greatest wetland and environmental losses to date, it is clear that the Restudy is the last hope of sustaining the reduced essence of the original Everglades. Because of the immense nature of restoring the Everglades and because so much is still unknown, it will take four years to complete the study. The overall restoration effort is the most ambitious in the world and it will take 10 to 15 years to complete. The goal is to establish a sustainable Everglades which accommodates the hominids; solving water quality problems is essential for success.

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The U.S. Army Corps of Engineers 11th Seminar on Water Quality was held in Seattle, WA, 26 February - 1 March 1996, to provide an opportunity for the presentation of work efforts involving water quality activities. Papers presented at the meeting are included in this report.

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