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Environmental Protection
Agency

Office of Water
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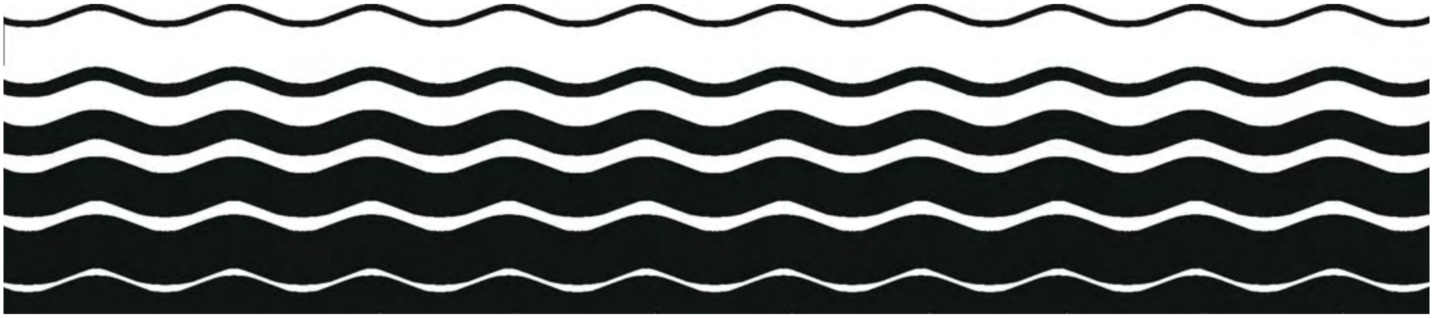
September 1995

Technical Guidance Manual for Developing Total Maximum Daily Loads

Book II: Streams and Rivers

Part 1: Biochemical Oxygen Demand/Dissolved Oxygen and Nutrients/Eutrophication



TITLE: Technical Guidance Manual for Performing Wasteload Allocations,
Book II: Streams and Rivers –
Part 1: Biochemical Oxygen Demand/Dissolved Oxygen and
Nutrients/Eutrophication

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ABSTRACT

As part of ongoing efforts to keep EPA's technical guidance readily accessible to water quality practitioners, selected publications on Water Quality Modeling and TMDL Guidance available at http://www.epa.gov/waterscience/pc/watqual.html have been enhanced for easier access.

This document is part of a series of manuals that provides technical information related to the preparation of technically sound wasteload allocations (WLAs) that ensure that acceptable water quality conditions are achieved to support designated beneficial uses. The document:

- Emphasizes the need for water quality managers to consider key water quality interactions and ecological responses to point and nonpoint source loadings in streams and rivers;
- Provides technical guidance on modeling, reaction rate coefficients, and field measurement techniques and
- Provides the recommended TMDL procedures for biochemical oxygen demand (BOD), dissolved oxygen (DO), and nutrients discharged into streams and rivers.

Book II Part 1 presents the technical basis for analysis of BOD, DO, nutrient, and eutrophication impact. It also discusses some of the mathematical models available to perform TMDL calculations, provides guidance on model selection, and uses case studies to illustrate key steps in constructing a site-specific for a TMDL. Detailed appendices provide additional discussions of important fate and transport processes, quality assurance for field monitoring, and uncertainty analysis.

KEYWORDS: Wasteload Allocations, Rivers, Streams, Biochemical Oxygen Demand, Dissolved Oxygen, Eutrophication, Nutrients, Modeling, Water Quality Criteria

MEMORANDUM

SUBJECT: Final Technical Guidance Manual for Developing Total Maximum Daily Loads (TMDLs)

FROM: Tudor T. Davies, Director
Office of Science and Technology (4301)

TO: Regional Water Division Directors
Regional Environmental Services Division Directors
Regional TMDL Coordinators

Attached for national use, is the final Technical Guidance manual for Developing Total Maximum Daily Loads, Book II: Streams and Rivers, Part 1: Biochemical Oxygen Demand/Dissolved Oxygen and Nutrients/Eutrophication. We are sending extra copies of this manual to the Regional TMDL coordinators for distribution to the States to use on performing TMDLs.

Section 303(d) of the Clean Water Act requires States to perform wasteload allocations (WLAs) and Total Maximum Daily Loads (TMDLs) for waters where technology-based treatment is found to be inadequate to meet State water quality standards (WQS). As a part of our technical assistance effort in performing WLAs, primarily involving controls of point source discharges, the Office of Water issued a series of technical guidance manuals. More recently, we issued guidance for the 303(d) program *Guidance for water quality-based decisions: The TMDL Process*, 1991 in response to the U.S. General Accounting Office (GAO) report *Water Pollution - Greater EPA Leadership Needed to Reduce Nonpoint Source Pollution*, October 1990.

We are issuing this TMDL technical guidance manual to support the implementation of the 1991 TMDL guidance mentioned above. This document provides guidance on how to assess water quality impacts of point and nonpoint source discharges of biochemical oxygen demanding (BOD) pollutants and nutrients to streams and rivers. More details of what this guidance includes are stated in the document under the heading “To the Reader” on page iii.

The earlier drafts of this document have been reviewed by your staff, and some of them made significant contribution to its development. Also, the document has been peer reviewed by technical experts. This final guidance reflects all comments and suggestions received on the earlier drafts.

If you have any questions, comments or desire additional information, please contact Hiranmay Biswas, Standards and Applied Science Division (4305), Telephone: (202) 260-7012.

Attachment

TO THE READER

This guidance manual represents the consolidation of the following two documents in the U.S. Environmental Protection Agency's series of *Technical Guidance Manuals for Performing Waste Load Allocations*:

> USEPA. 1983. *Technical Guidance Manual for Performing Waste Load Allocations, Book II: Streams and Rivers, Chapter 1: Biochemical Oxygen Demand* and *Dissolved Oxygen.*
>
> USEPA. 1983. *Technical Guidance Manual for Performing Waste Load Allocations, Book II: Streams and Rivers, Chapter 2: Nutrients and Eutrophication.*

The revised single manual, renamed *Technical Guidance Manual for Developing Total Maximum Daily Loads, Book II: Streams and Rivers, Part I: Biochemical Oxygen Demand/Dissolved Oxygen and Nutrients/Eutrophication*, eliminates duplicative information on hydrodynamics and physical characteristics of streams and rivers, and on the interactions of nutrients and dissolved oxygen dynamics, that was included in the above- cited manuals. The availability of a single manual also helps to meet the needs of water quality managers to adequately consider the key water quality interactions and ecological responses to pollutant loadings in streams and rivers. In addition, this manual includes updated information on modeling, reaction rate coefficients, field measurement techniques, etc. and includes several examples using EPA-supported models. More specifically, these changes and updates include:

Integration of principles and concepts on waste load and load allocations for nutrients/eutrophication with those for carbon (BOD) and oxygen balances in aquatic ecosystems (see Chapter 2 - *Basic Principles and Concepts*).

- Update of model identification and selection, with emphasis on the EPA-supported water quality model QUAL2E (see Chapter 3 - *Model Selection and Review*) and additional mention of watershed models.
- Update of water quality reaction rate coefficients and field measurement techniques (see Chapter 4 - *River and Stream Modeling Procedures*, Appendix A - *Development of Model Coefficients and Constants,* and Appendix C - *Quality Assurance for TMDL Studies*).
- *Update of technical literature citations (see Chapter 5 - References and Appendix E, Supplemental Bibliography).*
- *Inclusion of a TMDL example using QUAL2E and WASP5 (see Appendix B - Example Total Maximum Daily Load Analys*is).
- Inclusion of an uncertainty analysis example using QUAL2E-UNCAS (see Appendix D - *Uncertainty Analysis*).

Comments and suggestions from the user community help us in improving our guidance manuals, and we invite the user community to send their comments and suggestions to:

Hiranmay Biswas
U.S. EPA
Office of Science and Technology
Standards and Applied Science Division (4305)
Washington, DC 20460

ACKNOWLEDGEMENTS

The contents of this section have been removed to comply with current EPA practice.

TABLE OF CONTENTS

APPENDICES

LIST OF FIGURES

LIST OF TABLES

1. INTRODUCTION

1.1 PURPOSE

The purpose of this guidance manual is to present the most recent information and techniques for use in preparing total maximum daily loads (TMDLs) when excessive biochemical oxygen demand (BOD), low dissolved oxygen (DO), and excessive nutrients and eutrophication impair the water quality of streams and rivers. This manual reflects current policy on TMDL development as outlined in the *Guidance for Water Quality-based Decisions: The TMDL Process* (USEPA, 1991a), and represents the consolidation of the following two documents in the U.S. EPA's series of *Technical Guidance Manuals for Performing Waste Load Allocations*:

> USEPA. 1983. *Technical Guidance Manual for Performing Waste Load Allocations, Book II: Streams and Rivers, Chapter 1: Biochemical Oxygen Demand and Dissolved Oxygen*

and

> USEPA. 1983. *Technical Guidance Manual for Performing Waste Load Allocations, Book II: Streams and Rivers, Chapter 2: Nutrients and Eutrophication*.

This revised manual, renamed *Technical Guidance Manual for Performing Total Maximum Daily Loads, Book II: Streams and Rivers, Part 1: Biochemical Oxygen Demand/Dissolved Oxygen and Nutrients/Eutrophication*, eliminates duplicated information on hydrodynamics and physical characteristics of streams and rivers. The objectives of the manual are (1) to emphasize the needs of water quality managers to adequately consider the key water quality interactions and ecological responses to both point and nonpoint source loadings in streams and rivers; (2) to provide technical guidance on modeling, reaction rate coefficients, and field measurement techniques; and (3) to provide the recommended TMDL procedures for biochemical oxygen demand, dissolved oxygen, and nutrients for point sources and nonpoint sources discharging into streams and rivers. This manual includes several examples using EPA-supported models. Much of the information needed by the water quality analyst to design and develop a TMDL for a stream or river (i.e., model selection and design, field measurements, assignment of reaction rate coefficients, and analysis of the TMDL) is contained within this manual.

This guidance reflects the current policy on TMDL development for streams and rivers, which requires the consideration of pollutant loads from all sources within a watershed. TMDLs should be developed to provide more stringent water quality-based controls when technology-based controls are inadequate to achieve water quality standards (USEPA, 1991a). TMDLs are composed of waste load allocations (WLAs) for point sources, load allocations (LAs) for nonpoint sources, and a margin of safety (MOS). The MOS accounts for scientific uncertainty involved in establishing the TMDL. This uncertainty can be caused by insufficient or poor-quality data or a lack of knowledge about the receiving water and pollution effects. The TMDL process consists of the following steps:

(1) Identifying water quality-limited waters still requiring TMDLs.

(2) Prioritizing and targeting water quality-limited waters.

(3) Developing the TMDL.

(4) Implementing the TMDL through control actions.

(5) Assessing whether step 4 actions are sufficient to meet water quality standards.

The *Guidance for Water Quality-based Decisions: The TMDL Process* (USEPA, 1991a) discusses these procedures for TMDL development within the context of a water quality-based watershed approach. The following guidance is intended primarily to assist the water quality analyst in the third step of the TMDL process with regard to developing TMDLs to control BOD, DO, and nutrients in streams and rivers.

The level of effort required to develop a TMDL is highly dependent on the complexity and magnitude of the receiving water problems. In general, to assess the anticipated level of effort, site-specific conditions need to be evaluated in terms of the type and composition of loads, the variability and characteristics of pollutant sources and their response to local hydrologic events, and the characteristics of receiving water. Additional considerations may also involve the local or regional value of the resources being protected and the phase of the TMDL process. Since the TMDL program has directed water and watershed managers toward adoption of a phased approach to address controls on both point and nonpoint source loads under both dry-weather and wet-weather conditions, simplified modeling techniques for low-flow conditions may be of limited use for developing first-phase TMDLs. As water quality goals for a watershed are more clearly defined by first-phase assessments and additional monitoring efforts, intermediate or complex modeling techniques may be required for advanced phases of the TMDL process.

This manual calls for an intermediate level of effort to develop a TMDL for typical cases associated with oxygen depletion and nutrient loadings. Although the models reviewed in this manual accommodate multiple discharges and complex inflow characteristics, the emphasis is limited to less complex scenarios. More detailed modeling techniques are described in the *Compendium of Watershed-scale Models for TMDL Development* (USEPA, 1992b)*, Principles of Surface Water Quality Modeling and Control* (Thomann and Mueller, 1987)*,* and *Water Quality Modeling, Volume 1, Transport and Surface Exchange in Rivers* (McCutcheon, 1989). In special cases where a level of effort less than that presented in this document is deemed necessary, the following documents may be of interest: *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water* (Mills et al., 1985), and *Simplified Analytical Method for Determining NPDES Effluent Limitations for POTWs Discharging into Low Flow Streams* (USEPA, 1980).

1.2 RELATIONSHIP TO OTHER GUIDANCE DOCUMENTS

Table 1-1 summarizes the relationship of the various documents that make up the series of technical guidance manuals for waste load allocations. These manuals describe approaches for allocating point source waste loads in rivers and streams, lakes and impoundments, and estuaries. The pollutants addressed in the manuals listed in Table 1-1 include biochemical oxygen demand/dissolved oxygen, nutrients, and toxic substances (ammonia, organic chemicals, and metals). The manual Simplified Analytical Method for Determining NPDES Effluent Limitations for POTWs Discharging into Low-Flow Streams (Table 1-1) may be used to assist in waste load allocation procedures when such simplifications are valid. In addition, ammonia toxicity is addressed in more detail in the Simplified Methods Manual, which now includes methods for evaluating the interactions of multiple discharges.

Table 1-2 lists available guidance for TMDL development. These documents include guidance on the allocation of nonpoint source loads. These documents assist the water quality analyst in selecting and using appropriate models for development of TMDLs. In addition, an EPA report entitled Technical Guidance for Estimating Total Maximum Daily Loads (TMDLs): Integrating Nonpoint and Episodic Point Source Loading from Stormwater and Combined Sewer Overflows (CSOs) is currently in development and should be available by October 1994. This document is intended to provide technical guidance on the integration of point and nonpoint, steady state and episodic discharges into a waterbody. The guidance will provide several examples of evaluating these discharges within the TMDL process.

Users of this manual also can consult the latest State water quality standards before developing TMDLs. These standards provide applicable water quality criteria for pollutants of concern in the state. Federal water quality criteria for many pollutants are listed in the EPA "Gold Book":

> *USEPA. 1987. Quality criteria for 1986* (with updates 1 and 2 included). "Gold Book." EPA 440/5-86-001. U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Washington, DC.

The "Gold Book" is available from:

U.S. Government Printing Office
Superintendent of Documents
North Capitol and H Street, NW
Washington, DC 20401
(202) 783-3238
Order No. 955-002-00000-8

Usually, however, TMDL developers consult state water quality standards first. Several manuals on modeling and parameter selection are also available. These documents, listed in Table 1-2, are available from:

TABLE 1-1. TECHNICAL GUIDANCE MANUALS FOR PERFORMING WASTE LOAD ALLOCATIONS

Technical Guidance Manual for Performing Waste Load Allocations - Book II Streams and Rivers - Chapter 1 Biochemical Oxygen Demand/Dissolved Oxygen (EPA 440/4-84-020)
This chapter presents the underlying technical basis for performing WLA and analysis of BOD/DO impacts. Mathematical models to calculate water quality impacts are discussed, along with data needs and data quality.

Technical Guidance Manual for Performing Waste Load Allocations - Book II Streams and Rivers - Chapter 2 Nutrient/Eutrophication Impacts (EPA 440/4-84-021)
This chapter emphasizes the effect of photosynthetic activity stimulated by nutrient discharges on the DO of a stream or river. It is principally directed at calculating DO concentrations using simplified estimating techniques.

Technical Guidance Manual for Performing Waste Load Allocations - Book II Streams and Rivers - Chapter 3 Toxic Substances (EPA 440/4-84-022)
This chapter describes mathematical models for predicting toxicant concentrations in rivers. It covers a range of complexities, from dilution calculations to complex, multi-dimensional, time-varying computer models. The guidance includes discussion of background information and assumptions for specifying values.

Technical Guidance Manual for Performing Waste Load Allocations - Simplified Analytical Method for Determining NPDES Effluent Limitations for POTWs Discharging into Low-Flow Streams
This document describes methods primarily intended for “desk top” WLA investigations or screening studies that use available data for streamflow, effluent flow, and water quality. It is intended for circumstances where resources for analysis and data acquisition are relatively limited.

Technical Guidance Manual for Performing Waste Load Allocations - Book IV Lakes and Impoundments - Chapter 2 Nutrient/Eutrophication Impacts (EPA 440/4-84-019)
This chapter discusses lake eutrophication processes and some factors that influence the performance of WLA analysis and the interpretation of results. Three classes of models are discussed, along with the application of models and interpretation of resulting calculations. Finally, the document provides guidance on monitoring programs and simple statistical procedures.

Technical Guidance Manual for Performing Waste Load Allocations - Book IV Lakes, Reservoirs and Impoundments - Chapter 3 Toxic Substances Impact (EPA 440/4-87-002)
This chapter reviews the basic principles of chemical water quality modeling frameworks. The guidance includes discussion of assumptions and limitations of such modeling frameworks, as well as the type of information required for model application. Different levels of model complexity are illustrated in step-by-step examples.

Technical Guidance Manual for Performing Waste Load Allocations - Book VI Design Conditions - Chapter 1 Stream Design Flow for Steady-State Modeling (EPA 440/4-87-004)
Many state water quality standards (WQS) specify specific design flows. Where such design flows are not specified in WQS, this document provides a method to assist in establishing a maximum design flow for the final chronic value (FCV) of any pollutant.

Final Technical Guidance on Supplementary Stream Design Conditions for Steady State Modeling
WQS for many pollutants are written as a function of ambient environmental conditions, such as temperature, pH, or hardness. This document provides guidance on selecting values for these parameters when performing steady-state WLAs.

Technical Guidance Manual for performing Waste Load Allocations - Book VII: Permit Averaging (EPA 440/4-84-023)
This document provides an innovative approach to determining which types of permit limits (daily maximum, weekly, or monthly averages) should be specified for the steady-state model output, based on the frequency of acute criteria violations.

Handbook - Stream Sampling for Waste Load Allocation Applications (EPA 625/6-86/013)
This handbook provides guidance in designing stream surveys to support modeling applications for waste load allocations. It describes the data collection process for model support, and it shows how models can be used to help design stream surveys. In general, the handbook is intended to educate field personnel on the relationship between sampling and modeling requirements.

Technical Guidance Manual for Performing Waste Load Allocations - Book III Estuaries - Part 1 - Estuaries and Waste Load Allocation Models (EPA 823/R-92-002)
This document provides technical information and policy guidance for preparing estuarine WLAs. It summarizes the important water quality problems, estuarine characteristics, and the simulation models available for addressing these problems.

Technical Guidance Manual for Performing Waste Load Allocations Book III Estuaries - Part 2 - Application of Estuarine Waste Load Allocation Models (EPA 823-R-92-003)
This document provides a guide to monitoring and model calibration and testing, and a case study tutorial on simulation of WLA problems in simplified estuarine systems.

Technical Support Document for Water Quality-based Toxics Control (EPA 505/2-90-001)
This document discusses assessment approaches, water quality standards, derivation of ambient criteria, effluent characterization, human health hazard assessment, exposure assessment, permit requirements, and compliance monitoring. An example is used to illustrate the recommended procedures.

Technical Guidance Manual for Performing Waste Load Allocations - Book III - Estuaries - Part 4 - Critical Review of Coastal Embayment and Estuarine Waste Load Allocation Modeling (EPA 823-R-92-005)
This document summarizes several historical case studies of model use in one freshwater coastal embayment and a number of estuarine discharge situations.

Technical Guidance Manual for Performing Waste Load Allocations - Book III: Estuaries - Part 3 - Use of Mixing Zone Models in Estuarine Waste Load Allocations (EPA 823-R-92-004)
This technical guidance manual describes the initial mixing wastewater in estuarine and coastal environmental and mixing zone requirements. The important physical processes that govern the hydrodynamic mixing of aqueous discharges are described, followed by application of available EPA-supported mixing zone models to four case study situations.

These documents are available from the Office of Science and Technology (4305), Washington, DC 20460. See Standards and Applied Science Division Clearinghouse Request Form for document completion dates.

TABLE 1-2. AVAILABLE GUIDANCE AND OTHER REFERENCES FOR TMDL DEVELOPMENT

Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling (Bowie et al., 1985, EPA/600/3-85/040)
This report provides formulations used in surface water quality modeling along with accepted values for rate constants and coefficients. Topics covered include dispersion, heat budgets, dissolved oxygen saturation, reaeration, alkalinity, nutrients, algae, zooplankton, and coliform bacteria.

Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Groundwater, Parts I and II (Mills et al., 1985, EPA 600/6-85/002a and EPA 600/6-85/002b)
Part I of this series describes the aquatic fate of toxic organic substances, waste loading calculations, and the assessment of water quality parameters in rivers and streams. Part II describes the assessment of impoundments, estuaries, and groundwater.

Compendium of Watershed-scale Models for TMDL Development (USEPA, 1992b, EPA 841-R-92-002)
This document identifies and summarizes the most widely used watershed-scale models and is intended to assist in model selection.

Modeling of Nonpoint Source Water Quality in Urban and Non-urban Areas (Donigan and Huber, 1991, EPA/600/3-91/039)
This document presents detailed reviews of established nonpoint source assessment procedures, methodologies, and modeling techniques. Simple procedures (e.g., constant concentration, regression, statistical, and loading function approaches) and complex models (e.g., SWMM, HSPF, CREAMS, SWRRB) are described.

A Quick Reference Guide: Developing Nonpoint Source Load Allocations for TMDLs (USEPA, 1992a, EPA 841-B-92-001)
This document directs TMDL developers to existing technical guidance from other programs while more detailed TMDL technical guidance is developed.

TMDL Case Study Series
This series of case studies published by EPA illustrates real-world TMDL applications that the user may consult when appropriate.

The Enhanced Stream Water Quality Models QUAL2E and QUAL2E-UNCAS: Documentation and User Manual (Brown and Barnwell, 1987, EPA/600/3-87/007)
This manual describes the water quality models QUAL2E, which can be operated as a steady-state or dynamic model, and QUAL2E-UNCAS, which is an enhancement of QUAL2E that includes uncertainty analysis. QUAL2E allows the user to model the effects of diurnal variations and to examine diurnal dissolved oxygen variations caused by algal growth and respiration.

The water quality analysis simulation program, WASP5, Part A: Model documentation, Version 5.10 and The water quality analysis simulation program, WASP5, Part B: The WASP5 input dataset, Version 5.10 *(Ambrose, et al., 1993a and 1993b)*
This manual describes the use of the Water Quality Analysis Simulation Program Version 5.10 (WASP5). The WASP5 modeling system covers hydrodynamics, conservative mass transport, eutrophication-dissolved oxygen kinetics, and toxic chemical-sediment dynamics.

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1.3 ORGANIZATION AND SCOPE OF MANUAL

The remainder of this document is organized into three chapters and five appendices, as summarized below.

Chapter 2, Basic Principles and Concepts, presents the underlying technical basis for analyzing stream biochemical oxygen demand (BOD), dissolved oxygen (DO), nutrient, and eutrophication impacts. The basic theory on transport and fate and the nature of stream system responses to oxygen-demanding loads are described using equations and basic relationships.

Chapter 3, Model Selection and Review, discusses mathematical models available to perform TMDL calculations, with emphasis given to EPA-supported models including Multi-SMP, QUAL2E-UNCAS, and WASP5. Guidance is also provided to assist in identifying and selecting appropriate models for varying levels of complexity (e.g., steady-state vs. dynamic).

Chapter 4, *River and Stream Modeling Procedures,* presents the following procedures to construct a site-specific model for a TMDL: initial assessment, site-specific stream survey, model calibration, model validation, and model application. Examples from actual case studies are given to illustrate key steps in the procedures.

Appendix A, Development of Model Coefficients and Constants, provides a detailed discussion on various fate and transport processes and reaction rates affecting biochemical oxygen demand, dissolved oxygen, and nutrients in rivers and streams such as reaeration, oxidation, nitrification, photosynthesis, respiration, settling, sediment oxygen demand, and ammonia flux. Environmental factors that influence fate and transport processes and technical approaches for determining model parameters are presented. This appendix supplements the overview material presented in Chapter 2.

Appendix B, *Sample Total Maximum Daily Load Analysis,* presents an example that illustrates the TMDL process applied in settings using an analytical solution and the EPA-supported QUAL2E and WASP5 models. The example TMDL problem includes problem setting, river characteristics, treatment plant and effluent characteristics, ambient river water quality data review, model calibration, and model projections. The example illustrates an analysis of the same water quality problem using two different models: an analytical, screening level model and the WASP5 model. The solutions consider nonzero background sources, nonpoint source inputs, and eutrophication problems. The second example problem is based on a study of the Willamette River basin in Oregon. The Willamette River example highlights the use of the QUAL2E model in assessing water quality for a large river.

Appendix C, Quality Assurance for Field Monitoring Programs, provides an overview of objectives and components of a quality assurance plan for field monitoring.

Appendix D, Uncertainty Analysis, provides a discussion on uncertainty analysis as it applies to waste load allocation modeling. An example demonstrating various aspects of uncertainty analysis using QUAL2E-UNCAS is included. Also, through this example, techniques in uncertainty analysis, first-order error analysis, and Monte Carlo simulation are described.

Appendix E, Supplementary Bibliography, includes additional references relevant to oxygen depletion, nutrient enrichment, and eutrophication processes in freshwater and marine ecosystems. These references are not cited in the guidance manual.

Appendix F presents a glossary of technical terms related to the guidance document.

Appendix G presents a list of abbreviations used in the document. Appendix H provides a list of conversion factors for metric and US equivalent units.

Appendix I provides a list of symbols used as nomenclature in the document.

Appendix J provides an attached MS-DOS 3.5 inch diskette containing input files for the QUAL2E and WASP5 example problems presented in Appendix B.

2. BASIC PRINCIPLES AND CONCEPTS

2.1 PURPOSE

This chapter provides an introductory discussion of the primary concepts in conducting analyses of river and stream dissolved oxygen responses to loadings of BOD and nutrients. Section 2.2 provides an overview of the basic principles of the total maximum daily load (TMDL) process. Section 2.3 presents a discussion of the key relationships (i.e., loading, fate and transport by physical and chemical/biological processes) that determine the effect of a pollutant load on oxygen demand and eutrophication in a stream or river. Section 2.4 presents the mass balance principle and governing equations that form the basis for most water quality models used to simulate the key processes of interest.

2.2 OVERVIEW

EPA defines the total loading capacity (LC) or total maximum daily load (TMDL) as the greatest amount of pollutant loading that a waterbody can receive without violating water quality standards. A TMDL is the portion of the LC or TMDL that is allocated to one of its existing or future point sources of pollution. A load allocation (LA) is the portion of the TMDL that is allocated to one of its existing or future nonpoint sources of pollution and natural background. The sum of the individual WLAs for point sources and LAs for nonpoint sources (including natural background sources and tributaries) plus the margin of safety (MOS) is equivalent to the TMDL (i.e., TMDL= LC = WLA+LA+MOS). TMDL studies utilizing field monitoring data and predictive models provide quantitative information to assist managers in making effective decisions to protect water quality. Models and water quality equations are used to establish cause-and-effect relationships correlating incremental changes in stream water quality to changes in pollutant loading. From this correlation, optimum and desirable, but not required cost-effective treatment levels can be specified to achieve water quality standards and criteria. The MOS can be included implicitly in the TMDL model calculations to account for the uncertainty about the relationship between the allocated waste loads and loads and the predicted quality of the receiving waterbody. A reserve capacity for future development can be included in the TMDL at this stage. Wastewater treatment plant designers can then evaluate various combinations of alternative unit processes to select an optimum treatment scheme to meet the requirements of the WLA. Likewise, land use planners and engineers may need to analyze various management scenarios to meet the requirements of the nonpoint source LA. This analysis may include an evaluation of the cost-effectiveness of different combinations of best management practices (BMPs).

Knowledge of the quantitative cause-and-effect relationship between receiving water quality and pollutant loads is the key to making reliable determinations of the total loading capacity. This relationship is quite sensitive to natural environmental conditions. These conditions include physical characteristics such as stream flow, velocity, depth, slope, time of travel, and temperature and chemical/biological characteristics such as in-place sediment oxygen demand, algal photosynthesis and respiration, and nitrification. The determination of the rates at which various water quality reactions take place in the receiving waterbody introduces additional complications in establishing cause-and-effect relationships and projecting water quality impacts. In some instances, the water quality response can be as sensitive to the reaction rates as it is to the total amount of pollutant loadings. This is particularly important in BOD/DO reactions where the resulting dissolved oxygen concentration is determined by competing reactions of oxygen consumption from BOD, nitrification, and sediment oxygen demand (SOD) and oxygen replenishment from reaeration and photosynthesis.

Models not only are used to determine rigorous relationships between pollutant loads and the resulting water quality response, but also are necessary to predict future water quality conditions and conditions that may not have been monitored for in the past (e.g., 7Q10 critical low-flow conditions).[1] Models are also useful to

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[1] There is a 10 percent chance that the 7Q10 critical low flow (7-day average low flow that occurs once in 10 years) will occur during a 1-year monitoring period. In a 10-year monitoring period, there is only a 65 percent chance that critical low-flow conditions will occur.

evaluate the array of variables (temperature, stream flow, load, reaction rates, etc.) that simultaneously influence water quality response, especially where the system is relatively complex as a result of multiple sources, varying stream geometry, flow changes due to tributaries and storm events, and other factors.

2.3 CONCEPTS IN BIOCHEMICAL OXYGEN DEMAND, DISSOLVED OXYGEN, AND NUTRIENT ANALYSES

This section examines the relationships between pollutant fate and transport processes in natural waters and the response of dissolved oxygen in streams and rivers to point and nonpoint source loads. An appreciation of these relationships and related factors should help regulatory staff and water quality and watershed managers to assess the technical complexity associated with the development of a given site-specific TMDL and to recognize the level of modeling and monitoring effort that may be required. Factors that govern the fate and transport processes of pollutant loadings in streams and rivers and determine the effects on dissolved oxygen include those factors related to the magnitude and variability of the pollution sources, the hydrologic conditions of the receiving water, and the in-stream transport of pollutants. Detailed discussion of these factors can be found in Appendix A. In addition, detailed derivation of BOD, DO, and nutrient relationships can be found in Thomann and Mueller (1987).

2.3.1 Pollution Source Characteristics

An important task in watershed and water quality modeling is to characterize the pollution sources and estimate the associated pollutant loadings. Pollution sources can be characterized as either point sources or nonpoint sources. These two categories of sources are governed by different mechanisms, resulting in different impacts on the receiving water. Table 2-1 identifies typical differences between point and nonpoint source characteristics.

TABLE 2-1. COMPARISON OF TYPICAL POINT AND NONPOINT SOURCES

Point Sources	Nonpoint Sources
Fairly steady flow	Highly dynamic flow occurring at random intermittent intervals
Variability changes less than one order of magnitude	Variability changes several orders of magnitude
Most severe impacts occur during low flow conditions	Most severe impacts occur during or following storm events
Fairly predictable concentrations	Unpredictable, variable concentrations

Point Sources - Point source pollutant loads include effluent discharges from municipal and industrial wastewater treatment plants. Point sources are also characterized by pollutant inputs to surface waters from tributaries and other watercourses that aggregate into major surface water systems. Point sources are defined in the Clean Water Act) as ". . . any discernible, confined, and discrete conveyance, including but not limited to, any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, landfill leachate collection system, vessel or other floating craft from which pollutants are or may be discharged. This term does not include return flows from irrigated agriculture or agricultural storm water runoff "(40 CFR Vol. 1 7-1-1990 edition).

Point source loading rates from permitted discharges, such as publicly owned treatment works (POTWs) and combined sewer systems, may be determined from discharge monitoring reports (DMRs) available from many state regulatory agencies or EPA regional offices. Most of these DMRs contain information on many conventional pollutants such as BOD, ammonia, Kjeldahl nitrogen, suspended solids, and fecal coliform bacteria. Not all nutrients are reported in DMRs. For example, facilities that do not have phosphorus removal (e.g., secondary treatment plants) may not measure or report total phosphorus and/or orthophosphate concentrations in their effluents. In this case, the field monitoring program should include measuring these effluents. When plant-specific data are not available, literature values (see Appendix A) can be used for first approximations.

Nonpoint Sources - Nonpoint loading of pollutants results from the transport of pollutants into receiving waters via overland surface runoff within a drainage basin. Land use and hydrologic characteristics of a basin are major determinants of the magnitude of pollutant loading contributed from nonpoint source inputs. The general long-term trend of deforestation and the subsequent transition to agricultural, urban, and suburban land uses has resulted in large-scale changes in nonpoint source pollutant loading to the Nation's rivers and coastal waters.

As the magnitude of the nonpoint source pollution problem has become better understood over the past 10-20 years, a number of urban and agricultural management practices have been proposed, investigated, and implemented to reduce pollutant loading from these very diverse sources. A major emphasis of ongoing and future national water quality management objectives for EPA and the states will be the basin-scale implementation of best management practices to reduce nonpoint source pollutant loads.

Several modeling techniques have been developed for estimating pollutant loadings from diffuse and storm-driven sources. Recent reviews of these techniques are presented in USEPA (1992b) and Donigian and Huber (1991). Although watershed modeling techniques were originally developed to estimate loading and to provide input to receiving water models, the TMDL program has widened the range of application of these models to include the development and evaluation of various components of watershed management plans. Since not all decision situations regarding the development of TMDLs are the same, some models are more suitable than others under certain conditions. Simple and screening watershed models have been extensively used to support preliminary assessment and planning-level activities, while applications of detailed simulation models are most cost-effective when dealing with development of design criteria or evaluating management programs. Table 2-2 presents a set of commonly encountered decision situations associated with TMDL development for which the use of a watershed model may be considered. These situations are presented in increasing order of complexity and modeling requirements.

TABLE 2-2. DECISION SITUATIONS REQUIRING WATERSHED MODELS

- *Screening Level* (simple models)
    - Relative comparison of watersheds
    - Preliminary estimates of discharge quantity and quality
    - Delineation of the geographical extent and analysis of the temporal variability of major pollution sources
    - Identification of pollutants and governing processes of concern
    - Identification of modeling and monitoring needs
- *Planning Level* (mid-range to detailed models)
    - Prioritization and targeting of specific watersheds or pollution sources
    - Evaluation and selection of control strategies
- *Post-Planning Level* (detailed models)
    - Siting criteria for implementation of management measures
    - Design criteria for sizing control practices

Other nonpoint pollution sources that may influence the development of TMDLs include groundwater seepage, atmospheric deposition, and natural weathering of rocks and soils. These sources are difficult to control, and under natural conditions they represent the background concentrations of the waterbody. When developing TMDLs, uncontrollable sources need to be identified and their magnitude evaluated to characterize the available assimilative capacity of the waterbody. The weathering and dissolution processes of rocks and soil are natural mechanisms and should be considered as part of the uncontrolled loads. Atmospheric deposition is in part a result of industrial and development activities at the regional or national scale. Therefore, their control at the site-specific or watershed scale is not possible. The control of atmospheric deposition is usually addressed in regional and national programs and should be considered as part of the uncontrollable load for typical TMDL development.

Groundwater contributions to the nonpoint source loads are a main concern if the groundwater is considered contaminated. In general, since groundwater interfaces natural geological formations that undergo dissolution and weathering processes under natural conditions, pollutant discharges caused by noncontaminated groundwater seeping to surface waterbodies should be considered a part of the uncontrollable load. When dealing with contaminated groundwater, seepage to surface water can represent a major concern requiring an identification of contamination sources and the pollutant(s) of concern and an evaluation of the magnitude of the discharge. Potential groundwater assessment methods are summarized in *A Review of Methods for Assessing Nonpoint Source Contaminated Ground-water Discharges to Surface Water* (USEPA, 1991c).

2.3.2 In-Stream Fate and Transport of Pollutants

When a pollutant load is discharged into a flowing stream or river, it is subject to fate and transport processes that modify stream concentrations. The principal factors determining stream concentrations are advection, dispersion, and reaction.

Advection - Advection represents the primary transport process of pollutant inflow in the downstream direction. Lateral advective transport across a stream is typically neglected. Usually complete mixing between the pollutant load and the ambient stream flow in the vertical and lateral direction has been achieved within a relatively short distance downstream of the outfall.

Dispersion - If all water elements in a stream were traveling at a uniform speed over each cross-section of the river, they would arrive at a given location at the same time. In reality, however, lateral velocity differences cause each element to arrive at a different time, resulting in an apparent mixing due to vertical and lateral velocity gradients. For example, the center of the stream near the surface moves faster than the flow near the banks and streambed. This phenomenon is called longitudinal dispersion (see Fischer et al., 1979). When analyzing the effects of a continuous pollutant load, the effect of dispersion may be ignored since the contribution of dispersion to the resulting in-stream pollutant concentration is usually small in comparison to the contribution from advection. On the other hand, when analyzing transport of storm-driven loadings during wet-weather periods, longitudinal dispersion also must be considered since the pollutant loading is represented as a single "pulse" input rather than a continuous series of "pulse" inputs. When a water quality-analysis is conducted over a "long" distance with a short time "pulse" interval of discharge, then longitudinal dispersion must be considered in the analysis (Thomann and Mueller, 1987).

Reaction - The biodegradable materials discharged to a stream or river (e.g., oxygen-demanding organics) undergo decomposition by bacteria in the water column. In the presence of dissolved oxygen, bacteria convert organic materials to end products such as CO2, NO3, and H2O, stabilizing the pollutant load. In addition, algae take up nutrients such as inorganic phosphorus and nitrogen during photosynthesis and reduce the nutrient concentrations in the stream. Algal biomass is then recycled back into inorganic nutrients. A number of chemical, biological, and biochemical reactions contribute to the flux and attenuation of waste material concentrations.

The interactions of these factors are shown schematically in Figure 2-1, which presents what would be observed if a single slug of waste load were injected and could be followed downstream over a period of time. Conservative materials in the waste (those not subject to reaction and decay, such as chloride) would track as shown in the sketch of advection, or advection and dispersion. Reactive materials, such as oxygen-consuming materials, would behave as shown in the sketches that include reaction. Thus, the behavior of a dissolved substance in the stream is the result of the velocity and mixing action of the water and the resulting transformation from biological and chemical reactions.

2.3.3 Receiving Water Conditions

The local impact of pollutant loadings to receiving waters is largely determined by the relative magnitudes of the loading and the receiving water flow rate. Assimilation of the pollutant load is a function of the hydrologic conditions of the stream or river defined in terms of flow rate, transport characteristics, and background water quality. As a result, seasonal and storm-driven variations in the watershed hydrology and pollutant buildup and washoff characteristics can exert very different impacts on the receiving water during and immediately following wet-weather episodes. The significance of storm-driven discharges from nonpoint sources on dissolved oxygen is very site-specific and difficult to characterize explicitly. During wet-weather conditions, stream flow rate is generally high, allowing for higher transport and assimilative capacity. However, pollutant deposition and accumu-

TABLE 2-3. NONPOINT SOURCE MODELING OPTIONS

Stochastic/Probabilistic models: Mathematical models that include consideration of hydrologic uncertainty and probability. Models rely on statistical characteristics of the hydrologic process to predict the behavior of the hydrologic system. Transport and fate processes are represented using compounded parameters describing multiple processes.

Deterministic models: Model variables are thought to follow a predictable, certain behavior, and the probability of hydrologic data is generally ignored. Models rely on series of algorithms that simulate actual physical or chemical processes, and are sometimes referred to as physically based models.

Design storm simulation: Focuses on a detailed simulation of a single storm event, often selected as representative, with a given frequency and duration of rainfall (e.g., 1-year, six -hour storm).

Continuous simulation: Simulates the behavior of a system over an extended period of time (months to years) with relatively short time steps, providing continuous runoff and loading values.

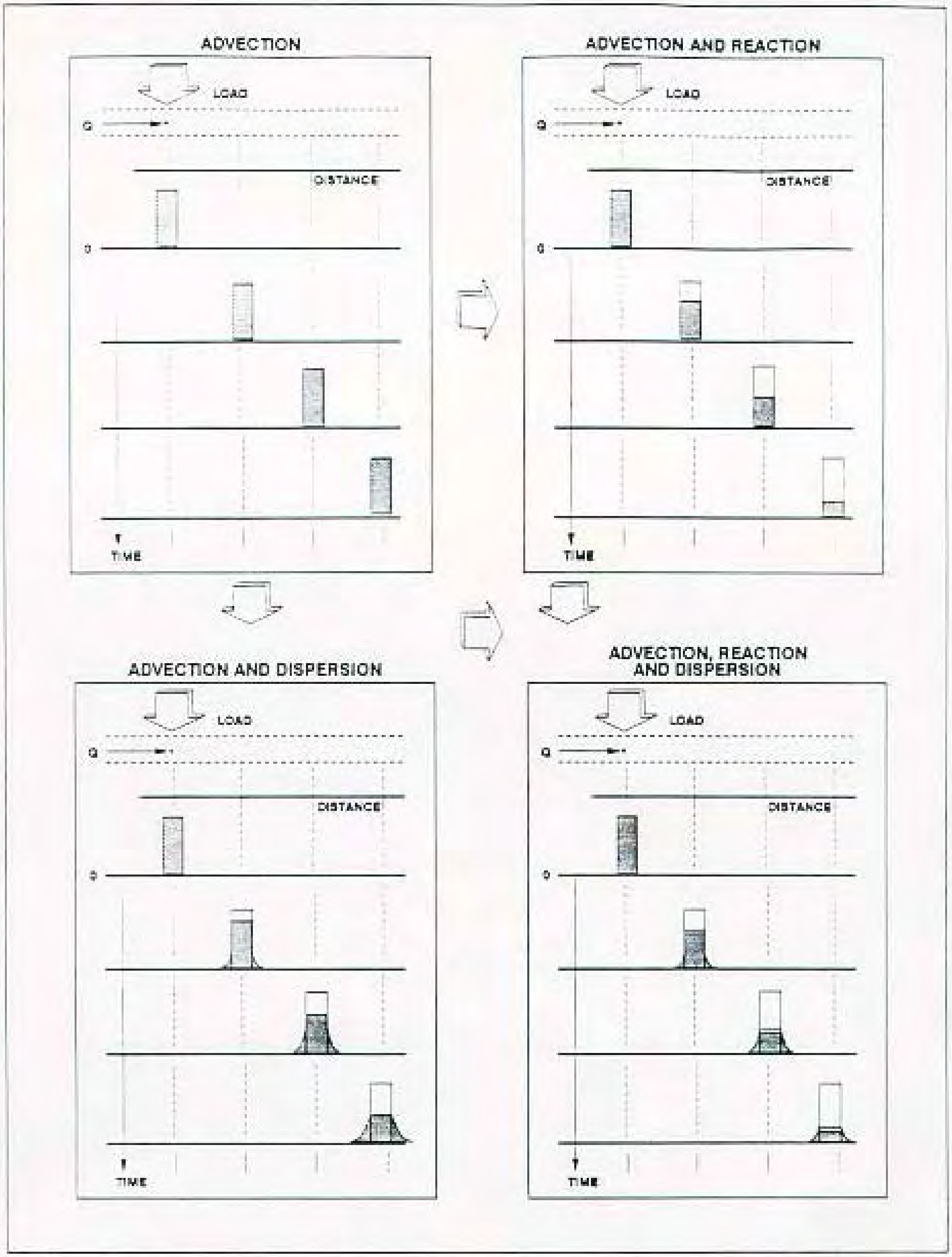


FIGURE 2-1. INTERACTION OF TRANSPORT MECHANISMS FOR LOADS IN A STREAM (After USEPA, 1983a)

lation on streambed and pool areas may exert a critical demand on dissolved oxygen during low flow conditions. When nonpoint sources are identified as the major sources of oxygen depletion, storm event or continuous and time-dependent modeling may be required. Nonpoint source modeling options that capture, to a certain extent, the impacts of wet-weather conditions are summarized in Table 2-3.

Often when oxygen demand is the mechanism of concern, the modeling effort focuses primarily on pollutant discharges during dry periods and the base flow rate, which depends on the magnitude of the flow recession following wet-weather recharge conditions. Although pollution discharges during dry periods are characterized by fairly constant rates and composition, the base flow rate fluctuates seasonally and annually. In the continental United States, seasonally high flow normally occurs during the colder period of winter and in early spring from snowmelt and spring rains, while seasonally low flow typically occurs during the warmer summer and early fall drought periods. Because of these seasonal hydrologic and climatological patterns of low flow, minimum dilution, and high temperature, summer and early fall are typically the critical periods for evaluating the worst-case impact of pollutant loads on water quality. During this period, flow conditions approaching steady state are achieved. The analysis and evaluation of data collected during this period become more meaningful because the mathematical assumption of steady state is frequently made when evaluating dissolved oxygen in streams and rivers due to fairly constant point sources.

Rapid transport of pollutants by high flow and mixing conditions results in a short residence time with typically minimal ecological damage. Conversely, slow removal of pollutants in waters characterized by a long residence time because of low-flow conditions can result in adverse ecological impacts such as severe oxygen depletion, nutrient enrichment, and eutrophication problems.

Factors that affect the time of travel in a natural stream include stream depth, width, cross-sectional area, and bed slope-hydraulic geometry. In some cases, stream hydraulic geometry and time-of-travel information are available from studies performed by the U.S. Geological Survey (USGS), the U.S. Army Corps of Engineers, or other State and Federal agencies. Sometimes a field program may be needed to collect the hydraulic geometry data and measure the time of travel (usually by a dye study).

Projections of water quality impacts for some future critical low-flow condition are normally required in TMDL studies. The change in hydraulic geometry caused by flow fluctuation must be predicted. Flow variance, in turn, results from changes in stream velocity and depth (both of which strongly affect the stream reaeration capacity).

Basically, two approaches are used to quantify hydraulic geometry and time of travel for future conditions. First, Leopold and Maddock (1953) have examined various rivers and developed empirical relationships between flow (Q), velocity (U), depth (H), and width (W) using the following functions with flow as the independent variable.

$$U \approx aQ^{n} \tag{2-1}$$

$$H \approx bQ^{m} \tag{2-2}$$

$$W \approx cQ^{f} \tag{2-3}$$

where a, b, and c are constants for the stream and n, m, and f are exponents defining the basic relationships.

These constants vary with size of the river basin. More detailed information on these constants can be found in Appendix A.

The second approach is to independently calculate stream velocity, depth, and width for different flows. Typically, hydrodynamic models based on momentum and continuity equations are used. Many stream water quality models (see Chapter 3) include hydraulic components that can be used to model flow in addition to water quality.

A simpler modeling approach is to use the Manning equation relating velocity, depth, slope, and a channel roughness coefficient (Brown and Barnwell, 1987; McCutcheon, 1989) to simulate the velocity and stream flow rate for different depths. It is typically easier to estimate and extrapolate the Manning roughness coefficient than to estimate the hydraulic constants and exponents in Equations 2-1 through 2-3. However, the Manning equation is an empirical formulation that may not reflect actual conditions of natural streams. While both equations are semi-empirical, the Manning equation involves only one coefficient (vs. six in the Leopold and Maddock equations), and that coefficient is well understood by hydraulic engineers. The Manning equation also provides better physical insight by integrating the effect

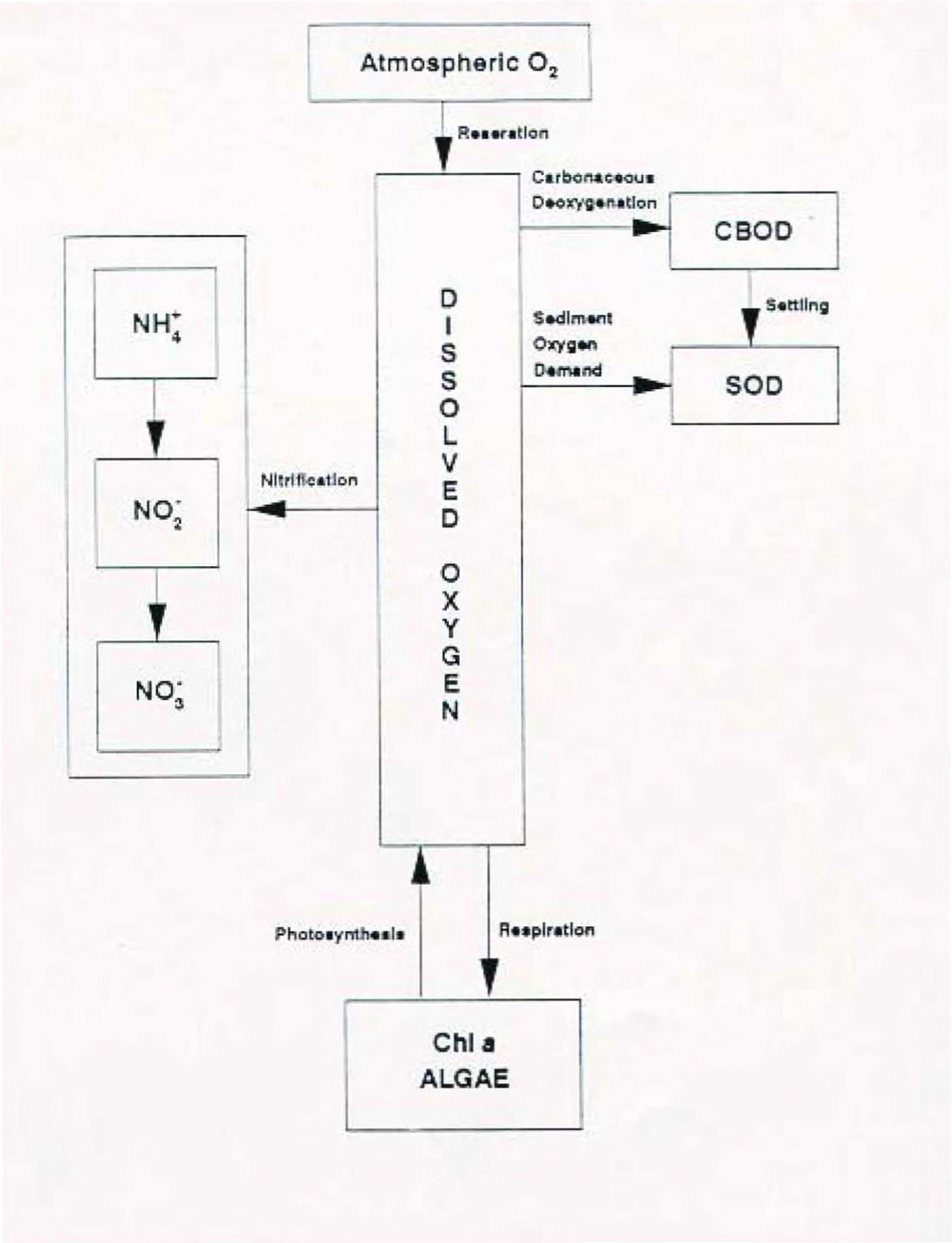


FIGURE 2-2. INTERRELATIONSHIP OF MAJOR KINETIC PROCESSES FOR BOD AND DO AS REPRESENTED BY WATER QUALITY MODELS (After McCutcheon, 1989)

of slope and elaborate stream geometry information. EPA Region 6, however, has shown Manning's n to greatly overestimate velocity. Although not extensively used in TMDLs, other backwater and dynamic routing simulations are feasible (see McCutcheon (1985) and French (1985) for a review).

2.3.4 Biochemical Oxygen Demand and Dissolved Oxygen Reaction Kinetics

Figure 2-2 shows the interrelationship of the following major BOD/DO kinetic processes for a water column as commonly represented by water quality models:

- Carbonaceous deoxygenation
- Nitrogenous deoxygenation (nitrification)
- Reaeration
- Sediment oxygen demand
- Photosynthesis and respiration

Prior to describing these processes, a brief discussion of the biochemical oxygen demand concept is necessary. BOD is a measure of the amount of oxygen required to stabilize organic matter in wastewater. As such, BOD is an equivalent indicator rather than a true physical or chemical substance. It measures the total concentration of dissolved oxygen that would eventually be demanded as wastewater degrades in the stream. The validity of BOD as a gauge of wastewater quality has often been questioned; nevertheless, the concept of BOD remains the standard for dissolved oxygen modeling analysis.

BOD is determined from a standardized test measuring the amount of oxygen available after incubation of the sample at 20 °C for a specific length of time, usually 5 days. The oxidation process is usually carried out in two stages: carbonaceous and nitrogenous (nitrification). The first stage is accomplished by saprophytic organisms, which derive their energy from the breakdown of organic carbon compounds; the second stage, by autotrophic bacteria, which require simple inorganic nitrogen compounds.

Each stage is characterized by two steps: synthesis and respiration. In the carbonaceous stage, the energy required for synthesis is obtained from the destruction of complex organic carbon compounds, liberating carbon dioxide and water. After the organic matter has been converted to bacterial cells, the endogenous respiration of the synthesized organisms occurs, also yielding carbon dioxide, water, and usually ammonia. In the BOD test, there is a pronounced lag between the carbonaceous oxidation and the nitrification step, the latter following by as much as 10 days. The lag is less for treated (stabilized) wastewaters and is on the order of 1 to 2 days for highly treated effluents. In streams, the two stages frequently proceed simultaneously, although there may be lags in the nitrification stage for highly polluted streams or those with low dissolved oxygen.

2.3.4.1 Carbonaceous Deoxygenation

The first phase of the BOD reaction involves the oxidation of the carbonaceous organic material. The reaction is approximated by a first-order reaction. The oxygen required, y, approaches the total demand of the overall process, Lo, and the rate is assumed to be proportional to the amount of oxygen-demanding material (Lo - y), either substrate or cells:

$$\frac{dy}{dt} = K_1 \left(L_o - y\right) \tag{2-4}$$

Integration of this expression yields:

$$y = L_o \left(1 - e^{-K_1 t}\right) \tag{2-5}$$

or, if the relationship is put in terms of the organic matter remaining,

$$L = L_o \, e^{-K_1 t} \tag{2-6}$$

where

K1 = BOD reaction rate coefficient (day-1)
y = oxygen consumed
L = oxygen equivalence of the organic matter remaining
Lo = total oxygen demand
t = time (day)

Equation 2-4 indicates that the rate at which the oxygen is consumed (dy/dt) is proportional to the concentration of biologically degradable organic material, as well as chemically oxidizable substances. The coefficient, K1, depends on the state of the material and the degree of treatment. A typical domestic wastewater may have the following values: raw sewage (0.35 day^{-1}), settled sewage (0.40 day^{-1}), and treated effluent (0.25 day^{-1}). Because of the nature and composition of wastewater, these values vary significantly. Industrial wastewaters are known for their widely varied K_1 rates. Coefficients determined from samples taken from rivers indicate that many factors affect the rate at which the reaction proceeds. In many natural settings, the reaction rate coefficient for river water is usually less than that of an undiluted wastewater sample and decreases with distance downstream (Thomann and Mueller, 1987). Decreasing coefficients indicate the progressive resistance to the oxidation of the more stable (refractory) end products.

By U.S. convention, BOD measurements are typically conducted for 5 days. In fact, regulatory agencies write wastewater discharge permits (NPDES) in terms of 5-day BOD. In addition, many of the tests are run with a nitrification inhibitor so that the test measures the oxidation of carbonaceous material only. When total BOD is measured after 5 days (an inhibitor is not used), these tests are designated as BOD_5. When the 5-day BOD test employs a nitrification inhibitor, the results are designated as $CBOD_5$ (Hall and Foxen, 1984). More and more frequently, long-term tests of 20 to 30 days are employed to measure ultimate BOD (BOD_u) to reflect the potential strength of the oxygen consumption. Some pulp and paper mill wastewater samples are analyzed for much longer periods (in excess of 100 days), but measurements over an extended period of time are of limited value in streams where the time of travel from the waste source to the dissolved oxygen sag is only a few hours or days, or where the stream is diluted by tributaries within a few hours or days. Such tests are very useful, though, in converting model results in $CBOD_u$ to a $CBOD_5$ NPDES limit. In a standard test, the values of oxygen used, y, at the end of specified intervals of time, t, are determined. Given a set of such values, the coefficient, K_1, and the ultimate value, L_0, may be determined (Metcalf and Eddy, 1991).

Another important concept for stream BOD is illustrated in Figure 2-3. When water samples are taken from a stream to the laboratory for analysis of their biochemical oxygen demand, the results may be represented by a family of curves (Equation 2-5) of oxygen consumed vs. time of incubation (see Figure 2-3b). Each of these curves has a different K_1 value. As suggested above, the K_1 value decreases in the downstream direction. If $CBOD_5$ values (either measured during the analysis or calculated using Equation 2-5) are plotted against the longitudinal stream distance (Figure 2-3a), a decreasing trend for $CBOD_5$ is obtained. This trend follows an exponential decay and usually can be approximated by the following equation in terms of ultimate CBOD:

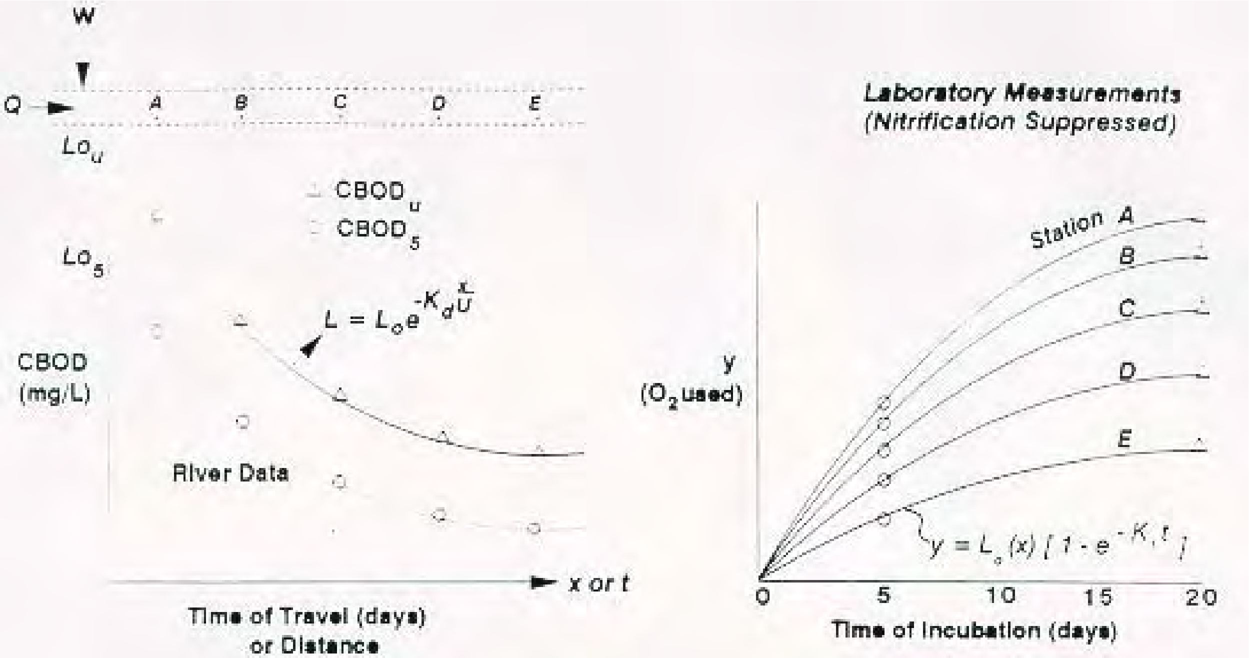
$$L\,(CBOD_u) = L_o\,(CBOD_u)\; e^{-K_r \frac{x}{U}} \qquad (2\text{-}7)$$

where

$L(CBOD_u)$ = oxygen equivalence of the organic matter at any given location in the stream (measured as $CBOD_u$)
$L_o(CBOD_u)$ = total oxygen demand measured at the source of waste load following complete mixing (measured as $CBOD_u$)
K_r = $CBOD_u$ removal rate in the stream (day^{-1})
x = distance below the wastewater discharge
U = average stream velocity

The time of travel, *t*, is equal to *x/U*. The meaning of Equation 2-7 is that the oxygen-consuming materials are removed from the water column at an overall loss rate of K_r. It should be noted that K_r is used to characterize the overall loss of dissolved organic materials in the water column due to biochemical oxidation and settling. It is highly empirical and, therefore, is usually quantified by fitting an exponential decay curve through the field data.

The rate of removal of organic material from the water column is not necessarily equal to the rate at which dissolved oxygen is utilized. The coefficient describing this oxygen utilization may be identified as K_d. If a difference exists between the rate of BOD removal (K_r) and oxygen utilization (K_d), it may be attributed to factors such as sedimentation, flocculation, scour, or volatilization. For example, K_r may be significantly greater than K_d for effluents with high concentrations of suspended solids, which readily settle to the streambed. In most of the highly treated effluents, suspended solids concentrations are low and the waste has been stabilized. It is usually found that the rate at which the organic matter is removed is equal to the rate at which the dissolved oxygen is utilized ($K_r = K_d$). Furthermore, the coefficients K_r and K_d may be quite different from the laboratory BOD test rate, K_1, primarily because of the various physical and biochemical characteristics of the two environments. It is useful to compare filtered versus unfiltered to determine if $K_d = K_r$. Biological growths on the streambed, nutrients, turbulence, and acclimation



(Figure 2-3a)

(Figure 2-3b)

FIGURE 2-3. COMPARISON OF STREAM BOD AND LABORATORY BOD FOR VARIOUS INCUBATION TIMES (Manhattan College, 1983)

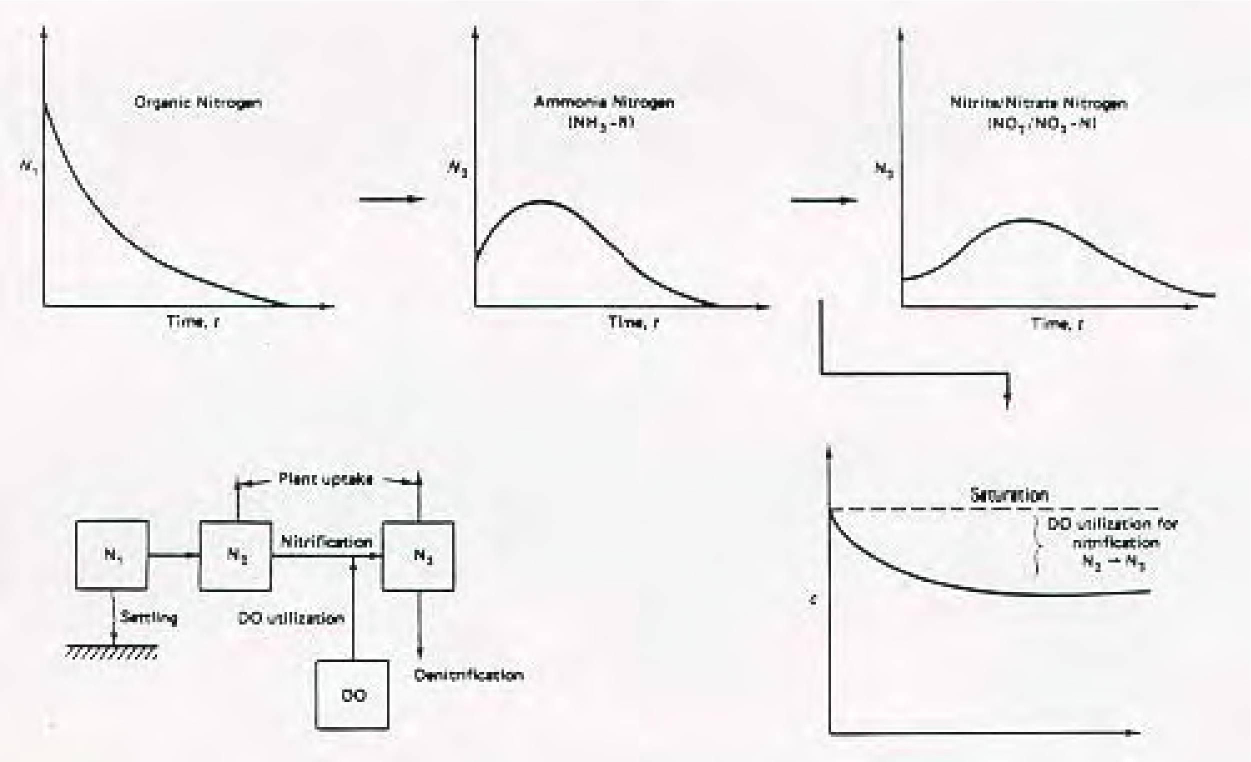


FIGURE 2-4. STEPS IN NITRIFICATION AND UTILIZATION OF DISSOLVED OXYGEN (After Thomann and Mueller, 1987)

may also contribute to the difference between the laboratory rate, K_1, and the field rates, K_r and K_d.

Both $CBOD_5$ and $CBOD_u$ can be used to describe the decay of CBOD in streams when the ratio of the two remain relatively constant in the downstream direction. However, only $CBOD_u$ can be used to simulate the loss of dissolved oxygen in stream TMDL studies. For this reason, it is necessary to employ $CBOD_u$ measurements for model calibration. If measurements of $CBOD_5$ are to be used, it is necessary to determine the relationship between $CBOD_5$ and $CBOD_u$. One purpose of evaluating K_1 (in Equation 2-5; $CBOD_5 = y$, $CBOD_u = L_o$; and $t = 5$ days) is to convert $CBOD_5$, as usually reported, to $CBOD_u$, which is required for the dissolved oxygen modeling analysis (e.g. QUAL2E or WASP5) (see Thomann and Mueller, 1987).

2.3.4.2 Nitrogenous Deoxygenation (Nitrification)

The nitrogenous stage of the BOD test includes conversion of organic nitrogen to ammonia and the subsequent oxidation of ammonia (Figure 2-4). Many wastewaters contain organic nitrogen, such as urea, and/or ammonia. The former is hydrolyzed to ammonia, under aerobic or anaerobic conditions, without the use of oxygen. Ammonia is successively oxidized through nitrite to nitrate by the organisms Nitrosomonas and Nitrobacter, respectively.

$$NH_4^+ + 1.5\,O_2 \longrightarrow NO_2^- + H_2O + 2H^+ \qquad (2\text{-}8)$$

$$NO_2^- + 0.5\,O_2 \longrightarrow NO_3^- \qquad (2\text{-}9)$$

Stoichiometrically, 3.43 and 1.14 grams of oxygen are required to transform each gram of ammonia nitrogen to nitrite nitrogen (Equation 2-8) and nitrite nitrogen to nitrate nitrogen (Equation 2-9), respectively. The decay of organic nitrogen indirectly requires oxygen after the organic nitrogen is hydrolyzed into ammonia. Some researchers (e.g., Wezernak and Gannon, 1967; Adams and Eckenfelder, 1977) have suggested that the oxygen requirement could be reduced to 3.22 and 1.11 grams, respectively, due to cell synthesis.

The most common approach to modeling nitrification is to use first-order kinetics (similar to BOD described earlier) to characterize Equations 2-8 and 2-9. That is, the rate of accumulation or depletion is linearly dependent on the amount of nitrogen available in a specific pool. Factors affecting the rate of nitrification include temperature, pH, nitrogen concentrations, dissolved oxygen, suspended solids, and organic and inorganic compounds.

Because of the ease of measuring organic nitrogen, ammonia, nitrite, and nitrate, waste load allocation modeling of nitrification involves a mass balance and a description of each species decay. Nitrification is best simulated as a cascade process involving hydrolysis of organic nitrogen, oxidation of ammonia, and oxidation of nitrite. In some models, the intermediate step of nitrite oxidation is combined with the overall oxidation of ammonia to nitrate, but only little computational efficiency is gained. Furthermore, the conversion of nitrite to nitrate is very rapid; therefore, the combination of the corresponding rates is not unreasonable.

A number of studies have demonstrated that nitrification and denitrification in the water column may be dominated by benthic processes, particularly in fast-moving shallow streams and rivers. Enumerations of nitrifier organisms have demonstrated that benthic populations can be two to three orders of magnitude greater than water column populations (Williams and Lewis, 1986). Several studies have shown that up to 80 to 95 percent of total nitrification can be accounted for by benthic processes. Selected studies include the James River in Virginia (Cerco, 1981), shallow streams in North Carolina (Kreutzberger and Francisco, 1977; Lewis, 1983), the Trent River in England (Curtis et al. 1975; Garland, 1978), and the Passaic River in New Jersey (Matulewich and Finstein, 1978).

The sequential forward reactions of mineralization of organic nitrogen and nitrification suggest that nitrate should accumulate as an end product of the reactions. Several data sets, however, suggest the removal of nitrate from the water column along with the conversion of ammonia to nitrite and nitrate (see Seitzinger, 1988). Simultaneous benthic nitrification and denitrification have been observed in the James River (Cerco, 1981) and in shallow streams in North Carolina (Williams and Lewis, 1986) and incorporated into water quality models of oxygen and nitrogen distributions (Cerco, 1981; Williams and Lewis, 1986).

Seitzinger (1988) has observed that measured rates of denitrification in most river, lake, estuarine, and coastal sediments (i.e., production of N_2O gas) are higher than the corresponding rates of nitrate loss to the sediments. The major source of nitrate for sediment denitrification underlying an aerobic water column is nitrate produced in the sediments during nitrification rather than nitrate diffusing from the overlying water column into the sediments.

Some earlier stream models made the cascade process a single process by combining Equations 2-8 and 2-9 and combining all nitrogenous oxygen demands (3.43 + 1.14 = 4.57 grams of oxygen per gram of nitrogen) as NBOD. Modeling NBOD and CBOD as separate demands is not as useful as modeling CBOD, organic nitrogen, ammonia, nitrite, and nitrate as separate demands to track the sequential reactions of the nitrogen cycle, which is widely used in waste load allocation studies. Nevertheless, NBOD modeling has been determined to be useful. For example, the Simplified Analytical Method for Determining NPDES Effluent Limitations for POTWs Discharging into Low-Flow Streams (see Table 1-1) uses this approach after ensuring that none of the constraints of the method are violated.

2.3.4.3 Reaeration

In general, oxygen may be removed from or added to water by various physical, chemical, or biological reactions. If oxygen is removed from the water column and the concentration drops below the saturation level, there is a tendency to make up this deficit by the transfer of the gas from the atmosphere through the surface into the stream at a certain rate. If oxygen is added and the water column concentration is greater than the saturation level, the supersaturation is reduced by the transfer of oxygen from the stream to the air. Such interactions between the gas phase and liquid phase are driven by the partial pressure gradient in the gas phase and the concentration gradient in the liquid phase (see Thomann and Mueller, 1987). In general, oxygen transfer in natural waters depends on:

- Internal mixing and turbulence due to velocity gradients and fluctuation.
- Temperature.
- Wind mixing.
- Waterfalls, dams, and rapids.
- Surface films.
- Water column depth.

The rate of transfer to be quantified in stream BOD/DO modeling analyses is expressed as:

$$\frac{dC}{dt} = K_a (C_s - C) \tag{2-10}$$

where

dC/dt = rate of change of oxygen concentration

C_s = saturation concentration of dissolved oxygen

C = dissolved oxygen concentration in stream

K_a = stream reaeration rate coefficient (day^{-1})

Many empirical formulations have been developed for estimating stream reaeration rate coefficients. For example, the QUAL2E model offers eight different formulations reported in the literature. Stream reaeration rate coefficients span a wide range of values (typically 0.1 to 10 day^{-1} or even greater) and have a greater magnitude than BOD reaction rate coefficients. Appendix A presents reaeration rate coefficients reported in the literature for a number of waterbodies with guidance for selection of an appropriate equation.

2.3.4.4 Sediment Oxygen Demand

Benthic decomposition of organic material is defined as the stabilization of the volatile suspended solids that have settled to the streambed. These deposits are stabilized by the biological activity of many different organisms including bacteria. As these organic materials are associated with suspended solids, the discharge of settleable waste components may form a sludge blanket below a wastewater outfall. After a period of time, organic materials may accumulate, since the deposition rate of particulate material is greater than the decomposition and physical loss rate.

The demand of oxygen by sediment and benthic organisms can, in some instances, be a significant fraction of the total oxygen demand. This is particularly true in small streams. The effects may be particularly acute during low-flow and high-temperature conditions. Decomposition of organic matter and respiration of resident invertebrates form the major oxygen demands from the sediment. In addition to biological decomposition and respiration of benthic invertebrates, net photosynthetic oxygen production of attached benthic algae (periphyton) can also be a significant component of the total SOD. The oxygen balance of shallow streams, in particular, can be influenced by this process since attached algae are frequently present in shallow streams (see Terry and Morris, 1986; Jeppesen and Thyssen, 1984). Although these processes are distinct, they are typically quantified together because in situ measurements combine oxygen uptake and separation of the processes would result in added model complexity.

Because of its complexity, it is difficult to estimate SOD analytically and independently. In situ measurements of SOD are usually conducted using a chamber at the bottom of the stream. Continuous measurement of oxygen uptake over a certain period of time provides data to derive the oxygen consumption rate. In some cases, samples of river sediments (undisturbed) are taken to the laboratory to measure the oxygen uptake of bottom muds. The amount of oxygen used over the test period is calculated as grams of oxygen per square meter per day (g O_2/m^2-day). In a modeling analysis, SOD is typically formulated as a zero-order process:

$$\frac{dC}{dt} = -SOD/H \tag{2-11}$$

where

dC/dt = rate change of oxygen concentration (g O_2/m^3-day)

SOD = sediment oxygen demand (g O_2/m^2-day)

H = average river depth (m)

Appendix A presents various values of SOD reported in the literature for a number of streams and rivers.

Like many other reaction rate coefficients, the SOD values can be determined by model calibration if direct measurements from the field are not available. The difficulty arises when SOD values need to be predicted for future conditions. In recent years, credible interactive sediment-water column models have appeared to independently quantify the oxygen uptake rates of sediments. For example, Di Toro et al. (1990) have developed a SOD model based on diagenesis of particulate organic materials to predict the production of hydrogen sulfide (H_2S), ammonium (NH_4), phosphate (PO_4), and silicon (Si). Such a framework explains some, but not all, of the processes associated with SOD and is still being tested. As a result, EPA recommends that conservative estimates of SOD be used for future conditions in TMDLs.

2.3.4.5 Photosynthesis and Respiration

Through photosynthesis and respiration, phytoplankton, periphyton, and rooted aquatic plants (macrophytes) could significantly affect the dissolved oxygen levels in the water column. Because phytoplankton growth requires sunlight and nutrients, quantifying photosynthetic oxygen production would need to address phytoplankton-nutrient dynamics. That is, phytoplankton and nutrients should be modeled concurrently to address this problem. In many simple stream BOD/DO models, however, the oxygen production rate due to photosynthesis and consumption rate due to respiration are assigned, thereby uncoupling the calculation from the phytoplankton-nutrient dynamics. In this section, the simple approach is presented; the full discussion of phytoplankton-nutrient dynamics is provided in Chapter 4.

In a stream water quality model, the daily average oxygen production due to photosynthesis and reduction due to respiration is formulated as follows:

$$\frac{dC}{dt} = P - R \tag{2-12}$$

where

dC/dt = rate of change of oxygen concentration (mg O_2/L-day)

P = average gross photosynthesis production (mg O_2/L-day)

R = average respiration (mg O_2/L-day)

Note that *R* is considered to be plant respiration only, excluding microbial respiration for carbonaceous deoxygenation and nitrification. In a model such as QUAL2E, the mass flux term for the aquatic plant contributions to the oxygen balance is typically modeled as a zero-order process:

$$\frac{dC}{dt} = (\alpha_3 \, \mu - \alpha_4 \, \rho) \, A_g \tag{2-13}$$

where

dC/dt = rate of change of oxygen concentration (mg O_2/L-day)

A_g = algal biomass concentration (mg/L)

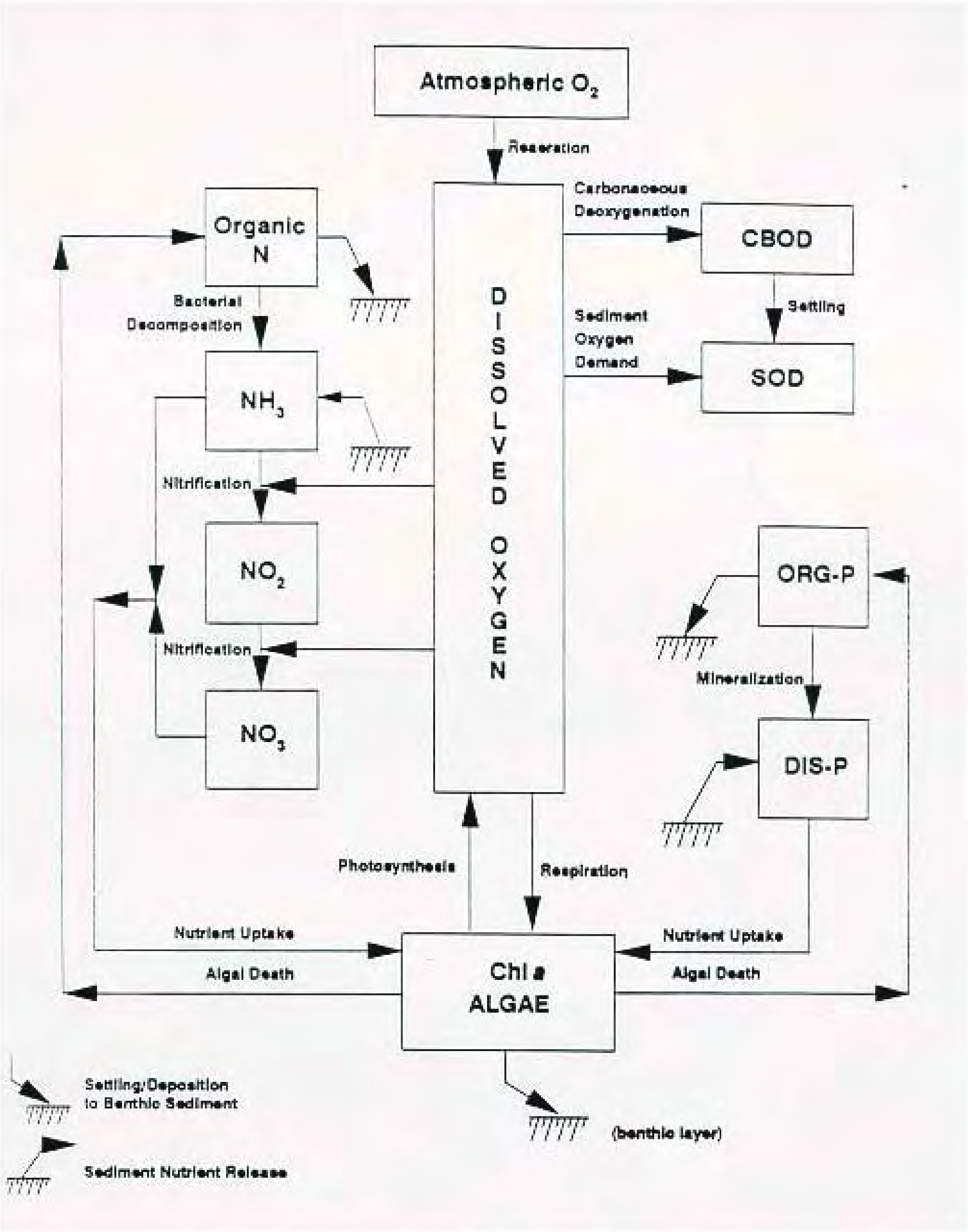


FIGURE 2-5. INTERRELATIONSHIP OF MAJOR KINETIC PROCESSES FOR BOD, DO, AND NUTRIENT ANALYSES AS REPRESENTED BY WATER QUALITY MODELS
(After McCutcheon, 1989)

μ = algal growth rate coefficient (day^{-1})

ρ = algal respiration rate coefficient (day^{-1})

α_3 = the stoichiometric ratio of oxygen production per unit of algal photosynthesis (mg/ mg)

α_4 = the stoichiometric ratio of oxygen uptake per unit of algae respired (mg/mg)

2.3.5 Eutrophication Kinetics

Figure 2-5 shows the major kinetic processes usually considered in a complete DO, BOD, and nutrient analysis. The reader should note the similarities between Figures 2-2 and 2-5. The following processes are discussed in this section:

- Algal growth and nutrient uptake
- Algal death and settling
- Nutrient mineralization
- Sediment nutrient release

2.3.5.1 Algal Growth and Nutrient Uptake

Phytoplankton growth is directly related to temperature in moderate climates, nutrient effect, and light intensity up to a saturating condition:

$$G_p = G_T \; r_L \; r_n \qquad (2\text{-}14)$$

where

G_p = phytoplankton growth rate (day^{-1})

G_T = temperature dependent growth rate (day^{-1})

r_L = light effect (dimensionless)

r_n = nutrient effect (dimensionless)

Temperature dependence, G_T, is approximated by:

$$G_T = G_{max} \, (1.066)^{T-20}$$

where

G_{max} = maximum growth rate(day^{-1})

T = temperature (°C)

The value suggested for G_{max} under average conditions for a mixed phytoplankton population is approximately 1.8 day^{-1} (Thomann and Mueller, 1987).

Auer and Canale (1980) and Canale and Vogel (1974) summarized data from phytoplankton growth experiments conducted at various temperatures. These results, plotted in Figure 2-6, illustrate the different temperature optimums for different phyla (species groups) of phytoplankton (diatoms, green and blue-green algae) as well as the influence of temperature on the growth rate. The growth rate at saturating light condition can be expected to be species group dependent, as shown in Figure 2-7. Because light energy available to phytoplankton varies with depth and time of day, an appropriate expression of light availability for use in models should account for these changes.

Averaging relative photosynthesis as a function of light intensity over a given depth of water and over a fixed interval of time yields

$$r_L = \frac{2.718 \, f}{K_e \, H T} \left[e^{-\alpha_1} - e^{-\alpha_2} \right] \qquad (2\text{-}15)$$

$$\alpha_1 = \frac{I_T}{I_s \, f} e^{-K_e H} \qquad (2\text{-}15a)$$

$$\alpha_2 = \frac{I_T}{I_s \, f} \qquad (2\text{-}15b)$$

where

r_L = light effect

I_s = saturating light intensity (ly/ day)

I_T = total daily solar radiation (ly)

f = photoperiod (day)

K_e = extinction coefficient (m^{-1})

H = depth (m)

T = average period (day)

The extinction coefficient, K_e, is dependant on inorganic solids, detrital particles, and phytoplankton biomass in the water body. Values of K_e in natural water bodies typically vary from 0.05 to 6.9 m^{-1}. Typical values for I_T range from 250 to 500 ly. The corresponding range of values for r_L is 0.1 to 0.5, so the overall daily effect of light extinction with depth is to reduce the growth rate by about 50-90 percent (Thomann and Mueller, 1987).

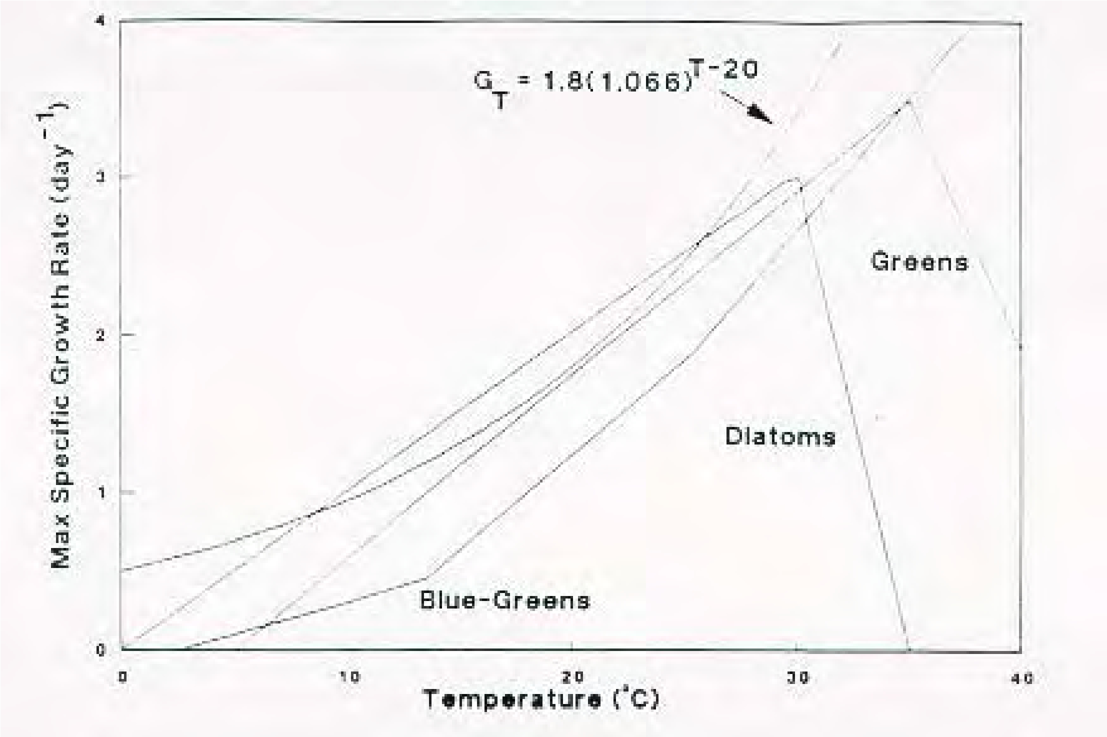


FIGURE 2-6. SPECIFIC ALGAL GROWTH RATE AS A FUNCTION OF TEMPERATURE
(After Canale and Vogel, 1974)

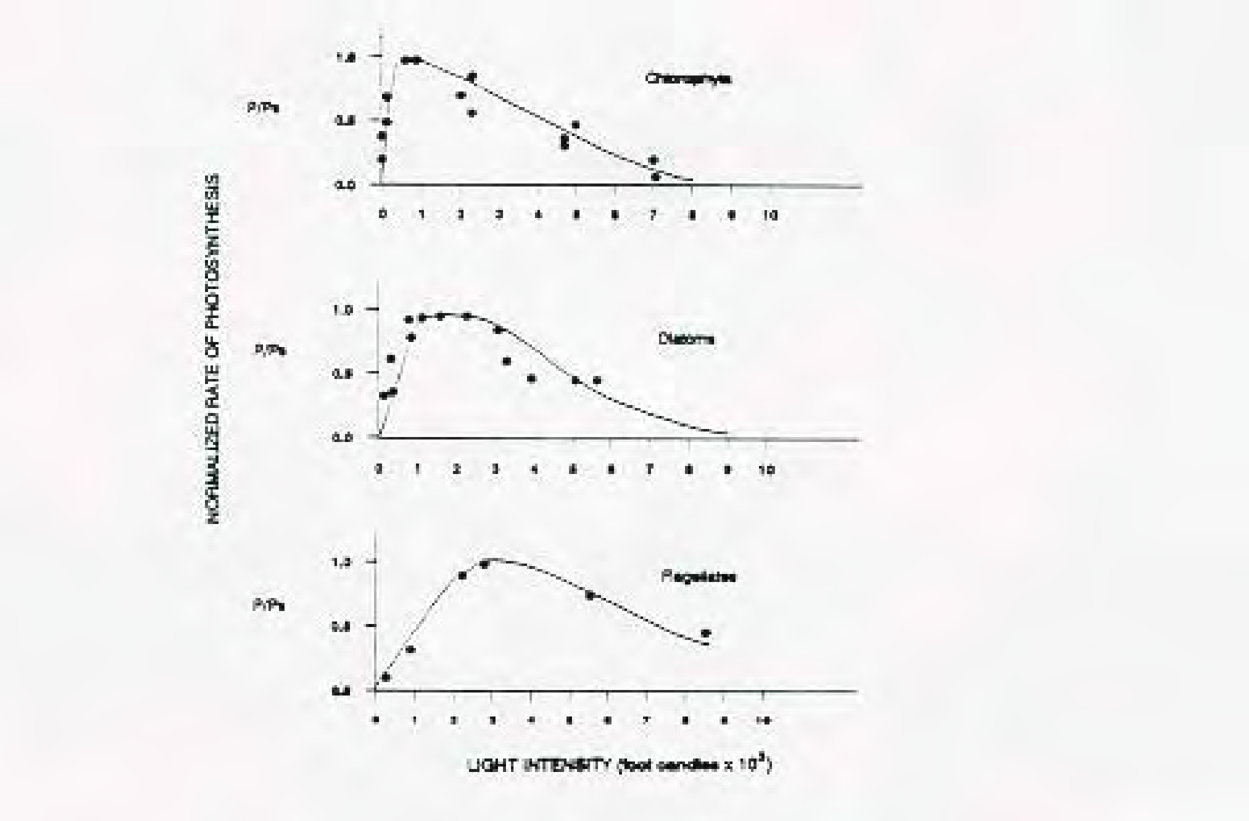


FIGURE 2-7. EFFECT OF LIGHT INTENSITY ON ALGAL GROWTH
(After Ryther, 1956)

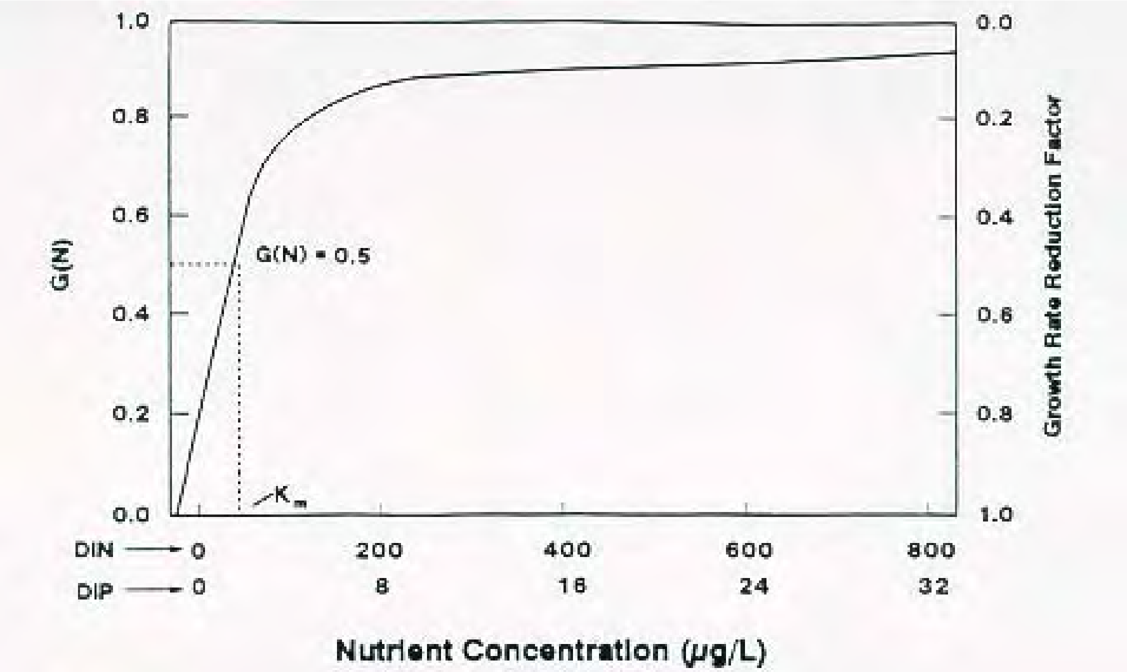


FIGURE 2-8. EFFECT OF NUTRIENTS ON ALGAL GROWTH
(After Ambrose et al., 1993a)

The phytoplankton growth rate is also a function of nutrient concentrations up to a saturating condition, greater than which it remains constant with nutrient concentration (Figure 2-8). At zero nutrient concentration, there is no growth. As the nutrient level is increased, the growth rate is linearly proportional to the availability of nutrients. However, as nutrient levels continue to increase, the effect on the growth rate of the phytoplankton is saturated. Such a relationship is described by a Michaelis-Menton formulation where the nutrient reduction factor, or nutrient effect, for algal growth, r_n, is:

$$r_n = \frac{Nut}{K_m + Nut} \tag{2-16}$$

where

Nut = the nutrient concentration (μg/L)

K_m = half saturation (Michaelis) constant (μg/L)

The Michaelis half-saturation constant, a function of the algal species group, is the nutrient concentration for which the nutrient reduction factor is 0.5 or half the maximum growth rate. The value usually ranges from 5 μg/L to 25 μg/L for nitrogen and from 1 μg/L to 5 μg/L for phosphorus, depending on the species. With more than one nutrient accounted for in the model (i.e., nitrogen, phosphorus, silica), the nutrient effect is given by:

$$r_n = \min\left(\frac{DIN}{K_{mn} + DIN}; \frac{DIP}{K_{mp} + DIP}; \frac{S_i}{K_{S_i} + S_i}; \ldots\right) \tag{2-17}$$

where

r_n = limiting nutrient reduction factor

DIN = inorganic nitrogen concentration (sum of ammonia, nitrate, and nitrite)

DIP = dissolved inorganic phosphorus concentration (μg/L)

K_{mn} = Michaelis-Menton constant for nitrogen (μg/L)

K_{mp} = Michaelis-Menton constant for phosphorus (μg/L)

S_i = dissolved inorganic silica concentration (μg/L)

K_{Si} = Michaelis-Menton constant for silica (μg/L)

The minimum ratio of the nutrients considered in the model thus controls the computation of the nutrient

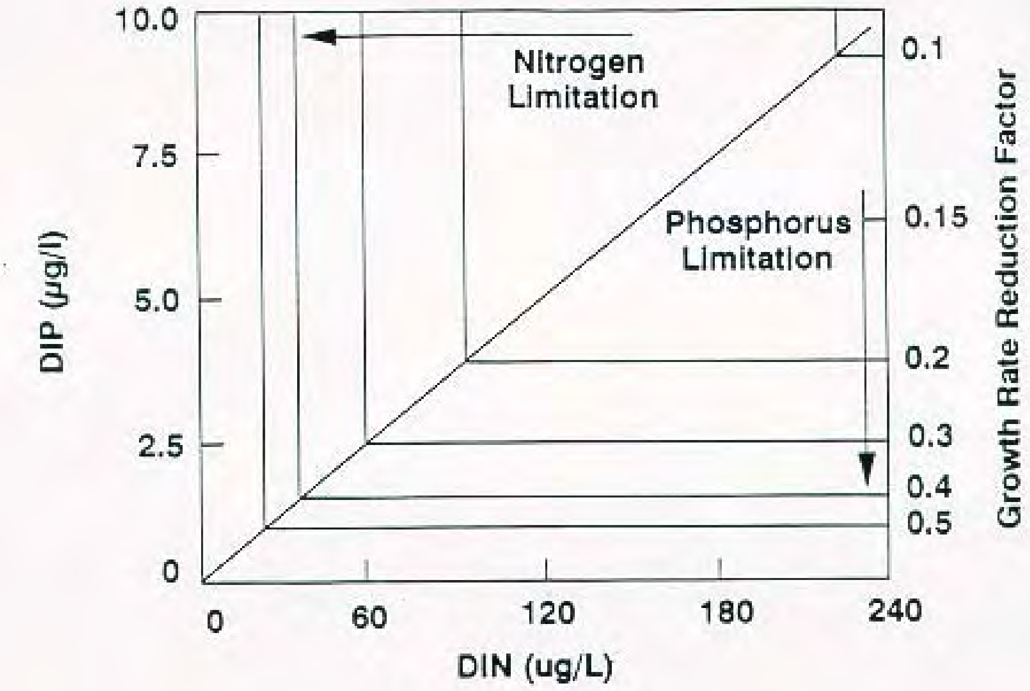


FIGURE 2-9. EFFECTS OF NUTRIENT LIMITATION ON ALGAL GROWTH (After Ambrose et al., 1993a)

reduction factor and is described as the nutrient limiting algal growth. Nitrogen and phosphorus are required by all algal species while silica is required only by diatoms.

Figure 2-9 shows the Michaelis-Menton formulation in a slightly different format. In this figure, K_{mn} = 25 µg/L and K_{mp} = 1 µg/L are used. For a stream with a DIN concentration of 100 µg/L, this corresponds to a 20 percent reduction in the growth rate (r_n = 0.8). For phosphorus to become the limiting nutrient in the stream, dissolved inorganic phosphorus must reach a level of 4 µg/L or less. It should also be pointed out that if upstream nitrogen controls were instituted such that DIN was reduced to 60 µg/L for the same stream reach, a further reduction in DIP to 2.5 µg/L would be required to keep phosphorus as the limiting nutrient. In other words, as the water column concentrations of DIP begin to approach growth-limiting levels due to continued reduction in point source phosphorus effluents, any nitrogen control strategies that might be instituted would require additional levels of phosphorus removal to keep phosphorus as the limiting nutrient by keeping an upstream N/P ratio below 10.

2.3.5.2 Algal Death

Decreases in algal biomass are brought about by two processes: algal respiration and death. Algal respiration is caused by endogenous respiration, in which algal biomass is oxidized to generate CO_2. Algal death includes grazing by zooplankton (for diatoms and greens only) and cell destruction through bacterial attack, disease, physical damage, the natural aging process, or other mechanisms. The distinction between phytoplankton reductions through death and reductions through respiration, grazing by zooplankton, or settling is that upon death all the carbon, nitrogen, and phosphorus contained in the algal biomass is returned to the carbonaceous BOD (CBOD), organic nitrogen, and organic phosphorus pools, respectively. During respiration, carbon is given off as CO_2 rather than CBOD; through grazing, only a portion of the organic contents of the algal cells is returned to the respective organic pools. (The remaining portion is lost from the phytoplankton mass balance as zooplankton biomass.)

The algal reduction rate, D_p, can be expressed as:

$$D_p = D_{p_1}(T) + D_z \tag{2-18}$$

where

$D_{p1}(T)$ = temperature-dependent endogenous respiration rate (day^{-1})

D_z = death rate (day^{-1}) (grazing and natural mortality)

The phytoplankton death rate, D_z, is a function of zooplankton population and zooplankton grazing rate. Zooplankton control the population of phytoplankton through predation. They also recycle the nutrient content of their prey. Limited data on zooplankton usually do not allow elaborate formulation of zooplankton grazing. Given the concentration of zooplankton and their filtering rate, the loss rate of phytoplankton due to zooplankton grazing may be formulated.

2.3.5.3 Algal Settling

Phytoplankton are lost from the water column through settling. In a vertically mixed water column, the net settling rate (i.e., settling to the bottom less resuspension from the bottom) is expressed as:

$$S = \frac{V_s}{H} \tag{2-19}$$

where

S = net settling rate (day^{-1}),

V_s = phytoplankton settling velocity (m/day)

H = average depth (m)

Through settling, none of the organic cell material is returned to the organic nutrient pools in the water column unless the model incorporates an explicit dynamic link between the sediments and the water column (e.g., Di Toro et al. 1990). As with other parameters in a eutrophication model, the phytoplankton settling velocity is dependent on the algal species group and size of the organism. Settling velocities can range from 0.1 to 10.0 m/day with small cells (e.g., chlorophytes) characterized by low velocities of 0.1 m/day and larger cell size diatoms by higher velocities (1-10 m/day).

2.3.5.4 Nitrogen Components

The major components of the nitrogen system are detrital organic nitrogen, ammonia, nitrite, and nitrate. In natural waters, there is a stepwise transformation from organic nitrogen to ammonia, nitrite, and nitrate, yielding nutrients for phytoplankton growth as shown in Figure 2-5. The kinetics of the transformations are temperature-dependent.

During algal respiration and death, the cellular nitrogen is returned to the organic nitrogen pool. Organic nitrogen undergoes a bacterial decomposition whose end product is ammonia. Ammonia, in the presence of nitrifying bacteria and oxygen, is oxidized to nitrite and to nitrate (nitrification). Both ammonia and nitrate are available for uptake and use in algal growth; however, for physiological reasons the preferred form of nitrogen is ammonia (Conway, 1977; Garside, 1981). The ammonia preference term is characterized in Figure 2-10. As the available nitrate increases above approximately the Michaelis limitation, for a given ammonia concentration, the preference for ammonia reaches a plateau. Also, as the concentration of available ammonia increases, the plateau levels off at values closer to unity, i.e., total preference for ammonia.

2.3.5.5 Phosphorus Components

In many stream water quality models, phosphorus is accounted for in two forms: dissolved and particulate. A fraction of the phosphorus released during phytoplankton respiration and death is in the inorganic form and readily available for uptake by other viable algal cells. The remaining fraction released is in the organic form and must undergo a mineralization or bacterial decomposition into inorganic phosphorus before it can be used by phytoplankton.

There is an adsorption-desorption interaction between dissolved inorganic phosphorus and suspended particulate matter in the water column. The subsequent settling of the suspended solids together with sorbed inorganic phosphorus can act as a significant loss mechanism in the water column and is a source of phosphorus to the sediment. Compared with the reaction rates for the algal and biological

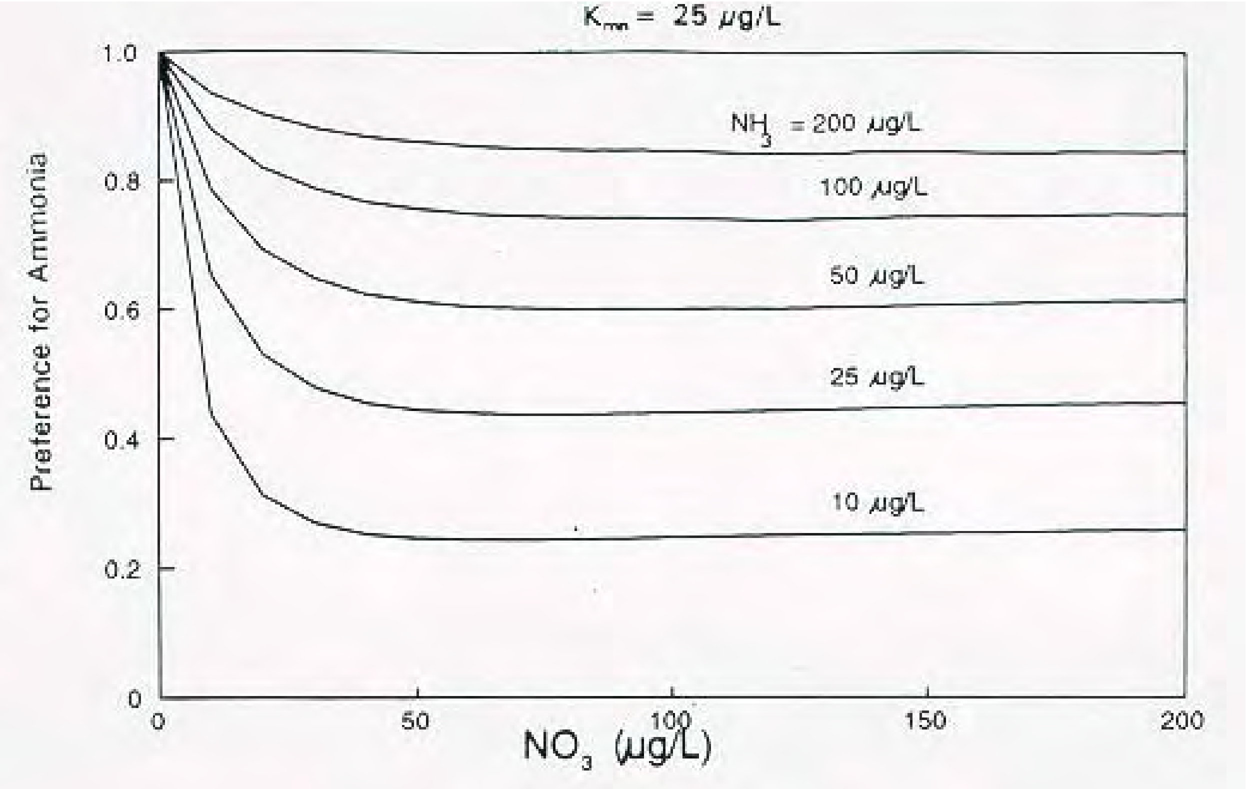


FIGURE 2-10. AMMONIA PREFERENCE STRUCTURE FOR ALGAL GROWTH (After Thomann and Fitzpatrick, 1982)

kinetics, which are on the order of days, the adsorption-desorption rates are much faster, permitting an instantaneous equilibrium assumption for the calculation. In the model formulation, the concentrations of dissolved and particulate phosphorus need to be repartitioned at every time step. A wide range of partition coefficients for phosphorus have been found in the literature. Schreiber and Rausch (1979) reported partition coefficients ranging from 4,540 to 15,900 for a flow detention reservoir (see also Thomann and Fitzpatrick, 1982).

2.3.5.6 Sediment Nutrient Release

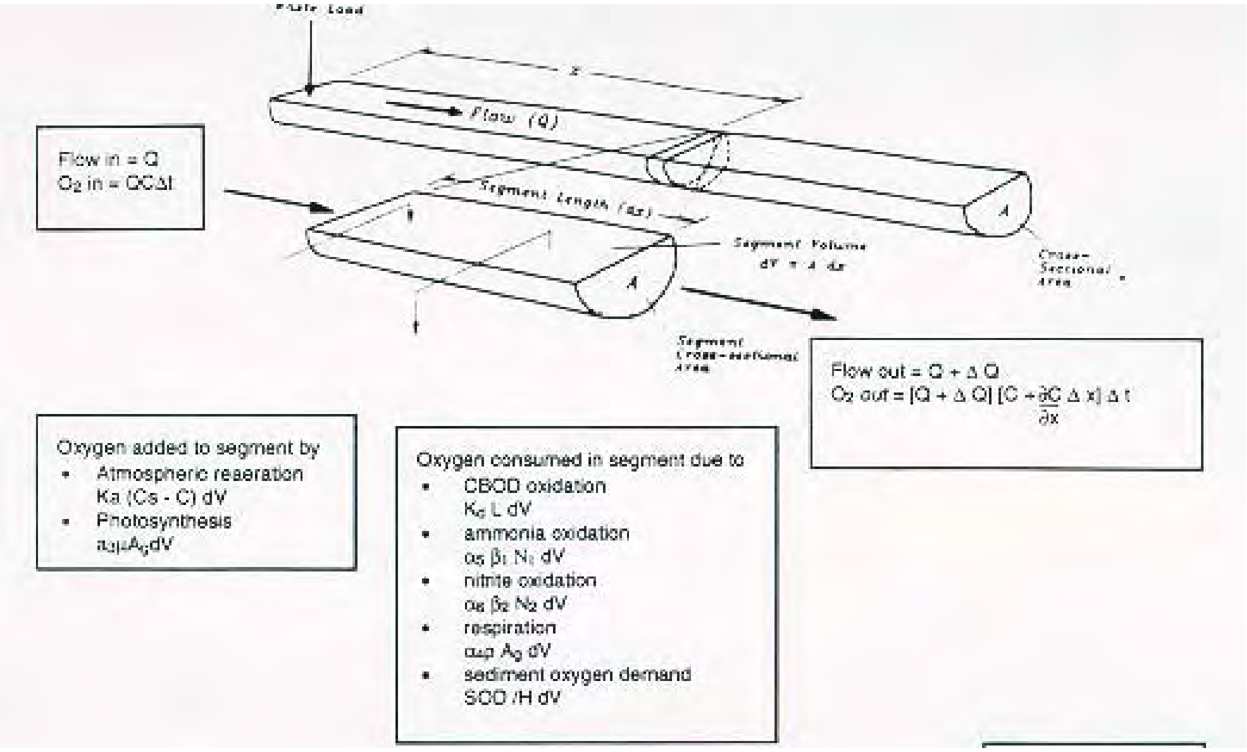
In addition to the external sources of nutrients, the release of nutrients from the sediments may also be important. Such releases occur as a result of a gradient in nutrient concentration between the overlying water and the interstitial water of the sediment. In some systems, the impact of sediment nutrient release can be significant and can result in continuing eutrophication problems even after point sources have been substantially reduced through control measures. Sediment nutrient releases can be treated as nutrient sources to the stream in waste load allocation modeling studies. In the absence of site-specific field data describing sediment nutrient release, approximations can be made on the basis of sediment oxygen demand estimates (see Appendix A).

2.4 GOVERNING EQUATIONS

2.4.1 Mass Balance Principle

The basic principle used to formulate a stream water quality model is mass balance. That is, for a given segment of the stream, the accumulation of a water quality constituent over a finite period of time is equal to the mass entering the segment plus the mass added to the segment, less the mass leaving the segment and the mass lost within the segment (Figure 2-11).

$$\begin{aligned} \text{Accumulation} &= \text{Mass In} - \text{Mass Out} \\ &\quad + \text{Source} - \text{Sink} \end{aligned} \qquad (2\text{-}20)$$



Dimension Code

L = length
M = mass
T = time

Q = river flow rate (L^3/T)
C = concentration of dissolved oxygen (M/L^3)
Cs = saturation concentration of dissolved oxygen (M/L^3)
L = CBOD concentration (M/L^3)
H = mean water depth (L)
dV = segment volume $A\Delta x$(L^3)
α_3 = the stoichiometric ratio of oxygen production per unit of algal photosynthesis (M/M
α_4 = the stoichiometric ratio of oxygen uptake per unit of algae respired (M/M)
α_5, α_6 = the stoichiometric ratio of oxygen uptake per unit of ammonia and nitrite-nitrogen oxidation, respectively (M/M)
β_1, β_2 = ammonia and nitrite oxidation rate coefficient, respectively (T^{-1})
μ = algal growth rate coefficient (T^{-1})
ρ = algal respiration rate coefficient (T^{-1})
N_1, N_2 = ammonia and nitrite-nitrogen concentration, respectively (M/L^3)
A_g = algal biomass concentration (M/L^3)
SOD = temperature-adjusted rate constant for SOD (M/L^2T)
K_a = atmospheric reaeration rate: reflects first-order reaction whereby a fraction of oxygen deficit is satisfied, $e^{-K_a t} = e^{-K_a x/U}$ (T^{-1})
$K_a(C_s\text{-}C)$ = change in dissolved oxygen concentration in a segment that when, multiplied by segment volume (dV), yields change in dissolved oxygen mass in segment (M/L^3T)
K_d = BOD oxidation rate where oxidation accounts for all CBOD removal (T^{-1})

FIGURE 2-11. MASS BALANCE EQUATIONS FOR DISSOLVED OXYGEN
(See Equations 2-21 and 2-24)
(After USEPA, 1983a)

Applying the mass balance principle and considering a small segment of a stream, one may develop:

$$dV\Delta C = QC\Delta t - [Q + \Delta Q]\,[C + \frac{\partial C}{\partial x}\,\Delta x]\,\Delta t$$
$$+ W\,\Delta t - dV\,KC\,\Delta t \qquad (2\text{-}21)$$

where

dV = volume of the segment and is equal to $A\Delta x$ (L^3)

ΔC = change of concentration (M/L^3)

Q = flow rate (L^3/t)

C = concentration (M/L^3)

Δt = small increment of time (t)

ΔQ = change of flow rate over the length

$\frac{\partial C}{\partial x}$ = concentration gradient over Δx (M/L^4)

W = direct loading rate (Mt^{-1})

K = first-order reaction rate (t^{-1})

Dividing Equation 2-21 by $dV\Delta t$ results in

$$\frac{\partial C}{\partial t} = \frac{-Q}{A}\frac{\partial C}{\partial x} - \frac{C}{A}\frac{\partial Q}{\partial x} + \frac{W}{dV} - KC \qquad (2\text{-}22)$$

Assuming steady-state conditions and neglecting the flow gradient, the above equation becomes

$$0 = -\frac{Q}{A}\frac{\partial C}{\partial x} + \frac{W}{dV} - KC \qquad (2\text{-}23)$$

Note that the reaction term KC may represent formulations for carbonaceous deoxygenation, nitrogenous deoxygenation, reaeration, or any other first-order reactions.

2.4.2 Dissolved Oxygen Equation

Using the notation in Figure 2-11, the distribution of dissolved oxygen may be formulated by including all dissolved oxygen sources and sinks described in Section 2.3:

$$0 = -\frac{Q}{A}\frac{dC}{dx} + K_a\,(C_s - C) - K_d L$$
$$-\ \alpha_5\,\beta_1\,N_1 - \alpha_6\,\beta_2\,N_2$$
$$+\ (\alpha_3\mu - \alpha_4\,\rho)A_g - \frac{SOD}{H} \qquad (2\text{-}24)$$

The terms on the right side of Equation 2-24 represent, respectively: the downstream transport of oxygen with the stream flow, atmospheric reaeration, biological oxidation of CBOD, biological oxidation of ammonia, biological oxidation of nitrite photosynthesis less respiration, and the biological oxidation of sediment materials. If CBOD is removed only by direct oxidation, the deoxygenation rate coefficient, K_d, reflecting actual oxygen reduction in the system, is equal to the CBOD removal rate coefficient, K_r. Equation 2-24 may be transformed into the time domain by substituting the relationship Adx/Q is equal to dt, or

$$\frac{dC}{dt} = K_a\,(C_s - C) - K_d\,L$$
$$-\ \alpha_5\,\beta_1\,N_1 - \alpha_6\,\beta_2\,N_2$$
$$+\ (\alpha_3\,\mu - \alpha_4\,\rho)A_g - \frac{SOD}{H} \qquad (2\text{-}25)$$

Equation 2-25 is the differential equation that is numerically solved by QUAL2E to describe the rate of oxygen change in one-dimensional streams and rivers.

In some stream BOD/DO models, the dissolved oxygen deficit, D (=C_s-C), is used instead of dissolved oxygen to formulate the dissolved oxygen profile, and Equation 2-24 can be expressed as:

$$0 = \frac{-Q}{A}\frac{dD}{dx} - K_a D + K_d L + \alpha_5\,\beta_1\,N_1 + \alpha_6\,\beta_2\,N_2$$
$$-\,(\alpha_3\,\mu - \alpha_4\,\rho)\,A_g + \frac{SOD}{H} \qquad (2\text{-}26)$$

Because of zero-order and first-order kinetics formulated in the model, the dissolved oxygen deficit terms due to different sources and sinks may be added (i.e., superimposed). For a simple case where nitrification, SOD, algal photosynthesis, and algal respiration are not significant and can be neglected, the solution to Equation 2-26 is:

$$D = \frac{K_d\,L_o}{K_a - K_r}\,(e^{-K_r\frac{x}{U}} - e^{-K_a\frac{x}{U}}) + D_o\,e^{-K_a\frac{x}{U}} \qquad (2\text{-}27)$$

Figure 2-12 shows the dissolved oxygen profile obtained by subtracting the dissolved oxygen deficit (Equation 2-27) from the saturated dissolved oxygen concentration. Also shown in Figure 2-12 is the $CBOD_u$ profile represented by Equation 2-7. At x =

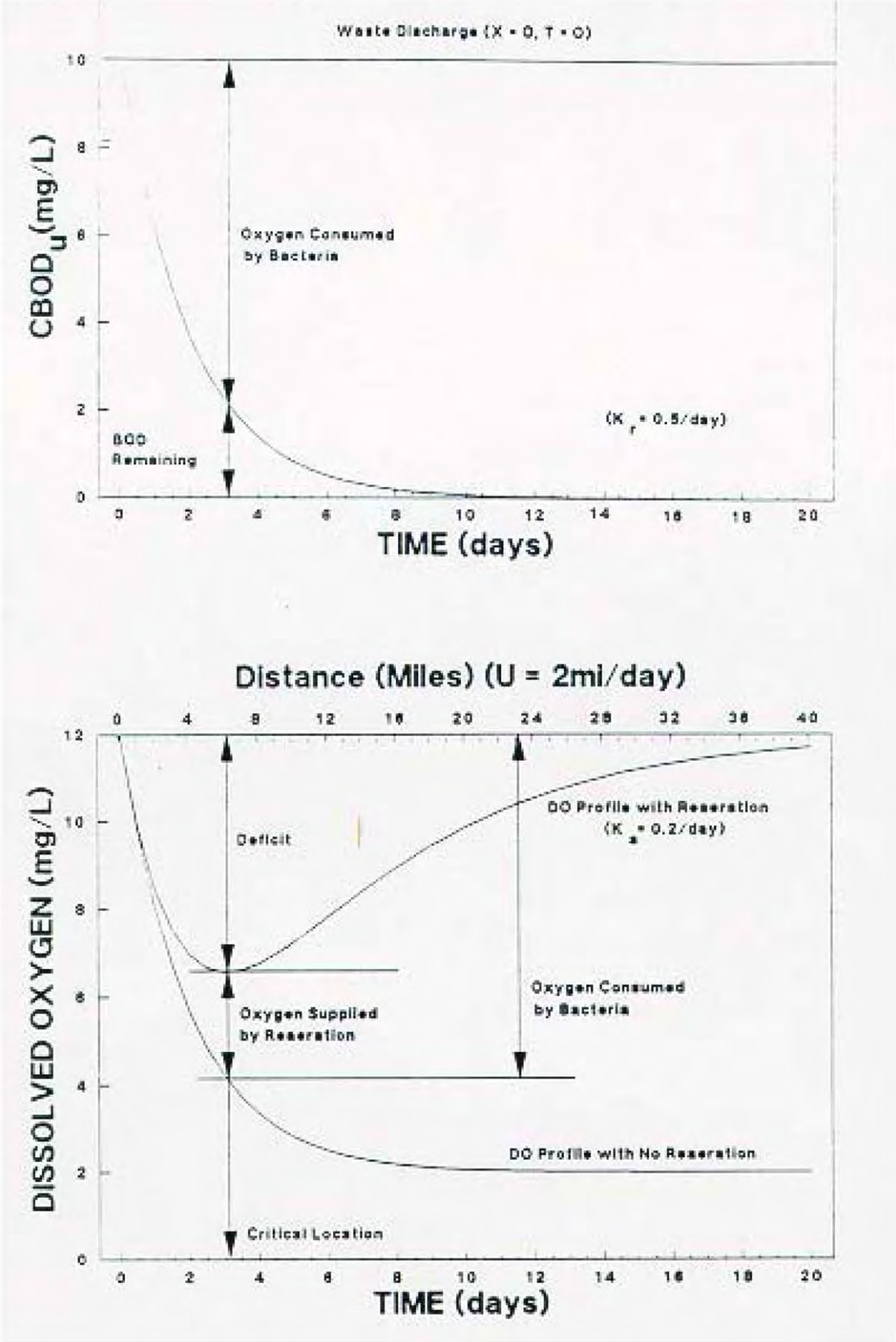


FIGURE 2-12. COMPONENTS OF DO PROFILE (SAG CURVE) DOWNSTREAM OF WASTE DISCHARGE

0, the initial $CBOD_u$ concentration is 10 mg/L following complete mixing between the waste load and stream flow. After 10 days all of the $CBOD_u$ has been exerted. Since the CBOD test measures the amount of organic material present in terms of the amount of oxygen required for its stabilization by bacteria, the reduction of CBOD concentration is equivalent to the dissolved oxygen consumption. The bottom plot in Figure 2-12 shows two calculated dissolved oxygen profiles associated with the CBOD profile in the top plot. The lower profile represents the dissolved oxygen concentration in the river if oxygen were not replenished by reaeration. In this case, the assumed initial dissolved oxygen concentration of 12 mg/L is ultimately reduced to 2 mg/L to compensate for the CBOD reduction (in top plot). The upper profile indicates the net effect of reaeration providing a source of oxygen.

The characteristic shape of the stream dissolved oxygen profile (called the DO sag curve) is the result of interplay of the biological oxidation and reaeration rates. Each is represented by first-order kinetics. In the early stages, oxidation greatly exceeds reaeration because of high CBOD concentrations and river dissolved oxygen concentrations close to saturation (i.e., small deficit). Oxygen is used faster than it is resupplied, and stream dissolved oxygen concentrations decrease. As the waste moves downstream, the consumption of oxygen decreases with the stabilization of waste and the supply of oxygen from the atmosphere increases because of greater deficits. The driving force to replenish oxygen by atmospheric reaeration is directly proportional to the oxygen deficit, (i.e., low oxygen concentration). At some point downstream from the waste discharge, the decreasing utilization and the increasing supply are equal. This is the critical location, where the lowest concentration of dissolved oxygen occurs. Further downstream, the rate of supply exceeds the utilization rate, resulting in a full recovery of the dissolved oxygen concentration. The above discussion is a simple illustration of the BOD/DO modeling analysis concept when it is assumed that organic decomposition and reaeration are the dominant pro-cesses affecting the organic balance. In reality, many other factors such as nitrification and SOD can significantly change the shape of the profile. Many streams receive nonpoint sources upstream or other point sources that depress the upstream dissolved oxygen below a saturation value. Natural background loading also may depress dissolved oxygen in certain streams. Note that constant hydraulic geometry is also assumed in the above illustration. In a natural stream, it is difficult to find constant hydraulic geometry for more than a few miles. In this case, the stream is divided into a number of reaches with uniform geometry.

2.4.3 Separate Mass Balance Equations by Constituent

Dissolved oxygen dynamics depend on the interactions of several constituents and processes. The constituents that directly influence oxygen include BOD, ammonia nitrite, and nitrate. Nitrogen and phosphorous determine growth of phytoplankton, periphyton, and aquatic plants and subsequently affect dissolved oxygen via photosynthesis and respiration. For each constituent that is in the dissolved oxygen mass balance, a separate mass balance equation is used to account for the reactions of that parameter. Using the notation developed thus far, these constituents may be modeled by the mass balance equations summarized in Table 2-4. The mass balance equations in Table 2-4 can be found in many stream water quality models (e.g., QUAL2E) that have been used inTMDL studies. Thomann and Mueller (1987) present a simplified version of the eutrophication equation for river and stream eutrophication analysis.

One should note that the major difference between the BOD/DO modeling and nutrient/eutrophication modeling is in terms of the model formulations. That is, the equations governing phytoplankton growth are nonlinear functions of nutrients and light availability, whereas the BOD/DO equations are all linear. In fact, the phytoplankton/nutrient problems are the most difficult models to work with because of the complexity of the algal biology, the nonlinear interactions between nutrients and aquatic plants, and the interactions of the sediment-water column interface. As a result, the superposition of results from BOD/DO equations is appropriate to isolate the effects of the various linear reaction terms, whereas the same is not true of the eutrophication results.

TABLE 2-4. SEPARATE MASS BALANCE EQUATIONS USED FOR EACH CONSTITUENT IN BOD, DO, AND NUTRIENT ANALYSES

Carbonaceous BOD (CBOD)	$\frac{dL}{dt} = -(K_d + K_s)\, L = -K_r\, L$
Ammonia Nitrogen	$\frac{dN_1}{dt} = \beta_3\, N_4 - \beta_1\, N_1 + \frac{\sigma_3}{H} - F\alpha_1\, \mu A_g$
Nitrite Nitrogen	$\frac{dN_2}{dt} = \beta_1\, N_1 - \beta_2\, N_2$
Nitrate Nitrogen	$\frac{dN_3}{dt} = \beta_2\, N_2 - (1 - F)\alpha_1\, \mu\, A_g$
Organic Nitrogen	$\frac{dN_4}{dt} = \alpha_1\, \rho A_g - \beta_3\, N_4 - \sigma_4\, N_4$
Algae	$\frac{dAg}{dt} = (\mu - \rho)\, A_g$
Organic Phosphorus	$\frac{dP_1}{dt} = \alpha_2\, \rho\, A_g - \beta_4\, P_1 - \sigma_5\, P_1$
Dissolved Phosphorus	$\frac{dP_2}{dt} = \beta_4\, P_1 + \frac{\sigma_2}{H} - \alpha_2\, \mu\, A_g$

Variables and coefficients not previously identified in Figure 2-11:

N_3 = nitrate nitrogen concentration (M/L^3)

N_4 = organic nitrogen concentration (M/L^3)

P_1 = organic phosphorus concentration (M/L^3)

P_2 = dissolved phosphorus concentration (M/L^3)

α_1 = fraction of algal biomass that is nitrogen (M/M)

α_2 = phosphorus content of algae (M/M)

K_s = effective loss rate due to settling (T^{-1})

F = fraction of algal nitrogen uptake from ammonia pool

β_3 = organic nitrogen hydrolysis rate coefficient (T^{-1})

β_4 = organic phosphorus decay rate (T^{-1})

σ_2 = benthos source rate for dissolved phosphorus (M/L^2T)

σ_3 = benthos source rate for ammonia nitrogen (M/L^2T)

σ_4 = rate coefficient for organic nitrogen settling (T^{-1})

σ_5 = rate coefficient for organic phosphorus settling (T^{-1})

3. MODEL SELECTION AND REVIEW

3.1 PURPOSE

The purpose of this chapter is to provide general guidance and some specific procedures for selecting an appropriate model(s) to support the development of TMDLs for BOD, DO, and nutrients in streams and rivers. Section 3.2 presents an overview of Chapter 3. Section 3.3 identifies and discusses the steps of model selection. A brief review of selected models is presented in Section 3.4. As stated earlier, the models reviewed in this guidance emphasize the fate and transport of BOD, DO, and nutrients in streams and rivers.

3.2 OVERVIEW

The success of a modeling effort to support the development of TMDLs is highly dependent on understanding the complexity of the water quality problems. This understanding will assist in defining the required accuracy, analyzing the implication of various simplifying assumptions, and eventually selecting an appropriate modeling strategy and modeling tools. It is generally known that the preferred and most cost-effective approach is to use the simplest model that includes all the important processes affecting water quality in the stream or river. Problem understanding, normally gained through characterization studies using available data, provides answers to questions such as the following: Are nonpoint sources an important contributor to water quality impairment? Are nonpoint sources or a portion of nonpoint sources controllable? Is watershed modeling necessary, and if so, what is the sufficient level of detail? What are the temporal and spatial boundaries of impaired waters? In general, the results of watershed and water quality characterization define the modeling needs as well as the need for monitoring, field surveys, and other support activities. The selection of too simple a model may result in inaccurate predictions of management needs and their implications on water quality. The cost implications of decisions are also important factors. Furthermore, inaccurate projections from present to future conditions may be caused by an insufficient sensitivity of the selected model(s) to changes in watershed or water quality processes such as the balance of sediment oxygen demand in specific reaches or seasons.

On the other hand, the selection of too complex a model can result in misdirected resources, delays in the study, and unnecessary costs. Predictive uncertainty may increase because of extra “free” model parameters that cannot be estimated with available data and resources. Study costs will increase because of the additional data needs, as well as model calibration and validation requirements. When water quality impairment is characterized as the result of both controllable point and nonpoint loadings, the selection of the modeling tools should be compatible so that watershed modeling results provide the data necessary for analysis of the water quality in the receiving water.

3.3 MODEL SELECTION

Successful model selection results from achieving a close match of the primary site-specific physical, hydrologic, and water quality features of interest to a model’s capabilities to simulate these features. Two categories of models are available for use throughout the TMDL process. The first category consists of watershed models that can be used to derive pollutant loadings from both point and nonpoint sources. Watershed models rely on (1) hydrologic processes and water balance over the watershed and (2) the physiographic characteristics of the watershed including land use and land cover, soils, topography, water uses, and discharges from municipal and industrial facilities. A detailed review of these models in terms of their potential application in the development of TMDLs is presented in USEPA (1992b). The second category consists of receiving water models that can be used to assess the impact of pollutant loadings on the waterbody. These models rely on (1) transport characteristics of the receiving water including flow rate, stream morphology and boundaries, and reaeration and dispersion parameters and (2) fate of the pollutant within reaches of the receiving water. Available watershed and water quality models range from simple empirical and statistical procedures to more deterministic and multidimensional models. In addition, these models can be differentiated based on a number of criteria including:

- Water quality constituents modeled.
- Spatial and temporal resolution of the results.
- Level of detail used to simulate hydrologic and water quality processes.
- Level of effort and data requirements for the specific application.
- Ease of application including input and output data processing, user support, documentation, and operating requirements.

Most importantly, the selection process should focus on determining which watershed and water quality processes closely match the site-specific characteristics. As mentioned earlier, the results of the characterization of watershed and water quality conditions can facilitate this selection process by providing information for (1) establishing the study objectives and constraints, (2) determining the needed detail to represent the pollutant loadings and the problem boundaries and identifying the critical conditions in terms of their temporal and spatial resolution, and (3) determining the pollutant of concern and the required mathematical formulations of hydrologic and water quality interactions.

A preliminary step of the characterization study consists of reviewing existing information about the water body so that the dominant physical and chemical processes can be defined. This information includes available site-specific analyses, monitoring data, past modeling studies that identify pollution sources and their magnitude, stream flow data, hydrologic and statistical characteristics, and ambient water quality impairment. In certain instances, simple calculations and statistical procedures may be required prior to the modeling selection process. These procedures may include pollutant loading estimates using simple loading functions, pollutant transport predictions using analytical steady-state methods, and prediction of water quality violations using standard excursion and trend analysis techniques. The result of this step is a detailed description of the modeling objectives and the potential and anticipated constraints. The magnitude of nonpoint source loadings and the significance of their impact on the receiving water may dictate whether a watershed model is necessary or whether a steady-state water quality model is sufficient to represent the dominant transport processes.

A second step may involve further evaluation of the variability of pollution sources and the hydrologic regime to assess the level of modeling effort required to represent the temporal and spatial resolution of the water impairment under consideration. Analysis of the variability of pollution loadings from various sources and identification of critical water quality impairment conditions will result in identification of the temporal and spatial resolution to be considered in model development in order to ensure an accurate representation of the system. At this stage, the analysis should ensure that the watershed model and the receiving water quality model are properly addressing the key decisions and that all assumptions are within the acceptable range. Although most streams and rivers can be represented using a one-dimensional steady-state model, certain wide or deep reaches may exhibit significant lateral and vertical water quality gradients, therefore requiring a two-dimensional configuration. In both of these cases, a simple watershed loading model may be sufficient to provide input data for the water quality model. However, a more detailed continuous or design storm watershed simulation model may be required if water impairment is characterized as storm-driven events, if a dynamic representation of water quality is required to capture daily variabilities, or if conditions where specific pollutant concentrations violate certain criteria must be defined.

In a third step, an initial assessment of the dominant water quality interactions is necessary to ensure that the proper combination of constituent and kinetics formulation is represented by the model. For example, where algal photosynthesis and respiration are a small component of the dissolved oxygen balance, the corresponding terms and rate coefficients can be ignored in the model equations. Similarly, sequential reaction of the various forms of organic and inorganic nitrogen may be highly nonlinear, resulting in time and space lags in the resultant dissolved oxygen profile and therefore making it inadvisable to select a model that combines all nitrogen reactions in a single term.

The steps listed above are addressed in more detail in the following sections, with special emphasis on BOD, DO, and nutrients in streams and rivers. Model selection may require a phased approach in which simple formulations are considered initially in Phase 1 of TMDL development. As new monitoring data and characterization studies become available, a more detailed modeling effort can be considered.

3.3.1 Study Objectives and Constraints

The first step in selecting an appropriate model to support the development of TMDLs is to review the existing data on pollutant loadings, stream flows, and ambient water quality regarding the designated uses of the stream and the applicable water quality standards. Estimation of pollutant loadings from a watershed may also require land use distribution data, soil characteristics, information on existing management practices, and pollutant buildup and washoff parameters in addition to climatic and hydrologic characteristics. These data should be reviewed to indicate whether standards violations or water quality problems are associated with diel fluctuations, storm events, flow variations, or seasons of the year.

In selecting a receiving water quality model, the modeler can use this information to determine the temporal resolution (steady-state, quasi steady-state, real time) and to specify the magnitude and variability of point and nonpoint sources that must be included in the selected modeling approach. Ambient water quality data should also indicate where violations or impairment problems are occurring and whether significant spatial gradients in concentrations exist. The combined information collected on the watershed and hydrologic characteristics and water quality problems will help determine the level of effort needed and the type of water and water quality processes that must be considered. Examples of processes of major concern when modeling BOD, DO, and nutrients in stream and rivers include CBOD oxidation, nitrite oxidation, sediment oxygen demand, ammonia oxidation, atmospheric reaeration, and algal photosynthesis and respiration.

The modeling framework should include preliminary mass balance calculations using simple models or analytical equations to help define water quality processes. These simple models provide analytical solutions for various load scenarios under varying flow conditions. In-stream sinks and sources can also be represented using simplified formulations such as zero- and first-order decay equations.

It is also desirable to anticipate the technical issues associated with pollution control scenarios, overall control levels, and other changes in watershed characteristics such as changes in land use and the addition of new sources of pollution. These issues can be summarized in terms of how these changes affect the magnitude of pollutant problems and therefore the modeling and monitoring needs. These needs actually represent the project objectives and define a number of criteria to assist in selecting the appropriate model.

3.3.2 Pollutant Loadings, Spatial and Temporal Resolution, and Transport Mechanisms

3.3.2.1 Pollution Sources

Various loads, sources, and sinks influence the dissolved oxygen distribution in streams and rivers. Upstream sources of oxygen demand or dissolved oxygen deficit can be caused by point source discharges from municipal and industrial waste treatment plants; combined and separate sewer system discharges and urban runoff; and runoff from forested, agricultural, and suburban drainage areas. In-stream processes that affect dissolved oxygen distribution include sediment oxygen demand, benthic regeneration, and oxygen production and utilization by phytoplankton and other aquatic plants.

All sources that are explicitly included in the TMDL analysis require direct measurements on appropriate time and space scales to define the magnitude of the individual source by contaminant. Under diverse conditions, receiving water quality data are required to evaluate the effects of both point and nonpoint sources.

The primary contaminants of concern associated with point sources are organic carbon compounds that produce carbonaceous biochemical oxygen demand (CBOD) and the reduced forms of nitrogen that result in nitrification. For each source type, it is necessary to define the magnitude of the ultimate oxygen demand for both classes of contaminants. In addition, field and laboratory data may be required to distinguish between the forms of organic nitrogen that can hydrolyze to ammonia and the nitrogen that is effectively refractory. This distinction can be important when nitrification is a concern. The effluents from treatment plants without nitrification can contain potentially significant concentrations of organic nitrogen. The degree of nitrification required can be influenced by the organic nitrogen level in the effluent that can be transformed to ammonia and subsequently oxidized in the stream or river. Although considered point sources, combined sewer overflows (CSOs) are storm-driven and contribute additional pollutants of concern to stormwater discharges. Groundwater may also contribute a significant portion of nitrate to surface water, although unless proven contaminated, this portion of the loading is uncontrollable within the short term.

Nonpoint source pollutants come primarily from agricultural lands, forested watersheds, and urban stormwater runoff. Agricultural areas can contribute a significant amount of nutrients depending on fertilization programs. Organic loads also can be significant during certain periods of the year. The main concern associated with agricultural sources is the storm-driven aspect of the pollutant loadings and the direct relationship between the occurrence of storm events and agricultural practices (e.g., timing and rate of fertilizer application, soil plowing and tillage techniques, etc.). Many researchers recognize that forested watersheds represent pristine conditions and that pollutant loadings from these areas are the result of natural processes. These loads represent the background condition and should be considered uncontrollable. However, some silvicultural activities (e.g., road construction, timber harvesting, prescribed burning) may result in soil disturbance and erosion processes that represent concerns similar to those for agricultural areas. Pollutants of concern include high suspended sediment and organic debris from destruction of topsoils. Urban stormwater runoff is also a complex, storm-driven source. Stormwater runoff transports significant amounts of metals. Build-up of dust and dirt on impervious areas represents a major process characterizing urban sources. Where combined sewer overflows (CSOs) are contributing to stormwater runoff, additional pollutants of concern involve those associated with municipal waste.

3.3.2.2 Model Dimensions

Most receiving water modeling projects that address dissolved oxygen in streams and rivers under summer, low-flow conditions can be adequately represented as one-dimensional, steady-state calculations. Both theory and practice demonstrate that dissolved oxygen gradients in streams and rivers are most significant along the longitudinal axis. There are only relatively minor vertical and lateral gradients except in the initial mixing zone at the point of discharge. Near the shore of streams and rivers, lateral gradients can occur from low oxygen conditions resulting from groundwater inflow depleted of oxygen, photosynthetic and respiratory processes of attached algae, elevated temperature, and reduced velocity (and atmospheric reaeration) of the shallower near-shore area. Most, if not all, State laws and regulations have defined initial mixing zones that alleviate the need to simulate the three-dimensional mixing of an effluent in a stream. This is especially true for CBOD and dissolved oxygen, for which it can be shown that only small errors result from treating the effluent plume as being immediately mixed at the point of discharge (McCutcheon, 1989). For ammonia toxicity, as well as other toxic constituents, it is the usual practice to meet chronic toxicity criteria at the edge of the mixing zone and acute criteria at the end of the discharge pipe and in the mixing zone of initial dilution (ZID). However, it is also important to account for the far-field effect of a potentially toxic discharge, since the extent of the toxicity is related to ambient and effluent levels of pH, temperature, and hardness. An effluent discharge from an activated sludge municipal wastewater plant that is characterized by low pH in the effluent may not cause an ammonia toxicity problem in the near-field mixing zone. However, further downstream, after the pH has returned to ambient conditions, ammonia toxicity can occur in the far-field region. Refer to the *Technical Support Document for Water Quality-based Toxics Control* (USEPA, 1991b) for additional information.

Certain rivers may require a framework that encompasses a two-dimensional analysis. These situations are generally associated with deep rivers or run-of-the-river impoundments where vertical or lateral gradients can be significant. Depending on the geomorphology, the upstream regions of lakes and impoundments may be characterized by significant lateral, as well as longitudinal, variations in dissolved oxygen that would require a two-dimensional analysis.

If a second dimension (i.e., depth or width) is required, the analyst should provide justification in terms of the specific decision-making elements relating to controls and treatment. This requirement is necessary since the additional dimension in the analysis for streams or rivers will usually require substantially more data collection efforts and generally will result in a more complex model whose parameter values cannot be determined reliably in all cases. Thus, if the study is not done well, the additional dimension can tend to weaken the analysis and may adversely affect the ability to make decisions. However, if significant vertical or lateral oxygen gradients are apparent in observed data, a two-dimensional model should be used.

Three-dimensional analysis of stream and river systems is still under development. These complex models are recommended only for TMDL decisions that cannot be addressed in any other fashion. If three-dimensional models are required, they must be developed by experts in the field.

3.3.2.3 Spatial Extent

The spatial extent of the modeling analysis should extend downstream of the dissolved oxygen recovery zone (see Figure 2-12). This spatial coverage is necessary for several reasons:

- Reaeration is a dominant factor in the zone of recovery, and analysis can provide information on the value of the reaeration coefficient.
- In many situations, a key issue is the presence of nitrification and the rate at which it may occur following treatment upgrades. Observations of nitrification in the zone of dissolved oxygen recovery could be valuable in defining bounds for nitrification rates to be considered in making projections under future conditions.
- Indications of phytoplankton or other aquatic plant growth can be obtained by examining the dissolved oxygen recovery zone after treatment upgrades.

The information obtained from the zone of dissolved oxygen recovery will depend, to a large extent, on the uniformity of the stream.

3.3.2.4 Time Scale

The time scale selected for the analysis should be a function of both the observed water quality and the dissolved oxygen standards or criteria for the system being analyzed. Dissolved oxygen analysis in streams and rivers usually can be performed on a seasonal time scale, employing either a steady-state or time-variable analysis. It is desirable to evaluate water quality data collected during several seasons to determine the critical period to be analyzed. The most frequent critical period is the low-flow, high-temperature summer period. Winter periods, however, may also be critical because of ice cover (physical restriction of reaeration). Fall may be significant if upstream organic carbon sources from phytoplankton and/or aquatic plants result in large depressions in dissolved oxygen. Also, spring floods that pick up large amounts of organic debris from adjacent floodplains can result in severe dissolved oxygen depletion or phytoplankton blooms. Some Great Plains streams experience a low flow in spring before snowmelt occurs. A rigorous hydraulic analysis is needed to define the critical flow periods in relation to the oxygen balance.

Next, the analyst must determine the time interval to be used in the water quality analysis. Several choices available are listed below in order of increasing complexity:

- Steady-state.
- Quasi-steady-state, including
    - Constant loads—constant stream flow and diel dissolved oxygen production by phytoplankton or aquatic plants
    - Constant loads—variable stream flow
    - Variable loads—constant stream flow
    - Other combinations of the above
- Fully time-variable dynamic analysis.

In a steady-state analysis, a spatial profile of concentration is calculated, such as would result at equilibrium (stream flows, waste loads, temperature, etc.). To the extent that actual variations in pollutant load, stream flow, and other factors can be realistically approximated by constant conditions for the period covered by the analysis, the calculated receiving water concentration profile will approximate an average of the actual concentrations during that period.

A fully time-variable analysis performs successive calculations at relatively short time steps and accepts variable input values for parameters such as stream flow, pollutant load, and temperature. The results are a record of both temporal and spatial fluctuations in the calculated water quality concentrations. Practical considerations of cost and operating time usually limit the duration that can be covered by such an analysis to critical conditions.

“Continuous” versions of time-variable models extend the calculations over longer periods of time by using larger time steps and averaging the variable input over that period. As a result, the calculated receiving water concentrations will not reflect short-term variations but will reproduce the longer-term fluctuation trends. Also available are complex kinetic systems that relate oxygen levels to phytoplankton populations (chlorophyll *a*), which in turn are controlled by light, nutrients, zooplankton, and other factors. These latter frameworks are time-variable and require extensive data for model calibration and validation.

Quasi-steady-state analyses usually have one time-variable element incorporated into what basically remains a steady-state calculation. For example, the QUAL2E program assumes that flow and loads are constant and simulates the dynamic effect of sunlight, temperature, and wind. Quasi-steady-state analyses that use steady-state calculations supplemented by time-variable calculations of diel oxygen variations are also available for streams and rivers (Chapra and Di Toro, 1991).

Continuous monitoring data (usually dissolved oxygen and flow measurements) are useful to determine the time resolution required for TMDL modeling. For example, if dissolved oxygen levels reach a constant low value for long periods and if flow and loads are relatively constant, then a steady-state analysis should be employed. Accordingly, if dissolved oxygen levels fluctuate, then a quasi-dynamic or dynamic analysis may be in order. However, steady-state averaging using a daily averaging period should be attempted before complex dynamic models are applied. If an average condition is investigated, then the fluctuations about the mean and relationships between the standard (i.e., daily or hourly average dissolved oxygen standard) and the mean values must be investigated as well. If necessary, some safety factor must be incorporated when using a steady-state analysis to estimate the mean trends in a dynamically varying stream.

Variability in loads and upstream conditions does not necessarily dictate a dynamic analysis. Only a significant variable response in water quality during critical low dissolved oxygen periods justifies a dynamic analysis to determine the exact critical conditions.

Diel variability due to photosynthesis also does not necessarily justify a dynamic or quasi-dynamic simulation. It has been consistently proven possible to simulate the average net effect of dissolved oxygen production minus respiration (P-R). However, the fluctuation about the mean must be measured or estimated by alternative means and taken into account (see Thomann and Mueller, 1987).

In general, a steady-state analysis should be widely useful. Point sources, sediment oxygen demand, groundwater inflows, and upstream background loads are approximately constant or can be adequately averaged. A dynamic analysis may be justified only if standards require that minimum dissolved oxygen levels be maintained at all times or for a significant portion of the time (i.e., 95 percent of the time) and loads are known to cause variable dissolved oxygen levels in the stream. The effects of photosynthesis can normally be taken into account with a steady-state analysis, or a dynamic analysis may occasionally be useful. The dynamic simulation is expected to provide more reliable predictions but will require more data collection and more computations. The increased amount of data input to perform a dynamic analysis also creates difficulty with proper interpretation of the results.

The issue of the time interval of the analysis is in part controlled by the major pollution sources. Point sources, sediment oxygen demand, and upstream conditions usually can be represented by steady-state modeling, which employs time-averaged values for the loads from these sources. The same type of analysis can be appropriate for some nonpoint sources, such as those associated with groundwater inflow, leaching from bottom deposits, and drainage not directly related to transient events such as storm runoff or spills. By contrast, event-related inputs of mass, such as those associated with storms that produce urban runoff and runoff from other land use types, can require either a time-variable analysis or a quasisteady-state analysis. The quasi-steady-state analysis often can be considered in situations when the receiving water is large and the incremental flow associated with the study area being modeled is small. For most of these situations, however, a time-variable analysis has been necessary.

The time-variable analysis can be applied satisfactorily if sufficient data exist or can be obtained. Projections present a special set of problems in terms of identifying the storms or storm sequences to be used to develop the TMDL. Furthermore, the event-related dissolved oxygen problem can be influenced strongly by the hydrograph after the event and the geomorphology of the downstream segments of the waterbody. In addition, the basic technical, economic, and environmental issues associated with wet-weather standards for dissolved oxygen have not yet been addressed fully.

3.3.2.5 Transport Mechanisms

The transport mechanisms that influence the distribution of wastes discharged into free-flowing streams and tidally mixed streams include advective transport and dispersive transport. Advective transport represents the bulk transport, by flow, and is often the dominant net transport mechanism except in certain tidally mixed streams where strong flow reversals occur. Dispersive transport represents the mixing (lateral and longitudinal) caused by local velocity gradients within the bulk fluid and is normally a small portion of the net transport except in tidally mixed streams where the net advective transport is overshadowed by the longitudinal mixing caused by periodic, strong tidally reversing flows.

Dispersion is present, to some extent, in all bodies of water. However, water quality profiles, such as dissolved oxygen concentrations, may not be influenced when the dispersive mixing is small and/or the advective transport is large. In these situations, decisions will not be influenced by inclusion of dispersion in the analysis. For certain slow-moving streams with complex configurations (e.g., bayous), the dispersion process may be a major transport component. Consequently, the complexity of the calculations and data collection programs will be reduced. The importance of dispersion is site-specific and can be estimated by the following procedure:

STEP 1 - Calculate the approximate longitudinal dispersion coefficient (Fischer et al., 1979).

$$D_x = 0.011\, U^2 W^2 / H U_* \qquad (3\text{-}1)$$

where

D_x = longitudinal dispersion coefficient (ft^2/sec)

U = average stream velocity (ft/sec)

W = stream width (ft)

H = stream depth (ft)

U_* = shear velocity (ft/sec)

The Shear velocity (U_*) for many streams is approximately one-tenth of the average stream velocity and can be estimated by:

$$U_* = \sqrt{gHS}$$

where

g = gravitational constant (32.2 ft/sec^2)

S = stream slope (ft/ft)

STEP 2 - Calculate the estuary number (n) as defined by O'Connor (Hydroscience, 1971). The longitudinal dispersion coefficient can be employed with stream velocity and oxidation rate (K_d) to develop this dimensionless number.

$$n = \frac{K_d D_x}{U^2} \qquad (3\text{-}2)$$

The estuary number (n) and the ratio (Φ) of the reaeration rate coefficient (K_a) to the oxidation rate coefficient (K_d) or

$$\Phi = K_a / K_d \qquad (3\text{-}3)$$

can be used, with Figure 3-1, to provide a basis for judging the significance of dispersion in calculations of dissolved oxygen concentration.

Figure 3-1 indicates that for advective streams with values for n of about 0.1 or less, neglecting dispersion effects will affect the calculation of the maximum dissolved oxygen deficit (critical deficit, D_c) by less than 10 percent. When considering steady-state conditions, dispersion can be ignored. Where reaeration is high relative to deoxygenation rates (high values of Φ), the lack of sensitivity to dispersion extends to higher values of n, as indicated by the essentially horizontal lines for the higher values of Φ.

It should be noted that the estimates of the dispersion coefficient and the ratio of the maximum DO deficit to the initial BOD concentration (D_c/L_o) incorporate several simplifying assumptions. The foregoing approach must therefore be considered to be an approximation. It should, however, be adequate for use in most studies.

There may be situations where dispersion is considered significant by the investigator even though the foregoing analysis suggests otherwise. Examples could include swamps, tidal rivers, or upstream segments of impoundments. If the computational framework employed in the analysis introduces dispersion due to spatial segmentation or numerical approximations (called numerical dispersion or numerical mixing), the study should contain an evaluation of the influence of dispersion on calculations of water quality. Finally, the influence of dispersion on TMDL decisions should also be supplied in this situation. The requirement relating to numerical mixing can often be met by comparisons of analytical solutions with computer output under comparable conditions.

A flow balance is required for the modeling effort;

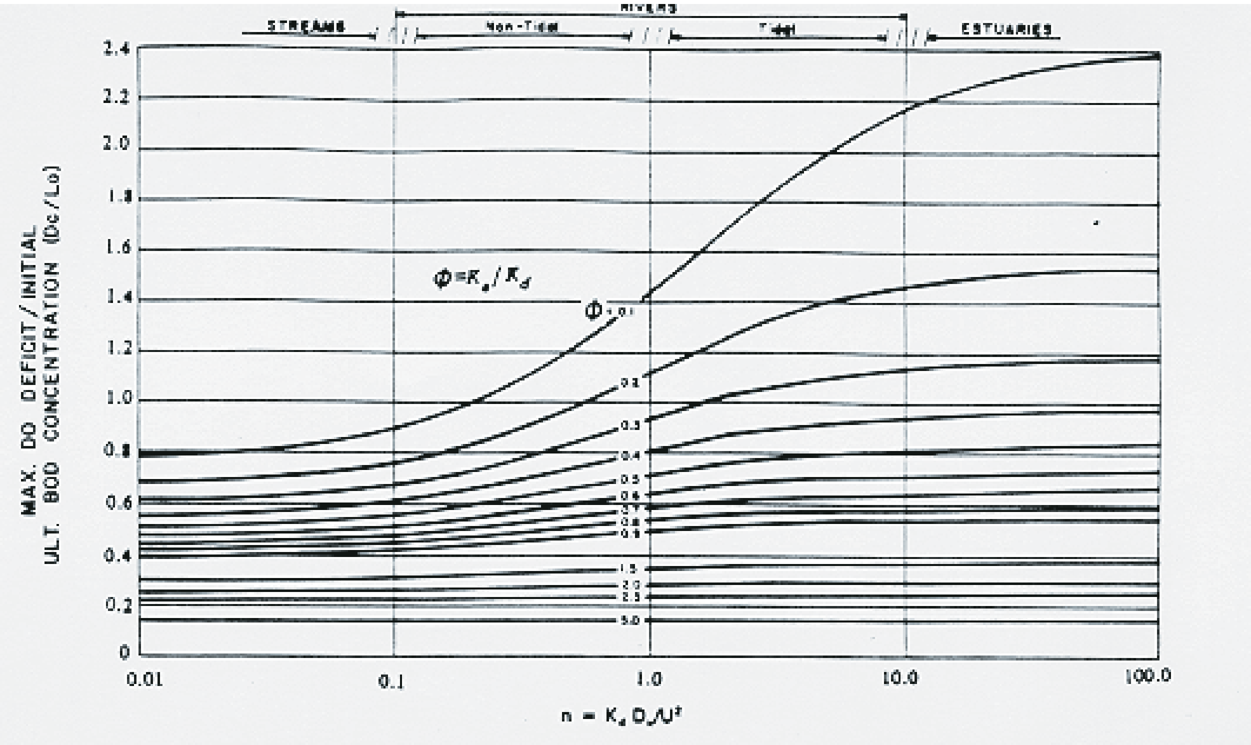


FIGURE 3-1. DISSOLVED OXYGEN RESPONSE AS A FUNCTION OF ESTUARY NUMBER, $n = K_d D_x / U^2$ (Equation 3-2), $\Phi = K_a / K_d$ (Equation 3-3) (Hydroscience, 1971)

tial importance of groundwater inflow and outflow. In addition, flow from significant tributaries and waste sources must be included in the model. The computation of a flow balance is not a trivial aspect of model development. Since USGS stream-flow gauge data are often used to compute time-averaged steady-state streamflow (e.g., monthly) at successive downstream station locations, the flow balance can become somewhat difficult because of the downstream travel time required for propagation of the flow wave and the transient response of a drainage basin to precipitation events between two successive stream gauges. Any discrepancy in the downstream balance of the USGS gauge time-averaged streamflow data with known (or estimated) point source inputs from waste discharges and gauged tributaries is usually attributed to either ungauged tributaries or groundwater flow. Each of the sources included in the model must also be supported by data (or best estimates) to characterize the concentrations of significant constituents, such as dissolved oxygen, BOD, and NH3, to compute the pollutant mass flux rate.

Data on the cross-sectional area, depth, and time of travel (or velocity), as a function of flow, are required for the flows at which observed water quality data are collected and at the critical flow regimes used for projections.

3.3.3 Water Quality Pollutant Interactions

Dissolved oxygen dynamics depend on the interactions of several constituents and processes. The constituents include dissolved oxygen, carbonaceous BOD, ammonia, nitrite, nitrate, temperature, and in some cases phytoplankton, periphyton, and aquatic plants.

These constituents and processes may be modeled by a set of coupled mass balance equations such as those in Table 2-4. The selection of constituents and processes should be based on site- and problem-specific factors. Documentation of the rationale for selection of a particular combination of variables should be provided in an early stage of the study and should include an examination of observed water quality data, considering each variable supplemented by calculations and relating the selected analysis framework to the decisions to be made at the conclusion of the project.

Ranges of the specific first-order rates for the various reactions are discussed in Appendix A, as are the procedures for defining site-specific reaction rates for various levels of treatment. The necessity for departures from this norm should be documented at an early point in the project and should particularly address the additional information required in the decision-making process.

There are circumstances, particularly in systems with low dissolved oxygen, in which nonlinear kinetic formulations can be considered. The nonlinear formulation usually employed is Michaelis-type kinetics, in which the overall rate of reaction decreases as a chemical species is depleted. Dissolved oxygen is one of the chemical species that controls these types of kinetic formulations. In particular, the rate of nitrification has been shown to be controlled by dissolved oxygen levels at and below 2 mg/L (Hydroscience, 1971; Carlucci and MacNally, 1969).

One method of approximating the nonlinear nitrification reactions has been to use lower values for first-order reaction rates in areas of low dissolved oxygen concentrations. Low dissolved oxygen concentrations can also reduce the rate of BOD oxidation and sediment oxygen utilization and increase the release of contaminants from the benthos. These latter reactions are influenced only at very low dissolved oxygen levels such as 0.1 mg/L or lower. In bodies of water with large detention times, feedback reaction sequences have occasionally been employed (Thomann, 1972). For example, the death and decomposition of algal cells returns organic nitrogen to the system. Feedback reactions can utilize first-order kinetics in dissolved oxygen analysis and have been used to model larger estuaries (see Thomann and Mueller, 1987; O'Connor et al., 1973). The usual reaction sequence employed for dissolved oxygen investigation is shown in Figures 2-2 and 2-5. This feedback reaction sequence may be appropriate for larger river systems.

Many models use a simplified framework that ignores or combines some of the processes in Figure 2-5. For example, in systems where photosynthesis and respiration are small components of the overall oxygen balance, the corresponding terms and equations can be left out of the analysis (e.g., Wu and Ahler, 1979). Simple models and hand calculation techniques often represent the nitrogen cycle using a single nitrification equation (e.g., Wu and Ahler, 1979) or combine the nitrogenous and carbonaceous BOD into a single constituent representing total BOD.

Even when the nitrogen cycle is not combined into a single BOD equation, models differ in the number of stages included in the cycle. The complete sequence should include hydrolysis of organic nitrogen to ammonia and oxidation of ammonia to nitrite and nitrite to nitrate. However, many models do not include organic nitrogen as a separate constituent (e.g., Wu and Ahler, 1979; Johanson et al., 1984). It will be important in many situations to distinguish between organic nitrogen and ammonia concentrations, rather than to define the nitrogenous oxygen demand (NOD or NBOD) load on the basis of total Kjeldahl nitrogen (TKN) concentrations, which are composed of both these forms. Time and space lags in the resultant dissolved oxygen profile, due to this sequential reaction, may be significant. If the two species of nitrogen are combined in the calibration and validation effort, the apparent nitrification rate (K_n) will be lower than the actual first-order nitrification rate of ammonia. The ratio of TKN to NH_3-N affects the value of the overall oxidation rate. Where this ratio changes after treatment, the modeler is faced with additional uncertainty. Several models (e.g., Brown and Barnwell, 1987; Ambrose et al., 1988) include both organic nitrogen and organic phosphorus capability. Many models also leave out nitrite so that ammonia is oxidized directly to nitrate in the model equations (e.g., Ambrose et al., 1988). In many situations, the NO_2 concentration level observed and calculated is very low or tends to be uniform, reducing the uncertainty of this simplification. It should also be noted that where algal problems are severe, NH_3 may be taken up directly by algae.

Several levels of analysis can be used for considering the influence of phytoplankton and other aquatic plants. These are summarized in Table 3-1. Level A, which uses measured values of photosynthesis and respiration (P-R) and diel dissolved oxygen data may be satisfactory in many cases. When significant changes in nutrients or light extinction coefficient are anticipated, the Level B analysis should be considered.

Level C represents a full-scale eutrophication approach, which increases the project costs for data and modeling by several orders and should be used when the problem is dominated by photosynthetic oxygen

TABLE 3-1. METHODS OF ANALYSIS FOR PHYTOPLANKTON AND AQUATIC PLANTS

A. Measure P-R and/or diurnal swings in DO: employ measured value in steady-state or quasi-steady-state models.

B. Measure chlorophyll *a*, light, light extinction, nutrients: employ the results in steady-state or quasi-steady-state models.

Calculate P-R.

Compare to P-R data and diurnal swings.

C. Model chlorophyll *a*, nutrients, dissolved oxygen, etc. With calibration and validation, a time-variable, nonlinear modeling framework is required.

production and utilization and where environmental or control costs are significant.

Eutrophication analyses require models that simulate nutrient and algal dynamics. Phosphorus and nitrogen are generally the only nutrients considered, although silica can be considered if diatoms are a dominant component of the phytophankton community. The major processes include algal uptake, algal excretion, sediment release, and nitrification. Periphyton and aquatic plants are rarely included in water quality models because of the difficulty in predicting biomass of these parameters although these photosynthetic organisms can be significant components of the oxygen and nutrient balance, specifically in shallow rivers (e.g., Jeppesen and Thyssen, 1984; Horner and Welch, 1981). An analytical framework described by Level A in Table 3-1 can be used to estimate the diel fluctuations in dissolved oxygen due to aquatic plants. More quantitative modeling approaches would require equations analogous to those used for algae except that the settling term is replaced by a sloughing or nonpredatory mortality term (e.g., Welch et al. 1989). The alternative approach is to use field data to account for the net photosynthetic contributions of water column algae, periphyton, and rooted aquatic plants as a combined (P-R) term in the oxygen balance model.

In addition to dissolved oxygen analyses, ammonia toxicity may be important. Ammonia toxicity is due to the un-ionized form of ammonia. The un-ionized fraction of total ammonia increases with pH and temperature. Figure 3-2 shows this relationship. Most currently available water quality models do not simulate un-ionized ammonia or pH. Therefore, TMDLs that involve ammonia toxicity must usually be based on total ammonia simulations in combination with field measurements of pH and temperature (e.g., Szumski et al., 1982; Yake and James, 1983). Un-ionized ammonia concentration can be calculated from model-projected total ammonia and a relationship such as that shown in Figure 3-2. There are models available for ammonia toxicity (e.g., STREAM DO from EPA Region VIII).

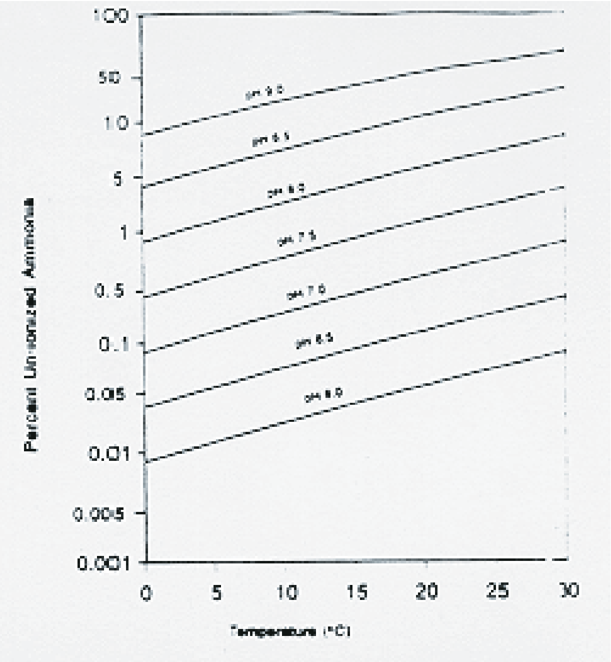


FIGURE 3-2. EFFECT OF pH AND TEMPERATURE ON UN-IONIZED AMMONIA (Novotny and Krenkel, 1975)

3.4 MODEL REVIEW

In this section the term *model*, following commonly used terminology, is used to describe computer programs. However, computer programs are not models until the user structures them with site-specific boundaries, topography, hydrology, pollution buildup and washoff, stream configuration, and pollutant interactions representative of the contributing watershed, sources, and the receiving waterbody being analyzed.

As stated earlier, TMDL development may require the development of a watershed or water quality model or both, depending on the results of the characterization study. The TMDL process creates a framework for considering the management of both point and nonpoint pollution sources that contribute to waterbody impairment. Although in most cases dissolved oxygen problems are observed during low-flow conditions in streams and rivers where point sources are the major pollutant load contributor, special consideration may be required in instances where nonpoint sources have a significant impact on dissolved oxygen levels. In these cases, the review and selection of appropriate watershed models are necessary. The model selected should represent the dominant processes in the waterbody, should provide the necessary management information on the magnitude and variability of pollutant loading, and eventually should allow for an evaluation of the implications of various watershed management alternatives. Watershed models are not considered in the present review. However, interested readers are referred to USEPA (1992b) and Donigian and Huber (1991), where detailed reviews of these models are presented to assist water quality analysts in selecting the appropriate model for a specific TMDL problem.

Selected receiving water quality models with potential application to analysis of dissolved oxygen variations in streams and rivers are reviewed in this section. The criteria used for reviewing these models as part of this document are as follows:

- They are in the public domain.
- They are available at a minimal cost from various public agencies.
- They are supported on a limited basis by Federal and/or State agencies. The form of support is generally telephone contact to a staff of engineers and programmers who have experience with the model and provide guidance (usually free of charge).
- They have been used extensively for various purposes and are generally accepted professionally.
- They represent a wide range of complexity. The more complicated models take into account additional processes and simulate a given process in more detailed manner.

The selection procedure should not be limited to those models discussed in this document. Other computer programs (models) that are available to a project or organization should be given consideration. USEPA (1979c) and Hinson and Basta (1979) describe many other available water quality models. The discussions and criteria presented in this document can be employed as major elements in the selection process. One additional consideration in this process is the experience and familiarity of the technical staff with a particular computer program.

It is suggested, however, that where project staffs do not have access to or familiarity with other computer programs, effort would be most effectively focused on the computer programs selected for discussion in this document. A brief description of the selected computer programs follows. The models are listed in order of increasing complexity. Source code, executable files and sample input files for EPA-supported models can be downloaded from EPA's CEAM electronic Bulletin Board Service (BBS); the phone number for the BBS is (706) 546-3402. The BBS system operator (SYSOP) can be contacted by telephone at (706) 546-3524.

Simplified Method Program for Multiple Dischargers (Multi-SMP) (USEPA, 1992d) is a steady-state, one-dimensional water quality model that implements EPA's *Simplified Analytical Method for Determining NPDES Effluent Limitations for POTWs Discharging into Low Flow Streams* (see Table 1-1). The model predicts four water quality variables: dissolved oxygen, CBOD, NBOD, and un-ionized ammonia. Water quality processes include reaeration, deoxygenation, nitrification, and sediment oxygen demand. The model considers up to 10 point source discharges. Multi-SMP can be obtained from the Center for Exposure Assessment Modeling (CEAM), Athens, Georgia (requires one diskette).

Enhanced Stream Water Quality Model QUAL2E and QUAL2E-UNCAS (Brown and Barnwell, 1987) are one-dimensional (longitudinal) water quality models that assume steady flow (steady-state hydraulics) but allow simulation of diel variations in temperature or algal photosynthesis and respiration. QUAL2E simulates a series of piecewise, nonuniform segments that make up a river reach. The effects of withdrawals, branches, and tributaries can also be included. Water quality variables simulated include conservative substances; temperature; bacteria; CBOD; DO; ammonia; nitrite, nitrate, and organic nitrogen; phosphate and organic phosphorus; and algae. QUAL2E is widely used for stream TMDLs and discharge permit determinations in the United States and other countries. It has a 15-year history of application and is a proven, effective analysis tool (e.g., Crabtree et al., 1986). QUAL2E Version 3 incorporates several uncertainty analysis techniques useful in risk assessment. This model can be obtained from CEAM (requires four diskettes).

Water Quality Analysis Simulation Program (WASP5) is a dynamic compartment modeling system that can be used to analyze a variety of water quality problems in one, two, or three dimensions (Ambrose et al., 1993). WASP5 simulates the transport and transformation of conventional and toxic pollutants in the water column and benthos of ponds, streams, lakes, reservoirs, rivers, estuaries, and coastal waters. The WASP5 modeling system covers four major subjects: hydrodynamics, conservative mass transport, eutrophication-dissolved oxygen kinetics (EUTRO5), and toxic chemical-sediment dynamics (TOXI5). The modeling system also includes a stand-alone link-node hydrodynamic program, DYNHYD5, that simulates the transport of water. WASP5, along with the associated programs TOXI5, EUTRO5, and DYNHYD5, can be obtained from CEAM.

Hydrological Simulation Program-FORTRAN (HSPF) is a comprehensive package that performs continuous simulation of watershed hydrology and water quality for both conventional and toxic organic pollutants. HSPF incorporates the watershed-scale Agriculture Runoff Model (ARM) and Non-Point Source model into a basin-scale analysis framework that includes fate and transport and transformation in one-dimensional stream channels (Johanson et al., 1984). It is the only comprehensive model of watershed hydrology and water quality that allows the integrated simulation of land and soil contaminant runoff processes with in-stream hydraulic and sediment-chemical interactions. HSPF, however, is an extremely complex model that requires enormous resources for development and application. HSPF can be obtained from CEAM (requires six diskettes).

CE-QUAL-RIV1 (US Army Corps of Engineers, 1990) is a fully dynamic one-dimensional riverine water quality model. The model comprises two submodels: a hydrodynamic model, RIV1H, which can stand alone, and a water quality model, RIV1Q, which requires output from RIV1H or another routing model to drive it. Ten water quality variables can be simulated: temperature, DO, carbonaceous BOD, organic nitrogen, ammonia, nitrate, phosphate, dissolved iron, dissolved manganese, and coliform bacteria. Additionally, algae/macrophyte photosynthesis, respiration, and nutrient interactions are included. CE-QUAL-RIV1 can be obtained from the US Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi.

RIVMOD is a numerical, hydrodynamic, and sediment transport riverine model that describes the longitudinal distributions of flows and sediment concentrations in a one-dimensional waterbody through time. It can be used as an alternative to the EPA-supported link-node model, DYNHYD5. RIVMOD is based on a 4-point implicit numerical integration scheme whereas DYNHYD5 is based on an explicit numerical scheme. RIVMOD is available from CEAM, although EPA does not currently provide support or documentation for the model. RIVMOD has been used by CEAM to link the transport output data files as input to the previous version of the general WASP model, WASP5.

Three of the models discussed (WASP5, HSPF, and CE-QUAL-RIV1), when operated in the fully dynamic modes, are quite complex and require well-trained analysts.

The salient features of the first five models selected for discussion are summarized in Tables 3-2 through 3-10. Since QUAL2E is probably the most widely used computer model for predicting the effects of conventional pollutants in streams, the tables use QUAL2E as a reference point against which other models can be compared. The tables presented are as follows:

Table 3-2	Constituents Modeled
Table 3-3	Summary of Capabilities
Table 3-4	Reaeration Formulations
Table 3-5	Input Data Requirements
Table 3-6	Ease of Application—Output Form and Content
Table 3-7	Ease of Application—Sources, Support, and Documentation
Table 3-8	Ease of Application—Equipment and Programming Requirements
Table 3-9	Operating Costs
Table 3-10	Hierarchy of Models Based on Selected Features

Information presented under the first four table subjects (Constituents, Capabilities, Reaeration Formulations, and Input Data Requirements) is primarily technical and is required to evaluate whether the model simulates the important physical and biochemical features of a problem. Information presented under the table subjects Ease of Application and Operating Costs is primarily nontechnical or related to operational features of the models. This information is needed to evaluate the cost associated with and the ease of acquiring the model, getting the model running on the user's system, calibrating and validating the model, and finally applying the model.

TABLE 3-2. COMPARISON OF MODELS: CONSTITUENTS MODELED

	DO	CBOD or Total BOD	NBOD	SOD	Temp	Tot P	Org P	PO_4	Tot N	Org N	NH_3	NO2	NO_3	Carbon	Algae or chl *a*	Zooplankton	pH	Alk	TDS	Coliform Bacteria	Other
QUAL2E	X	X	Δ	◆	X		X	X		X	X	X	X		X				X	X	Arbitrary conservative & nonconservative substances
CE-QUAL-RIV1	X	X		◆	X			X		X	X		X		X					X	Dissolved iron & manganese
WASP5	X	X	Δ	◆	◆	X	X	X	X	X	X		X			◆	◆		X		User can add own subroutines to model other constituents
HSPF	X	X	X	◆	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	Total inorganic carbon
Multi-SMP	X	X	X	◆	◆						X						◆	◆			

X Constituent modeled.
Δ NBOD simulated as nitrification of ammonia.
◆ Specified by model users.

DO	dissolved oxygen	NH_3	ammonia nitrogen
CBOD	carbonaceous BOD	NO_2	nitrite - nitrogen
NBOD	nitrogenous BOD	NO_3	nitrate - nitrogen
SOD	sediment oxygen demand	Carbon	carbon
Temp	water temperature	Algae	algae biomass
Tot P	total phosphorus	Zooplankton	zooplankton biomass
Org P	organic phosphorus	pH	pH
PO_4	inorganic phosphorus	Alk	alkalinity
Tot N	total nitrogen	TDS	total dissolved solids
Org N	organic nitrogen		

TABLE 3-3. COMPARISON OF MODELS: SUMMARY OF CAPABILITIES

								Process Simulated		
	Time Scale	Space Scale	Event or Continuous	Waterbody	Types of Loads Accepted	Loading Rate Accepted	Water Quality Parameter Modeled	Chemical/ Biological	Physical	Calibration/ Verification Parameters
QUAL2E	Steady state hydraulic, steady & diel quality	1D	-	Stream network	Multiple point source & nonpoint source	Constant	DO, CBOD, temperature, ammonia, nitrate, nitrite, algae, phosphorus, coliform, arbitrary nonconservative substances, three conservative substances	1st-order decay of NBOD, CBOD, coupled DO, benthic demand(s), CBOD, settling(s), nutrient-algal cycle	Dilution, advection, reaeration, heat balance	Flow, velocity, concentration
CE-QUAL-RIV1	Dynamic	1D	Event or continuous	Stream network	Multiple point source & nonpoint source	Constant	DO, CBOD, temperature, ammonia, nitrate, algae, coliform, phosphate, organic nitrogen	1st-order decay of BOD, coupled DO, CBOD, settling(s), nutrient-algal cycle	Dilution, advection, reaeration, heat balance	Flow, velocity, concentration
WASP5	Dynamic	1D, 2D, or 3D	Event or continuous	Stream network	Multiple point source & nonpoint source	Constant or time variable	DO, CBOD, chlorophyll a, organic nitrogen, ammonia, nitrite, nitrate, organic phosphorus, orthophosphate	1st-order decay of CBOD, coupled DO, settling, benthic oxygen demand, photosynthesis, respiration, nutrient algal cycle	Dilution, advection, dispersion, reaeration	Flow, velocity, concentration
HSPF	Dynamic	1D	Event or continuous	Stream network	Multiple point source & nonpoint source	Constant or time variable	Temperature, DO, BOD, nutrient, pesticide, sediment, and organic chemical	1st-order decay of NBOD, CBOD, coupled DO, hydrolysis, oxidation, photolysis, biodegradation, volatilization, adsorption, settling, benthic demand, photosynthesis, respiration, nutrient-algal cycle	Dilution, advection, reaeration, heat balance	Flow, velocity, concentration
Multi-SMP	Steady state	1D	–	Stream network	Multiple point source	Constant	DO, CBOD, NBOD, un-ionized ammonia	1st-order decay of NBOD, CBOD, coupled DO, benthic demand(s)	Dilution, advection reaeration	Flow, velocity, concentration

TABLE 3-4. COMPARISON OF MODELS: REAERATION FORMULATIONS

Model	Number of Options	Options
QUAL2E	8	A, B, F (after L), G, H, I, J, K, model accounts for C
CE-QUAL-RIV1	3	E (after M), F (after H), C (after N)
WASP5	3	A, D, E (after M)
HSPF	4	A, D, E, K
Multi-SMP	3	A, F (after H), J

Sources of Stream Reaeration Options

A	Input directly
B	As a power function
C	Structural reaeration due to dams
D	Covar's method (automatic selection among H, I, and L)
E	Wind-driven reaeration
F	Calculated as a function of velocity and depth
G	Langbien and Durum (1967)
H	O'Connor and Dobbins (1958)
I	Owens et al., (1964)
J	Thackston and Krenkel (1969)
K	Tsivoglou-Wallace method (Tsivoglou and Wallace, 1972)
L	Churchill et al., (1962)
M	O'Connor (1983)
N	Wilhelms and Smith (1981)

The information provided in these tables is primarily qualitative and sufficient to determine whether a model may be suitable for a particular application. For some of the models, more quantitative information is given in *Evaluation of Water Quality Models: A Management Guide for Planners* (USEPA, 1976b). For complete information the potential user must consult the appropriate user's manuals and other supporting documentation. The Center for Exposure Assessment Modeling (CEAM, EPA Environmental Research Laboratory, Athens, Georgia) is a source of information and limited technical support. Brief descriptions of the contents of Tables 3-2 through 3-10 follow.

Table 3-2, Constituents Modeled. As a basis for comparison, QUAL2E simulates the following constituents:

- Dissolved oxygen
- Carbonaceous biochemical oxygen demand
- Temperature
- Algae (as chlorophyll *a*)
- Organic nitrogen
- Ammonia
- Nitrite
- Nitrate
- Organic phosphorus
- Dissolved phosphorus
- Coliforms
- Arbitrary nonconservative constituents
- Three conservative constituents

Table 3-2 compares QUAL2E with other models commonly used in TMDL analyses with respect to the constituents simulated. The models vary significantly in terms of the number and type of constituents for which calculations are performed. The number of constituents analyzed usually reflects the number and complexity of biochemical processes simulated and is shown in Table 3-3. In the more complex models (e.g., QUAL2E, WASP5), provision is made for selecting only those constituents (and therefore processes) of interest.

Table 3-3, Summary of Capabilities. The model equations and process formulations in QUAL2E are the same as those discussed in Section 2.3 for dissolved oxygen, nutrients, and phytoplankton. Figure

2-5 shows the interactions of the various constituents in QUAL2E.

Table 3-3 compares the general features of QUAL2E (i.e., temporal and spatial resolution, hydraulics, types of loads, and processes simulated) with other computer models used in TMDL analyses. Multi-SMP is limited to steady-state DO/BOD analyses, whereas QUAL2E, WASP5, and CE-QUAL-RIV1 can be used for eutrophication analyses as well as dissolved oxygen analyses. The latter three models simulate the effects of photosynthesis, respiration, and temperature on diel variations of dissolved oxygen. WASP5, HSPF, and CE-QUAL-RIV1 are truly dynamic since they simulate continuous temporal variations in stream hydraulics and waste loadings. QUAL2E assumes these features remain constant, but allows the meteorology and water quality conditions downstream of the upstream boundaries to vary, making it a quasi-dynamic model.

Table 3-4, Reaeration Formulations. Most models permit direct input of the reaeration coefficient or selection from several commonly used correlations or methods. Appendix A provides a discussion of this parameter.

Table 3-5, Input Data Requirements. All models require data for input, calibration, and validation. It is best if model selection is not restricted by availability of data and the decision is made to acquire the specific type of data required for the model. On the other hand, if data availability is a constraint, selection of a less sophisticated model than would be warranted on technical grounds may be appropriate. Table 3-5 compares the input data requirements for the models discussed. Input data requirements increase with the complexity of the stream hydraulics and water quality mathematical formulations. For example, Multi-SMP, QUAL2E, and WASP5 (descriptive transport mode) assume steady-state hydraulics formulas which then require specification of regression coefficients (see Equations 2-1 through 2-3) to estimate velocity and depth required in the reaeration formulas. The more complex models such as WASP5 (linked with DYNHYD5) or CE-QUAL-RIV1 solve a form of the momentum equation, which requires more detailed characterization of the stream geometry and roughness. Similarly, extensive data are required to simulate the nonlinear nutrient-algal-DO linkage.

Table 3-6, Ease of Application—Output Form and Content. All of the computer programs print results of the simulation and the input data to standard ASCII files. The more complex programs require scratch disks or tapes for storing intermediate results to be read subsequently in submodels or for storing information to be plotted. Post-processing of model output is a major task in the application of a model, requiring software for statistical data summaries and graphical display of observed and modeled data sets (Stoddard, 1988; Stoddard et al., 1990).

Table 3-7, Ease of Application—Sources, Support, and Documentation. Two of the most important factors in facilitating the use of a new model are the adequacy of the documentation and the adequacy of the support available. The documentation should state the theory and assumptions in adequate detail, describe the program organization, and clearly present the input data requirements and format. A well-organized input data scheme is essential. Limited technical support is typically provided by agencies responsible for distribution of models. For example, EPA's Center for Exposure Assessment Modeling (CEAM) in Athens, Georgia and the U.S. Army Corps of Engineers, Waterways Experiment Station in Vicksburg, Mississippi will provide very limited technical consultation to users experiencing problems with operations of models. It may be possible that special support arrangements (including short courses or informational or personnel exchanges) are available under existing intra- or interagency agreements or could be made available to the potential user. The support agency may also be able to provide the potential user with a list of local users who could be contacted for information regarding their past or current experience with the computer program associated with the model.

Table 3-8, Ease of Application—Equipment and Programming Requirements. The models are written in FORTRAN 77, with the exception of Multi-SMP, which is written in Turbo Pascal. Most models are machine-independent though pre- and post-processors are important to ease of application. Storage requirements increase with program complexity.

Table 3-9, Operating Costs. It is difficult to estimate overall costs involved in a model application because applications differ in scope and complexity and the ability to solve or avoid certain problems is highly dependent on the experience and technical background of the analysts involved. However, machine requirements and costs associated with typical runs are usually estimated in the program documentation. As a rule, the simpler the model, the less expensive it is to apply. It is essential that the support agency and other experienced professionals be contacted for information or assistance.

Once the cost of application has been estimated, it should be compared with the benefits of using the program as part of the water quality modeling effort and the overall importance of the problem. The TMDL study costs should be consistent with the economic, social, or environmental values associated with the problem and its solution.

Table 3-10, Hierarchy of Models Based on Selected Features. To assist in initial model selection, Table3-10 shows a hierarchy of models based on important distinguishing features. As shown in this table, the programs increase in complexity. One of these programs should be adequate for most TMDL studies and, in general, the simpler program should be chosen if it contains all the features needed to simulate the important processes in the prototype. On the other hand, use of a more complex model may be justified. Often, a complex model can be used with no more additional effort than that required for a simple model by “turning off" processes (i.e., set coefficients to zero values). This procedure allows easy upgrading of the model as more information becomes available. QUAL2E, for example, can be used at the same analysis level as Multi-SMP and requires no additional information.

TABLE 3-5. COMPARISON OF MODELS: INPUT DATA REQUIREMENTS

Model	Geometric	Meteorologic	Hydrologic	Hydraulic	Water Quality	Effluent	Reaction Rates	Other
QUAL2E	Stream length, connection scheme, length of computational element	(optional) Cloud cover, barometric pressure, dry and wet bulb temperature, wind speed, evaporation coefficients, net solar radiation, dust attenuation coefficient, reach-variable climatology input for steady-state, temperature simulation	Headwater and tributary inflows, withdrawals	Coefficients for velocity-flow regression, or bottom roughness (Manning's n), bottom width, side and channel slope for trapezoidal cross-sectional option	Inflow concentration, temperature, DO, BOD, conservative constituents, initial conditions and constituent concentrations at upstream boundary	Flow rates and concentration	Reaeration, oxidation, reaction, and settling rate coefficients; nutrient half- saturation coefficients; and temperature correction factors	Latitude, longitude, and elevation of basin, day of year, computational time-step, target minimum DO concentration and sources of flow augmentation, incremental inflow and outflows, metric/English units option
CE-QUAL-RIV1	Stream lengths, connection scheme	(optional) Cloud cover, barometric pressure, dry and wet bulb temperature, windspeed, dust attenuation coefficient	Headwater and tributary inflows, withdrawals	Bottom roughness (Manning's n), time series of flow, cross-sectional geometry, bottom elevation	Inflow concentration, temperature, DO, BOD, initial and boundary conditions for all conditions for all modeled state variables	Flow rates and concentration	Reaeration, oxidation, reaction, and settling rate coefficients; nutrient half-saturation coefficients; and temperature correction factors	Flows, computational time steps, latitude, longitude, elevation of basin
WASP5	Channel length, width and direction, connection scheme, segment surface area and depth	Time series of solar radiation, wind speed and direction, photoperiod, temperature	Time series of headwater and tributary inflows	Coefficients for velocity-flow regression (steady-state, time, cross-sectional geometry, bottom elevation, series of segment volume, flow, and bottom roughness (Manning's n)	Inflow concentration, temperature, DO, BOD, initial and boundary conditions for all modeled state variables	Time series of loadings for all water quality state variables	Reaeration, oxidation, reaction, and settling rate coefficients; nutrient half-saturation coefficients; and temperature correction factors	Flows, computational time step, scale/conversion factor, sediment oxygen demand, benthic nutrient fluxes, extinction coefficient

TABLE 3-5. COMPARISON OF MODELS: INPUT DATA REQUIREMENTS (Continued)

Model	Geometric	Meteorologic	Hydrologic	Hydraulic	Water Quality	Effluent	Reaction Rates	Other
HSPF	Rating tables, length of computational element, drainage area	Solar radiation, cloud cover, air temperature, dew point temperature, wind-speed, precipitation, and evaporation rate	Internally computed	Volume and discharge/depth for each reach, average channel slope, flowthrough time for mean flow, velocity at mean flow, roughness (Manning's n)	Inflow and runoff concentration, temperature, benthic demand, and toxic organic pollutant	Flow rates and concentration	Flow rates and concentration	
Multi-SMP	Stream lengths, uniform reaches			Headwater and tributary inflows	Inflow concentration, stream temperature, and photosynthetic demand	Flow rates and concentration	Reaeration and reaction rate coefficients, and temperature correction factors	Water quality standard

Note: Meteorologic input is optional.

TABLE 3-6. COMPARISON OF MODELS: EASE OF APPLICATION—OUTPUT FORM AND CONTENT

Model	Output Form	Output Content
QUAL2E	Computer printout, plots, screen graphics	a) Listing of input data b) Concentrations and temperature by reach and computational elements at specified time steps c) Maximum, minimum, and average concentration, temperature, flow, velocity, and depth at reaches d) Final summary, which includes components of DO deficit and plot of DO and BOD e) Local climatological data f) Detailed summary of hydraulic calculations g) Plots of observed and predicted DO concentrations (option)
CE-QUAL-RIV1	Computer printout tables and output plot file	a) Listing of input data b) Concentration of a constituent at any time/location
WASP5	Computer printout/ASCII file, screen graphics, and screen tables	a Listing of input data b) Concentration of constituent and other water quality variables at any time/location c) Post-processor is available to create tables for import to graphics routines d) Values of transport and variables at any time/location e) Mass balance table for selected constituents
HSPF	Computer printout	a) Time history of runoff flow rate sediment load, nutrient concentrations and loads b) Water quality and quantity at any point in a watershed c) Display of one or more time series on a plotter and tabular display of input/output frequency tables
Multi-SMP	Computer printout of graphs and tables displayed to screen or printer	a) Listing of input data b) DO, CBOD, and NBOD concentrations at multiple points along stream reach

TABLE 3-7. COMPARISON OF MODELS: EASE OF APPLICATION—SOURCES, SUPPORT, AND DOCUMENTATION

Model	Source(s) of Model	Nature of Support	Documentation (Reference)	Source(s) of Documentation	Adequacy of Documentation
QUAL2E	Center for Exposure Assessment Modeling U.S. Environmental Protection Agency Athens, GA 30605 (706) 546-3549	Software maintenance, workshops, limited technical assistance through official EPA channels	Brown and Barnwell (1987)	CEAM	Excellent discussion of theory and assumptions; adequate user's manual; includes well-organized input data format
CE-QUAL-RIV1	US Army Engineer Waterways Experiment Station Vicksburg, MS 39180 (601) 634-3670	Telephone contact: limited technical support	U.S. Army Corps of Eng. (1990)	NTIS (703) 487-4650 ADA 230794	Excellent discussion of theory and assumptions; adequate user's manual; includes well-organized input data format
WASP5	Center for Exposure Assessment Modeling U.S.Environmental Protection Agency Athens, GA 30605 (706) 546-3549	Software maintenance, workshops, limited technical assistance through official EPA channels	Ambrose et al. (1993)	CEAM	Good discussion of theory and assumptions; adequate user's manual; includes good input data information
HSPF	Center for Exposure Assessment Modeling U.S.Environmental Protection Agency Athens, GA 30605 (706) 546-3549	Software maintenance, workshops, limited technical assistance through official EPA channels	Johanson et al. (1984)	CEAM	Good discussion of theory and assumptions; adequate user's manual; includes good input data information
Multi-SMP	Center for Exposure Assessment Modeling U.S.Environmental Protection Agency Athens, GA 30605 (706) 546-3549		USEPA (1992)	CEAM	Adequate user's manual; includes input data information

TABLE 3-8. COMPARISON OF MODELS: EASE OF APPLICATION–EQUIPMENT AND PROGRAMMING REQUIREMENTS

Model	Requirements
QUAL2E	QUAL2E is written in FORTRAN 77 and is compatible with both mainframe and personal computer systems equipped with 640 KB RAM. Can be executed from floppy disk. User interface and capabilities to interface with graphics display, laser printers or dot matrix printers, or pen plotters.
CE-QUAL-RIV1	The program is written in FORTRAN 77 and is compatible with mainframe and personal computers.
WASP5	The program is written in FORTRAN 77 and is compatible with mainframe and personal computer systems equipped with 640 KB RAM. A 10-MB hard disk and a printer are generally required. A math coprocessor is highly recommended.
HSPF	The program is written in FORTRAN 77 and is compatible with mainframe and personal computer systems equipped with 640 KB RAM. A 10-MB hard disk and a printer are generally required. A math coprocessor is needed.
Multi-SMP	Executable Turbo Pascal file. Requires IBM-compatible PC and EGA card for graphic capability.

TABLE 3-9. COMPARISON OF MODELS: OPERATING COSTS

Dimensionality	Water Quality Problem	Approximate Level of Effort
1-D steady state	DO, BOD, nutrients	1-6 person-months
1-D, 2-D, steady state	DO, BOD, nutrients, phytoplankton, toxics	0.5-1 person-years
1-D, 2-D time variable	DO, BOD, nutrients, phytoplankton, toxics	0.5-2 person-years
3-D time variable	DO, BOD, nutrients, phytoplankton, toxics	1-5 person-years

TABLE 3-10. HIERARCHY OF MODELS BASED ON SELECTED FEATURES

Models (Ordered from Least to Most Complex)	Multiple Point Sources of BOD	Distributed Sources of CBOD	Benthic Oxygen Demand	Net Algal Production	Longitudinal Dispersion	BOD Settling	Time-variable Waste Loads and Water Quality	Time-variable Flow
Multi-SMP	X	∇	♦			·		
QUAL2E	X	X	♦	D	X	•	*	
CE-QUAL-RIV1	X	X	♦		♦		X	X
WASP5	X	X	♦	Δ	¨	X	X	X
HSPF	X	X	X	X		X	X	X

X	Available feature
♦	Specified (i.e., input to the model as forcing function)
Δ	Simulated in a nutrient-algal cycle
∇	Can be simulated approximately by input of load at beginning of each multiple segment
•	Can be simulated by making $K_r > K_d$
*	Meteorology only

4. RIVER AND STREAM MODELING PROCEDURES

4.1 PURPOSE

The purpose of Chapter 4 is to review briefly the various steps associated with the development of a site-specific water quality model. These steps are generic and can be modified according to available data and the type and complexity of the water impairment being analyzed. Furthermore, the level of detail required within each step may depend on the phase of the TMDL. Simple analyses are usually sufficient during the first phase of TMDL development, whereas more detailed analysis may be required for later phases. Examples illustrating the use of these steps using the EPA-supported water quality models QUAL2E and WASP5 are provided in Appendix B. The five steps suggested for model development, illustrated in Figure 4-1, are as follows:

- Initial assessment
- Site-specific stream survey
- Model calibration
- Model validation
- Model application

4.1.1 Modeling Goals

Prior to detailing each step, it is necessary to present the overall goals of model development. A phased TMDL may require only simple modeling tools in the early stages of development. However, an increasing level of model complexity may be needed in later phases when additional data become available. At either stage, the overall modeling goals should remain consistent. For determining a TMDL for streams and rivers, the following goals are applicable:

- Development of a technically credible quantitative cause-effect representation of in-stream processes.
- Ensuring that modeling results are defensible for use in determining the loading capacity and load and TMDLs.
- Provision of analytical or modeling tools sufficient for evaluating the implications of various pollution reduction alternatives.
- Definition of the level of uncertainty for determination of the margin of safety.

When developing TMDLs, it is reasonable to separate the water quality impacts caused by a given pollution source. This is best achieved through the use of one model or a combination of models (watershed and/or receiving water models). For example, in some systems where nonpoint source loadings are significant, control of point sources may provide only a marginal improvement in water quality.

Consider a typical case in which the dissolved oxygen profile does not meet a water quality standard of 5 mg/L using summer temperatures and a 7Q10 low flow. Separation of the dissolved oxygen profile into component responses may show that the discharger is responsible for a minor portion of the predicted dissolved oxygen depression. The major depression in dissolved oxygen could be created by upstream conditions and sediment oxygen demand, for example. This perspective is important because it demonstrates that the discharger may have a minor impact on the dissolved oxygen resources, and increased treatment at the point source may have only a minor effect on the dissolved oxygen balance.

Consider a second case in which critical conditions for both dissolved oxygen and un-ionized ammonia occur during the summer when the flow is low and the river temperature is high, and where nitrification is occurring in the river. In this case, it is necessary to be able to separately evaluate the effects of carbonaceous and nitrogenous BOD on dissolved oxygen and the effects of nitrification on ammonia in order to optimize decisions on controlling nutrient loads and on selecting wastewater treatment schemes (i.e., nitrification facilities vs. advanced CBOD removal). Appendix B presents an example TMDL modeling analysis using QUAL2E that illustrates this type of optimization.

In this context, typical questions to be addressed in a stream BOD/DO and nutrient/eutrophication TMDL may include the following:

- How can the effects of two or multiple pollutant loads be differentiated?

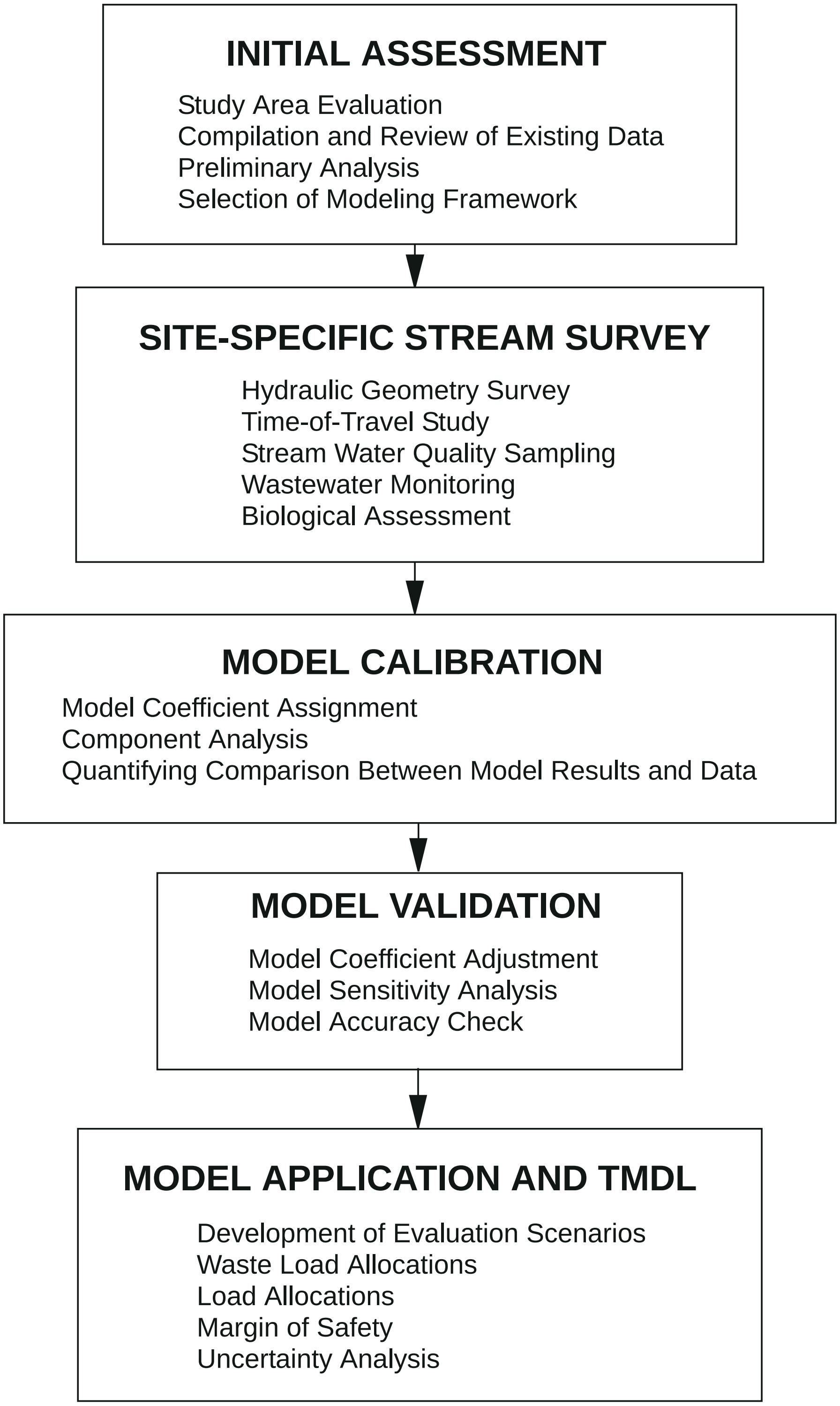


FIGURE 4-1. STEPS IN THE USE OF A WATER QUALITY MODEL FOR A SITE-SPECIFIC TMDL APPLICATION

- How can the individual impacts of sediment oxygen demand, nonpoint sources, and point sources be quantified?
- Is nitrification occurring in the stream, and would it occur under future conditions?
- Which nutrient should be controlled to reduce the algal biomass?
- What is the magnitude of dissolved oxygen fluctuations that cannot be accounted for by the present analysis?
- How do these fluctuations vary with time and space?
- Is dissolved oxygen or ammonia toxicity (un-ionized ammonia) the limiting water quality parameter?

In summary, the goal of the TMDL modeling analysis is to obtain a quantitative assessment of system behavior that will support decision making. To accomplish this task, a number of general requirements are listed below.

4.1.2 General Requirements of a Stream Water Quality Modeling Analysis

The following are examples of the basic requirements of a TMDL study:

- A quantitative analysis of all pollutant loads and inputs.
- Sufficient data to support the derivation of model coefficients.
- A consistent set of model coefficients determined from independent derivation, model calibration, and validation using available data.
- Assignment of reasonable values for model coefficients in model projections under future conditions.

The suggested requirements should be flexible to meet site-specific needs.

4.2 INITIAL ASSESSMENT

An essential element of a TMDL study is a quantitative assessment of the relative impacts of different types and sources of pollutant loads on specific water quality parameters. This assessment will help to ensure that all participants in the TMDL process understand the relative importance of various pollutant sources at an early stage and that appropriate priorities are defined. Another advantage of an initial assessment is to provide a check that all significant waste loads have been identified. It will also help to ensure that subsequent field monitoring programs are cost-effective and responsive to planning and decision-making requirements. The following analysis particularly addresses water quality in streams and rivers.

4.2.1 Study Area Evaluation

The study area evaluation defines the study area and problem by determining applicable water quality standards as well as existing and potential water quality problems. A more detailed description of the evaluation is presented in the following subsections.

4.2.1.1 Water Quality Standards

First, a desirable water use, or uses, for the stream system (e.g., recreation, water supply, agriculture) must be designated. State regulatory agencies should be consulted to define the designated uses, as well as specific water quality criteria. In addition, EPA has published a series of water quality criteria since the first “Green Book” issued by the Federal Water Pollution Control Administration in 1968. The current edition is EPA’s “Gold Book,” *Quality Criteria for 1986* (USEPA, 1987). In all cases, however, State criteria should be consulted first.

In the United States there are no standards regulating CBOD concentrations in streams. Instead, there are extensive standards for dissolved oxygen levels that are affected by CBOD deoxygenation. As a result, CBOD and oxidizable nitrogen (NBOD) are regulated on the basis of dissolved oxygen standards. Dissolved oxygen standards have been set by State regulatory agencies to protect designated use(s) for individual streams or segments of streams. State dissolved oxygen standards may be expressed as any one or all of the following:

- Average daily concentration.
- Minimum or lower percentile concentration (usually used for streams that have significant diel variations due to algae).
- Percent saturation.

There are no specific algal biomass standards for eutrophication analyses since it is difficult to determine whether a particular chlorophyll *a* concentration will be a problem. Figure 4-2 compares chlorophyll *a* concentration ranges with perceived water quality conditions and target objectives for several different waterbodies. These cases may be used as a guide to regulate nutrient inputs for eutrophication control.

4.2.1.2 Identifying Existing and Potential Water Quality Problems

Table 4-1 summarizes the constituents, waste sources, and consequences associated with dissolved oxygen, nutrient enrichment, and eutrophication problems. Nutrient enrichment and subsequent algal growth are a concern in rivers and streams because of their effect on dissolved oxygen concentrations. Growing plants provide a net addition of dissolved oxygen to the stream on an average daily basis, yet respiration can cause low dissolved oxygen levels at night that can affect the survival of less tolerant fish species. Also, if environmental conditions cause a die-off of either microscopic or macroscopic plants, the decay of biomass can cause severe oxygen depressions. Therefore, excessive plant growth can affect a stream's ability to meet both average daily and instantaneous dissolved oxygen stream standards.

Biological assessments can also be designed to establish baseline conditions and assess impacts from point and nonpoint pollution sources. They can play a fundamental role in establishing biocriteria, which are numerical or narrative expressions that describe the reference biological condition of aquatic communities inhabiting waters of a given designated aquatic life use (Barbour et al., 1992). Biocriteria are often presented as measures such as species composition, abundance, and diversity (Gallant et al., 1989). Biological communities reflect overall biological integrity and integrate the effects of different pollutant stressors. In cases where specific impacts are absent or unknown (e.g., nonpoint source impacts that degrade habitat), bioassessments may be the only practical assessment tool.

The individual water quality problems can be associated with specific time and space scales, which can be used to identify the most appropriate method of analysis. Several different time and space scales are required for effective water quality evaluation (see Figure 4-3). In general, the dissolved oxygen problem associated with organic waste discharges has a significant time scale of days to weeks, with a significant space scale of impacts up to 20 miles. Nutrients are usually associated with a longer time scale of seasons to years and a space scale of up to 100 miles. It is essential to recognize these time and space scales in order to address questions and problems in the most economical manner and to provide meaningful analysis. The selection of a steady-state or time-variable model should be determined on the basis of the water quality variable, the available data base, and the major mechanisms affecting that variable.

In evaluating dissolved oxygen water quality effects, including situations where algal influences are important, a steady-state analysis can be used. Phytoplankton chlorophyll *a* concentrations will commonly be sufficiently constant over the period covered by a steady-state analysis to justify this approach. In such cases, a steady-state analysis of dissolved oxygen response to point source BOD discharges is superimposed over the algae-induced diel fluctuations. These fluctuations can be calculated by simplified analytical approximations. QUAL2E uses steady-state hydrology and allows simulation of diel variations in temperature or algal photosynthesis and respiration.

Time-variable approaches to eutrophication problems are sometimes employed when a time-variable data base exists (or can be developed) to calibrate the model dynamically over a range of conditions. Models such as HSPF, CE-QUAL-RIV1, and WASP5 are run in the time-variable mode. When using these models, the computation can be continued, using constant input values, until a steady-state condition is reached.

A general guideline for determining the appropriateness of a steady-state vs. a time-variable approach is summarized below:

- If phytoplankton chlorophyll *a* concentrations are relatively constant over a time period of 1 or 2 weeks, then a steady-state approach is justified. This period should coincide with the critical season in terms of stream flow and temperature for dissolved oxygen analyses. Spatial variations in algal biomass can be handled by averages over appropriate river reaches.
- Where the principal water quality issue is the level of biomass rather than oxygen depletion, longer time periods (covering one or more seasons) are usually selected. On such a time scale, expected changes are large and time-variable eutrophication models are the most appropriate modeling approach.

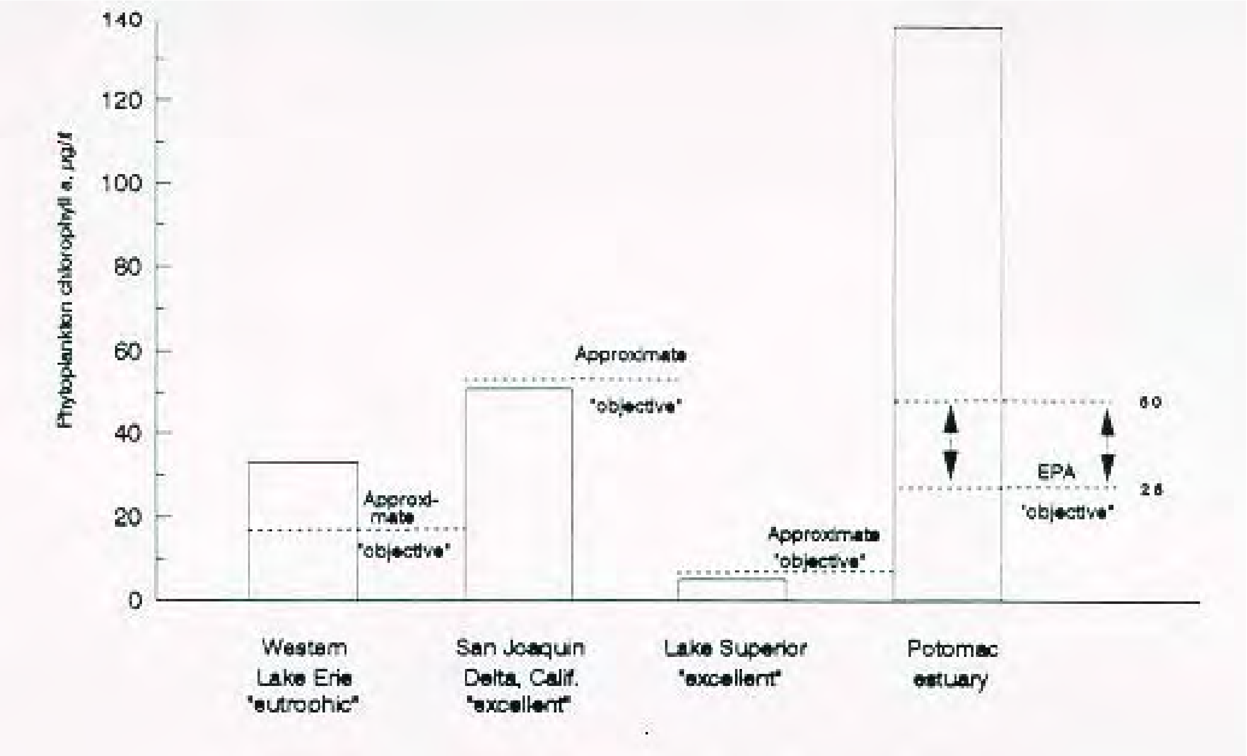


FIGURE 4-2. RANGE OF CHLOROPHYLL *a* AVERAGE CONCENTRATIONS AND TARGET "OBJECTIVES" TO REGULATE NUTRIENT INPUTS FOR EUTROPHICATION CONTROL FOR VARIOUS WATER BODIES (After Thomann and Mueller, 1987)

TABLE 4-1. IDENTIFICATION OF POTENTIAL WATER QUALITY PROBLEMS: DISSOLVED OXYGEN DEPLETION, NUTRIENT ENRICHMENT, AND EUTROPHICATION

Sources:	Organic material, ammonia in: wastewater runoff and CSOs benthic oxidation algal production marinas, boating heated effluent	Nitrogen, phosphorus, carbon in: wastewater runoff and CSOs atmospheric deposition benthic recycling marinas, boating
Consequences:	Fish kills Reduced fish productivity Less desirable aquatic community	Nuisance levels of phytoplankton Less desirable aquatic community Large dissolved oxygen fluctuations Dissolved oxygen depletion

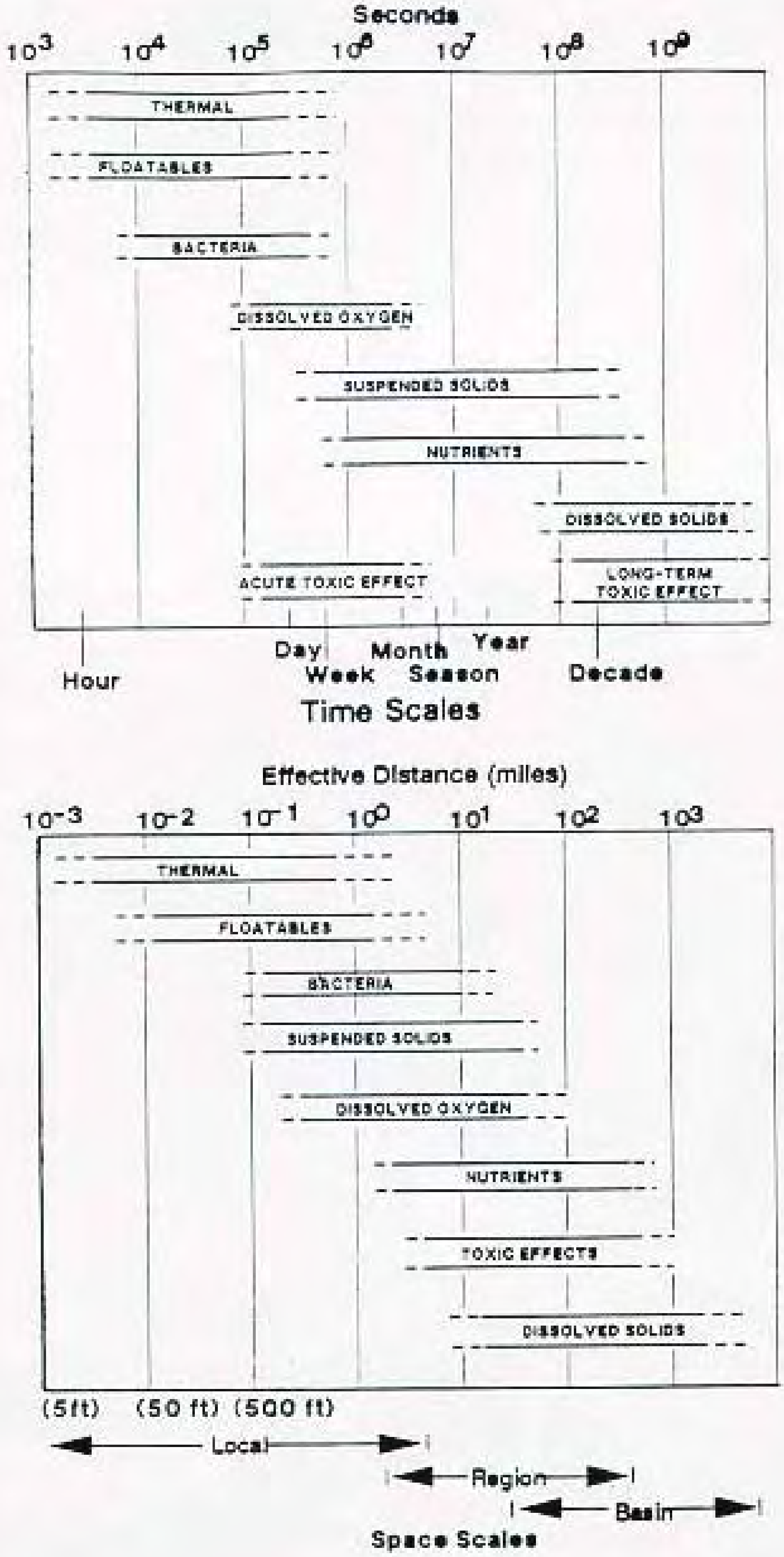


FIGURE 4-3. TIME AND SPACE SCALES FOR ASSESSMENT OF WATER QUALITY PROBLEMS
(After USEPA, 1983b)

4.2.2 Compilation and Review of Existing Data

A thorough characterization of the river or stream is necessary for any water quality study. Table 4-2 lists the types of data to be gathered and their possible sources.

Stream flow and geometry are typically available at specific gaging stations for large rivers through the USGS and/or the US Army Corps of Engineers. For analysts with access to EPA's mainframe computer, USGS streamflow data available can be easily queried using interactive tools and the Reach File.

The EPA STORET system is a good source for water quality data. STORET data are usually appropriate to describe long-term trends for water quality problem identification. However, STORET does not always have synoptic surveys of the stream system, which are most useful for water quality modeling.

EPA's Permit Compliance System (PCS) is the repository of NPDES permitted loads and all reported

TABLE 4-2. DATA TYPES AND POSSIBLE DATA SOURCES FOR STREAM TOTAL MAXIMUM DAILY LOADS

	Source		
Data Type	**Federal Agencies**	**State Agencies**	**Local Groups**
Stream Geometry	USGS US Army Corps of Engineers' Division/District Offices EPA	Special studies	Planning agencies
Stream Flow	USGS gage records and low flows (available through EPA)	Publications on low flows Basin plans	Universities Planning agencies
Water Quality Data	EPA STORET USGS US Fish & Wildlife Service	Regulatory agencies' TMDL studies State Dept. of Health	Studies by regional planning groups Discharger's studies Universities
Wastewater Loads	EPA Permit Compliance System (PCS)	Discharge Monitoring Reports (DMR)	Municipal and industrial discharger's plant records
Nonpoint Loads	EPA STORET, USGS and US Fish & Wildlife Service; urban runoff data available from EPA National Urban Runoff Program (NURP); precipitation and meteorological data available from NOAA National Climatic Data Center; land use data available from USGS; soils characteristics data available from USDA Soil Conservation Service.	Urban runoff data from special studies; precipitation and meteorological data from State planning agencies and local airports; land use and soils characteristics data from State planning, agricultural and geological agencies.	Urban runoff data from regional, city and county studies; precipitation and meteorological data from local and county planning agencies and local airports; land use and soils characteristics data from regional and county planning, agricultural and geological agencies.

information from NPDES discharge monitoring reports (DMRs).

4.2.3 Preliminary Analysis

Preliminary screening analysis of available information described in the previous section may be performed by employing analytical equations, simplified models, or a preliminary version of the water quality model. Several simplified analyses are presented in the following paragraphs.

4.2.3.1 Screening Procedure for Determining Algal-Nutrient Relationship

This simple procedure may be used to provide an appropriate indication of a *nonproblem*. That is, if the maximum possible chlorophyll *a* level that could be achieved is extremely low, it will usually be safe to conclude that nutrients do not pose a problem in relation to water column algae. The guidance in Section 2.3.4.5 and Section A.9 of Appendix A, which relates chlorophyll *a* levels to dissolved oxygen effects, can be used to determine how low the concentration of chlorophyll *a* must be in a particular situation to be considered insignificant.

On the other hand, it is not appropriate to use this screening procedure to conclude that there is a problem. In most natural systems, especially flowing streams, the actual chlorophyll *a* levels that occur will be substantially less than the maximum potential under a combination of ideal conditions. Collection of chlorophyll *a* data could be used to verify the estimated chlorophyll *a* levels and to determine whether a problem exists.

Stoichiometric ratios can be used in preliminary screening analyses to make two useful initial assessments that can help focus subsequent data acquisition, testing, and analysis activities. The first of these is to determine the limiting nutrient (nitrogen or phosphorus) and therefore the most appropriate for control. The second is an estimate of the maximum potential chlorophyll *a* level that could result and the implications on the need for nutrient control. In either case, it should be recognized that such a screening is relatively imprecise and results should be interpreted with care. When indicated conditions are marginal rather than being dramatically in favor of one result over another, additional analyses should be performed as indicated in the discussion that follows.

Algae require inorganic carbon, nitrogen, phosphorus, silica (for diatoms), and various trace elements in the presence of light to synthesize algal protoplasm. Nitrogen and phosphorus are the only essential elements that can be controlled since carbon is often (but not always) readily available in solution and the various trace elements are usually plentiful in natural systems. When considering cell stoichiometry of aquatic plants or phytoplankton, for example, cells contain approximately 0.5 - 2.0 µg phosphorus per µg chlorophyll *a* and 7 - 10 µg nitrogen per µ*g chlorophyll a. Although the weight ratio of each nutrient to chlorophyll a* varies with the age of an algal population, species composition, and nutritional state, the following ratios are commonly used to represent typical conditions:

7 µg N/ µg chlorophyll *a*
1 µg P/ µg chlorophyll *a*

The chlorophyll *a*-to-carbon and carbon-to-nutrient stoichiometry of algal cells is not precise, and ratios that are somewhat different from those used in this manual may be preferred by other analysts (see Bowie et al., 1985). Such preferences are usually based on local data, which should be used whenever possible.

For example, consider the following nutrient concentrations:

N = 0.35 mg/L = 350 µg N/L
P = 0.02 mg/L = 20 µg P/L

Using the above stoichiometric ratios, the maximum potential chlorophyll *a* concentration would be either:

$$350\ \mu g\ N/L)\ \frac{1}{7\ \mu g\ N/\mu g\ Chl\ a} = 50\ \mu g\ \ Chl\ a/L$$

(Nitrogen)

or

$$(20\ \mu g\ P/L)\ \frac{1}{1\ \mu g\ P/\mu g\ Chl\ a} = 20\ \mu g\ \ Chl\ a/L$$

(Phosphorus)

Since each concentration represents a maximum potential, the lower of the two is the maximum result and phosphorus is therefore the limiting nutrient. The maximum possible chlorophyll *a* concentration that could result from the waste discharge in combination with the background stream concentration is 20 µg/L.

This level might be achieved if there is adequate residence time in the study area, optimal environmental conditions (i.e., temperature and light) exist, and all of the phosphorus is in a form available for algal uptake. Stream conditions, however, are usually considerably less than optimal. Stream turbidity, shading by a forest canopy, or self-shading by the algae usually restrict the available light.

If the ratio of ambient nitrogen (mg N/L) to phosphorus (mg P/L) is greater than 12 to 1, phosphorus is considered to be the limiting nutrient; if the N-to-P ratio is less than 5 to 1, nitrogen is considered limiting. However, a number of factors must be considered when interpreting the results of the type of analysis illustrated above, particularly when the outcome is not at one extreme or the other.

- Nutrient availability is an important issue. Organic and particulate forms of the nutrients cannot be used directly by algae. Although a relatively slow conversion to available forms takes place in natural water systems, the residence time in most stream systems is too short to make this a significant factor.
- The lack of precise stoichiometric ratios can be an important consideration when N-to-P ratios are only marginally in favor of one or the other as a limiting nutrient.
- Nitrogen-fixing blue-green algae may negate the impact of a control program based on nitrogen being the limiting nutrient because they can draw on a source (atmospheric) other than the wastewater discharge.

The first two of these issues can be addressed more reliably by the use of algal growth potential (AGP) tests to supplement or substitute for the simple analysis based on stoichiometric ratios. Properly performed AGP tests are generally preferred because they will provide more accurate results than the use of stoichiometric ratios. The Selenastrum capricornutum Printz Algal Assay Bottle Test described by Miller et al. (1978) is an example of a suitable AGP test.

4.2.3.2 Phytoplankton Analysis of Short Streams

The previous section describes the estimation of maximum chlorophyll *a* concentrations based on given nutrient concentrations under optimum light and temperature conditions. This section illustrates procedures to develop estimates of maximum chlorophyll *a* concentrations in a stream under specific light, temperature, and nutrient conditions. A *short* stream is defined as one in which nutrients are in excess of growth-limiting concentrations over the entire length of interest. The distance and hence the time of travel for stream nutrient problem contexts generally tend to be short, perhaps on the order of less than 10 days. This travel time is equivalent to distances of less than 160 miles for streams with a velocity of about 1 ft/sec (0.3 m/sec). As a result, the phytoplankton biomass may not have enough time to grow to the maximum level calculated from the N-to-P ratio. The rate of growth of the phytoplankton and the travel time of the stream length are therefore of specific importance.

Thomann and Mueller (1987) describe a simplified set of differential equations for chlorophyll *a* and inorganic phosphorus and nitrogen under a steady-state condition:

$$\frac{dA}{dt*} = G_n \ A \tag{4-1}$$

$$\frac{dP}{dt*} = a_P \ G_p \ A \tag{4-2}$$

$$\frac{dN}{dt*} = a_N \ G_p \ A \tag{4-3}$$

where

A = concentration of chlorophyll *a* (µg/L)
P,N = concentrations of inorganic phosphorus and nitrogen (mg/L)
$t*$ = travel time in stream (= x/U) (days)
x = distance downstream of effluent (miles)
U = stream velocity (miles/day)
a_P = phosphorus:chlorophyll ratio (0.001 mg P/µg A)
a_N = nitrogen:chlorophyll ratio (0.007 mg N/µg A)
G_n = phytoplankton net growth rate (day $^{-1}$)
= $[G_p - D_p - V_s/H]$
G_p = phytoplankton growth rate (r_N = 1.0) (day $^{-1}$)
D_p = phytoplankton death rate (day $^{-1}$)

V_s = phytoplankton net settling velocity (ft/day)
H = average stream depth (ft)

In these equations, inorganic phosphorus is assumed not to settle and is not recycled from respired algae.

Solutions of Equations 4-1 through 4-3 are:

$$A = A_o e^{G_n t^*} \tag{4-4}$$

$$P = P_o - \frac{a_P G_p A_o}{G_n} (e^{G_n t^*} - 1)$$

$$(\text{ for } P > 0.025 mg/L) \tag{4-5}$$

and

$$N = N_o - \frac{a_N G_p A_o}{G_n} (e^{G_n t^*} - 1)$$

$$(\text{ for } N > 0.125 \ mg/L) \tag{4-6}$$

Note that these equations are valid only in the region where nutrients are in excess of phytoplankton growth needs. A_o, P_o, and N_o are the in-stream concentrations of chlorophyll *a* (µg/L), inorganic phosphorus (mg/L), and inorganic nitrogen (mg/L) at the outfall after mixing of the upstream and effluent flows. The travel time to the location in the stream where nutrients begin to significantly affect the phytoplankton growth rate can be calculated from Equation 4-5 or 4-6 by substituting P = 0.025 mg/L for inorganic phosphorus and N = 0.125 mg/L for organic nitrogen:

$$t^*_P = \frac{1}{G_n} \ln \left[\frac{A_o' + P_o - 0.025}{A_o'} \right] \tag{4-7}$$

$$t^*_N = \frac{1}{G_n} \ln \left[\frac{A_o'' + N_o - 0.125}{A_o''} \right] \tag{4-8}$$

where

t^*_P, t^*_N = travel times to stream locations where inorganic phosphorus and nitrogen concentrations begin to significantly limit phytoplankton growth (days)

$$A_o' = \frac{a_P \ G_p A_o}{G_n} \quad \text{(mg/L)}$$

$$A_o'' = \frac{a_N \ G_p A_o}{G_n} \quad \text{(mg/L)}$$

In summary, "short" streams are defined as those streams where actual travel times are less than t^*_P or t^*_N as calculated from Equations 4-7 and 4-8. For such streams, phytoplankton concentrations vary exponentially according to Equation 4-4 and are essentially independent of nutrient concentrations (which are in excess of growth-limiting concentrations). Nutrient removals at a point source will reduce the in-stream concentrations P_o and/or N_o and will decrease the travel times t^*_P and/or t^*_N. If t^*_P or t^*_N becomes less than the actual stream travel time, peak chlorophyll concentrations will be reduced.

For small streams, 10 to 20 miles long with velocities of 0.5 to 1.0 ft/sec (8 to 16 miles/day), resulting travel times are from 1 to 2.5 days. If a high-rate activated sludge (HRAS) plant flow with effluent P = 5 mg/L (75 percent of which is available for uptake) mixes with an equal upstream flow with ambient P = 0.02 mg/L, P_o = 25 µg/L, G_p = 1/day, and G_n = 0.5/day, t^*_P will equal approximately 7 days. If phosphorus removal were instituted and the effluent were reduced to 1 mg/L, t^*_P would become approximately 4 days. In both cases, t^*_P exceeds the actual travel time and the stream would be classified as a "short" stream, with phytoplankton concentrations varying exponentially throughout its length.

The following procedure for analysis is suggested:

1. Determine the limiting nutrient (inorganic phosphorus or nitrogen). Include an estimate for the fraction of the inorganic nutrients available for uptake (for example, 0.75).
2. For present conditions, estimate G_n, G_p, D_p, and V_s using observed phytoplankton data and empirical relationships.
3. Calculate t^*_P or t^*_N for present conditions from Equation 4-7 or 4-8.
    - If t^*_P (or t^*_N) is greater than the actual travel time in the stream reach under consideration (t^*_a), then nutrients are in excess and

$$A_{\max} \approx A_o e^{G_n t^*_a}$$

    - If t^*_P or t^*_N is less than t^*_a, nutrients have the potential to limit at t^*_P or t^*_N and

$$A_{\max} \approx A_o e^{G_n (t^*_P \text{ or } t^*_N)}$$

TABLE 4-3. DATA FOR STREAM EUTROPHICATION CALCULATION

Parameter	Unit	Present	Design
Flow Rates:			
Ambient Stream	cfs	20.0	12.0
Wastewater	cfs	0.39	0.49
Total Flow	cfs	20.39	12.49
Hydraulic Geometry:			
Stream Depth	ft	3.0	2.2
Velocity	ft/sec	0.5	0.4
Velocity	mi/day	8.2	6.56
Water Temperature	^oC	23.0	25.0
Solar Radiation:			
Daily Solar Radiation (I_T)	ly	600	600
Optimum Light Intensity (I_s)	ly/day	300	300
Photoperiod (f)	day	0.5	0.5
Averaging Period (T)	day	1.0	1.0
Light Extinction Coef. (K_e)	ft^{-1}	0.33	0.33
K_eH		0.99	0.73
Inorganic Phosphorus Conc:	mg/L		
Upstream		0.02	0.02
Wastewater		5.0	1.0
Maximum Limiting Phosphorus Conc	mg/L	0.025	0.025
Chlorophyll *a* Conc:			
Upstream ($x < 0$)	µg/L	25.0	25.0
Downstream ($x = 20$ mi)	µg/L	65.0	?
Algal Growth Rate, G_{max} (20 ^oC)	day^{-1}	1.8	1.8
Algal Respiration Rate, μ_R (20 ^oC)	day^{-1}	0.1	0.1
Net Algal Settling Rate, V_s	ft/day	0.327	0.327

4. Under projected conditions and future removal programs, repeat steps 1 through 3. If the new t^*_P (or t^*_N) is greater than the new t^*_a, nutrients would still be in excess.

The data given for the example calculation for a short stream are summarized in Table 4-3 and Thomann and Mueller (1987). The underlying assumption is that nutrients are not limiting the algal growth in the stream.

Analysis

- Estimate net phytoplankton growth rate (G_n):

Use observed chlorophyll *a* data at $x = 0$ and $x = 20$ miles and assume an exponential increase. $P(x=20) = P(x=0)\exp(+G_n\, x/U)$

Travel time for reach: $x/U = t^* = 20$ mi/8.2 *mi/day* = 2.44 *days*

Chlorophyll *a* at end of 20 miles: $P(x=20) = 65 = 25\, e^{(2.44)(Gn)}$

Net growth rate: $G_n = [\ln(65/25)] / 2.44 = 0.391\ day^{-1}$

- Determine algae population dynamics rate factors:

$$G_p = G_{\max}\, 1.066^{\,(T-20)}\left[\frac{2.718\, f}{K_e H T}\left(e^{-\alpha_1} - e^{-\alpha_2}\right)\right]\left[\frac{Nut}{K_m + Nut}\right]$$

$$= G_T\, r_L\, r_n \tag{2-14}$$

$$r_n = 1.0 \tag{2-17}$$

(assuming no nutrient limitation)

$$G_{T=} (1.8\,day^{-1})\,(1.066)^{\,(23-20)} = 2.18\ day^{-1} \quad \text{(Figure 2-6)}$$

$$\alpha_1 = \frac{600\ ly}{300\ ly/day\,(0.5\,day)}\quad e^{-(0.33)(3)} = 1.47 \tag{2-15a}$$

$$\alpha_2 = \frac{600\ ly}{300\ ly/day\ \ (0.5\,day)} = 4.00 \tag{2-15b}$$

$$r_L = \frac{(2.718)(0.5)}{(0.33)(3.0)(1.0)}\,(e^{-1.47} - e^{-4.00}) = 0.287 \tag{2-15}$$

$$G_p = (2.18\ day^{-1})(0.287)\ (1)\ =\ 0.626\ day^{-1}$$

$$D_p = (0.1\,day^{-1})1.08^{(23-20)} = 0.126\ day^{-1}$$

Since $G_n = [G_p - D_p - V_s/H]$, the net settling loss rate can be estimated from

$$V_s = H\,(G_p - D_p - G_n) = 3.0\,(0.626 - 0.126 - 0.371)$$

$$V_s = 0.327\ ft/day$$

Summary of population dynamics rates:

Specific growth rate, $G_p = 0.626\ day^{-1}$
Respiration loss rate, $D_p = 0.126\ day^{-1}$
Algal settling loss rate, $V_s/H = 0.109\ day^{-1}$
Net algal growth rate, $G_n = 0.391\ day^{-1}$

- Check for nutrient limitation:

  Using a phosphorus-to-chlorophyll *a* ratio (a_P) of 1.0, the amount of inorganic phosphorus required to generate a *net* 40 μ*g/L chlorophyll a is*

$$P_o' = A_o'/a_P = \left(\frac{a_P\, G_p\, A_o}{G_n}\right)\Big/ a_P = 40\ \mu g/L$$

  The initial phosphorus concentration following complete mixing between the waste input and stream flow is

$$P_o = \frac{(20)(0.02) + (0.39)(5.0)}{20 + 0.39} = 0.115\ mg/L = 115\ \mu g/L$$

  By the end of the 20-mile reach, the inorganic phosphorus concentration would be equal to 115 - 40 or 75 μ*g/L, which is much higher than the maximum limiting concentration of 25* μ*g/L (see Table 4-3). Thus, the above analysis is appropriate.*

- Estimate algal population dynamics rate factors under future design conditions:

  Assume that the phytoplankton settling rate (V_s) and light extinction coefficient (K_e) will not change under future design conditions.

  Modify pertinent rate factors for design stream flow, temperature, and depth.

  Using design conditions summarized in Table 4-3 and the pertinent relationships defined earlier, the rate factors for algal growth dynamics become:

  Light limiting factor, $r_L = 0.236$
  Nutrient limiting factor, $r_n = 1.0$
  (initial assumption)
  Specific algal growth rate, $G_p = 0.585\ day^{-1}$
  Algal respiration rate, $D_p = 0.147\ day^{-1}$
  Algal settling rate, $V_s/H = 0.149\ day^{-1}$
  Net algal growth rate, $G_n = 0.289\ day^{-1}$

  The projected algal chlorophyll *a* concentration at $x = 20$ miles would be

$$25e^{(0.289 day^{-1})(20 mi/6.56\ mi/day)} = 60.3\ \mu g/L$$

  which would require the following amount of inorganic phosphorus to support it:

$$P_o' = \frac{(1.0)(0.585)(25)}{0.289} = 50.6\ \mu g/L$$

  Yet, the inorganic phosphorus concentration following complete mixing at $x = 0$ is only

$$P_o = \frac{(12.0)(0.02) + (0.47)(1.0)}{12.0 + 0.49} = 0.0568\ mg/L = 56.8\ \mu g/L$$

  Although this inorganic phosphorus concentration is slightly more than the amount required for algal growth, it is not sufficient to maintain a *no limitation* condition while approaching the end of the 20-mile stream. [Note that an inorganic phosphorus concentration of 25 μg/L (see Table 4-3) is required for a no limitation condition in the water column.] In other words, phosphorus limitation will occur in the stream prior to the end of the 20-mile reach. Because of this limitation, the chlorophyll *a* concentration at x = 20 miles, 60.3 μg/L as calculated earlier, will be the upper bound for the algal biomass. A lower bound can be estimated by first calculating the time it takes to reach a potential phosphorus limitation (i.e., inorganic phosphorus conc. = 25 µ*g/L):*

$$\frac{1}{G_n}\,\frac{50.6+56.8-25}{50.6} = 1.628 \ days$$

The lower bound of the chlorophyll a concentration is therefore

$$25\ e^{(G_n)\ (1.68)} = 41\ \mu g/L$$

Based on the above analysis, the maximum chlorophyll a concentration at x = 20 miles would be between 41 and 60 µg/L. The analysis also indicates that the short stream assumption is violated under future design conditions. More rigorous analyses (i.e., using a computer model) are required to address this issue.

4.2.3.3 Diel Dissolved Oxygen Variation Due to Algae

If only average daily dissolved oxygen concentrations are of concern in a TMDL, the above analysis may be used to determine the daily average net dissolved oxygen production due to algal photosynthesis and respiration. In cases where minimum daily standards are of concern, an estimate of the diel variation in dissolved oxygen must be made. A brief theoretical analysis is presented in the following paragraphs and followed by an example using the data from the previous example in Section 4.2.3.1.

Algal oxygen production as a function of time during the day can be approximated as (Di Toro, 1975; Chapra and Di Toro, 1991):

$$P(t) = P_M \sin(\pi\ t/f) \qquad 0 < t < fT \qquad (4\text{-}9)$$

$$P(t) = 0 \qquad f\,T < t < T$$

where

$P_{(t)}$ = algal gross photosynthetic production of oxygen (mg/L–day)
P_M = maximum rate of photosynthetic oxygen production (mg/L–day)
t = time (days)
T = period (day)
f = photoperiod (fraction of day)

To extend *P(t)* for more than one day, a Fourier series can be used:

$$P(t) = P_M \frac{2f}{\pi T} + \sum_{n=1}^{\infty} b_n \cos\left[\frac{2\pi n}{T}(t - f/2)\right] \qquad (4\text{-}10)$$

where

n = number of days

$$b_n = \cos(\pi n f/T)\,\frac{4\pi T/f}{(\pi T/f)^2 - (2\pi n)^2}$$

and the average daily value of this function is equated to the average daily algal oxygen production calculated as:

$$P_{av} = P_M \frac{2f}{\pi T}$$

where

P_{av} = average daily rate of photosynthetic oxygen production (mg/L–day).

Assuming that phytoplankton oxygen production can be represented with the diel Fourier Series function given above and that algal respiration and sediment oxygen demand are constant, the periodic steady-state solution to the differential equation is given as (O'Connor and Di Toro, 1970; Di Toro, 1975):

$$C(t) = C_s + [(P_{av} - R)/K_a] - [S_b/(K_a H)] + P_m\left[\sum_{n=1}^{\infty} \frac{b_n}{[K_a^2 + (2\pi N/T)^2]^{1/2}} \cos\left[\frac{2\pi n}{T}(t - f/2) - \tan^{-1}\frac{2\pi n}{K_a T}\right]\right] \qquad (4\text{-}11)$$

where

$C(t)$ = time varying oxygen level (mg/L)
C_s = oxygen saturation value (mg/L)
P_{av} = daily average algal photosynthesis (mg O_2/L-day)
R = algal respiration (mg O_2/L-day)
K_a = atmospheric reaeration coefficient (day^{-1})
S_b = sediment oxygen demand (g O_2/m^2-day)
H = depth of the water column (m)
P_m = maximum algal photosynthetic production (mg O_2/L-day)
T = diel period = 1.0 day
t = time during day (fraction of day)
f = photoperiod (fraction of day)
b_n = periodic coefficient (n=1, n=2)

Using unique properties of the solution of the periodic equation, Di Toro (1975) and Chapra and Di Toro (1991) present an analytical expression (delta method) to estimate the diel range of oxygen attributed to algal photosynthesis as follows:

$$\frac{(C_{max} - C_{min})}{P_{av}} = \frac{[1 - e^{-K_a f T}]\ [1 - e^{-K_a T (1-f)}]}{f K_a [1 - e^{-K_a T}]} \tag{4-12}$$

where C_{max} and C_{min} represent the maximum and minimum daily (24 hr) oxygen levels.

Thomann and Mueller (1987) and Chapra and Di Toro (1991) present complete documentation of derivation of the diel oxygen production model and Di Toro's (1975) delta method of determining the diel range of oxygen from algal photosynthesis.

In shallow streams and rivers, attached epiphytic algae and benthic macrophytes can account for significant components of observed primary production and oxygen and nutrient distributions (Jeppesen and Thyssen, 1984). In particular, steep gradient reaches of rivers with high current velocity (ca. 50 cm/s) and sufficient nutrient supply are typically characterized by maximum rates of benthic primary productivity from periphyton (Hynes, 1970; Horner and Welch, 1981; Welch et al. 1989). Consistent with other studies reported in the literature, stream velocity increases of up to ~50 cm/s have been observed to result in enhanced biomass accumulation and productivity of attached periphyton (Horner and Welch, 1981). Velocities higher than ~50 cm/s tend to result in reduced biomass accumulation because of physical scouring and removal of attached biomass.

Since data to describe benthic biomass or benthic primary productivity in a river are typically not available, literature values can be used to estimate parameter values for gross benthic algae production and production/respiration ratios (P/R) (e.g., Bott et al., 1985). From these literature values, gross benthic algae productivity appears to be on the order of 0.5 to 5.0 g C/m^2-day. Based on photosynthetic efficiency and assuming that 1 mole O_2 = 112 kcal, gross benthic production is on the order of 0.5 to 5.0 percent of total incoming solar radiation (Thomann and Mueller, 1987).

In a comparative long-term seasonal study of four rivers across the United States (Oregon, Michigan, Pennsylvania, and Idaho), Bott et al. (1985) reported summer benthic algae productivity rates of 0.25 to 2.5 g C/m^2-day. Bott et al. (1985) also reported a range of values of P/R ratios consistent with the "River Continuum Concept" (Williams, 1981) where transitions in community metabolism (i.e., P/R) tend to occur over the domain of a river as small streams develop into larger rivers over a drainage basin. In general, the data of Bott et al. (1985) tend to support this hypothesis, as summarized below:

- Upper reach of river: predominant heterotrophy (P/R <1)
- Middle reach of river: predominant autotrophy (P/R > 1)
- Lower reach of river: predominant heterotrophy (P/R <1)

In shallow systems, accurate representation of the observed diel oxygen range might require a daily average and diel benthic algae component in addition to the phytoplankton component.

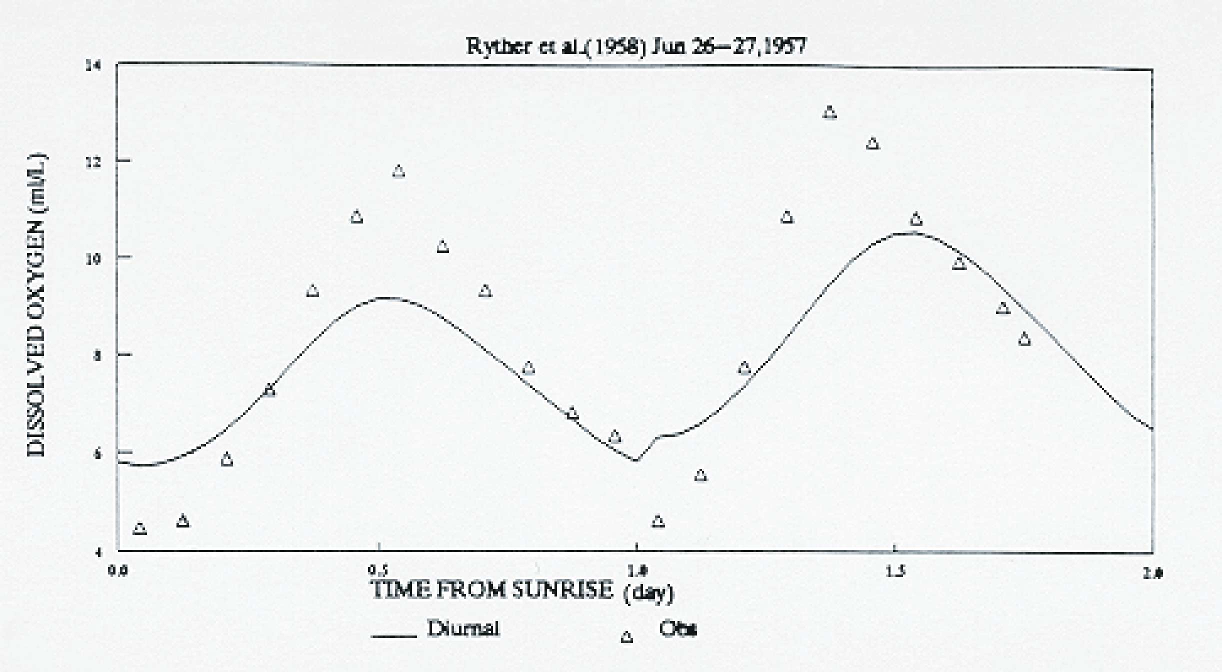
In order to account for benthic algae, or macrophytes, Equation 4-11 can be easily modified to include additional daily average and diel terms. This approach has been used with hydrographic data for Flanders Bay and Senix Creek (Tetra Tech, 1989; Morton et al. 1990), shallow estuarine ecosystems in eastern Long Island . Data from a 48-hour time series in Senix Creek (Ryther et al., 1958) are presented for comparison to the diel model results assuming (a) phytoplankton production only and (b) phytoplankton and benthic macrophyte production (Figure 4-4).

Using the hydrographic and nutrient data obtained for Senix Creek, the computed average phytoplankton primary production rate of 5.3 g C/m^2-day falls within the observed range of 3.2-6.2 g C/m^2-day. If phytoplankton were the dominant primary producer, then the diel analysis should have adequately reproduced the observed 48-hour time series of oxygen data where sediment oxygen demand was assumed constant at 4.1g O_2/m^2-day at the ambient temperature of 25 °C. Like the diel oxygen model results reported for the Shenandoah River (Deb and Bowers, 1983), the computed amplitude and phase of the phytoplankton diel oxygen model do not adequately reproduce the observed data (Figure 4-4(a)). Incorporation of an average benthic macrophyte production (4 g C/m^2-day) and respiration (3.75 g C/m^2-day), however, results in much better agreement with the observations from Senix Creek (Figure 4-4(b)).

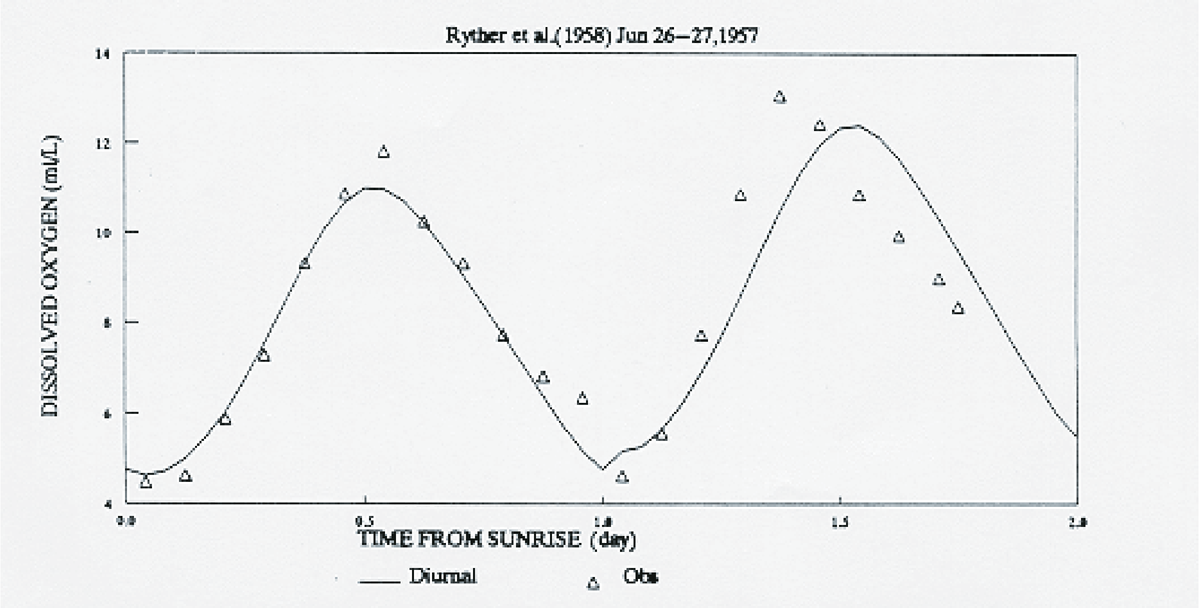
The results of the diel analysis clearly demonstrate that benthic photosynthetic oxygen production can be a significant factor in the observed diel variability of oxygen in shallow systems such as Senix Creek. It is also likely that benthic macrophytes would account for a significant component of total nutrient uptake in the water column.

4.2.4 Selection of Modeling Framework

Obtaining a simulation model that effectively implements the conceptual model is important. If availab



(a) ONLY ALGAL PRODUCTION INCLUDED IN MODEL (BENTHIC MACROPHYTE PRODUCTION AND RESPIRATION EQUAL TO ZERO)



(b) BOTH ALGAL AND MACROPHYTE PRODUCTION INCLUDED IN MODEL (BENTHIC MACROPHYTE PRODUCTION AND RESPIRATION GREATER THAN ZERO)

FIGURE 4-4. DIEL MODEL VS. OBSERVED OXYGEN IN SENIX CREEK, LONG ISLAND (After Tetra Tech, 1989) models do not fully implement a specific conceptual model, the analyst may:

- Refine the model code.
- Make calculations or assumptions external to the model code.
- Explore consequences with model sensitivity analyses.

It should be pointed out that how a modeling framework is used is typically more important to a TMDL modeling study than exactly which model is used. Selection of an appropriate modeling framework increases the probability of accurate results.

The nature of the problem and, specifically, the time and space scales of the problem dictate the simplicity or complexity of the modeling analysis. Given or assuming these scales, a specific question is posed, and the purpose of the modeling analysis is to answer this question in the simplest, most efficient, and most realistic manner. For example, if the dissolved oxygen depression in the stream is primarily due to point source BOD discharges and no significant algae have been detected in the study area, a simple model of BOD/DO without eutrophication is sufficient for the analysis. By contrast, if the dissolved oxygen problem is caused by decomposition of algal biomass following nutrient inputs, the next level of analysis, incorporating algal-nutrient dynamics, should be considered.

Furthermore, many stream water quality problems can be and have been answered by a steady-state analysis with linear kinetics and simple transport (one-dimensional) components. The simple dissolved oxygen models of streams are typical examples. Such problems may be approximated with sufficient accuracy to yield an analysis that is adequate for making decisions regarding the treatment level of wastewater. On the other hand, at this stage of development, an analysis of the eutrophication problem usually requires time-variable and nonlinear terms in order to determine the effect of nutrient removal from wastewaters.

The relative complexity of the model is an important factor in TMDL studies—the more complex, the greater the degree of model validation required. The complexity of the model is determined by the number of transport, kinetic, and input terms in the model equations. Compare the simplicity of the equations used to describe the steady-state distribution of dissolved oxygen in streams to the complexity of those describing the time-variable distribution of nutrients and phytoplankton in streams. For each additional component included in the analysis, an additional degree of validation is necessary. As a consequence, additional data are required. If data are not available on a specific component, it is questionable whether the component should be included in the analysis.

Most of the stream water quality models for TMDLs can be run on personal computers. Therefore, it is essential that the modeling framework selected be user-friendly. Technical support to operate the model is also crucial. Finally, graphic display of the model results can significantly increase the productivity of the TMDL study. With the rapid advancement of microcomputer hardware and software, user-friendly features should be considered in model selection.

4.3 SITE-SPECIFIC STREAM SURVEY

Following the initial assessment, including review of existing data, preliminary analyses, if any, and the selection of a modeling framework, a field survey may be conducted to fill any data gaps. The additional data are key to the model calibration and validation for the TMDL study. In fact, the initial assessment should determine:

- What pollution sources will be monitored?
- What is the extent of water quality data to be collected?

In general, a stream survey includes three basic components:

- Measuring stream physical parameters such as hydraulic geometry, velocity, flow, and time of travel.
- Receiving water quality (physical, chemical, and biological) data collection.

The following special studies may be conducted, as needed, if the budget and time schedule allow:

- Time-of-travel and dye dispersion studies.
- Measurement of reaeration coefficients.
- Light and dark bottle tests.
- Diel oxygen measurements.

- Nitrifying bacteria counts in the water column and sediments.
- Long-term BOD tests of wastewater and receiving water.
- Field measurements of sediment oxygen demand and nutrient fluxes.

It is extremely important that these components be synchronized to form a synoptic survey, making the data most useful for the water quality modeling analysis. The handbook on stream sampling for wasteload allocation applications by Mills et al. (1986) provides complete information for designing a stream survey program for point sources. A complete QA program for the stream survey should be developed in advance of any sampling activities to ensure documented evidence that a data product of known and acceptable quality is produced. Appendix C presents the basic elements of a quality assurance program for field monitoring programs.

4.3.1 Hydraulic Geometry Survey

Physical stream data include stream cross-sectional area, average stream depth, stream flow, and average stream slope. These data must be collected at each of the sampling stations selected. If the river is constricted or not reflective of the natural channel at these stations, then area and depth data should be measured slightly upstream of the location or where the channel is reflective of natural conditions.

About 5 percent accuracy should be required for cross-sectional area data, while 10 percent accuracy should be required for flow measurements. In addition, some self-checking procedure should be established so that any vandalism or change in staff gage elevation can be determined and corrected.

4.3.2 Time-of-Travel Study

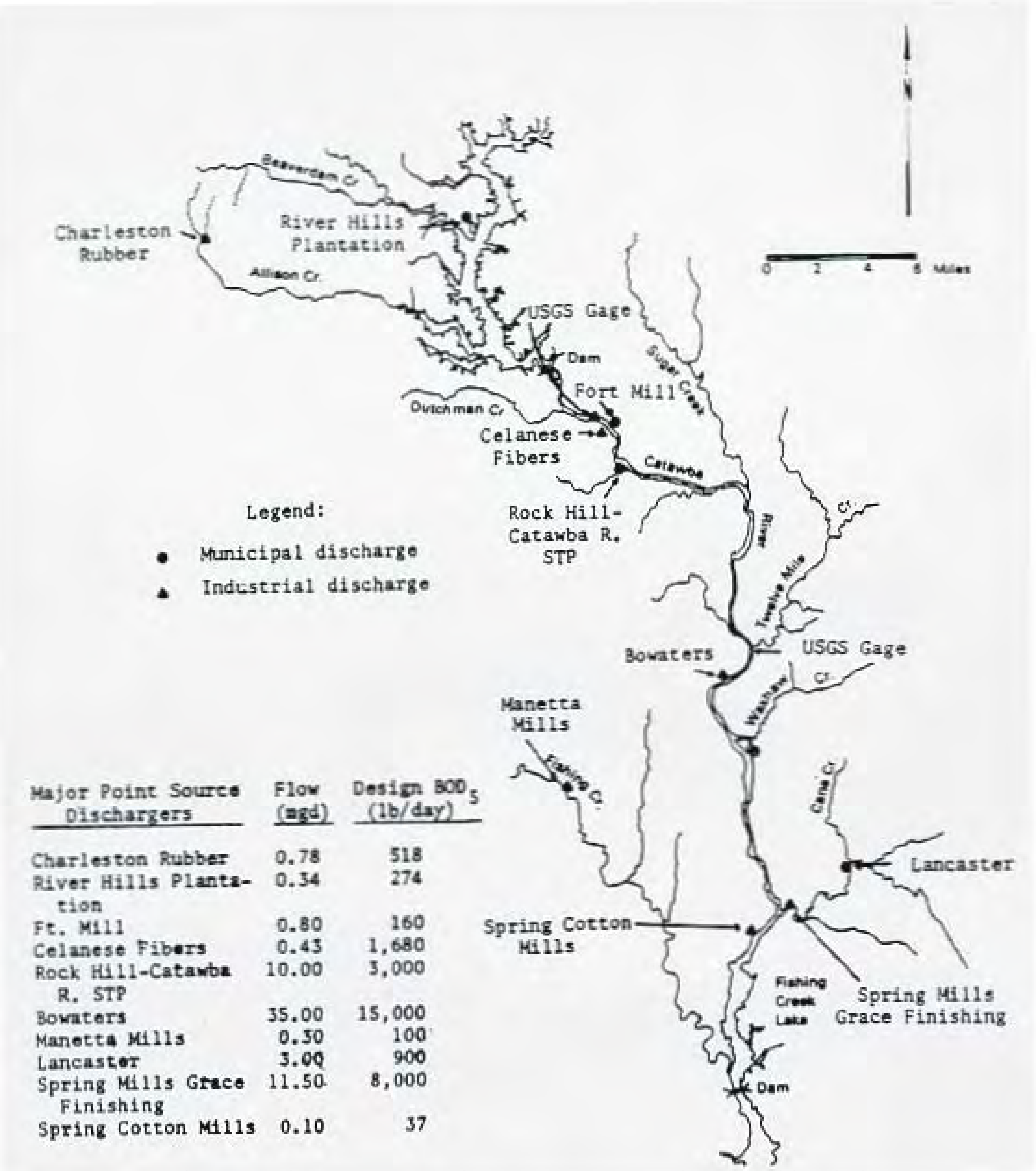
Time-of-travel data are useful to define the time of passage between various sampling stations and to determine the magnitude of longitudinal dispersion in the stream. Time-of-travel studies should be conducted under different flow conditions (see Figure A-4). These flow conditions could be low flow or dry weather flow, and high flow. Each time-of-travel study should take place during a constant flow period of a few days. During the survey, fluorescent dye is released at each selected location on the river and fluorescence is measured at the next downstream station. Stations should be selected to take into account any features in the river basin that might change the time of travel such as present and future wastewater inputs, tributary inputs, and changes in channel characteristics. The dye is released instantaneously at the upstream station and collected over time at the downstream station. Dye samples can be collected by an automatic sampler for fluoresence measurement. The sampling interval from start of sampling to end of sampling should be determined before the dye is released, based on the river flow and the length of the stream reach. Although most dye studies are conducted using a single instantaneous dye dump, continuous dye injection can also be used. When conducting a time-of-travel study for multiple reaches in a river, the analyst must always start with the most downstream reach and proceed in the upstream direction to avoid any influence from upstream dye releases.

Rhodamine WT dye is normally used in stream surveys. This dye should be diluted, one to one, with methyl alcohol to bring the solution to a specific gravity of approximately 1.0. The mass of dye released should be recorded, as well as the instantaneous flow at the upstream and downstream stations. After dye samples have been collected, they should be analyzed for percent transmittance. In addition, a calibration curve should be presented, showing the percent transmittance vs. dye concentration (Wilson et al., 1986).

4.3.3 Stream Water Quality Sampling

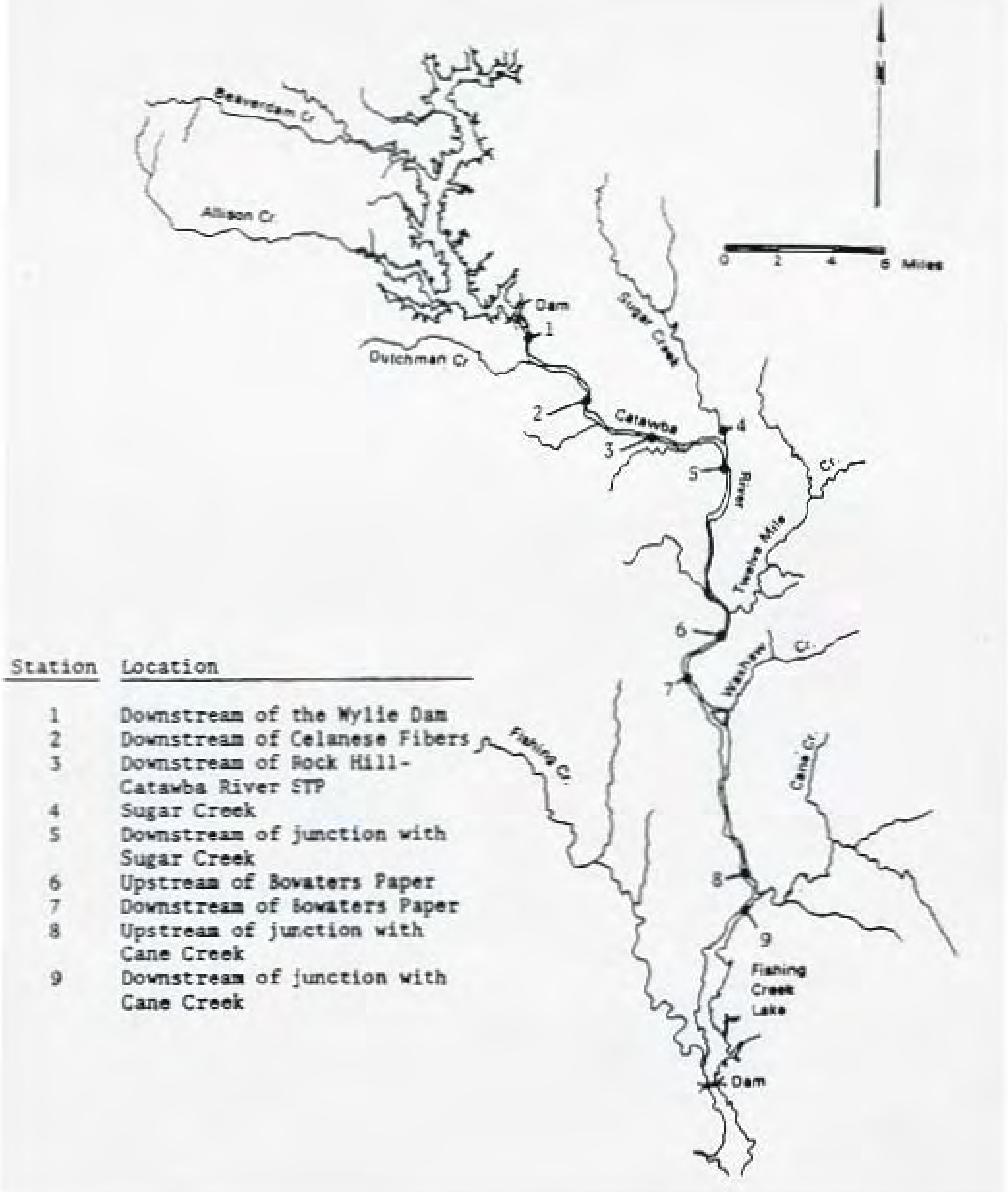
Water quality sampling should be conducted to assess the point and nonpoint source impacts on BOD/DO levels in a stream or river. When monitoring water quality, all samples should be checked for residual chlorine. If substantial levels are found, then BOD samples must be dechlorinated and reseeded. In addition, when planning a stream water quality survey, three key items should be determined: (1) sampling locations, (2) sampling time and frequency, and (3) sampling protocols.

A survey program designed for the Catawba River, South Carolina, illustrates the scope of a stream survey to support a BOD/DO TMDL study (Lung, 1990). Figure 4-5 shows the study area and major point sources along the Catawba River. Figure 4-6 presents the sampling network along the river. Note that the water quality sampling stations were selected to reflect the major point source discharges and nonpoint source tributary inputs. The receiving water quality parameters sampled are shown in Table 4-4, including the sampling frequency. The number and locations of the stations could vary slightly depending



Major Point Source Dischargers	Flow (mgd)	Design BOD_5 (lb/day)
Charleston Rubber	0.78	518
River Hills Plantation	0.34	274
Ft. Mill	0.80	160
Celanese Fibers	0.43	1,680
Rock Hill-Catawba R. STP	10.00	3,000
Bowaters	35.00	15,000
Manetta Mills	0.30	100
Lancaster	3.00	900
Spring Mills Grace Finishing	11.50	8,000
Spring Cotton Mills	0.10	37

FIGURE 4-5. CATAWBA RIVER STUDY AREA AND MAJOR POINT SOURCES (Lung, 1990)



Station	Location
1	Downstream of the Wylie Dam
2	Downstream of Celanese Fibers
3	Downstream of Rock Hill-Catawba River STP
4	Sugar Creek
5	Downstream of junction with Sugar Creek
6	Upstream of Bowaters Paper
7	Downstream of Bowaters Paper
8	Upstream of junction with Cane Creek
9	Downstream of junction with Cane Creek

FIGURE 4-6. PRELIMINARY WATER QUALITY SAMPLING NETWORK (Lung, 1990)

TABLE 4-4. WATER QUALITY SURVEY FOR THE CATAWBA RIVER

Parameter	Stations	Sampling Frequency
Temperature[a]	All	Twice a day for 2 days
pH[a]	All	Twice a day for 2 days
Dissolved Oxygen[a]	All	Twice a day for 2 days
Specific Conductivity[a]	All	Twice a day for 2 days
Diurnal Dissolved Oxygen[b]	3, 5, 7, 9	Grab samples every 2 hr for 24 hr
$CBOD_u$	3, 5, 7, 9	Grab samples[c]
$TBOD_u$[d]	3, 5, 7, 9	Grab samples[c]
$CBOD_5$	All	Twice a day for 2 days
Total Kjeldahl Nitrogen	All	Twice a day for 2 days
Ammonia Nitrogen	All	Twice a day for 2 days
Nitrite and Nitrate	All	Twice a day for 2 days
Total Suspended Solids	All	Twice a day for 2 days

---

[a] Using field sampling unit.
[b] Light and dark bottle rig was used.
[c] Both total BOD and nitrification inhibited BOD for days 1, 2, 3, 5, 7, 10, 15, 20, 30, 40, and 50.
[d] Total BOD.

TABLE 4-5. POINT SOURCE SAMPLING PROGRAM

Parameter	Sources	Sampling Frequency
Temperature	All	Instantaneous
pH	All	Instantaneous
Flow	All	24-hour composites taken 4 times during the survey
Dissolved Oxygen	All	Same as above
$CBOD_5$	All	Same as above
Total Kjeldahl Nitrogen	All	Same as above
Ammonia Nitrogen	All	Same as above
Nitrite and Nitrate	All	Same as above
Total Suspended Solids	All	Same as above
$CBOD_u$	All	Grab samples[a]
$TBOD_u$	All	Grab samples[a]

---

[a] Both total BOD and nitrification inhibited BOD for days 1, 2, 3, 5, 7, 10, 15, 20, 30, 40, and 50.

on the actual field conditions. A similar summary is shown in Table 4-5 for point source sampling. The survey lasted for approximately 4 days to cover the entire length of time required for dissolved substances to travel the designated section of the river. The sampling of the upstream stations began 2 days prior to the stream sampling to establish accurate boundary conditions for the modeling analysis.

4.3.4 Wastewater Monitoring

Different types of point sources require an analysis of wastewater characteristics to accurately determine the ultimate dissolved oxygen demand. The most important factor for this determination is the CBODu-to-CBOD5 ratio. In the Catawba River, a number of municipal and industrial point sources were sampled during dry weather conditions. Table 4-5 lists the water quality parameters sampled and the sampling frequency.

Because of insufficient CBODu and CBOD5 data, conservative values (high ratio) were used during the study to avoid criticism. An overly conservative CBODu-to-CBOD5 ratio would result in wasteload allocations more stringent than necessary. Therefore, if industries want to have adequate allocations, sufficient data are required to justify a lower CBODu-to-CBOD5 ratio. A long-term BOD test of the effluents and the stream samples is needed. Long-term BOD tests usually take about 90 to 100 days and are very straightforward, but time-consuming.

4.3.5 Biological Assessment^{t}

Several methods exist for evaluating the biological attributes of a stream system (USEPA, 1993a). Habitat Evaluation Procedures (HEPs) are used to document the quality and quantity of available habitat by providing information for comparing in-stream and riparian habitat in different areas or in one area under different conditions. Rapid Bioassessment Protocols (RBPs) for habitat are screening tools for determining whether a stream is supporting a designated aquatic life use. One component of these protocols is an in-stream habitat assessment procedure that measures physical characteristics of the stream reach. RBP III, an RBP for benthic macroinvertebrates, focuses on quantitative sampling in riffle/run habitat or on other submerged, fixed structures where riffles may not be available. The data collected are used to calculate various metrics pertaining to benthic community structure, community balance, and functional feeding groups. The Index of Biological Integrity (IBI) has been used in many States to assess a wide range of impacts in streams and rivers. The IBI includes 12 matrices in three major categories of fish assemblage attributes: species composition, trophic composition, and fish abundance and condition. Any of these methods can be useful in determining the effects of pollutant loadings on biological communities in streams and rivers.

4.4 MODEL CALIBRATION

Model calibration is the first stage of testing and tuning a model to a set of field data, preferably a set of field data not used in the original model construction (Thomann and Mueller, 1987). Given the external parameters of a modeled stream system, an initial estimate is made of the appropriate transport and reaction rate coefficients in the model. These coefficients may be determined from a fundamental analysis relating to each specific coefficient (i.e., hydrologic or hydraulic analyses). The coefficients may also be determined from a statistical analysis, as is usually done with biological and chemical kinetic terms. In any case, if a range of these values is known, a best estimate is made for each, the model is run, and the output is compared to the data. Successive iterations and adjustments are required to obtain a reasonable fit of the model and data. This procedure is known as model calibration.

4.4.1 Model Coefficient Assignment

Model calibration is also part of the process of determining model coefficients. A simple example is the derivation of the stream deoxygenation coefficient, Kd, using the measured CBOD5 or ultimate CBOD data (e.g., CBOD20).

In many stream BOD/DO modeling analyses, sediment oxygen demand was not included in the models. To calibrate the model to actual in-stream dissolved oxygen data, the effect of SOD was incorporated into other modeling rates such as Kd, Kn, and Ka. In cases where Ka was determined using a reaeration formula and was not adjusted, the oxygen demand from SOD could be incorporated into the Kd or Kn rates. This approach would have the effect of overestimating the dissolved oxygen impact from removal of nitrogenous or carbonaceous BOD. Subsequently, this derived value must be incorporated into the site-specific model to check the model-computed CBOD5 and dissolved oxygen profiles against the field data.

Incorrect calibration of models could also arise from wrong steps in model calibration. For example, Kd can be adjusted until the calculated dissolved oxygen matches the measured data, rather than adjusting Kd to correspond to the CBODu data. Sediment oxygen demand, not considered in the model, was measured at a certain value. In this case, the effect of SOD could inadvertently be included in the Kd rate in order to match the model output with the observed data. Thus, the Kd rate used in the model could be substantially higher than the Kd estimated from the CBOD data. Since the Kd used in the model exceeds its likely value, the dissolved oxygen increase resulting from CBOD removal would be overestimated.

In model calibration analyses, adjustments of model coefficients should not exceed a predetermined range for each individual rate constant. For example, if CBOD5 data show that Kd could range from 0.25 to 0.30 day-1 (depending on how the slope of the CBOD decay curve was drawn), Kd should not be adjusted beyond this range. Another approach in calibrating a model is to set all rates other than Ka equal to their most reasonable value based on available data, and then to vary Ka under various flow regimes within the range indicated by the applicable reaeration formulas. If adjustments within this range of Ka do not produce a good match with the data, then the other rate constants may be adjusted further within their range of uncertainty. If these adjustments still do not produce a good match, the analyst should reevaluate available data, the reaeration formulas used, and the receiving stream itself to identify factors that may be preventing model results from corresponding with actual in-stream data. It should be noted, however, that if Ka is calculated or measured using site-specific data, it is not advisable to vary Ka values and other estimated coefficients such as SOD should be adjusted.

A more difficult case is the assignment of the model coefficients involving eutrophication. As indicated in Section 2, many kinetic processes related to phytoplankton growth and nutrient recycling in the water column are difficult, if not impossible, to obtain independently because of cost or time constraints. The practical approach of assigning them in a TMDL analysis is to rely on model calibration and sensitivity analyses. That is, model coefficient values are selected from literature values, preferably from previous studies at the specific site location, or from water bodies with similar problem settings. Subsequent model runs are performed to fine tune these model coefficients by matching the field data. Although a number of model coefficient values are derived from literature data, independent estimates of the exogenous variables such as streamflows, time of travel, boundary conditions, and environmental conditions should still be derived from the field data to minimize the degree of tuning.

When field measurements are available, model coefficients can be determined using curve-fitting procedures. However, it should be pointed out that model calibration is not a curve-fitting exercise. The model coefficients (e.g., algal growth rate) are also adjusted through a series of model runs with reasonable and narrow ranges of their values derived from the literature. The model is designed to mimic the steady-state algal growth and nutrient dynamics and should be shown to accomplish the task of reproducing the algal biomass and nutrient concentrations in the stream. In model sensitivity analyses, adjusting the kinetic coefficients and constants (within their predetermined ranges) to improve the calibration of certain water quality constituents often results in adverse outcomes for other water quality constituents. These are the constraints in the model calibration process that would eventually lead to the determination of a unique set of credible model coefficients.

4.4.2 Component Analyses

In the steady-state stream BOD/DO modeling analysis, the amount of dissolved oxygen deficit produced by each of the oxygen-demanding components can be calculated. The dissolved oxygen deficit is calculated for each source of deficit and then is plotted. Figure 4-7 shows the component analysis results from the modeling analysis of Rock Creek, Pennsylvania, for the August 1979 conditions (Lung, 1990). The results suggest that the Gettysburg wastewater treatment plant and the sediment oxygen demand contributed the most oxygen deficit in the stream. All other sources had a much smaller effect on the dissolved oxygen concentrations. This type of analysis is important in both the model calibration/validation and wasteload allocation analyses. The component analysis gives the analyst a graphical presentation of the cause-and-effect relationship for in-stream water quality.

While component analyses are routinely performed to quantify the contribution of individual sources to dissolved oxygen deficits, a similar component analysis is not appropriate for eutrophication modeling analysis because of the nonlinear nature of the phytoplankton growth-nutrient dynamics in the model. That is, results from a component analysis would not predict algal biomass accurately in terms of the various

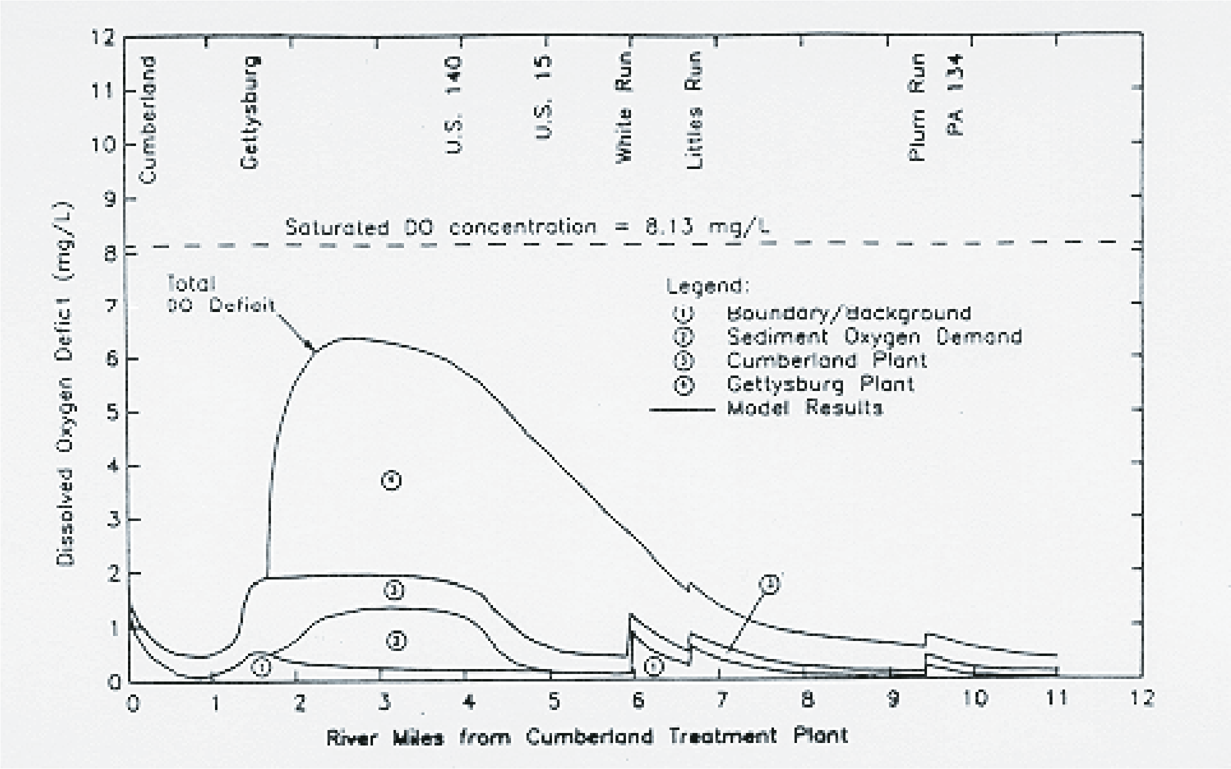


FIGURE 4-7. COMPONENT ANALYSIS OF DO FOR ROCK CREEK, PENNSYLVANIA (Lung, 1990)

sources of phosphorus without taking into consideration other factors that also control algal production. Lung and Testerman (1989) have demonstrated a technique called numerical tagging to determine how much phosphorus in the algal biomass at a certain location in the James River, Virginia, is from a particular wastewater source. The technique is similar to using a radioactive tracer in limnological studies to track the fate and transport of phosphorus in systems. Instead of using a radioactive tracer such as 32PO4, Lung and Testerman used a numerical tracer injected at the particular wastewater source studied. That is, that source of phosphorus was numerically labeled and added to the river. Figure 4-8 shows the numerical tagging results for the James River. The results show that POTWs in the James River are major contributors of orthophosphate as well as algal biomass. One interesting observation is that while the Richmond POTW contributes more than 80 percent of the orthophosphate to the river, its contribution to the biomass is only about 50 percent (a disproportionate share compared with its phosphorus input). Such a result further emphasizes the nonlinear nature of eutrophication models.

4.4.3 Quantifying the Comparison Between Model Results and Data

Research activities (Thomann, 1982, 1987) in modeling eutrophication in lakes have begun to explore the use of simple statistical comparisons in an attempt to quantify model adequacy. These techniques could be a supplement to the qualitative comparisons of observed and calculated water quality profiles. Three techniques that have been used are:

- Comparison of means
- Regression analysis
- Relative error

In the first technique, the mean of the observed data is compared to the mean of the computed profile for the comparable conditions of loading, transport, and temperature. The Student's t-probability density function is employed for the comparison of the means.

In regression analysis, calculated concentrations and observed data are considered as paired points in the test equation:

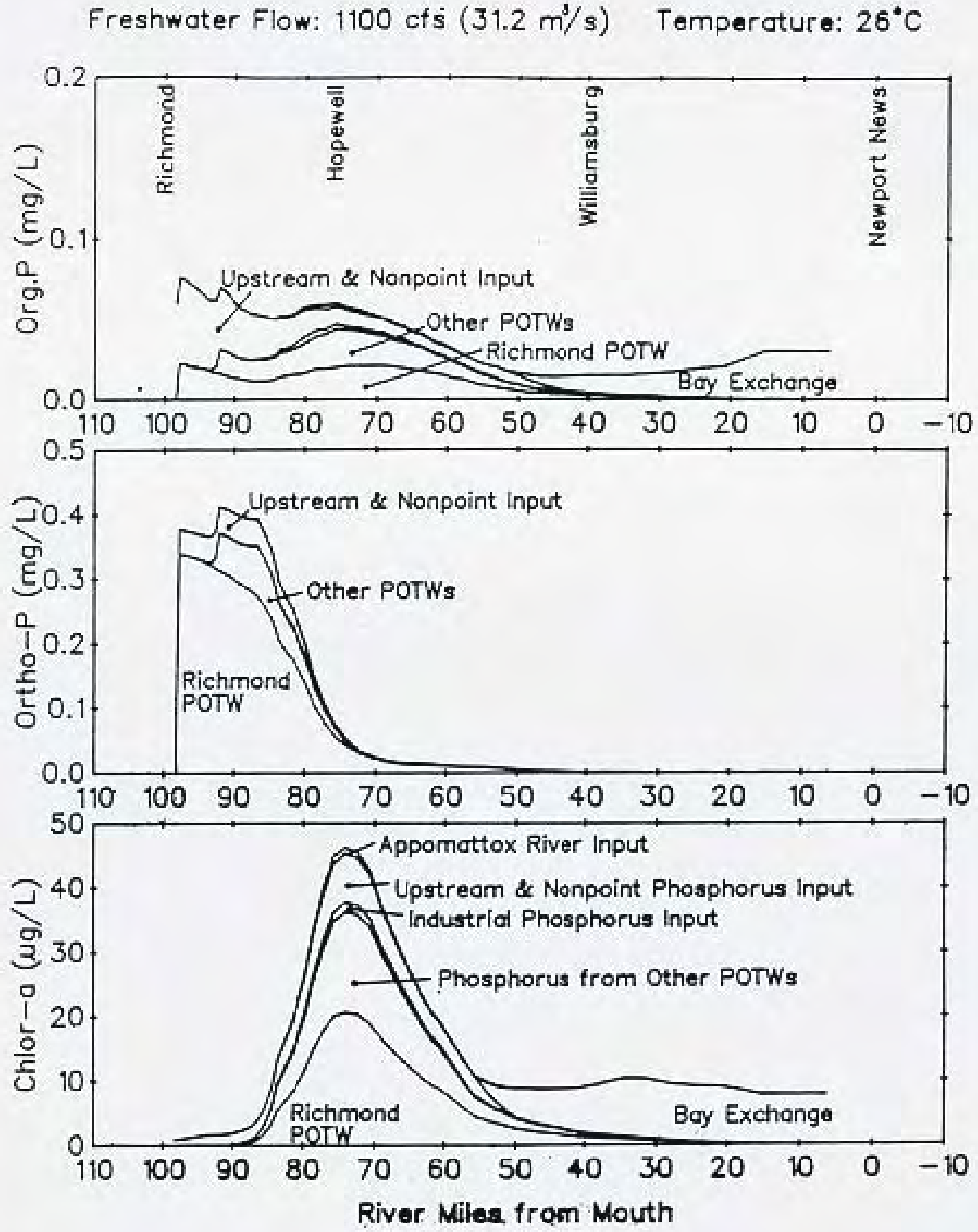


FIGURE 4-8. NUMERICAL TAGGING OF JAMES RIVER
(After Lung and Testerman, 1989)

$$X = \alpha + \beta C + E$$

where α and β are the true intercept and slope, respectively, between the calculated value C and the observed data X. E is the error associated with the observed data X. The regression analysis assumes that the calculated value C is known with certainty and that the error E is in the measured data, which is not necessarily a realistic assumption. Standard linear regression methods can be used to compute the square of the correlation coefficient (r^2) and the standard error of estimate representing the residual error between data and model. Estimates of the slope and intercept are calculated and a test of significance developed. Well-calibrated models would have a zero intercept and a slope of one.

Relative error is the absolute value of the difference between the observed and the calculated values divided by the observed value. The relative error may be aggregated across time or space, and the cumulative frequency of error can be computed. Estimates can be made of the median relative error as well as the 10 percent and 90 percent frequency of error. This statistic is poorly behaved at the upper tail and at low values of X. The median error can be easily understood; therefore, it is the suggested measure if statistical representations of model adequacy are to be employed in a TMDL study.

Statistical measures of adequacy are in the early stages of research and should be employed recognizing that they provide, at the very best, a lower bound on the magnitude of the error.

4.5 MODEL VALIDATION

4.5.1 Model Coefficient Adjustment

Once a set of coefficients representative of one set of external conditions (e.g., with respect to temperature, flow, and loading inputs) has been established, the model is rerun for a different set of input conditions. If the model output agrees in a reasonable, qualitative way with the second set of data, the model is considered to be validated in the first degree.

In some instances, if sufficient data are available, a quantitative comparison is possible. Then a measure such as the standard error of the mean can be used, and the model is considered validated if the model results fall within one standard error. Additional comparisons with different combinations of the exogenous parameters yield higher degrees of validation.

In some cases, a model coefficient value may have to be varied slightly to match the data in the validation process. Then the changed value must be tested again with the data set for the model calibration run. A good example of such an exercise is the calibration of the nitrification rate, K_n. When data from a winter survey are first used to calibrate a model, the dissolved oxygen balance is not sensitive to the nitrification rate and, therefore, K_n cannot be determined accurately. During model validation, another set of data collected in the summer months is used. Since the nitrification process is highly sensitive to temperature, the modeling analysis is able to tune the nitrification rate with greater accuracy. Now, this modified K_n should be checked again with the winter run. Because of the cold temperature in the winter months, the model results are not affected. This procedure is also valid for model calibration and validation analyses between two drastically different stream flow conditions. Under high flows, some kinetic coefficients may not be important and, therefore, cannot be accurately calibrated until the low-flow conditions, when the kinetics become much more significant in relation to stream flow.

4.5.2 Model Sensitivity Analysis

When validating or calibrating a mathematical water quality model, the analyst selectively determines some model input parameters that, when used in the model, yield reasonable simulations of observed water quality data. Some of these input parameters, such as stream geometry, cross-sectional areas, and depths, are directly measured. Other model parameters, such as system transport, oxidation rates, reaeration rates, and nitrification rates, are not directly measured. These parameters are determined from empirical formulations, literature searches, or iterative model simulations. The purpose of a sensitivity analysis is to test the sensitivity of the model to some of these input parameters. Some of the common model sensitivity analyses in a stream TMDL include:

- Sensitivity of model to transport coefficients.
- Sensitivity of model to K_d rates.
- Sensitivity of model to K_a rates.
- Sensitivity of model to K_n rates.
- Sensitivity of model to the net algal oxygen production rate *(P-R)*.
- Sensitivity to SOD

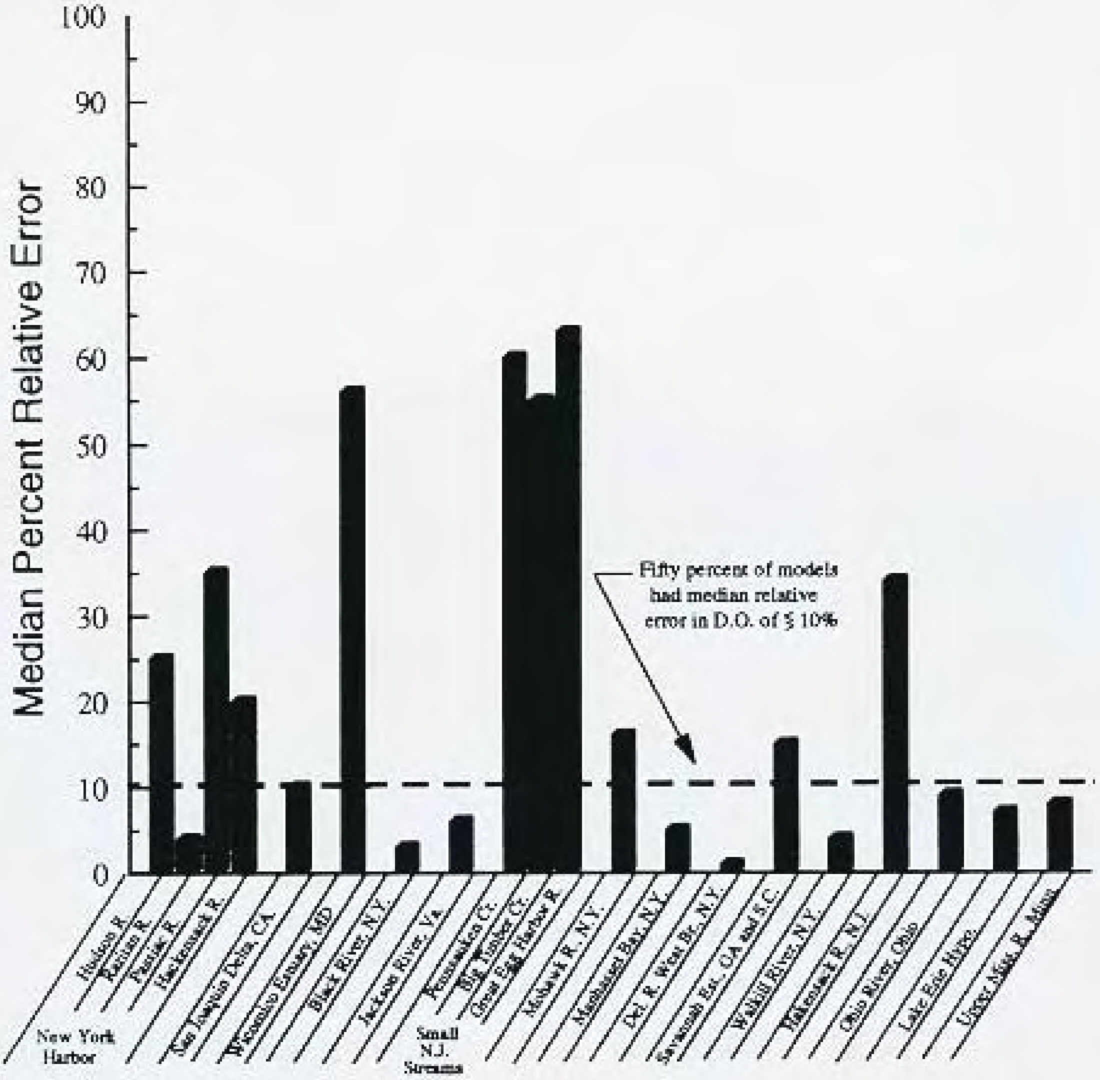


FIGURE 4-9. SOME RELATIVE ERRORS OF DISSOLVED OXYGEN MODELS
(Thomann, 1980)

4.5.3 Model Accuracy

The question of model accuracy is often crucial in situations where a given allocation is being negotiated or contested. Thomann (1980) has discussed this question and compiled a distribution of relative errors between model calibration results and the observed data. Figure 4-9 displays the median relative error in measured versus simulated (modeled) dissolved oxygen for waterbodies of varying complexity. The models represented in Figure 4-9 generally represent state-of-the-art models, applied by experienced practitioners using best judgment on loads, parameters, and model structure. That is, the calibrations were conducted based on defensible theoretical assumptions rather than simply an attempt to match the measured dissolved oxygen values by arbitrarily adjusting model coefficients. With this consideration in mind, Figure 4-9 indicates that for the 20 models represented, 50 percent had a median relative error in dissolved oxygen of 10 percent (plus or minus), with maximum errors of up to 60 percent occurring in the smaller streams/estuaries simulated. This comparison is useful in suggesting the present ability to reproduce the observed data with a credible model.

4.6 MODEL APPLICATION AND TOTAL MAXIMUM DAILY LOADS

An integral part of the TMDL process is the analysis of cause-effect relationships via a mathematical model of loading input and resulting water quality response. The TMDL rests heavily on the credibility and predictive capability of the mathematical modeling framework (Thomann and Mueller, 1987). However, the adequacy of the modeling framework is only one of many issues that must be considered in a TMDL process (Chadderton and Kropp, 1985). To develop an actual TMDL, a number of tasks need to be conducted. The following sections provide a brief description of these tasks.

4.6.1 Development of Management Scenarios

In many cases, management alternatives can be evaluated by using model applications, particularly in river systems that receive loadings from multiple sources. Usually a regional or State planning agency is responsible for soliciting input from dischargers, the public, and other interested parties to determine the most feasible management alternatives. In all cases, depending on scenario, all point and nonpoint sources should be considered when developing allocation scenarios.

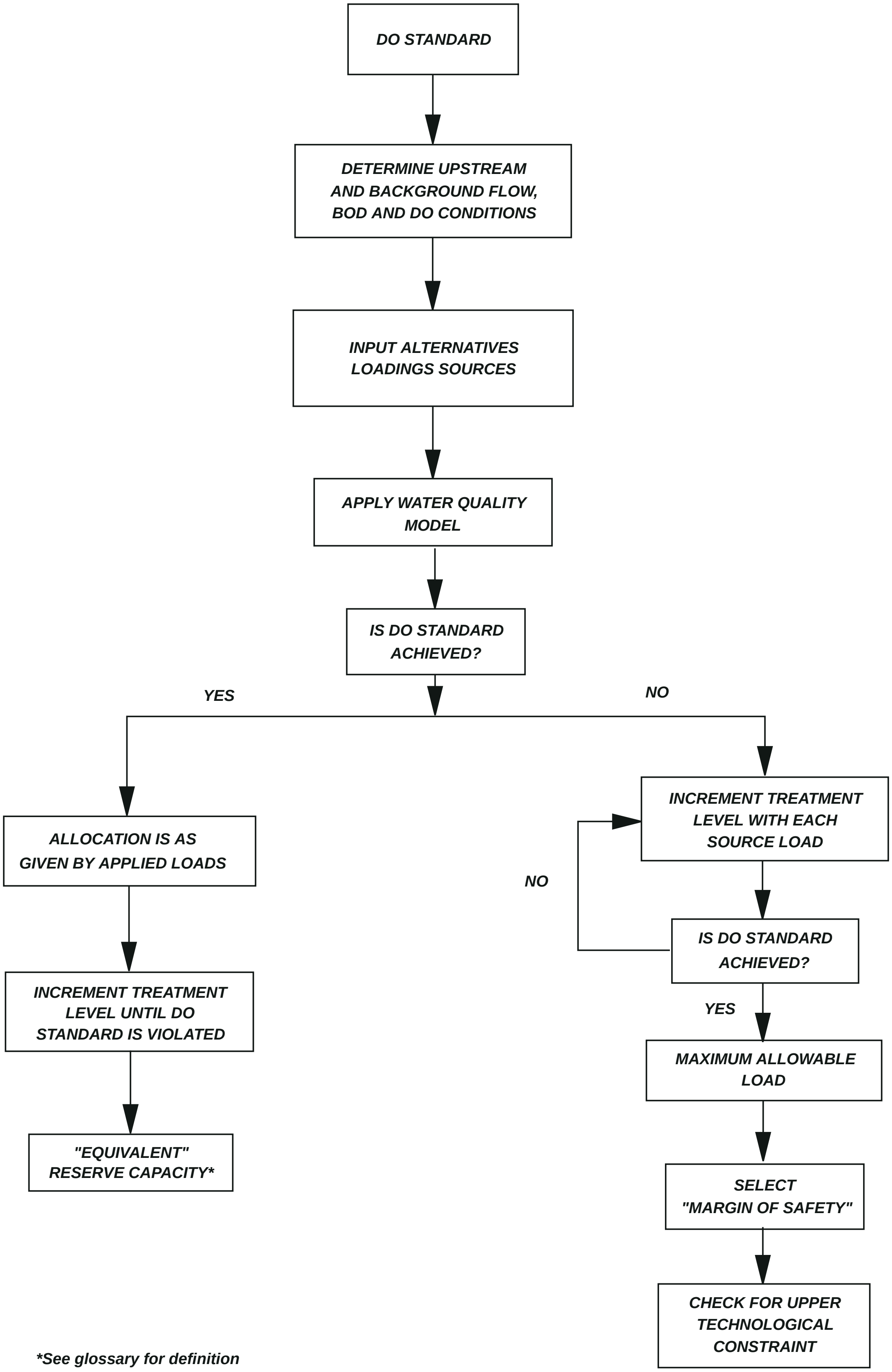
When developing an allocation scenario for a TMDL, the water resource manager should select the best combination of point and nonpoint source controls that achieves water quality standards. The selection of an allocation alternative largely depends on available technical and financial resources. The best combination of pollution reduction controls is that which is the most cost-effective and feasible to implement. Allocation scenarios typically reduce point source discharges through NPDES permitting, reduce nonpoint source loads through the implementation of best management practices (BMPs), or use a combination of both.

Cost trade-offs are an important consideration when developing alternative pollution allocation scenarios. Point and nonpoint source trading is one cost-effective alternative for meeting water quality criteria or other appropriate TMDL endpoints. Although it can be implemented in many different forms, essentially trading allocates pollutant loading reductions across point and nonpoint sources using least cost as the criterion (USEPA, 1992c). For example, in lieu of upgrading their pollution control technology, point source dischargers may be allowed to pay for equivalent or greater reductions in nonpoint source loadings within their watersheds. Trading is applicable when implementation of nonpoint source BMPs is less costly per unit of pollution reduction than upgrading point source treatment technology.

4.6.2 Total Maximum Daily Loads

Application of a model to allocate waste loads and nonpoint loads is usually done under 7-day, 10-year low-flow conditions depending on the WQS being implemented, and the type of waterbody (see Section A.3.2). A temperature condition needs to be established as well. There is no standard procedure in the model application analysis. Figure 4-10 is a suggested allocation procedure for BOD/DO in streams, the steps of which are discussed below. The procedure does not address cost/benefit issues.

The determination of the dissolved oxygen standard or endpoint as the first step includes an evaluation of the statistical requirements of the standard. Thus, if the standard indicates that the dissolved oxygen should never be less than 5 mg/L, then recognition should be given to random uncontrollable variations in dissolved oxygen. For streams and rivers, these fluctuations may be on the order of a standard deviation of 0.25 mg/L. Thus, if 0.5 mg/L is added to the standard, then the



See glossary for definition

FIGURE 4-10. TMDL PROCEDURE FOR BOD/DO PROBLEM

resulting level of 5.5 mg/L represents the target minimum level that, if attained, will meet the absolute minimum level of 5 mg/L with only a 2.5 percent chance of dropping below the standard. This does not imply that the short-term fluctuations may or may not be damaging to the ecosystem. That determination is part of the interpretation of the standard.

The selection of a background dissolved oxygen deficit is subject to wide variation depending on the specifics of the area, such as urban, suburban, or rural land use. Some States have determined background percent saturation for specific ecoregions. The deficit may be determined from upstream BOD and dissolved oxygen conditions and calculated through the region of interest. This approach requires assignment of BOD deoxygenation coefficients. A minimum effort analysis would simply assign a constant dissolved oxygen deficit throughout the river reach of 0 - 1 mg/L depending on the problem conditions. This step is clearly subject to potentially widely varying engineering judgment. It should be noted that the use of a 1-mg/L dissolved oxygen deficit may result in a significantly higher degree of required treatment than that resulting if no background is assigned.

The inputs from each of the point source discharges are then estimated following general guidelines for expected effluent concentrations. Nonpoint source loadings are estimated from existing or collected watershed data. Often, nutrient budget studies are conducted as part of the TMDL process to determine approximate pollutant loadings contributed by nonpoint sources. The remaining steps are as indicated. The application of the water quality model may also vary widely, depending on the level of effort involved, from simplified desktop calculations to full-scale field and calibration studies. If the dissolved oxygen standard is achieved with presently mandated effluent levels, then the allocation is as given by those levels and an equivalent reserve capacity can be estimated. In some cases the dissolved oxygen standard may be achieved by incrementing point source treatment by discrete levels. However, nonpoint source controls may be needed when further reductions in point source waste loads are not possible or are cost-prohibitive. The technological upper bound should be checked here. The maximum allowable discharge load is then the load needed to achieve the standard. However, this is not necessarily the load to be allocated.

If relatively rapid growth is forecasted for an area, then it is recommended that some fraction of the maximum allowable load be placed in reserve for future growth. A fraction of the maximum allowable load can be set aside explicitly, or implicitly, as a margin of safety to account for scientific uncertainty about whether the TMDL reflects the actual loading capacity of the waterbody. This uncertainty can be caused by insufficient or poor-quality data or a lack of knowledge about the water resource and pollutant effects. Thus, if a margin of safety of 0.8 is chosen, then 20 percent of the allowable load is placed in reserve. The allocation is given by the margin of safety times the maximum allowable load. However, a final check should be made to ensure that the required treatment level is technologically feasible. If an upper technological treatment bound has been exceeded, the margin of safety may have to be adjusted.

4.6.3 Uncertainty Analysis

Uncertainty analysis should be included as an integral component of water quality modeling. One of the primary purposes is to quantify the error in predicting water quality and evaluate the effect of input parameters on model output. Better management decisions can be made by quantifying this error. Such quantification also facilitates subsequent studies, such as risk assessments, to evaluate alternative allocations.

In addition, uncertainty analysis may provide insight into the need for additional data collection to refine the estimate of certain loads, initial conditions, or reaction rates. For example, if the model is sensitive to the reaeration rate (that is, a small change in reaeration rate results in large changes in the prediction of critical water quality parameters such as dissolved oxygen), it may be appropriate to allocate resources to more accurately estimate the reaeration rate of that stream or river.

Appendix D presents a discussion of the three techniques for performing uncertainty analysis: sensitivity analysis, first-order error analysis, and Monte Carlo simulation. Each technique has advantages and disadvantages in terms of applicability and computational burden that will make one method more suitable than another for a particular analysis. In many instances, the modeler may need to explore the results from all three procedures. The three methods may produce discrepancies in their results because the methodologies and assumptions differ. Each of these techniques is available in QUAL2E-UNCAS, and the discussion and example in Appendix D is limited to the features available in that model.

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APPENDIX A. DEVELOPMENT OF MODEL COEFFICIENTS AND CONSTANTS

A.1 OVERVIEW

As demonstrated in Chapter 2, a number of model coefficients and constants are formulated in a stream water quality model. Coefficient values can be obtained in four ways:

- Direct measurement
- Estimation from field data
- Literature values
- Model calibration

Model calibration is usually required regardless of the approach selected. However, coefficients that are site-specific or those that can take on a wide range of values should be measured directly or estimated from field samples. The purpose of Appendix A is to provide sufficient information and data to develop a consistent set of model coefficients and parameter values for a TMDL model analysis. For some model coefficients, additional discussions are presented to address subtle technical issues associated with the determination. This appendix is organized to follow the materials presented in Chapter 2 and is summarized below.

A.2 LOADS

A.2.1 Effluent Concentrations

As suggested in Chapter 2, point source inputs from municipal wastewater treatment plants or publicly owned treatment works (POTWs) and industrial facilities should be measured for site-specific situations. However, if the data are not readily available, typical effluent characteristics reported for POTWs in the literature may be used as a first approximation. In a study by Leo et al. (1984), an extensive amount of POTW effluent information was gathered and compiled to assess effluent BOD_5, $CBOD_5$, ammonia, $CBOD_u$-to-BOD_5 ratios, and $CBOD_u$-to-$CBOD_5$ ratios for various treatment levels. In total, information on these parameters was available from approximately 114 treatment facilities. Table A-1 presents a summary of the effluent BOD_5, $CBOD_5$, and ammonia concentrations for POTWs with various treatment levels.

Effluent BOD_5 and $CBOD_5$ concentrations are significantly different (see Table A-1), reinforcing the findings by Hall and Foxen (1984) that significant nitrification occurs during BOD tests for many POTWs with secondary treatment. For the 26 secondary treatment facilities in the above study, the ammonia data were gathered during intensive summer water quality surveys, indicating that many secondary POTWs achieve some nitrification during summer periods. It is likely that with in-plant nitrification occurring, nitrifying bacteria present in the effluent can cause oxygen consumption during the BOD_5 test. The BOD_5 test would therefore tend to underestimate the ability of the POTW to remove carbonaceous oxidizing materials.

In general, only POTWs that practice phosphorus removal to meet their NPDES permit effluent limits measure and report phosphorus concentrations in the effluent. Lung (1986b) reported an average total phosphorus concentration of 6.25 mg/L in the effluents of 18 secondary POTWs in the Chesapeake Bay area (plants in Virginia, Maryland, and Pennsylvania) that did not have phosphorus removal. Phosphate detergents have been progressively banned in the Chesapeake Bay region since that study. Recent data collected from a number of POTWs in Hampton Roads Sanitation District, Virginia, indicated up to a 50 percent phosphorus load reduction following the phosphate detergent ban, which became effective on January 1, 1988 (Virginia Water Control Board, 1989). One should note that phosphate detergent bans would have no effect on the effluent concentrations at POTWs that remove phosphorus to meet NPDES permits.

A number of texts, technical reports, and other literature document influent and effluent characteristics of municipal wastewater for various treatment levels. Tables A-1 through A-17 present data summarized by a number

TABLE A-1. SUMMARY OF EFFLUENT CHARACTERISTICS (After Leo et al., 1984)

		POTW Effluent Concentrations (mg/L)					
		BOD_5		$CBOD_5$		Ammonia-N	
Treatment Type	Number of Locations[a]	**Mean**	**Standard Deviation**	Mean	**Standard Deviation**	**Mean**	**Standard Deviation**
Primary	2	101.0	21.2	—	—	—	—
Trickling Filter	13	41.2	27.8	—	—	16.6	12.2
Secondary	38	19.1	16.3	10.3	6.4	8.9	6.3
Secondary + P-Removal	9	16.2	14.0	14.6	9.3	7.9	8.9
Secondary + Nitrification	10	11.5	11.8	4.8	3.9	1.0	1.4
Secondary + P-Removal + Nitrificaton	3	13.6	18.6	—	—	0.9	0.7
Secondary + P-Removal + Filters	3	3.9	2.0	—	—	4.8	8.2

[a] Number of locations with BOD_5 data. In some cases, number with $CBOD_5$ or NH_3 data may be less.

of investigators that can be used to estimate effluent characteristics.

A.2.2 Effluent $CBOD_u$-to-BOD_5 or $CBOD_u$-to-$CBOD_5$ Ratios

The effluent $CBOD_u$-to-BOD_5 or $CBOD_u$-to-$CBOD_5$ ratio is required in dissolved oxygen modeling analyses to estimate POTW $CBOD_u$ from effluent BOD_5 or $CBOD_5$ data. This data is also needed to convert model output (as $CBOD_u$) to NPDES permit limits (as $CBOD_5$). A summary of this information is presented in Figure A-1 (Leo et al., 1984), suggesting a mean value of 2.47 for $CBOD_u$-to-BOD_5 and 2.84 for $CBOD_u$-to-$CBOD_5$. Thomann and Mueller (1987) summarize the $CBOD_u$-to-$CBOD_5$ ratios for municipal wastes as 1.2 for no treatment, 1.6 for primary/secondary, 3.2 for activated sludge, and 2.84 for advanced primary. In the absence of site-specific data, these ratios are reasonable approximations for a dissolved oxygen modeling analysis.

EPA strongly recommends that, whenever possible, data from existing plant or pilot plant effluents be used in the modeling analysis. In this case, long-term BOD tests should be run to determine the K_1 coefficient from Equation 2-5 and consequently the $CBOD_u$-to-$CBOD_5$ ratio. However, caution should be exercised when using data from an existing plant that has a treatment level significantly less than that of a proposed plant. In this case, the existing data should be used as a guide. A model sensitivity analysis of the final WLA with respect to the ratio should help the analyst to judge the need for additional data.

The $CBOD_u$-to-$CBOD_5$ ratio of industrial wastewater is highly dependent on the type of industry manufacturing processes, treatment schemes or operation, measurement techniques, and other factors. Pulp and paper mill effluent, for example, is characterized by very high ratios of $CBOD_u$ to $CBOD_5$ because of the refractory nature of the cellulose and compounds in the wastewater. For many industrial wastewaters, the ratios also may vary with BOD concentration.

A.2.3 Nonpoint Source Loads

Other loading rates for nonpoint loads such as combined sewer overflows, urban storm runoff, and upstream background loads vary from one study area to another. Thomann and Mueller (1987) and Novotony (1991, 1992) provide a brief summary of these loading rates. Mills et al. (1985) present information on determining these loads. Table A-2 lists some of the typical ranges as described in the literature.

TABLE A-2. TYPICAL RANGES OF POLLUTANT LOAD FOR SOURCES

Source	Range	Supplemental References
Domestic and Industrial Point Sources	NPDES Permits Compliance Reports	a, b, c, d, e
Upstream Background Levels:		
Dissolved Oxygen Deficit	0.5-2.0 mg/L	g, i, Use STORET
BOD_5	0.5-3.0 mg/L	f, g, h, Use STORET
NH_3	0.05-.27 mg/L	f, g, h, Use STORET
NO_3	0.07-0.37 mg/L	f, Use STORET
Organic N	0.05-0.50 mg/L	f, g, Use STORET
Combined Sewer Overflow, BOD	115 mg/L	d, h, j
Organic N	3.8 mg/L	d, h, j
TN	9.1 mg/L	d, h, j
Nonpoint sources (kg/ha/yr)		
Urban		
General		
TN	6.69	k
	6.37-8.00	l
	8.12	m
	5.62-7.14	n
	17.23	o
TP	1.57	k
	0.40-3.19	l
	1.20	m
	0.89-4.46	n
	1.33	o
BOD5	58.97	k
	34.23	m
Residential		
TN	0.87	k
	4.77-7.16	l
	6.69	m
TP	0.17	k
	0.48-0.79	l
	1.03	m
BOD5	28.55	k
	28.67	m
Commercial		
TN	7.34	k
	2.39-9.56	l
	17.83	m
TP	0.55	k
	0.07-0.71	l
	2.70	m
BOD5	13.03	k
	78.03	m

TABLE A-2. (Continued)

Source	Range	Supplemental References
Agricultural		
General		
	0.63-59.71	l
TP	0.08-7.16	l
Cropland		
TN	20.46	k
	4.77-47.77	l
	0.09-10.35	n
	18.71	o
TP	0.83	k
	0.24-5.58	l
	0.04-2.32	n
	5.45	o
Improved Pasture		
TN	5.02	k
	3.98-11.95	l
TP	0.92	k
	0.08-0.48	l
Pasture		
TN	4.17	k
	2.31	o
TP	0.24	k
	0.44	o
Forested TN	2.44	k
	0.79-6.37	l
	2.41-10.35	n

a Leo et al., 1984
b Metcalf and Eddy, 1972
c Mueller et al., 1976
d Thomann and Mueller, 1987
e Mueller et al., 1982
f Hydroscience, 1975
g Hydroscience, 1968
h Metcalf and Eddy, 1977
i Manhattan College, 1980
j USEPA, 1976a
k Shahane, 1982
l Novotny and Chesters, 1981
m NURP, 1983
n Sweeten and Melvin, 1985
o Haith and Shoemaker, 1987

TABLE A-3. REPORTED VALUES OF SELECTED WASTE INPUT PARAMETERS IN THE UNITED STATES
(after Thomann and Mueller, 1987)

Variable	**Units**[a]	Municipal Influent[b]	**CSO**[c]	**Urban Runoff**[d]	**Agriculture (lb/mi^2-day)**[e]	**Forest (lb/mi^2-day)**[e]	**Atmosphere (lb/mi^2-day)**[f]
Average daily flow	gcd	125					
Total suspended solids	mg/L	300	410	610	2500	400	
$CBOD_5$[g]	mg/L	180	170	27	40	8	
$CBOD_U$[g]	mg/L	220	240				
NBOD[g]	mg/L	220	290				
Total nitrogen	mg-N/L	50	9	2.3	15	4	8.9-18.9
Total phosphorus	mg-P/L	10	3	0.5	1.0	0.3	0.13-1.3
Total coliforms	10^6/100 mL	30	6	0.3			
Cadmium	mg/L	1.2	10	13			0.015
Lead	mg/L	22	190	280			1.3
Chrome	µg/L	42	190	22			0.088
Copper	µg/L	159	460	110			
Zinc	mg/L	241	660	500			1.8
Total PCB	mg/L	0.9	0.3	-			0.002-0.02

[a] Units apply to municipal influent, combined sewer overflow (CSO), and urban runoff sources; gcd = gallons per capita per day.

[b] Thomann (1972); heavy metals and PCB, HydroQual (1982).

[c] Thomann (1972); total coli, Tetra Tech, (1977); heavy metals Di Toro et al. (1978); PCB, Hydroscience (1978).

[d] Tetra Tech (1977); heavy metals, Di Toro et al. (1978).

[e] Hydroscience (1976).

[f] Nitrogen and phosphorus, Tetra Tech (1982); heavy metals and PCB, HydroQual (1982).

[g] CBOD5 = 5 day carbonaceous biochemical oxygen demand (CBOD); $CBOD_U$ = ultimate CBOD; NBOD = nitrogenous BOD.

TABLE A-4. APPROXIMATE COMPOSITION OF AN AVERAGE DOMESTIC WASTEWATER (mg/L)
(after Clark et al., 1977)

	Before Sedimentation	**After Sedimentation**	**Biologically Treated**
Total solids	800	680	530
Total volatile solids	440	340	220
Suspended solids	240	120	30
Volatile suspended solids	180	100	20
BOD	200	130	30
Ammonia nitrogen as N	15	15	20
Total nitrogen as N	35	25	20
Soluble phosphorus as P	7	7	7
Total phosphorus as P	10	8	7

TABLE A-5. MUNICIPAL WASTE CHARACTERISTICS BEFORE TREATMENT (after Thomann, 1972)

Variable	Unit	Approx. Average	Normal Range
Avg. Daily Flow	gal/cap/day	125	100-200
Solids -Total	mg/L	800	450-1200
Total Volatile	mg/L	400	250-800
Total Dissolved	mg/L	500	300-800
Total Suspended	mg/L	300	100-400
Volatile Suspended	mg/L	130	80-200
Settleable	mg/L	150	—
CBOD (5-day)[a]	mg/L	180	100-450
CBOD (ultimate)	mg/L	220	120-580
NBOD[b]	mg/L	220	—
Total Nitrogen	mg/L N	50	15-100
Organic Nitrogen	mg/L N	20	5-35
Ammonia Nitrogen	mg/L N	28	10-60
Nitrate + Nitrite	mg/L N	2	0-6
Total Phosphate	mg/L PO_4	20	10-50
Ortho Phosphate	mg/L PO_4	10	5-25
Poly-Phosphate	mg/L PO_4	10	5-25
Total Coliforms	million/100mL	30	2-50
Fecal Coliforms	million/100mL	4	0.3-17

[a] CBOD - Carbonaceous oxygen demand.
[b] NBOD - Nitrogenous oxygen demand, ultimate; exclusive of CBOD.

TABLE A-6. TYPICAL COMPOSITION OF RAW DOMESTIC SEWAGE (All values except settleable solids are expressed in mg/L) (after Metcalf & Eddy, 1972)

	Concentration Before Treatment		
Constituent	Strong	Medium	Weak
Solids, total	1200	700	350
Dissolved, total	850	500	250
Fixed	525	300	145
Volatile	325	200	105
Suspended, total	350	200	100
Fixed	75	50	30
Volatile	275	150	70
Settleable solids, (mL/L)	20	10	5
Biochemical oxygen demand, 5-day, 20 °C (BOD_5 @20 °C)	300	200	100
Total organic carbon (TOC)	300	200	100
Chemical oxygen demand (COD)	1,000	500	250
Nitrogen, (total as N)	85	40	20
Organic	35	15	8
Free Ammonia	50	25	12
Nitrites	0	0	0
Nitrates	0	0	0
Phosphorus (total as P)	20	10	6
Organic	5	3	2
Inorganic	15	7	4
Chlorides[a]	100	50	30
Alkalinity (as $CaCO_3$)[a]	200	100	50
Grease	150	100	50

[a] Values should be increased by amount in carriage water.

TABLE A-7. TYPICAL MUNICIPAL WASTEWATER CHARACTERISTICS
(After Mueller et al., 1976)

Parameter	**Concentration (mg/L)**		
	NYC Raw Sewage	**NJ** Primary Effluent	**NYC Secondary Effluent**
SS	139	93	43
ALK	190	190	170
BOD_5	131	158	36
COD	2.5 x BOD_5	2.5 x BOD5	4.7 x BOD_5
TOC	83	0.68 x BOD_5	0.94 x BOD5
MBAS	10	10	1.0
Oil & Grease	36	23	15
NH_3-N	10.6	0.58 x Tot.N	0.64 x Tot.N
Org-N	10.4	0.69 x NH_3-N	0.53 x NH_3-N
NO_2+NO_3-N	0.68	0.02 x Tot.N	0.02 x Tot.N
Total N	21.7	22	22
Ortho-P	3.27	0.7 x Tot.P	0.7 x Tot.P
Total P	4.70	6.14	3.30
Cd	0.018	0.012	0.012
Cr	0.15	0.057	0.057
Cu	0.23	0.105	0.105
Fe	2.5	0.70	0.70
Hg	0.033	0.025	0.025
Pb	0.26	0.190	0.190
Zn	0.39	0.185	0.185
Fecal Coli (cells/100mL)	0.44 x T.Coli	0.44 x T.Coli	0.44 x T.Coli
Total Coli (cells/100mL)	$50x10^6$	$15x10^6$	$2.5x10^6$
Total Coli after Chlorination (cells/ 100mL)		357	357

TABLE A-8. TYPICAL MUNICIPAL WASTEWATER CHARACTERISTICS FOR CONVENTIONAL POLLUTANTS AND MOST METALS

(after Mueller et al., 1982)

Parameter	**NYC Raw Sewage[a] (mg/L)**	NJ Primary Effluent[b] **(mg/L)**	**Middlesex County Secondary Effluent[c]**	**NYC Secondary Effluent[d] (mg/L)**
SS	110	105	43	20
BOD	104	218	39	15
TOC	93	151	128	39
NH_3-N	10	22	11	7.9
ORG-N	13	13	17	6.1
NO_2-N	0.07	0.06	0.16	0.19
NO_3-N	0.38	0.51	3.6	1.3
Ortho-P	2.0	7.7	3.5	1.6
Total-P	3.2	9.3	2.3	2.1
Fecal Coliform (MPN/100mL)[e]				
Winter	3×10^6	33	33	1.5 x 105
Summer	3×10^6	33	33	33
Cadmium	1.2	14	12	1.1
Chromium	42	68	34	16
Copper	159	185	334	93
Cyanide	92	92(f)	57	52
Lead	22	211	77	11
Mercury	1.3	0.62	0.2	0.57
Nickel	45	105	37	37
Zinc	241	365	4800	101

[a] Not including Newton Creek and Bowery Bay (high industry).
[b] 14 New Jersey plants.
[c] SS, BOD, TOC, ORG-N, and ortho-P from ISC data. All other from Middlesex County (NJ) quarterly reports.
[d] Not including Coney Island, Newton Creek, and Owls Head (intermediate treatment) and Bowery Bay (high industry).
[e] NYC raw from 208 study, NJ primary, Middlesex County, and NYC secondary from NYC summer 1980 and NYC winter from Lake Tahoe, California.
[f] NYC raw concentration assumed.

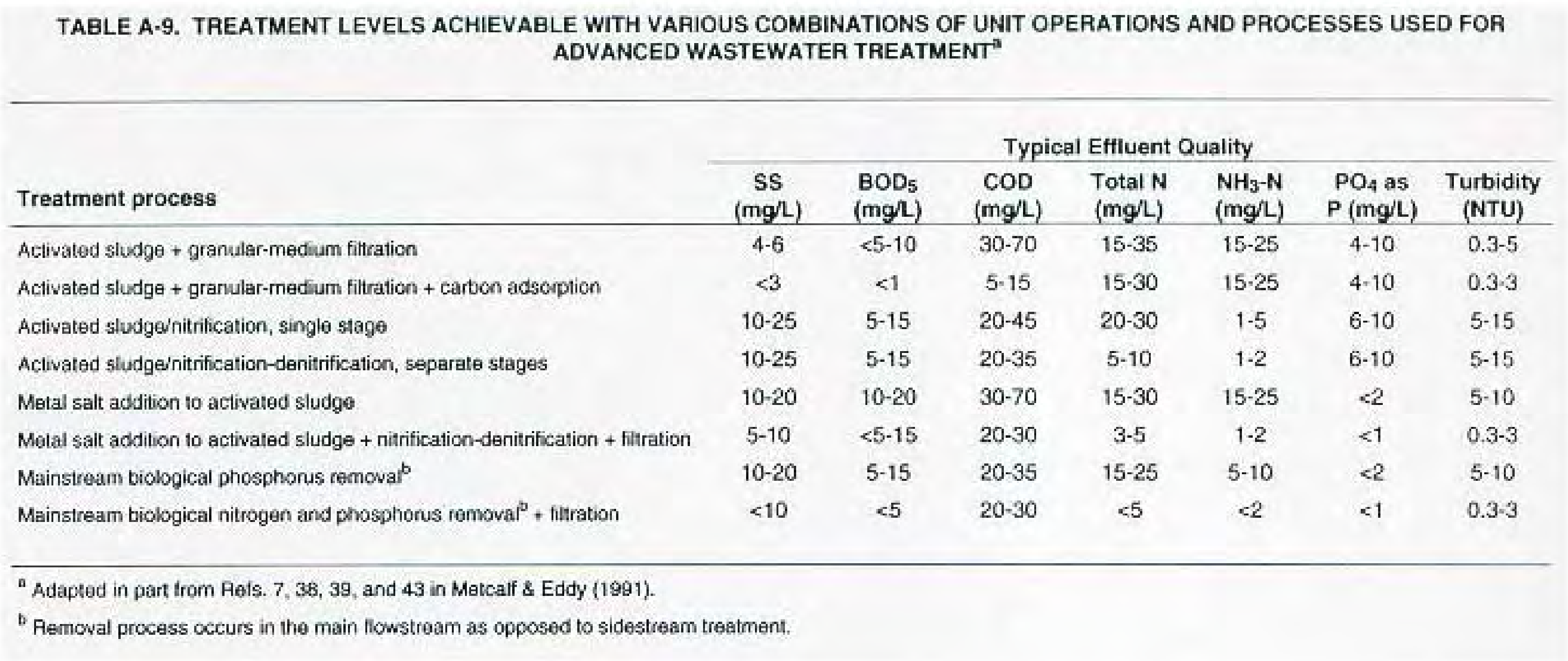

TABLE A-9. TREATMENT LEVELS ACHIEVABLE WITH VARIOUS COMBINATIONS OF UNIT OPERATIONS AND PROCESSES USED FOR ADVANCED WASTEWATER TREATMENT[a]

Treatment process	Typical Effluent Quality						
	SS (mg/L)	BOD_5 (mg/L)	COD (mg/L)	Total N (mg/L)	NH_3-N (mg/L)	PO_4 as P (mg/L)	Turbidity (NTU)
Activated sludge + granular-medium filtration	4-6	<5-10	30-70	15-35	15-25	4-10	0.3-5
Activated sludge + granular-medium filtration + carbon adsorption	<3	<1	5-15	15-30	15-25	4-10	0.3-3
Activated sludge/nitrification, single stage	10-25	5-15	20-45	20-30	1-5	6-10	5-15
Activated sludge/nitrification-denitrification, separate stages	10-25	5-15	20-35	5-10	1-2	6-10	5-15
Metal salt addition to activated sludge	10-20	10-20	30-70	15-30	15-25	<2	5-10
Metal salt addition to activated sludge + nitrification-denitrification + filtration	5-10	<5-15	20-30	3-5	1-2	<1	0.3-3
Mainstream biological phosphorus removal[b]	10-20	5-15	20-35	15-25	5-10	<2	5-10
Mainstream biological nitrogen and phosphorus removal[b] + filtration	<10	<5	20-30	<5	<2	<1	0.3-3

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[a] Adapted in part from Refs. 7, 38, 39, and 43 in Metcalf & Eddy (1991).

[b] Removal process occurs in the main flowstream as opposed to sidestream treatment.

TABLE A-10. MASSACHUSETTS WATER RESOURCES AUTHORITY (MWRA) BOSTON HARBOR PILOT PLANT WASTEWATER EFFLUENT LEVELS
(After Metcalf & Eddy, 1989)
(in mg/L unless otherwise specified)

Effluent Parameter	MWRA Primary Effluent	MWRA Secondary Effluent	Data Source
Total $CBOD_5$	112	9	MWRA Pilot
Soluble $CBOD_5$	44.80	3.05	Total*Particulate
Particulate $CBOD_5$	67.20	5.94	Total*Particulate/Total
Particulate/Total $CBOD_5$	60%	66%	MWRA Pilot
Total COD	253	50.50	MWRA Pilot
Soluble COD	93.61	37.87	Total*Particulate
Particulate COD	159.39	12.62	Total*Particulate/Total
Particulate/Total COD	63%	25%	MWRA Pilot
Total SS	136	15	MWRA Pilot
TKN	16	11	MWRA Pilot
NH_3-N	11	8.10	MWRA Pilot
Total N	16.33	11.22	TN=TKN/0.98
NO_3-N + NO_2-N	0.33	0.22	NO_3=TN*0.02
Total P	4.13	1.88	MWRA Pilot

TABLE A-11. EFFECT OF VARIOUS TREATMENT OPERATIONS AND PROCESSES ON PHOSPHORUS REMOVAL[a]
(after Metcalf & Eddy, 1991)

Treatment Operation or Process	Removal of Phosphorus Entering System (%)
Conventional treatment	
Primary	10-20
Activated-sludge	10-25
Trickling-filter	8-12
Rotating biological contactors	8-12
Biological phosphorus removal only	
Mainstream treatment	70-90
Sidestream treatment	70-90
Combined biological nitrogen and phosphorus removal	70-90
Chemical removal	
Precipitation with metal salt	70-90
Precipitation with lime	70-90
Physical removal	
Filtration	20-50
Reverse osmosis	90-100
Carbon adsorption	10-30

[a] Adapted in part from Ref. 24 cited in Metcalf & Eddy (1991).

TABLE A-12. EFFECT OF VARIOUS TREATMENT OPERATIONS AND PROCESSES ON NITROGEN COMPOUNDS[a]

(After Metcalf & Eddy, 1991)

Treatment Operation or Process	Nitrogen Compound			Removal of Total Nitrogen Entering Process, %[b]
	Organic Nitrogen	NH_3-NH_4	NO_3	
Conventional treatment				
Primary	10–20% removed	No effect	No effect	5-10
Secondary	15–50% removed[c] urea —> NH_3-NH_4[d]	< 10% removed	Slight effect	10-30
Biological processes				
Bacterial assimilation	No effect	40-70% removed	Slight	30-70
Denitrification	No effect	No effect	80-90% removed	70-95
Harvesting algae	Partial transformation to NH_3-NH_4[d]	—> Cells	—> Cells	50-80
Nitrification	Limited effect	—> NO_3	No effect	5-20
Oxidation ponds	Partial transformation to NH_3-NH_4	Partial removal by stripping	Partial removal by nitrification/denitrification	20-90
Chemical processes				
Breakpoint chlorination	Uncertain	90-100% removed	No effect	80-95
Chemical coagulation	50-70% removed	Slight effect	Slight effect	20-30
Carbon adsorption	30-50% removed	Slight effect	Slight effect	10-20
Selective ion exchange for ammonium	Slight, uncertain	80-97% removed	No effect	70-95
Selective ion exchange for nitrate	Slight effect	Slight effect	75-90% removed	70-90
Physical operations				
Filtration	30-95% of suspended organic N removed	Slight effect	Slight effect	20-40%
Air stripping	No effect	60-95% removed	No effect	50-90%
Electrodialysis	100% of suspended organic N removed	30-50% removed	30-50% removed	40-50%
Reverse osmosis	60-90% removed	60-90% removed	60-90% removed	80-90%

TABLE A-13. PERCENTAGE REMOVALS OF WASTE CONSTITUENTS BY PRIMARY, TRICKLING FILTER, AND ACTIVATED SLUDGE PLANTS (USEPA, 1977; Gunnerson et al., 1982)

	Primary Plants				Trickling Filter Plants				Activated Sludge Plants			
Parameter	Mean	Standard Deviation	Max./ Min.	No. of Plants	Mean	Standard Deviation	Max./ Min.	No. of Plants	Mean	Standard Deviation	Max./ Min.	No. of Plants
SS	51	18	92/17	47	75	19	97/20	66	75	22	99/9	62
BOD	30	22	89/0	52	77	18	96/5	60	84	15	99/18	65
TOC	24	19	56/0	30	64	18	84/8	23	73	12	89/42	13
COD	26	-	82/0	18	71	-	95/34	36	75	-	94/24	40
P-total	13	8	24/0	7	26	22	99/0	24	42	25	92/0	36
TKN	22	20	60/0	7	50	27	94/7	20	34	26	92/5	11
NH_3	20	16	64/0	42	41	30	99/0	48	49	31	99/4	47
Phenol	38	-	50/25	2	50	28	85/0	12	69	31	98/0	16
Metals												
Cd	8	17	76/0	31	20	25	75/0	35	17	27	88/0	44
Cr	26	26	80/0	36	37	30	99/0	48	46	34	98/0	54
Pb	24	26	88/0	34	37	31	93/0	41	39	32	95/0	49
Hg	27	29	75/0	21	30	23	67/0	20	39	32	99/0	34
Cu	26	24	77/0	44	54	24	95/0	49	57	24	95/0	63
Ni	6	18	92/0	28	21	23	86/0	32	20	21	80/0	44
Zn	31	22	88/0	38	46	22	89/0	52	58	25	99/0	58
Fe	40	22	89/0	27	50	26	90/0	30	63	27	98/8	35
Mn	15	20	81/0	16	31	23	72/0	21	38	32	93/0	19

NOTE: BOD-biochemical oxygen demand; COD-chemical oxygen demand; P-total phosphorus; SS-suspended solids; TKN-total Kjeldahl nitrogen; TOC-total organic carbon.

TABLE A-14. REMOVAL OF NITROGEN FROM SEWAGE EFFLUENTS (After Ketchum, 1982)

	Effect on Constituent			
Treatment process	**Organic-N**	**Ammonia-N**	**Nitrate-N**	**Removal of Total Nitrogen Entering Process (%)**
Conventional treatment processes				
Primary	10-20% removed	no effect	no effect	5-10
Secondary	15-20% removed urea —> NH_3/NH_4	<10% removed	nil	10-20
Advanced wastewater treatment processes				
Filtration	30-95% removed	nil	nil	20-40
Carbon sorption	30-50% removed	nil	nil	10-20
Electrodialysis	100% of suspended organic N removed	40% removed	40% removed	35-45
Reverse osmosis	100% of suspended organic N removed	85% removed	85% removed	80-90
Chemical coagulation	50-70% removed	nil	nil	20-30
Other nitrogen removal processes				
Selective ion exchange for nitrate	nil	nil	75-90% removed	70-90
Oxidation ponds	partial transformation to NH_3/NH_4	partial removal by stripping	partial removal by nitrification-denitrification	20-90
Algae stripping	partial transformation to NH_3/NH_4	—> cells	—> cells	50-85
Bacterial assimilation	no effect	40-70% removed	limited effect	30-70

TABLE A-15. TOTAL NITROGEN AND PHOSPHORUS CONCENTRATION (MEDIAN) IN WASTEWATER EFFLUENTS FOLLOWING FOUR CONVENTIONAL TREATMENT PROCESSES (After Ketchum, 1982)

	Treatment Type			
	Primary	**Trickling Filter**	**Activated Sludge**	**Stabilization Pond**
No. of plants sampled	55	244	244	149
Total P (mg/L)	6.6 ± 0.66	6.9 ± 0.28	5.8 ± 0.29	5.2 ± 0.45
Total N (mg/L)	22.4 ± 1.30	16.4 ± 0.54	13.6 ±0.62	11.5 ± 0 .84
N:P (weight) (mg N/mg P)	3.39	2.38	2.34	2.21
N:P (atoms) (mg-at N/mg-at P)	7.52	5.26	5.19	4.90

TABLE A-16. AVERAGE WASTE REDUCTION EFFICIENCIES OF VARIOUS CONTROLS OPERATING ON MUNICIPAL WASTE

(After Metcalf & Eddy, 1991)

	Percent Reduction[a]				
Control	**CBOD**	**Suspended Solids**	**Total Nitrogen**	**Dissolved Solids**	**Total Phosphorus**
Conventional Treatment					
Primary settling	25-40	20-60	10-20	–	–
Intermediate-chemical treatment	40-65	60-80	20-30	–	–
Secondary-primary and trickling filter	65-85	60-80	20-40	–	–
Secondary-primary and conventional activated sludge	85-90	90	20-40[b]	5	10
Advanced Treatment					
Secondary, nitrification and denitrification	90-95	95	90	5	10
Secondary and coagulation, settling filtration	95	99	50	10	95
Secondary and coagulation and adsorption	99[c]	99	55	15	95
Secondary and coagulation and adsorption and electrodialysis	99[c]	99	75	50	97

[a] Percent reduction based on raw waste concentration.

[b] Longer aeration times convert oxidizable nitrogen to nitrate.

[c] Minimum CBOD of about 2-5 mg/L.

TABLE A-17. MEDIAN AND MEAN PHOSPHORUS AND NITROGEN CONCENTRATIONS AND MEDIAN LOADS IN WASTEWATER EFFLUENTS FOLLOWING FOUR CONVENTIONAL TREATMENT PROCESSES

(After Gakstatter et al., 1978)

		Treatment Type			
		Primary	**Trickling Filter**	**Activated Sludge**	Stabilization Pond
Number of Sampled Plants		55	244	244	119
Total Population Served		1,086,784	3,459,893	4,357,138	270,287
Ortho-P Conc. (mg/l)	Median	3.5 ± 0.29[a]	5.1 ± 0.21	4.6 ± 0.24	3.9 ± 0.34
	Mean	4.0 ± 0.62	5.4 ± 0.38	5.3 ± 0.40	4.8 ± 0.62
Total-P Conc. (mg/l)	Median	6.6 ± 0.66	6.9 ± 0.28	5.8 ± 0.29	5.2 ± 0.45
	Mean	7.7 ± 1.19	7.2 ± 0.50	6.8 ± 0.51	6.6 ± 0.81
Total-P Load (cap · y)	Median	1.1 ± 0.10	1.2 ± 0.05	1.0 ± 0.06	0.9 ± 0.10
Total-N Load (kg/cap · y)	Median	3.7	2.9	2.4	2.0
Inorganic-N Conc. (mg/l)	Median	6.4 ± 1.00	7.1 ± 0.38	6.5 ± 0.45	1.3 ± 0.29
	Mean	8.3 ± 1.40	8.2 ± 0.60	8.4 ± 0.69	5.5 ± 1.95
Total-N Conc. (mg/l)	Median	22.4 ± 1.30	16.4 ± 0.54	13.6 ± 0.62	11.5 ± 0.84
	Mean	23.8 ± 3.48	17.9 ± 1.23	15.8 ± 1.16	17.1 ± 3.59
Total-N Load	Median	4.2 ± 0.40	2.9 ± 0.17	2.2 ± 0.15	2.0 ± 0.26
TN:TP Ratio	Median	3.4	2.4	2.4	2.2
Per Capita Flow (1/cap · d)	Median	473 ± 72	439 ± 19	394 ± 26	378 ± 38

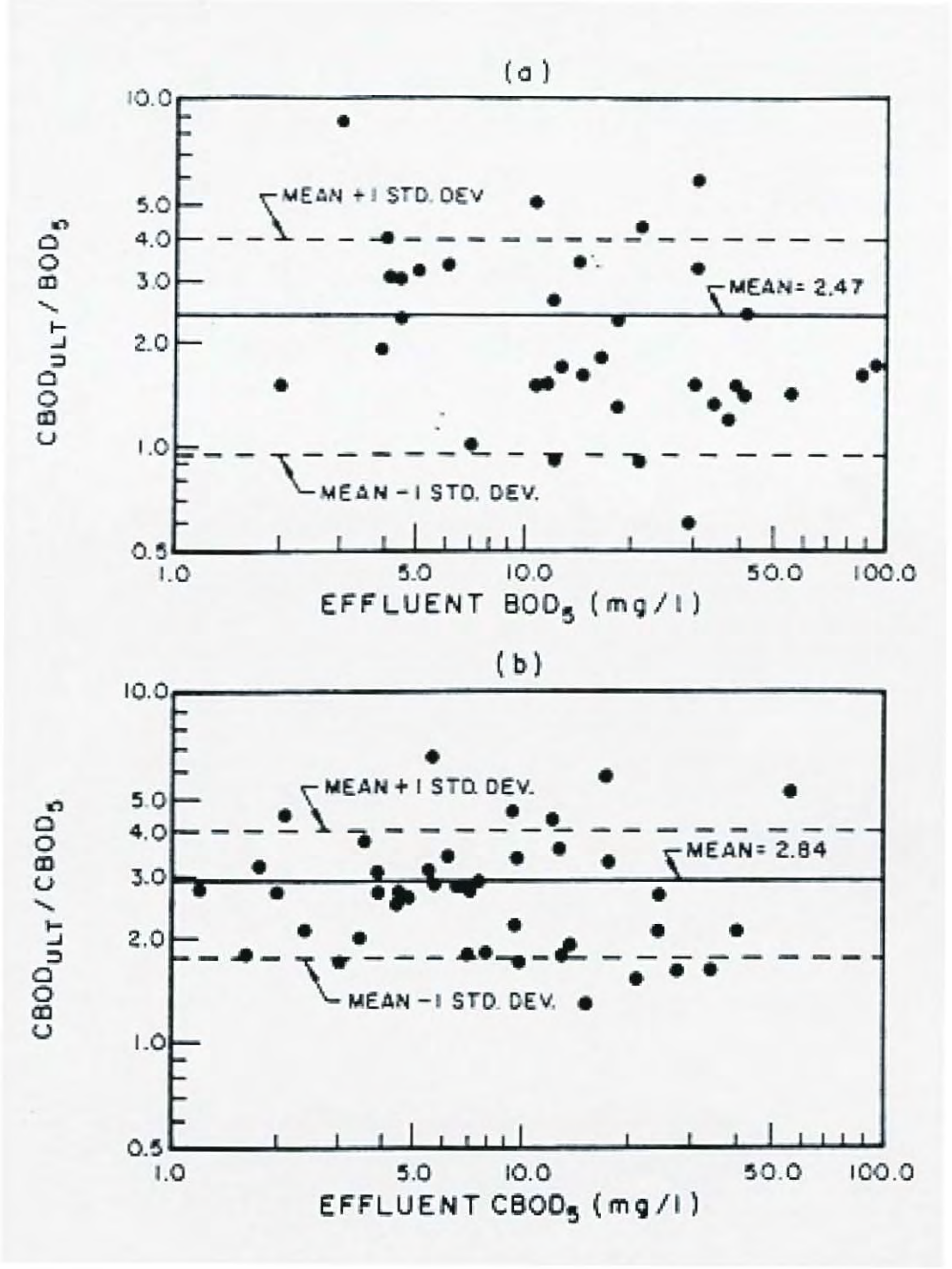


FIGURE A-1. POTW EFFLUENT ULTIMATE CBOD AS A FUNCTION OF $CBOD_5$ AND BOD_5
(After Leo et al., 1984)

These loading rates represent those rates found at specific sites across the country. In any water quality modeling analysis, site-specific data should be collected.

A.3 PHYSICAL PARAMETERS

A.3.1 Hydraulic Geometry Data

Hydraulic data include the velocities, flows, and water depths. Flows include the flows at upstream boundaries of all channels, as well as all significant tributary inflows, lateral inflow (from groundwater or runoff), flow diversions, return flows, and stage at some locations. If wastewater flows represent a significant portion (i.e., greater than 5 to 10 percent) of the total stream flow, they should be included in the hydraulic analysis.

While the upstream boundary, tributary, and diversion flows can be measured directly, lateral inflows from groundwater or runoff can be estimated from differences in measured flow at different locations along the stream channel. USGS official flow records are annually published several months to a year after the end of the water year, which ends September 30. All USGS flow data are available through EPA's mainframe computer. Under special contract, USGS is able to furnish the records for the period of a stream study as soon as the stream stage data can be converted to flow. These records may be designated as *tentative* or provisional, but are adequate for all but the strictest legal uses. The U.S. Army Corps of Engineers and the U.S. Bureau of Reclamation maintain stream flow records on streams for which they have special responsibilities. In addition, some State regulatory agencies only rarely make stream flow measurements for selected streams.

As indicated in Section 2.3.3, mathematical models such as Manning's Equation and stream hydrodynamic models can be used to quantify the stage-flow relationships for each channel reach in a stream. One of the simple stream hydrodynamic models is RIVMOD (USEPA, 1990). Power functions may also be developed relating flow with velocity, depth, and cross-sectional area (see Equations 2-1 through 2-3). It can be shown that the sum of the exponents ($m + n + f$) and the product of the coefficients (abc) from Equations 2-1 through 2-3 are unity because of the continuity equation. Using this and experience from a variety of streams, the value of the exponents can be approximated as follows (Barnwell et al., 1989):

n = range (0.4 - 0.6); typical 0.43

m = range (0.3 - 0.5); typical 0.45

f = range (0 - 0.2); typical 0.12

Impounded reaches in rivers have exponents m and $f = 0$, and $n = 1$. Where the analyst has more than one set of data, a log-log plot of area, depth, and velocity against stream flow will permit extrapolation to other flows of interest. The slope of such plots provides the local value of the exponents (see Figure A-2). When data at only a single flow regime are available, estimates for other flows can be developed by the following ratios, derived from the foregoing relationships:

velocity: $$U_2/U_1 = (Q_2/Q_1)^{0.43} \tag{A-1}$$

depth: $$H_2/H_1 = (Q_2/Q_1)^{0.45} \tag{A-2}$$

cross-sect. area: $$A_2/A_1 = (Q_2/Q_1)^{0.43} \tag{A-3}$$

travel time: $$T_2/T_1 = (Q_2/Q_1)^{0.43} \tag{A-4}$$

It should be recognized that these relationships exist only in free-flowing streams and care should be exercised when upstream dams or hydropower facilities are present that may interfere with the assumption of the analysis. It is also common to collect site-specific data since the exponents may vary by 50 percent for any river.

A.3.2 Low-Flow Analysis

Generally, the minimum average 7-day flow expected to occur once every 10 years is used in modeling analyses depending on the state WQS and water body assessed. However, several different types of flows can be estimated from a hydrologic record. For example, the minimum average 7-day flow occurring at any time in a year can be estimated, or the minimum average flow in a given month or season can be computed. Figure A-3 shows the low-flow frequency curves for the Potomac River at Point of Rocks developed using the USGS flow records from 1891 to 1981. Table A-18 presents a summary of data obtained from Figure A-3. The analysis to determine the low-flow frequency curve is simple and straightforward; Thomann and Mueller (1987) provide an easy-to-follow procedure.

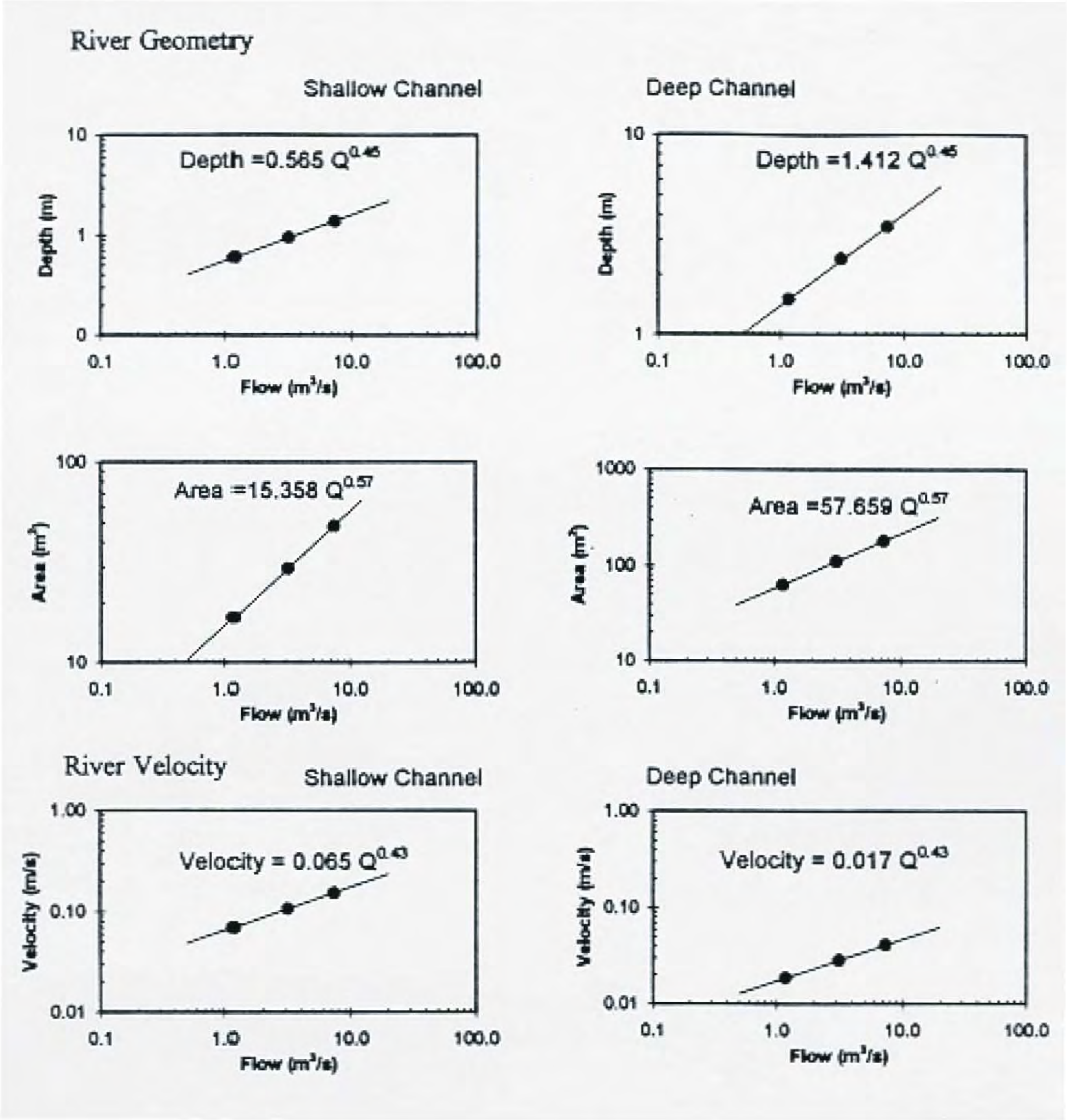
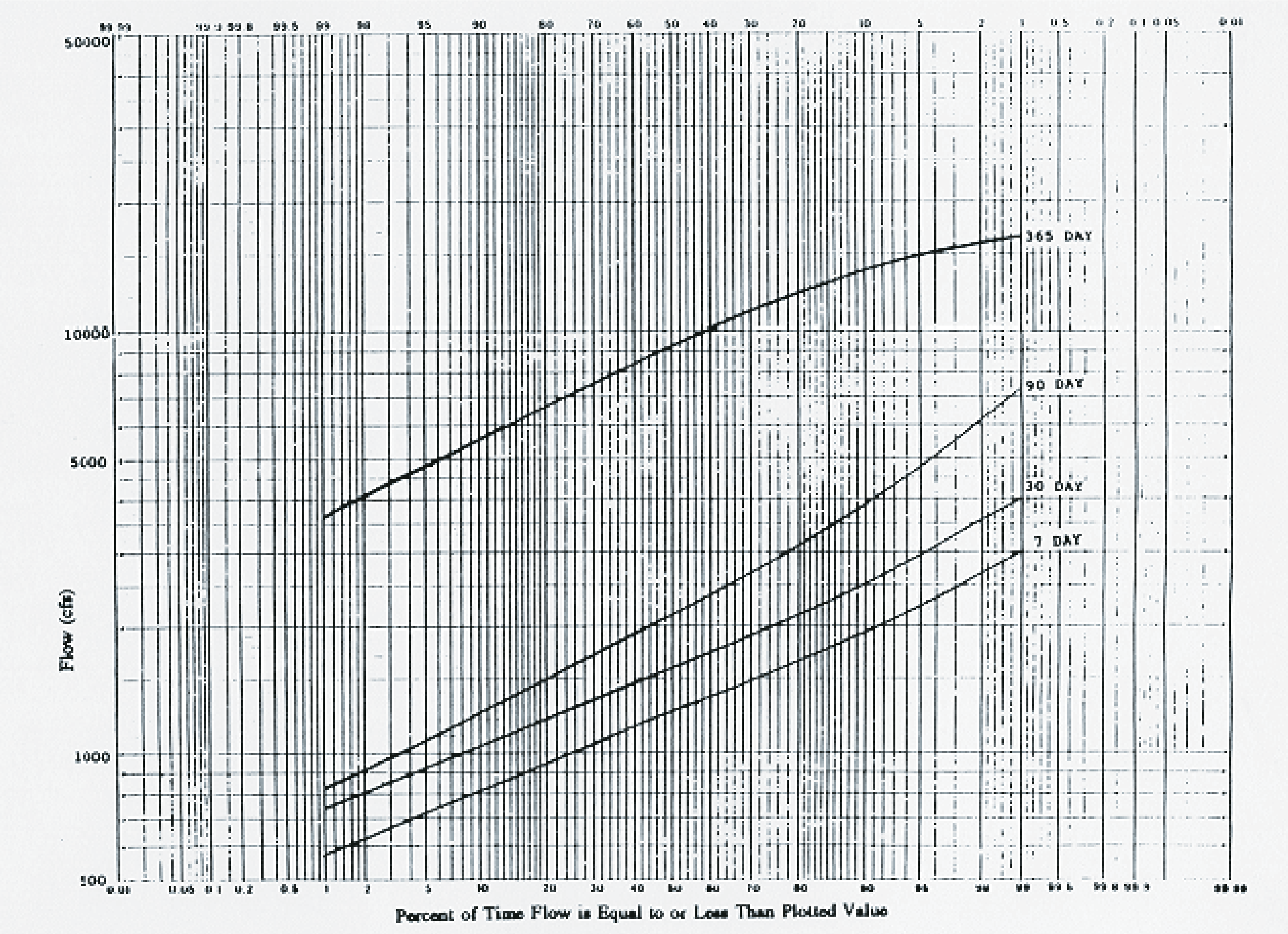


FIGURE A-2. EMPIRICAL RELATIONSHIPS FOR HYDRAULIC GEOMETRY



POTOMAC RIVER AT POINT OF ROCKS: 1891-1991
LOW FREQUENCY CURVES BY CLIMATE YEAR

FIGURE A-3.

TABLE A-18. LOW FLOW RECURRENCES, POTOMAC RIVER AT POINT OF ROCKS, MARYLAND (1891 - 1981)

Recurrence Interval (years)	Probability of Occurrence (% time flow ≤ flow plotted)	Mean Annual Low Flow (cfs) 7-day	30-day	90-day	365-day
5	20%	950	1200	1500	6600
10	10%	825	1005	1250	5600
50	2%	625	800	910	4000

A.3.3 Time-of-Travel Survey

Time of travel can be determined by several different methods. The three principal methods involve use of surface floats, use of a tracer such as a dye or radionuclide, and measurement of cross-sectional areas.

Surface floats can be followed downstream and timed for known distances to determine times of travel. This approach requires considerable judgment since floats tend to travel into eddies or become trapped on tree limbs, stream banks, or other obstacles. The floats must be frequently retrieved and returned to the current of the stream. The principal judgment factors are how long the floats should be left in quiet areas before retrieval and where they should be placed in the current. The surface water velocity is greater than the average for the entire stream, and a correction factor must be applied to the surface velocity; an average velocity equal to 85 percent of the surface velocity is a reasonable rule of thumb. Oranges make very good floats since their density is such that they float with only a small portion of their tops exposed to wind action and their color is easily detected and followed in the water.

Measurement of cross-sections at frequent longitudinal intervals and calculation of average velocity from the average cross-section and stream flow at the time of measurement constitute a time-consuming method of obtaining time of travel. This method does, however, produce information that is useful for other purposes. For example, reaeration coefficients may be calculated by one of the formulas based on average depths and velocities.

The most accurate method of measuring time of travel, and a good way to measure velocity, involves following a tracer downstream. An industrial waste may include an occasional discharge of some constituent that can serve as a tracer. Radioisotopes have given good results, but their safe handling can present problems. Several kinds of dyes have been used, including the recent trend of using Rhodamine WT. This dye can be detected in concentrations as low as 0.05 part per billion (μg/L) by a fluorometer. The dye is distributed across the stream at the upstream point, as nearly instantaneously as possible. Because of longitudinal mixing (see Figure 2-1), the dye arrives downstream in a wide band. The time of travel to a downstream point is the difference between the time the dye was released to the stream and the time the centroid of the dye mass arrives at the downstream point. Two handbooks by the USGS (Hubbard et al., 1982 and Wilson et al., 1986) provide step-by-step procedures to conduct a time-of-travel study. Figure A-4 shows the hydraulic geometry and time-of-travel data for the Kalamazoo River in Michigan under various stream flow conditions.

A.3.4 Longitudinal Dispersion Coefficient

Although the mechanics of longitudinal dispersion are well understood, general analytical treatment is extremely difficult, if not impossible. Theoretical development has led to a number of equations to calculate longitudinal dispersion in one-dimensional stream models. One equation suggested by Fischer et al. (1979) is:

$$D_x = \frac{0.011 U^2 W^2}{H U^*} \quad \text{(A-5)}$$

where

D_x = longitudinal dispersion coefficient (ft^2/sec)
U = average velocity (ft/sec)
W = river width (ft)
H = river depth (ft)
U^* = shear velocity (ft/sec)

Since the exact effect of irregularities cannot be included and, for most applications, results are insensitive to the exact value used, it is generally not

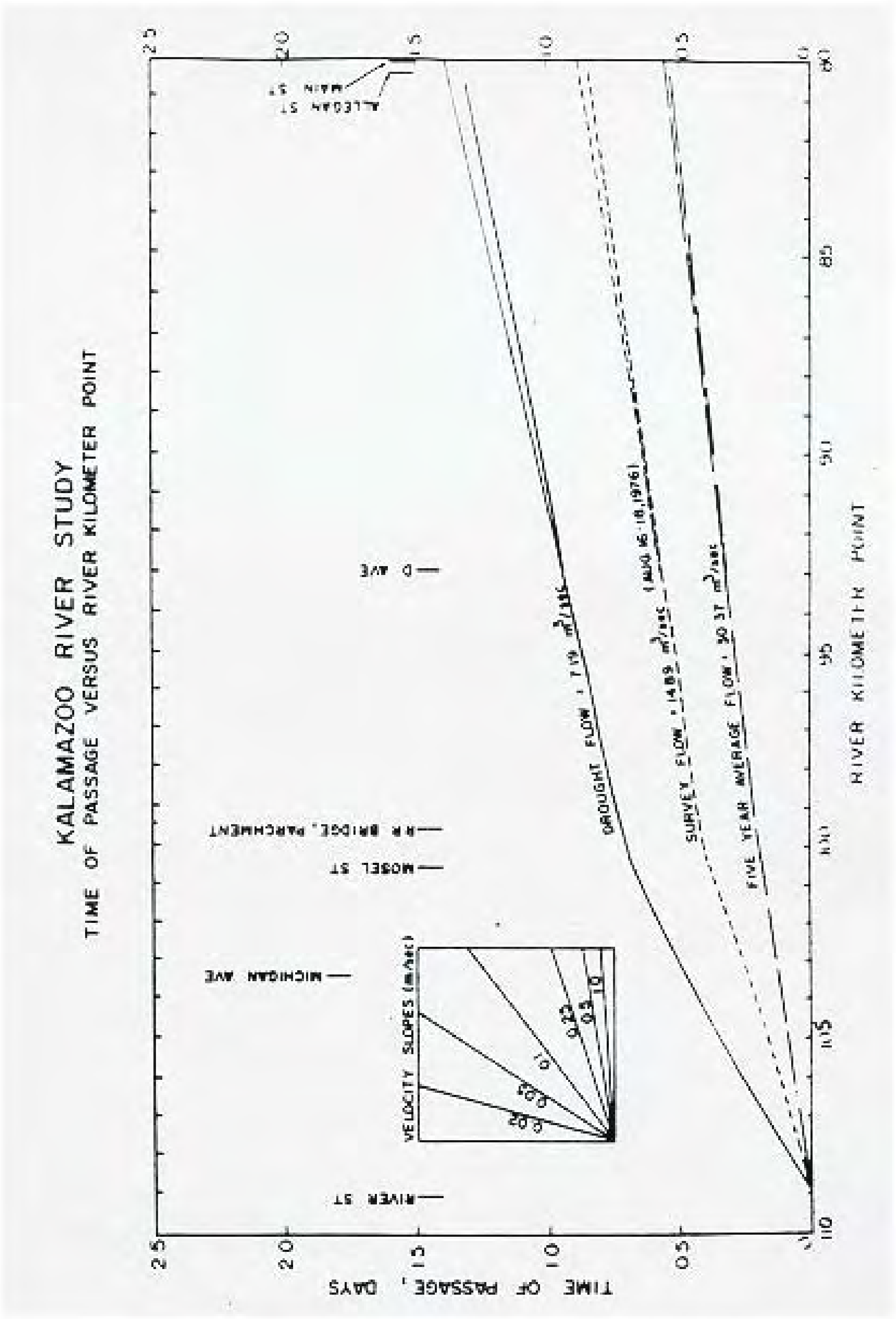


FIGURE A-4. HYDRAULIC GEOMETRY AND TIME OF TRAVEL necessary to achieve a high accuracy in predicting dispersion coefficients (Fischer et al., 1979). Equation A-5 provides a good approximation to longitudinal dispersion coefficients for a number of rivers, as shown in Table A-19.

A.4 CARBONACEOUS DEOXYGENATION RATE

A.4.1 Using Field Data to Determine K_r and K_d

The carbonaceous deoxygenation rate coefficient can be estimated from field data by plotting CBOD measurements versus time of travel on semi-log paper, based on Equation 2-7. The rate of deoxygenation is estimated as the slope of the line that best fits the data points:

$$K_r = -\frac{\ln(L/L_o)}{t} \qquad \text{(A-6)}$$

where

L = oxygen equivalence of the organic matter at any given location in the stream (mg/L)

L_o = total oxygen demand measured at the source of waste load following complete mixing (mg/L)

t = time of travel (day)

The above equation, based on concentrations, is applicable for constant hydraulic geometry. In practice, the river channel changes frequently and additional flow from tributaries may provide dilution to the river flow, all affecting the BOD concentrations. A more practical approach to estimating the deoxygenation rate coefficient is to plot the mass loading rate (kg/day) of BOD instead of concentration. The mass loading rate at any given location in the river is the product of measured BOD concentration and river flow. Figure A-5 shows such a plot for Rock Creek, Pennsylvania. Note that the tributaries provide dilution along the stream.

In Rock Creek, a two-stage curve is obtained. The first part of the curve is steep, showing that the total removal rate (K_r) results when both settling and biological oxidation are significant. The second part, a more gradual slope, generally represents the CBOD deoxygenation rate (K_d) after most of the settling has taken place. The settling rate (K_s) can then be estimated as the difference between K_r and K_d.

The above procedure is valid for either CBOD_5 or CBOD_u provided that the laboratory coefficients, K_1, are identical for all stations. If K_1 values vary from station to station, it is necessary to use CBOD_u.

Leo et al. (1984) evaluated the change in K_d following the installation of higher treatment levels at POTWs. The data indicate that K_d values associated with advanced treatment levels are generally lower than those determined for lesser treatment levels.

Algae can affect the CBOD data used to calculate K_d. Algal respiration and decay in the CBOD bottle can

TABLE A-19. LONGITUDINAL DISPERSION COEFFICIENT IN RIVERS (After Fischer et al., 1979)

River	*H* (m)	*W* (m)	*U* (m/sec)	*u** (m/sec)	D_x(m^2/sec) Measured	D_x(m^2/sec) Using Eq. A-5	References
Missouri River	2.70	200	1.55	0.074	1500	5290.8	Yotsukura et al. (1970)
Clinch River, Tennessee	0.85	47	0.32	0.067	14	43.7	
	2.10	60	0.94	0.104	54	100.2	Godfrey and Frederick (1970) (predicted D_x from Fischer, 1968)
	2.10	53	0.83	0.107	47	94.7	
Bayou Anacoco	0.94	26	0.34	0.067	33	13.6	
	0.91	37	0.40	0.067	39	39.5	
Nooksack River	0.76	64	0.67	0.27	35	98.6	
Wind/Bighom Rivers	1.10	59	0.88	0.12	42	224.6	McQuivey and Keefer (1974) (predicted D_x from Fischer, 1975)
	2.16	69	1.55	0.17	160	342.7	
John Day River	0.58	25	1.01	0.14	14	86.4	
	2.47	34	0.82	0.18	65	23.5	
Comite River	0.43	16	0.37	0.05	14	17.9	
Sabine River	2.04	104	0.58	0.05	315	392.4	
	4.75	127	0.64	0.08	670	191.2	
Yadkin River	2.35	70	0.43	0.10	110	42.4	
	3.84	72	0.70	0.13	260	66.0	

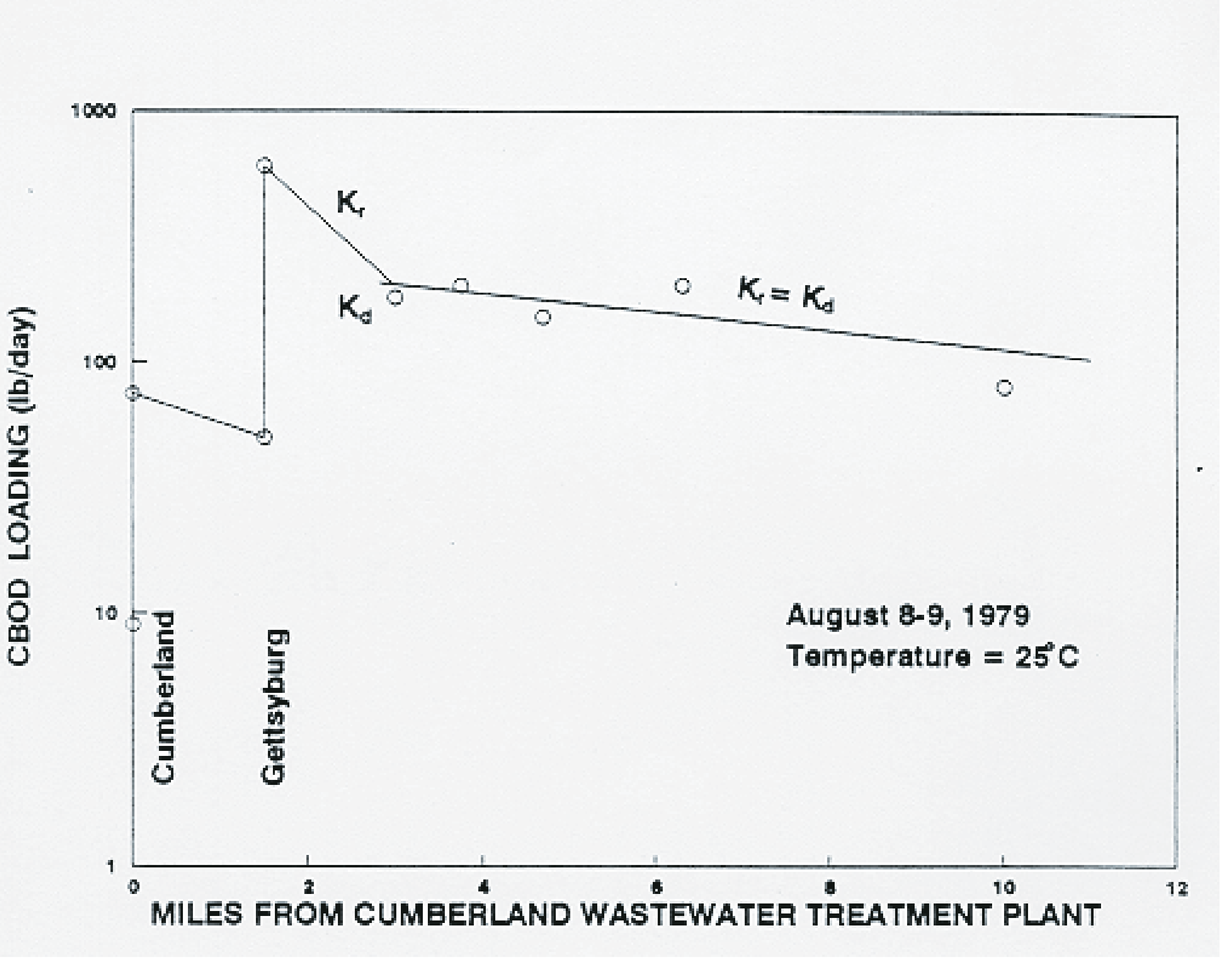


FIGURE A-5. DETERMINATION OF CBOD REMOVAL AND DEOXYGENATION RATES FOR ROCK CREEK, PENNSYLVANIA cause higher measured CBOD values and thus higher K_1 rates compared to samples without algae. In addition, if the concentration of suspended algae is not constant in the stream below the discharge, the measured CBOD concentrations would not indicate a defined K_d. Where the concentration of suspended algae increases in the stream, there may be a net increase of measured CBOD below the discharge. To minimize these effects, filtered CBOD and total CBOD samples should be analyzed at several locations downstream of the discharge. The oxygen demand resulting from settleable (i.e., filtered) organics is then accounted for separately in establishing the K_s (BOD settling) and SOD rates. However, even with filtered and unfiltered CBOD data, it is difficult to select rates for model calibration and projection analyses of algae-dominated streams. Where large concentrations of algae occur in the receiving stream, a range of K_d rates should be estimated based on filtered and unfiltered CBOD data. Some general rules of thumb follow:

- Algal impacts on K_d occur wherever high concentrations of chlorophyll *a* or large diurnal dissolved oxygen fluctuations are measured.
- 10 µg/L of chlorophyll *a* will increase the $CBOD_u$ concentration by 1 mg/L above levels without algae. Rough estimates can be obtained from multiples of this relationship; i.e., 100 µg/L of chlorophyll *a* may increase the concentration by 10 mg/L.
- If the stream is effluent-dominated with most of the CBOD originating from the discharge rather than algae, filtering may not be needed or the number of filtered CBOD analyses may be decreased. If the stream is not effluent-dominated and most of the CBOD is from algae, filtered and unfiltered samples should be run.

A.4.2 Projection of Carbonaceous Deoxygenation Rates

In wasteload allocations, K_d rates are projected for future conditions. Literature data have been compiled (Figure A-6) to correlate K_d with the stream depth in lieu of any other parameters. The rationale behind this correlation is that the greater the wetted perimeter, the greater the contact with the biological community in the streambed, the most important factor in natural oxidation processes. The tendency for this relation to hold is greater for rocky streambeds than for silty beds. However, the general trend appears reasonable up to depths of about 5 to 10 ft. Hydroscience (1971) developed the following relationship:

$$K_d = 0.3\left[\frac{H}{8}\right]^{-0.434} \quad \text{for } 0 < H < 8 \qquad \text{(A-7a)}$$

$$= 0.3 \quad \text{for } H > 8 \qquad \text{(A-7b)}$$

where H is the depth in feet. Wright and McDonnell (1979) suggested the following:

$$K_d = 10.3\, Q^{-0.49} \qquad \text{(A-8)}$$

where Q is stream flow in cfs. They also indicated that for flow conditions greater than 800 cfs, K_d rates were consistent with K_1 for the effluent. In other words, for larger and deeper streams (greater than 10 ft), the characteristics of the streambed become less of a factor and the level of treatment would dictate the following K_d values:

Primary	0.4 day^{-1}
Intermediate	0.3 day^{-1}
Secondary	0.2 day^{-1}
Advanced	0.1 day^{-1}

That is, for increasing levels of treatment, the residual waste contains a large proportion of refractory organisms and will be less easily oxidized since the treatment processes are designed to oxidize the labile components of the organic matter.

A.4.3 Literature Values of K_d

Bowie et al. (1985) summarized a large number of carbonaceous deoxygenation rate coefficients reported in the literature (Table A-20). Leo et al. (1984) provided a comparison of K_d rates before and after the improvement in stream dissolved oxygen conditions following treatment upgrades (Table A-21).

A.5 NITROGENOUS DEOXYGENATION (NITRIFICATION) RATE

A.5.1 Using Field Data to Determine K_n

The procedure described in Section A.4.1 (see Figure A-5) may be used to develop the nitrification coefficient, K_n, for NBOD. Either the nitrogenous components of the laboratory BOD test or the directly measured ammonia concentrations at the various locations in the stream may be used. Each should yield the same value of K_n following the first-order reaction. The displacement of the curves is due to the stoichiometric relation between oxygen and am-

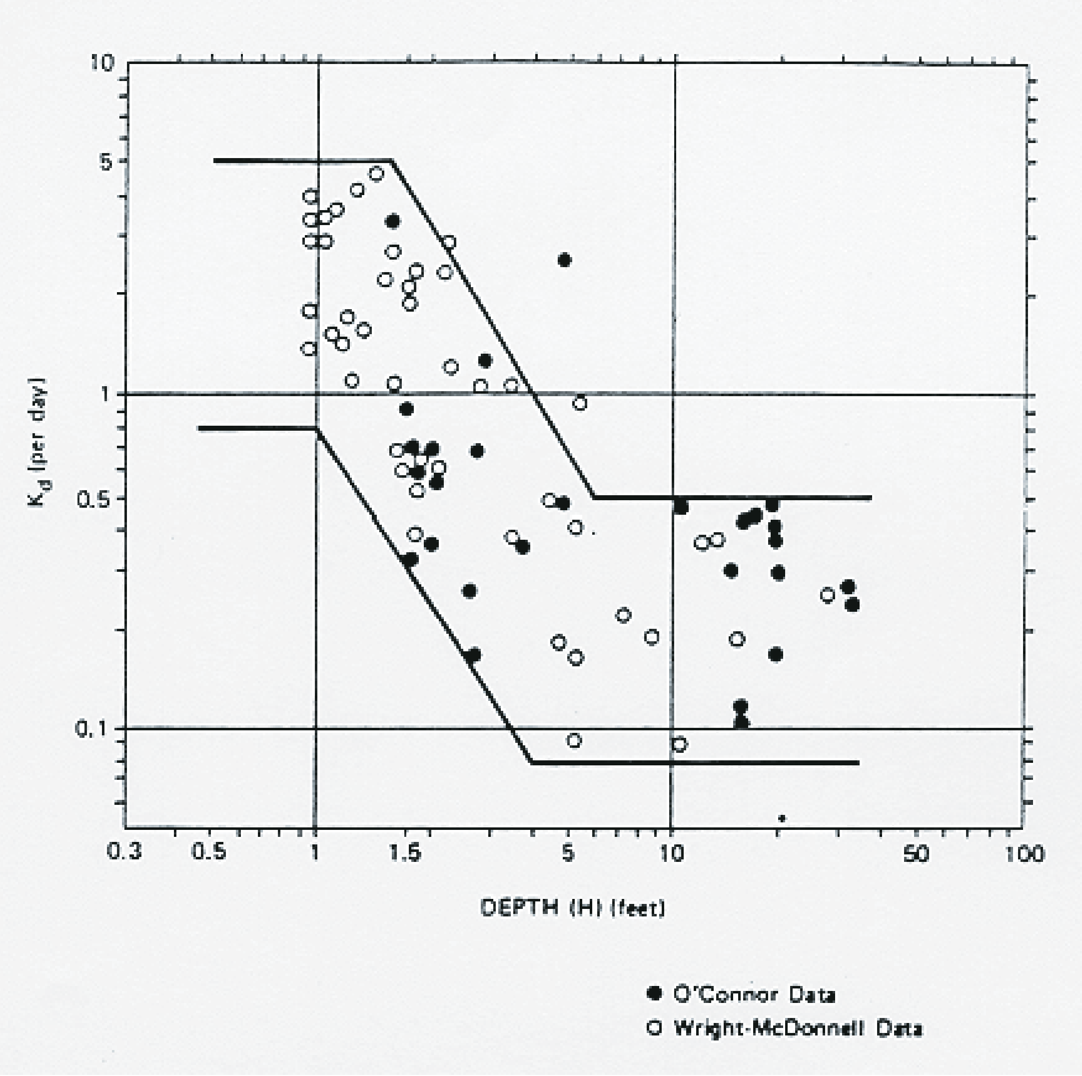


FIGURE A-6. DEOXYGENATION COEFFICIENT (K_d) AS A FUNCTION OF DEPTH
(A benthic deoxygenation component is included in these values)
(After Hydroscience, 1971)

TABLE A-20. VALUES OF KINETIC COEFFICIENTS FOR DECAY OF CARBONACEOUS BOD (After Bowie et al., 1985)

Location	K_d (day^{-1} @20 °C, base e)	Method Determining Coefficient	Reference
Potomac Estuary 1977	0.14±0.023	field study	USEPA (1979b)
Potomac Estuary 1978	0.16±0.05		USEPA (1979a)
Willamette River, OR	0.1-0.3		Baca et al. (1973)
Chattahoochee River, GA	0.16		Bauer et al. (1979)
Ganga River, India	3.5-5.6(K_r)	field study	Bhargava (1983)
Yamuna River, India	1.4		
S. Fork, Shenandoah River	0.4(Kr)	field study	Deb and Bowers (1983)
Merrimack River, MA	0.01-0.1	field study	Camp (1965)
Gray’s Creek, LA	1.44(K_r)	model calibration	Crane and Malone (1982)
Onondaga Lake, NY	0.10	model calibration	Freedman et al. (1980)
Yampa River, CO	0.40	model calibration	Grenney and Kraszewski (1981)
Skravad River, Denmark	0.15	field study	Hvitved-Jacobsen (1982)
	0.90(K_r)		
Seneca Creek	0.008		MWCOG (1982)
Kansas (6 rivers)	0.20-0.60	various methods	Reported by Bansal (1975)
Michigan (3 rivers)	0.56-3.37		
Truckee River, NV	0.36-0.96		
Virginia (3 rivers)	0.30-1.25		
N. Branch, Potomac, WV	0.4		
South Carolina (3 rivers)	0.3-0.35		
New York (2 rivers)	0.125-0.4		
New Jersey (3 rivers)	0.2-0.23		
Houston Ship Channel, TX	0.25		
Cape Fear R. Estuary, NC	0.23		
Holston River, TN	0.4-1.5	model calibration	Novotny and Krenkel (1975)
New York Bight	0.05-0.25		O’Connor et al. (1981)
White River, AR	0.004-0.66(K_1)	laboratory study	Terry et al. (1983)
N. Fork Kings River, CA	0.2		Tetra Tech (1976)
Lake Washington, WA	0.2		Chen and Orlob (1975)
Ouachita River, AR	0.15	calibration	Hydroscience (1979)
	0.17(K_1)		
	0.02(K_1)	laboratory study	NCASI (1982a)
36 U.S. river reaches plus laboratory flume	0.08-4.24	field studies	Wright and McDonnell (1979)
San Francisco Bay Estuary	0.2		Chen (1970)
Boise River, ID	0.75		Chen and Wells (1975)
W. Fork, Trinity River, TX	0.06-0.30		Jennings et al. (1982)
Wilamette River, OR	0.07-0.14	laboratory study	McCutcheon (1983)
Arkansas River, CO	1.5	lab and field study	
Lower Sacramento River, CA	0.41		Hydroscience (1972)
Delaware River Estuary	0.31		
Wappinger Creek Estuary, NY	0.31		
Potomac Estuary	0.16, 0.21		Thomann and Fitzpatrick (1982)
Speed River, Ontario	1.0	field study	Gowda (1983)

TABLE A-21. SUMMARY OF PRE- AND POST-IMPROVEMENT OXIDATION RATES (After Leo et al., 1984)

		Pre-Improvement			Post-Improvement	
River	Treatment	CBOD Oxidation Rate (day^{-1})	NBOD Oxidation Rate (day^{-1})	Treatment	CBOD Oxidation Rate (day^{-1})	NBOD Oxidation Rate (day^{-1})
Main Stem Patuxent	Secondary	0.61[a]	0.76[a]	Secondary and Nitrification	0.30[a]	0.46[a]
Wilsons Creek	Secondary	0.30	0.40	Secondary and Nitrification and Polishing	0.30	0.40
Hurricane Creek	Trickling Filter	0.10-0.50	0.10-0.50	Secondary[b]	0.35	0.70
Cibolo Creek	Secondary	0.18	0.25	Secondary[c]	0.18	0.25
Clinton River	Secondary	2.20	0	Secondary and P- Removal	0.20	2.5-3.8
Hudson River	Primary	0.25	0	Secondary	0.15	0
South River	Secondary	-	1.6-2.0	Secondary and Nitrification	-	1.6-2.0

[a] From reference (23) cited in Leo et al. (1984), not from calibration analysis.
[b] Oxidation ditch achieving nitrification.
[c] New facility achieving nitrification.

monia, which is approximately 4.57. Figure A-7 shows the determination of K_n for the Shenandoah River.

Benthic ammonia regeneration, benthic nitrification, and large concentrations of algae, either suspended or attached, markedly affect values obtained for K_n. The approach shown in Figure A-7 to determine K_n reflects only the net loss of ammonia. Such an approach can result in the overestimation of K_n where significant algal or attached periphyton effects occur. Algae consume ammonia as a nutrient; therefore, a K_n determination based only on the loss of ammonia would include uptake of ammonia by algae as well as ammonia oxidation. In some cases, using the rate of nitrate increase is a better approach for estimating K_n because nitrate increases result directly from ammonia oxidation. However, as a cautionary note, under some conditions algae can consume nitrate as well as ammonia. In addition, benthic denitrification can account for a significant component of the total nitrogen balance as a loss mechanism for nitrate (Seitzinger, 1988). Therefore, the K_n rate derived from nitrate data would represent the minimum K_n. A critical issue in determining K_n involves the time of year. Although ammonia and nitrate data may indicate relatively high K_n rates during July and August, K_n rates for the same stream may be negligible during the winter months and even during transitional periods such as April, May, June, September, October, and November. Seasonal adjustments in K_n using the temperature correction relationships can be found in Section A.12. During months when the temperature falls below 20 °C, nitrogen series data collected during these cooler periods may be necessary to select appropriate K_n rates to support cool-weather nitrification.

A.5.2 Projection of Nitrogenous Deoxygenation Rates

K_n rates applied to deep and slow-moving rivers, without site-specific data, are higher than those normally expected for such rivers. The availability of surfaces for nitrifier attachment can affect K_n rates. These surfaces include the stream bottom and suspended particles in the water column. Therefore, shallow streams with rocky bottoms favor the growth of nitrifying bacteria with associated high K_n rates. Deep rivers composed of sands, silts, or clays generally have fewer attached nitrifiers. Although these streams may support significant populations of nitrifiers in the water column, they tend to have lower K_n values than shallow streams.

One of the difficult issues related to the determination of K_n is the onset of the nitrification process. It is known that there are lags in the nitrification process in highly polluted streams or those with low dissolved oxygen. In projection analyses, a critical question is

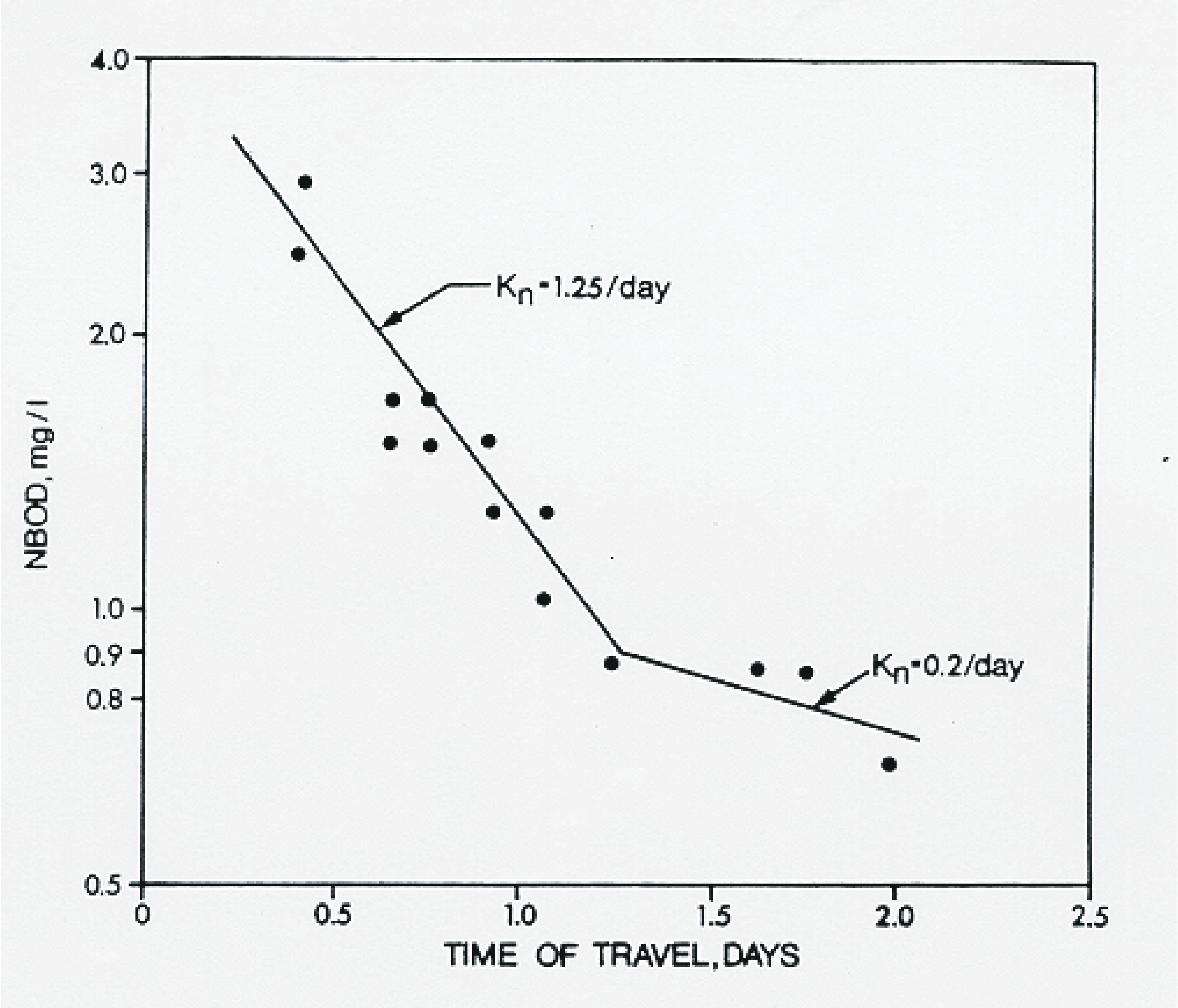


FIGURE A-7. NITROGENOUS BIOCHEMICAL OXYGEN DEMAND VERSUS TRAVEL TIME IN SHENANDOAH RIVER
(Deb and Bowers, 1983)

when and where nitrification should occur following improved treatment such as nitrification at the plant.

A.5.3 Literature Values of K_n Rates

A large body of literature exists for case studies of K_n rates in streams and rivers. Tables A-22 and A-23, taken from Bowie et al. (1985), summarize K_n rates measured in the field and used as parameter values for models from a number of investigations.

A.6 STREAM REAERATION RATE

A.6.1 Measuring Stream Reaeration Rate

Several methods of measuring K_a rates have been reported in the literature. These methods are discussed in detail by Bennett and Rathbun (1972). Churchill et al. (1962) used a dissolved oxygen balance method to measure reaeration in several Tennessee rivers. Owens et al. (1964) used a similar technique for small English streams. Both required that all other factors influencing the oxygen balance be known or negligible. Because of the difficulty in accurately measuring the other factors, these techniques are subject to considerable error. Tsivoglou (1967) developed a gas-tracer method for directly measuring gas transfer in streams, eliminating the need for the oxygen balance information. In this method, a fluorescent tracer is used for determining time of travel and longitudinal dispersion, tritium is used as an indicator of total dispersion, and krypton-85 is used as a gaseous tracer. Of the various methods, the gas-tracer method is superior since it does not require estimating any other factor affecting the oxygen balance. This method, however, requires the handling and the injecting of radioactive tracers into streams. Application of the method is limited because the use of radioactive tracers in the natural environment is subject to public health restrictions. Rathbun et al. (1975) modified Tsivoglou's method to use nonradioactive hydrocarbons (ethylene or propane) as the gaseous tracers. Additional information about the method is given in Rathbun and Grant (1978). Whittemore (1990a) has recently reported a nonradioactive technique for determining reaeration rates.

A.6.2 Predicting Stream Reaeration Coefficient

Various predictive equations are discussed in Bennett and Rathbun (1972) and Rathbun (1977). One of the most popular theoretical formulas was that of O'Connor and Dobbins (1958):

$$K_a = \frac{12.9\,U^{1/2}}{H^{3/2}} \tag{A-9}$$

where

U = average stream velocity (ft/sec)

H = average stream depth (ft)

This formulation was derived from theoretical considerations regarding surface renewal of the liquid film through internal turbulence. Table A-24 lists a number of other conceptual, empirical, and semi-empirical predictive equations found in the literature. The model QUAL2E offers eight different formulations.

The variability of K_a estimates using these formulas is illustrated in Figure A-8. As shown in this figure, the range of K_a values calculated with the O'Connor, Owens, Churchill, and Tsivoglou equations is significant at all flow levels in the Muskingum River. Although these formulas may not be equally applicable to this particular river, this result illustrates the differences in the K_a values calculated using these four equations. For some allocations, K_a values derived from empirical formulas could result in the degree of uncertainty exceeding the degree of projected water quality improvement from the proposed treatment plant upgrades. Using a predictive equation developed for the receiving water conditions similar to those being modeled is <u>extremely</u> important (see the applicability column in Table A-24).

In addition to the numerous studies compiled in Bowie et al., (1985), Whittemore (1990b) has compiled a data base of stream reaeration measurements obtained over a 30-year period under a wide range of environmental and hydraulic conditions. The data base and computer software for querying the data base are available to interested investigators from NCASI for a modest cost. Using a personal computer, the data base can be queried to extract field study measurements obtained under a range of environmental criteria specified by the analyst (i.e., depth, velocity, temperature, etc.).

A.6.3 Dam Reaeration

The QUAL2E model uses the following equation described by Butts and Evans (1983) to estimate oxygen input from dam reaeration:

$$D_a - D_b = \left[1 - \frac{1}{1 + 0.116\, a\, b\, H\,(1 - 0.034H)\,(1 + 0.046T)}\right] D_a \tag{A-10}$$

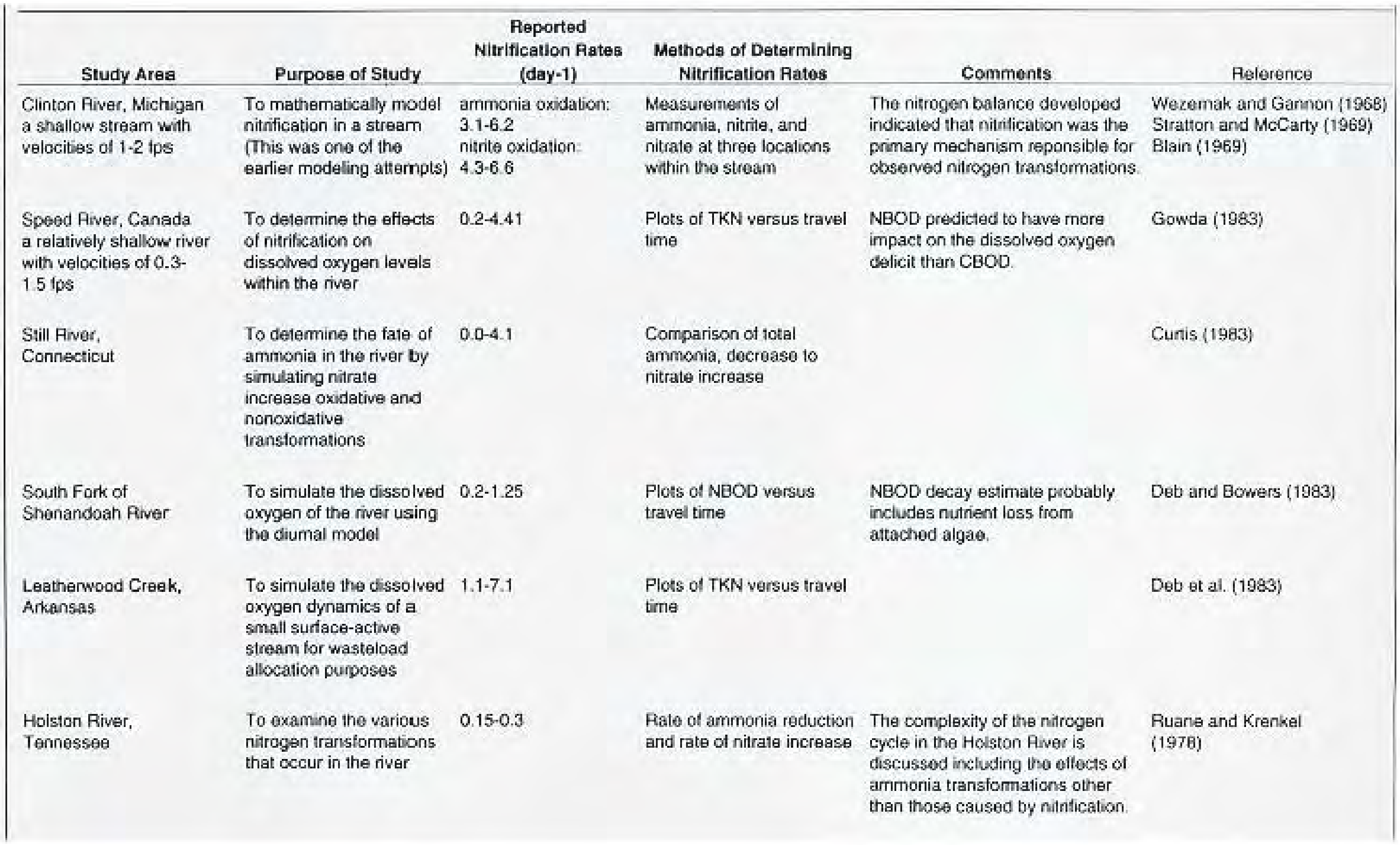
TABLE A-22. SUMMARY OF NITRIFICATION RATES (day^{-1})
(After Bowie et al., 1985)

River	Maximum	Average	Minimum	Reference
Grand River, MI	3.9	2.6	1.9	Courchaine (1968)
Clinton River, MI	15.8 4.0	5.7 1.9	2.2 0.4	Wezernak and Gannon (1968)
Truckee River, NV	2.4	1.9	-	O'Connell and Thomas (1965)
South Chickamaugo Creek, TN	1.9	-	1.1	Ruane and Krenkel (1978)
Oostanaula Creek, TN	0.8	-	0.1	Ruane and Krenkel (1978)
Town Branch, AL	-	0.7	0	Ruane and Krenkel (1978)
Chattahoochee River, GA	-	0.44	-	Stamer et al. (1979)
Willamette River, OR	0.7	-	0.4	Rinella et al. (1981)
Flint River, MI	2.5	1.4	0.1	Bansal (1976)
Upper Mohawk River, NY	0.3	0.25	0.25	Bansal (1976)
Lower Mohawk River, NY	0.3	0.3	0.3	Bansal (1976)
Barge Canal near Upper Mohawk River, NY	0.25	0.25	0.25	Bansal (1976)
Ohio River, OH	0.25	0.25	0.25	Bansal (1976)
Big Blue River, NB	0.25	0.11	0.03	Bansal (1976)
Delaware River Estuary, DE	0.54	0.3	0.09	Bansal (1976)
Willamette River, OR	-	0.75[a] 1.05[b]	-	Alvarez-Montalvo et al. (undated)
Ouachita River, AR and LA	-	0.1[a] 0.5[b]	-	NCASI (1982a)
Potomac Estuary	-	0.09-0.13	-	Thomann and Fitzpatrick (1982)
Lake Huron and Saginaw Bay	-	0.20	-	DiToro and Matystik (1980)
New York Bight, NY	-	0.025	-	O'Connor et al. (1981)

Note: Nitrification rates are in units of day^{-1}.

[a] Ammonia oxidation.

[b] Nitrate oxidation.



Study Area	Purpose of Study	Reported Nitrification Rates (day-1)	Methods of Determining Nitrification Rates	Comments	Reference
Clinton River, Michigan a shallow stream with velocities of 1-2 fps	To mathematically model nitrification in a stream (This was one of the earlier modeling attempts)	ammonia oxidation: 3.1-6.2 nitrite oxidation: 4.3-6.6	Measurements of ammonia, nitrite, and nitrate at three locations within the stream	The nitrogen balance developed indicated that nitrification was the primary mechanism reponsible for observed nitrogen transformations.	Wezernak and Gannon (1968) Stratton and McCarty (1969) Blain (1969)
Speed River, Canada a relatively shallow river with velocities of 0.3-1.5 fps	To determine the effects of nitrification on dissolved oxygen levels within the river	0.2-4.41	Plots of TKN versus travel time	NBOD predicted to have more impact on the dissolved oxygen deficit than CBOD.	Gowda (1983)
Still River, Connecticut	To determine the fate of ammonia in the river by simulating nitrate increase oxidative and nonoxidative transformations	0.0-4.1	Comparison of total ammonia, decrease to nitrate increase		Curtis (1983)
South Fork of Shenandoah River	To simulate the dissolved oxygen of the river using the diurnal model	0.2-1.25	Plots of NBOD versus travel time	NBOD decay estimate probably includes nutrient loss from attached algae.	Deb and Bowers (1983)
Leatherwood Creek, Arkansas	To simulate the dissolved oxygen dynamics of a small surface-active stream for wasteload allocation purposes	1.1-7.1	Plots of TKN versus travel time		Deb et al. (1983)
Holston River, Tennessee	To examine the various nitrogen transformations that occur in the river	0.15-0.3	Rate of ammonia reduction and rate of nitrate increase	The complexity of the nitrogen cycle in the Holston River is discussed including the effects of ammonia transformations other than those caused by nitrification.	Ruane and Krenkel (1978)

TABLE A-23. CASE STUDIES OF NITRIFICATION IN NATURAL WATERS (Bowie et al., 1985)

TABLE A-23. (Continued)

Study Area	Purpose of Study	Reported Nitrification Rates (day^{-1})	Methods of Determining Nitrification Rates	Comments	Reference
Iowa and Cedar Rivers, Iowa	To determine the locations and rates of nitrification downstream from two wastewater treatment plants	0.5-9.0	Rate of ammonia reduction and rate of nitrate increase	Algal assimilation of ammonia appeared to be an important transformation process. Laboratory rates of nitrification varied from 0.02 to 0.35.	Koltz (1982)
Morgan Creek, Ruin Creek, and Little Lick Creek, three shallow streams in North Carolina	To determine the distributions of nitrifying organisms and to examine the nitrogen transformation occurring in the streams	-	-	Nitrifying organisms were enumerated in the water column and in the top 1 cm of sediments. The populations were much larger in the sediments, which indicated that nitrification was occurring predominantly in the sediments and not in the water column.	Kreutzberger and Francisco (1977)
Passaic River, New Jersey	To determine whether nitrification was a significant process in the Passaic River	-	-	There were high ammonia nitrogen concentrations in the river with relatively little nitrification occurring. The potential for nitrification appeared high and was expected to be exerted if water quality within the river improved.	Cirello et al. (1979)
Passaic River, New Jersey	To determine the distribution of nitrifying bacteria in the river	-	-	Nitrifying bacteria were found to be from 21 to 140,000 times more abundant volumetrically in sediments than in the water column.	Finstein and Matulewich (1974)
Potomac Estuary	To determine factors important in the oxygen balance within the estuary	0.10-0.14	Thomas Graphical Method		Slayton and Trovato (1978, 1979)

TABLE A-24. REAERATION COEFFICIENTS FOR RIVERS AND STREAMS
(Bowie et al., 1985)

K_a, base e (day^{-1} at 20 °C)	Units	Applicability	Author(s)
$\frac{12.9U^{0.5}}{H^{1.5}}$	U-fps H-feet	Moderately deep to deep channels; 1 ft ≤ H ≤ 30 ft, 0.5 ≤ U ≤ 1.6 fps. 0.5 ≤ k_2 ≤ 12.2/day. O'Connor and Dobbins also developed a second formula for shallow streams, but O'Connor (1958) showed that the differences between the two formulas was insignificant and recommended that the first formula be used.	O'Connor and Dobbins (1958)
$\frac{11.6U^{0.969}}{H^{1.673}}$	U-fps H-feet	Based on observed reaeration rates below dams from which oxygen-deficient water was released. 2 ≤ H ≤ 11 ft; 1.8 ≤ U ≤ 5 fps. Churchill et al. also developed other formulas, but recommended this formula.	Churchill et al. (1962)
$\frac{21.7U^{0.67}}{H^{1.85}}$	U-fps H-feet	Oxygen recovery monitored for six streams in England following deoxygenation with sodium sulfite. 0.4 ≤ H ≤ 11 ft; 0.1 ≤ U ≤ 5 fps.	Owens et al. (1964)
$\frac{23.3\ U^{0.73}}{H^{1.75}}$	U-fps H-feet	This is a second formula developed by Owens et al., and it applies for 0.4 ≤ H ≤ 11 ft; 0.1 ≤ 1.8 fps.	Owens et al. (1964)
$\frac{7.6\ U}{H^{1.33}}$	U-fps H-feet	Based on synthesis of data from O'Connor-Dobbins (1958), Churchill et al. (1962), Krenkel and Orlob (1963), and Streeter et al. (1936). (Also known as USGS equation.)	Langbein and Durum (1967)
$\frac{8.62\ U}{H^{1.5}}$	U-fps H-feet	Developed using regression analyses from data collected using a recirculating cylindrical tank. 0.5 ≤ H ≤ 1.5 ft; 0.6 ≤ U ≤ 1.6 fps.	Issacs and Gaudy (1968)
$\frac{48.4\,(1+0.17F^2)(SU)^{3/8}}{H}$	U-m/s S-m/m H-meters	Developed from data collected in 12 sewers and in natural streams.	Parkhurst and Pomeroy (1972)
$10.9\left(\frac{U}{H}\right)^{0.85}$	U-fps H-feet	Developed from a recirculating flume with depths less than 0.5 feet.	Negulescu and Rojanski (1969)
$\frac{24.9\,(1+F^{0.5})u_*}{H}$	u*-fps H-feet	Based on measurements made in 2ft wide flume with deoxygenated waters. 0.5 ≤ H ≤ 0.23 ft.	Thackston and Krenkel (1969)
$2515\left(\frac{u_*}{u}\right)\frac{3u}{H}$	u*-fps U- fps H-feet	Based on reanalysis of the data of Thackston and Krenkel (1969), Krenkel (1960), and Churchill et al. (1962).	Lau (1972)
$\frac{234\,(US)^{.408}}{H^{.66}}$	U-fps S-ft/ft H-feet	Based on 1-ft-wide flume data. 0.08 ≤ H ≤ 0.2 ft	Krenkel and Orlob (1962)
$\frac{8.4D_L^{1.321}}{H^{2.32}}$	D_L-ft^2/min H-feet	Experiments performed in a 1-ft-wide flume by deoxygenating the water. Other similar formulas are also reported. The flume dispersion coefficient, D_L, was below the range expected in natural systems.	Krenkel and Orlob (1962)
$\frac{6.9U^{0.703}}{H^{1.054}}$	U-fps H-feet	Regression analysis performed on data where 9.8 < k_2 < 28.8/day.	Padden and Gloyna (1971)
$\frac{336\,(US)^{0.5}}{H}$	U-fps S-ft/ft H-feet	Based on multivariate analysis of reaeration data.	Cadwallader and McDonnel (1969)

TABLE A-24. (Continued)

K_a, base e (day^{-1} at 20 °C)	Units	Applicability	Author(s)
$\frac{4.67U^{0.6}}{H^{1.4}}$	U-fps H-feet	Based on reanalysis of reaeration data in numerous rivers.	Bansal (1973)
$\frac{106U^{0.413}S^{0.273}}{H^{1.408}}$ $\frac{20.2U^{0.607}}{H^{1.689}}$	U-fps S-ft/ft H-feet	These two equations are based on a reanalysis of historical data, with the second equation being almost as good a predictor as the first, but not having the slope term.	Bennett and Rathbun (1972)
$\frac{117\,(1+F^2(US)^{0.375})}{(0.9+F)^{1.5}H}\coth\left[\frac{4.10\,(US)^{0.125}}{(0.9+F)^{0.5}}\right]$	U-fps H-feet S-ft/ft	Theory combined with measurements in natural streams, and flume data of Krenkel and Orlob (1963).	Dobbins (1964)
$\frac{37W^{2/3}S^{1/2}U^{7/6}g^{1/2}}{Q^{1/3}}$	W-feet S-ft/ft U-fps Q-ft^3/sec	Based on data collected in several small Oregon streams.	Ice and Brown (1978)
$\frac{-1n\left[1-2\left(\frac{\alpha/24}{\pi(30.48H)^2}\right)^{0.5}\right]}{I}$ $I=\begin{cases}0.0016+0.0005H & H\leq 2.26ft\\ 0.0097\ \ln(H)-0.0052 & H\leq 2.26ft\end{cases}$	H-feet $\alpha = 1.42\,(1.1)^{T-20}$ T-°C	Based on the Velz method (1970) and replaces the iterative techniques. The expressions for the mix internal I are based on an accumulation of applications of the Velz technique.	McCutcheon and Jennings (1982)
$\frac{1.923\,U^{0.273}}{H^{0.894}}$	U-m/sec H-meters	Known as the Texas equation. Based on data collected on streams in Texas.	Long (1984)
$0.30+0.19S^{0.5}$ at $25°C$	S-ft/mi	Radioactive tracer technique used on small streams in Kentucky. 0 < S< 42 ft/mi.	Foree (1976)
$0.888\,(0.63-0.4S^{1.15})q^{0.25}$ at $25\ °C$ for $0.05\leq q\leq 1$ $0.888\,(0.63+0.4S^{1.5})$ at $25\ °C$ for $q>1$ $0.42\,(0.63+0.4S)^{1.15}$ at $25\ °C$ for $q<0.05$	S-ft/mi q-cfs/mi^2	Reanalysis of Foree's (1976) data.	Foree (1977)
$0.054\frac{\Delta h}{t}$ at $25\ °C$	Δh-feet t-days	Based on summary of radioactive tracer applications to 5 rivers.	Tsivoglou and Wallace (1972)
$0.11\left(\frac{\Delta h}{t}\right)$ for $1\leq Q\leq 10$ cfs $0.054\left(\frac{\Delta h}{t}\right)$ for $25\leq Q\leq 3000$ cfs	Δh-feet t-days	Based on data collected on 24 different streams using radioactive tracer method.	Tsivoglou and Neal (1976)
$0.09\left(\frac{\Delta h}{t}\right)$ at $25°\ C$	Δh-feet t-days	Based on data from 10 small streams in Wisconsin using radioactive tracer techniques: $2.1\leq k_2\leq 55$/day $1.2\leq S\leq 70$ ft/mile $0.3\ Q\leq 37\leq$ cfs	Grant (1976)

TABLE A-24. (Continued)

K_a, base e (day^{-1} at 20 °C)	Units	Applicability	Author(s)
$0.06\left(\frac{\Delta h}{t}\right)$ at 25 °C	Δh-feet t-days	Based on radioactive tracer data developed on Rock River, Wisconsin and Illinois: $0.01 \leq k_2 \leq 0.8$/day $0.25 \leq U \leq 1.6$ fps $0.2 \leq S \leq 3.5$ ft/mi $260 \leq Q \leq 1030$ cfs	Grant (1978)
$0.08\left(\frac{\Delta h}{t}\right)$ at 25 °C for $Q \leq 10$ cfs $0.06\left(\frac{\Delta h}{t}\right)$ at 25 °C for $10 \leq Q \leq 280$ cfs	Δh-feet t-days	Based on statistical analyisis of reaeration coefficients for rivers in 7 states, where the radioactive tracer method was used to find the reaeration rates.	Shindala and Traux (1980)

Eloubaidy and Plate (1972)	Wind effects analyzed. See text for discussion.
Mattingly (1977)	Wind effects analyzed. See text for discussion.
Gulliver and Stefan (1981)	Wind effects analyzed.
Frexes et al. (1984)	Wind effects analyzed.

Definitions of Symbols:

D_L = longitudinal dispersion coefficient
F = Froude number
$= \frac{U}{(gh)^{0.5}}$
g = acceleration due to gravity
Δh = change in stream bed elevation between two points
R = stream discharge divided by drainage area
S = slope
t = travel time between two points where Δh measured
U = stream velocity
u_* = shear velocity = $\sqrt{gRS}$
W = width

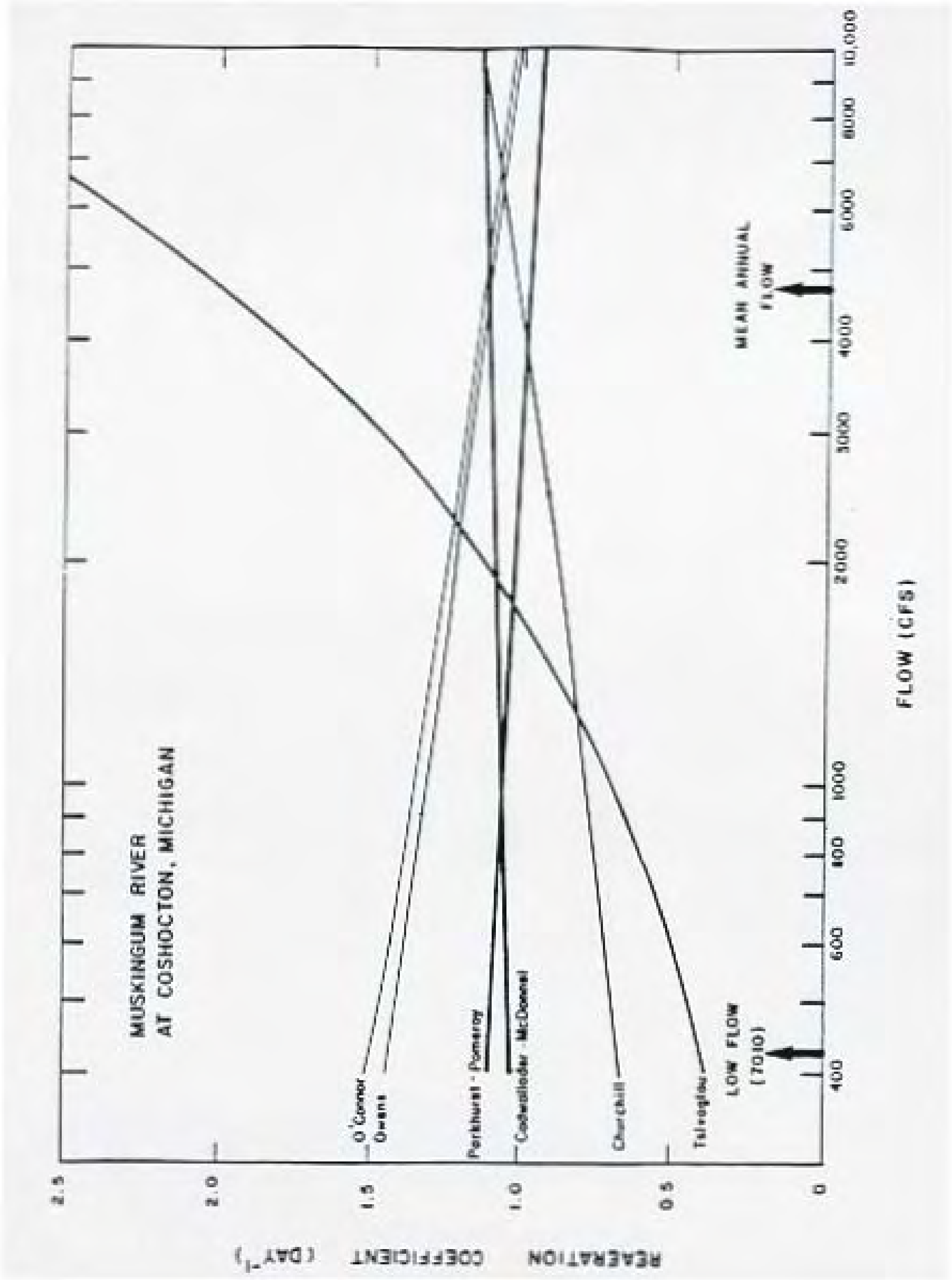


FIGURE A-8. STREAM REAERATION ACCORDING TO SIX FORMULAS where

D_a = dissolved oxygen deficit above dam (mg/L)
D_b = dissolved oxygen deficit below dam (mg/L)
T = temperature (^{o}C)
H = height through which water falls (ft)
a = 1.80 in clean water
= 1.60 in slightly polluted water
= 1.00 in moderately polluted water
= 0.65 in grossly polluted water
b = 0.70 to 0.90 for flat, broad crested weir
= 1.05 for sharp crested weir with straight slope face
= 0.80 for sharp crested weir with vertical face
= 0.05 for sluice gates with submerged discharge

The parameters *H, a,* and *b* need to be assigned for each dam.

A.6.4 Saturated Dissolved Oxygen Concentration

The solubility of oxygen in water is dependent on water temperature, barometric pressure (i.e., altitude), and the concentration of dissolved solids in the water. Oxygen saturation concentration decreases with the increase in temperature and salts and increases with barometric pressure. Freshwater streams, where temperature ranges from about 0 to 30 °C, typically have oxygen saturation concentrations ranging from 7.5 to 15 mg/L. The following equation is recommended to calculate the saturated dissolved oxygen concentrations as a function of temperature for freshwater streams:

$$C_s = \frac{468}{316.6 + T} \tag{A-11}$$

where T is water temperature in degrees Celsius (°C). This equation is accurate to within 0.03 mg/L compared with the Benson-Krause equation on which the Standard Methods tables are based (see McCutcheon, 1985).

A.7 SEDIMENT OXYGEN DEMAND

A.7.1 Field Measurement of SOD

Direct field measurement of sediment oxygen consumption upstream and downstream of the discharge is the preferred approach for obtaining model input data. Consistent field techniques for determining SOD in natural waters are evolving with investigators in the southeast United States using an approach developed by Murphy and Hicks (1986). The two basic measurement techniques are (1) in situ chambers and (2) sediment core extraction and laboratory measurement (see Bowie et al., 1985; Hatcher, 1986; and Whittemore, 1986). The in situ method requires submersion of a chamber on the bottom with periodic measurements of oxygen to determine the uptake rate in the chamber. Laboratory measurements are based on a sample core from the sediment (hopefully undisturbed) being placed in a well-oxygenated column with oxygen measurements taken over time to determine the uptake rate. Different investigators have varying opinions on the relative merits of each technique; however, the use of in situ chambers, with minimal disturbance of the natural sediments, appears to be the preferred technique (Murphy and Hicks, 1986).

A.7.2 Predicting SOD

Projections of the expected water quality impact of a waste discharge alternative for some future low-flow condition are normally required during waste load allocation studies. Projections of the expected change in SOD that might result from a change in waste loading to a stream is a complex evaluation (e.g., DiToro et al., 1990). HydroQual (1987), for example, demonstrated that a reduction of total organic carbon loading to the Potomac estuary from 92,540 lb/day (1.56 g C/m^2-day) in 1969 to 57,800 lb/day (0.98 g C/m^2-day) in 1985 resulted in a reduction of the mean SOD from 2.2 to 1.8 g O_2/m^2-day. The relationship used to infer the long-term coupling between carbon loading and SOD in the Potomac is not a simple formulation.

Until such time when models (such as that of Di Toro et al., 1990) are readily available to explicitly couple particulate carbon deposition and sediment oxygen demand, it is beyond the scope of most waste load allocation studies to predict future SOD rates with any credibility since SOD is *not* linearly proportional to the waste loading of organic carbon in freshwater systems. For projecting future water quality conditions, it is preferable to use the same SOD parameter values that were used in verification of the model. At the least, this approach will result in a somewhat conservative projection of future oxygen levels since SOD is likely to be reduced following improvements in waste management.

A.7.3 Literature Values of SOD

Model parameter values for SOD could be specified using field measurements reported for streams and rivers with similar hydraulic and waste loading characteristics. A fairly large body of literature (e.g., Phoel, 1982; Butts and Evans, 1978; Butts, 1974) is available for the analyst to review actual field measurements obtained under a wide range of conditions that might be similar to the study area for a waste load allocation. Table A-25, taken from Bowie et al. (1985), summarizes a large number of investigations of SOD rates that have been reported for streams and rivers in the literature. Table A-26, taken from Murphy and Hicks (1986), also summarizes a large number of in situ chamber SOD measurements obtained from 1977 to 1984.

A.8 PHOTOSYNTHESIS AND RESPIRATION

A.8.1 Estimation Techniques

Three methods for estimating photosynthesis (P) and respiration (R) in waste load allocation modeling studies are:

- Estimation from observed chlorophyll *a* levels.
- Measurements of diel variations of dissolved oxygen concentrations.
- Light and dark bottle measurements of dissolved oxygen.

The first method is addressed by the following problem: given the concentration of phytoplankton in a stream, estimate the average daily oxygen production. A technique for performing this estimate, developed by Di Toro (1975), can be found in Thomann and Mueller (1987). In summary, the following equations are used:

$$P = 0.25\ Chl\ a \tag{A-12}$$

and

$$R = 0.025\ Chl\ a \tag{A-13}$$

in which Chl *a* is the chlorophyll *a* concentration in μg/L.

For the second method, Di Toro (1975) has developed an analytical method to calculate P based on the measured diurnal dissolved oxygen range:

$$P = \frac{f\,K_a\,(1 - e^{-K_a T})}{(1 - e^{-K_a f T})\,(1 - e^{-K_a T(1-f)})}\,\Delta \tag{A-14}$$

where

f = photo period ($0 < f < 1$)

K_a = stream reaeration rate coefficient (day^{-1})

T = 1-day period

Δ = diurnal dissolved oxygen range (mg/L)(max-min)

Note that Equation A-14 can be used to estimate the diurnal range of dissolved oxygen with an estimate of P from the first two methods (Thomann and Mueller, 1987). Table A-27 presents a summary of stream photosynthesis studies compiled by Bowie et al. (1985).

The light and dark bottle method is described in detail by Standard Methods (APHA, 1989). As shown in Figure A-9, clear glass (light) and foil-wrapped glass (dark) bottles are stationed or suspended at various fixed depths in a stream and filled with water collected at their respective depths. Usually, an attempt is made in deep rivers to suspend the bottles at least to the depth of the euphotic zone, taken to be the 1 percent light penetration depth. Based on Equation A-20, the depth to 1 percent remaining light can be estimated as $4.6/K_e$. Since K_e is approximated by 1.7/*Secchi depth* (Equation A-21), the approximate depth of the euphotic zone is 2.9 times the *Secchi depth*.

Dissolved oxygen measurements are made at regular time intervals, with the light bottles that receive the solar radiation, measuring net photosynthetic oxygen production (P-R). The dark bottles, in the absence of light, measure gross respiration (R) as shown in Figure A-9. It should be noted that:

- In contrast to the diurnal method where water column and benthic algae or macrophytes contribute to the observed oxygen balance, only the photosynthetic activity of the algae in the water column (phytoplankton) is measured by this technique. If there are significant attached algae or rooted plants, no measurement of their photosynthetic contribution is made.
- The estimate of respiration (R) made from the dark bottle studies includes both algal respiration and bacterial respiration from oxidation of carbonaceous and nitrogenous compounds.
- Both P and R are temperature-dependent. Since they are essentially expressions of growth rate and respiration rate in oxygen equivalents, the temperature-rate relationships discussed in Sections 2.3.5 and A.9 for growth and respiration apply directly to P and

TABLE A-25. MEASURED VALUES OF SEDIMENT OXYGEN DEMAND IN RIVERS AND STREAMS
(After Bowie et al., 1985)

SOD, g0$_2$\m^2 day	Environment	Experimental Conditions	References
0.022-0.92	Upper Wisconsin river	60-hour laboratory core incubation periodic mixing, 4 °C, dark	Sullivan et al. (1978)
0.09 ± 0.02 (@12°C) 0.15 ± 0.04 (@20°C) 0.20 ± 0.03 (@28°C) 0.29 ± 0.07 (@36°C)	Eastern U.S. river	45-day incubation of 0.6 liter sediment in 3.85-liter BOD dilution water, light	NCASI (1981)
0.18 ± 0.05 (@12°C) 0.55 ± 0.22 (@20°C) 0.60 ± 0.28 (@28°C) 0.87 ± 0.23 (@36°C)	Southeastern U.S. river		
3.2-5.7	Fresh shredded tree bark	10-liter incubations in aged tap water, room temperature, light	NCASI (1971)
0.52-3.6	Aged shredded tree bark		
2-33	Four eastern U.S. rivers downstream of paper mill discharges	In situ chamber respirometers, 22-27 °C, light, stirred at varying rates	NCASI (1978)
0.9-14.1		Open-ended tunnel respirometer, in situ 22-27 °C, dark	
<0.1-1.4 (@20°C)	Eastern U.S. river downstream of paper mill discharge	In situ respirometer stirred at various rates, 9-16 °C, dark, $\Theta = 1.08$	NCASI (1979)
0.27-9.8	Northern Illinois rivers (N = 89 stations)	In situ respirometer, dark, T = 5-31°C, time = 1 1/2-3 hours	Butts & Evans (1978)
0.1-5.3 (@20°C)	Six stations in eastern Michigan rivers	In situ respirometer in stirred chambers, 15-27 hours, dark, 19-25 °C, $\Theta = 1.08$ Θ = constant for temperature adjustment	Chiaro & Burke (1980)
1.1-12.8	New Jersey rivers (10 stations)	In situ respirometer, dark 30 minutes-8 hours, stirred. Temperature unknown	Hunter et al. (1973)
0.3-1.4	Swedish rivers	In situ respirometer, light, stirred, 0-10 °C	Edberg & Hofsten (1973)
0.2-1.2	Swedish rivers	Laboratory incubations, stirred, dark, 20 °C	Edberg & Hofsten (1973)
1.7-6.0	Spring Creek, PA	Laboratory incubators in dark, stirred, 20 °C	McDonnell & Hall (1969)
1.5-9.8	74 samples from 21 English rivers	Laboratory incubation of cores; 15 °C	Rolley & Owens (1967)
4.6-4.4	Streams	Oxygen mass balance	James (1974)

TABLE A-26. SOD RATES MEASURED USING EPA IN SITU CHAMBERS, 1977-1984
(After Murphy and Hicks, 1986)

		SOD Rates			
Location	**Description**	**Mean g 0_2/m^2-hr**	**Range g 0_2/m^2-hr**	**C,V %**	**Mean Temp. °C**
Indian River, FL	High-salinity lagoon	.12	.10-.14	23.6	30.0
Sykes Creek at Merritt ls, FL	Saltwater tidal creek subject to urban runoff and STP	.31	.12-.69	81.0	31.8
Turkey Creek at Melbourne, FL	Density-stratified tidal creek stream, residential development, heavy organic deposit	.54	.49-.60	14.3	32.4
Inidan River at Turkey Creek, Melbourne, FL	Estuary	.12	.13-.22	36.4	32.0
Sugarloaf Key, FL	Dead-end canal, hypersaline	.12	.10-.14	23.6	25.0
Wilson Creek, SC	Shallow, flashy, piedmont creek	.10	.09-.11	10.7	23.4
Wilson Creek, SC	Shallow, flashy, piedmont creek	.08	.05-.14	36.0	24.6
Wilson Creek, SC	Shallow, flashy, piedmont creek	.12	.07-.18	30.4	25.1
Mobile Bay, AL	Low-salinity estuary	.12	.10-.13	16.7	28.0
St. Andrews Bay, FL	Estuary	.05	.03-.04	40.0	20.0
Savannah River at Savannah, GA	Density-stratified, high-velocity river/estuary	.027	.21-.32	22.2	24.8
Savannah River at Savannah, GA	Density-stratified, high-velocity river/estuary	.061	.38-.78	29.5	21.2
Savannah River at Savannah, GA	Density-stratified, high-velocity river/estuary	.036	.25-.44	19.4	23.2
Hillsborough River at Tampa Bay, FL	Density-stratified river/estuary	.41	.37-.45	8.37	22.8
Hillsborough River at Tampa Bay, FL	Density-stratified river/estuary	.18	.16-.19	6.55	22.8
Hillsborough River at Tampa Bay, FL	Density-stratified river/estuary	.11	.09-.14	19.72	22.9
Hillsborough River at Tampa Bay, FL	Density-stratified river/estuary	.17	.12-.19	14.92	23.2
Sarasota Bay, FL (Summer)	Shallow bay, grass flat	.165	.148-.183	14.9	29.6
Sarasota Bay, FL (Summer)	Open bay, sandy bottom	.227	.172-.357	34.3	29.0
Sarasota Bay, FL (Summer)	Open bay, sandy bottom	.156	.145-.166	9.6	28.2
Sarasota Bay, FL (Summer)	Deep bay channel, coarse sand	.128	.107-.152	17.5	28.9
Sarasota Bay, FL (Winter)	Shallow bay, grass flat	.122	.070-.202	39.6	20.3
Sarasota Bay, FL (Winter)	Open bay, sandy bottom	.106	.094-.116	9.5	21.1
Sarasota Bay, FL (Winter)	Shallow bay, grass flat	.077	-	-	21.1
Sarasota Bay, FL (Winter)	Deep bay channel, coarse sand	.086	.063-.110	27.2	22.0
Whitaker Bayou, FL (Summer)	Density-stratified creek, thick organic deposits. Subject to urban runoff and STP	.155	.050-.121	43.0	28.9
Whitaker Bayou, FL (Summer)	Density-stratified creek, thick organic deposits. Subject to urban runoff and STP	.140	.117-.157	12.9	27.7
Sarasota Bay, FL at Whitaker Bayou (Summer)	Bay near mouth of tidal creek. Subject to urban runoff and STP	.264	.098-.604	66.0	28.4
Whitaker Bayou, FL (Winter)	Density-stratified creek, thick organic deposits. Subject to urban runoff and STP.	.154	.138-.169	14.3	22.0
Whitaker Bayou, FL (Winter)	Density-stratified creek, thick organic deposits. Subject to urban runoff and STP	.240	.134-.300	35.4	20.5
Sarasota Bay, FL, at Whitaker Bayou (Winter)	Bay near mouth of tidal creek. Subject to Urban runoff and STP	.160	.123-.214	26.3	20.5
Lake Myakka, FL (Summer)	Shallow freshwater lake with bottom of dense organic matter	.049	.020-.064	51.0	28.5
Lake Myakka, FL (Winter)	Shallow freshwater lake with bottom of dense organic matter	.070	.063-0.75	7.1	19.7
Lake Myakka, FL (Winter)	Shallow freshwater lake with bottom of dense organic matter	.124	.081-.172	36.3	19.2
Gulf Shores, AL	Gulf Intracoastal Waterway	.086	.070-.109	18.8	22.5
Gulf Shores, AL	Gulf Intracoastal Waterway	.070	.066-.078	9.9	21.5
Gulf Shores, AL	Gulf Intracoastal Waterway	.112	.072-.154	32.7	22.0
Guntersville Reservoir, AL	TVA lake	.163	.136-.222	20.0	25.5
Guntersville Reservoir, AL	TVA lake	.099	.078-.120	18.4	24.8
Pickwick Reservoir, AL	TVA lake	.037	.028-.056	27.0	23.5
Pickwick Reservoir, AL	TVA lake	.099	.093-.104	5.7	23.0

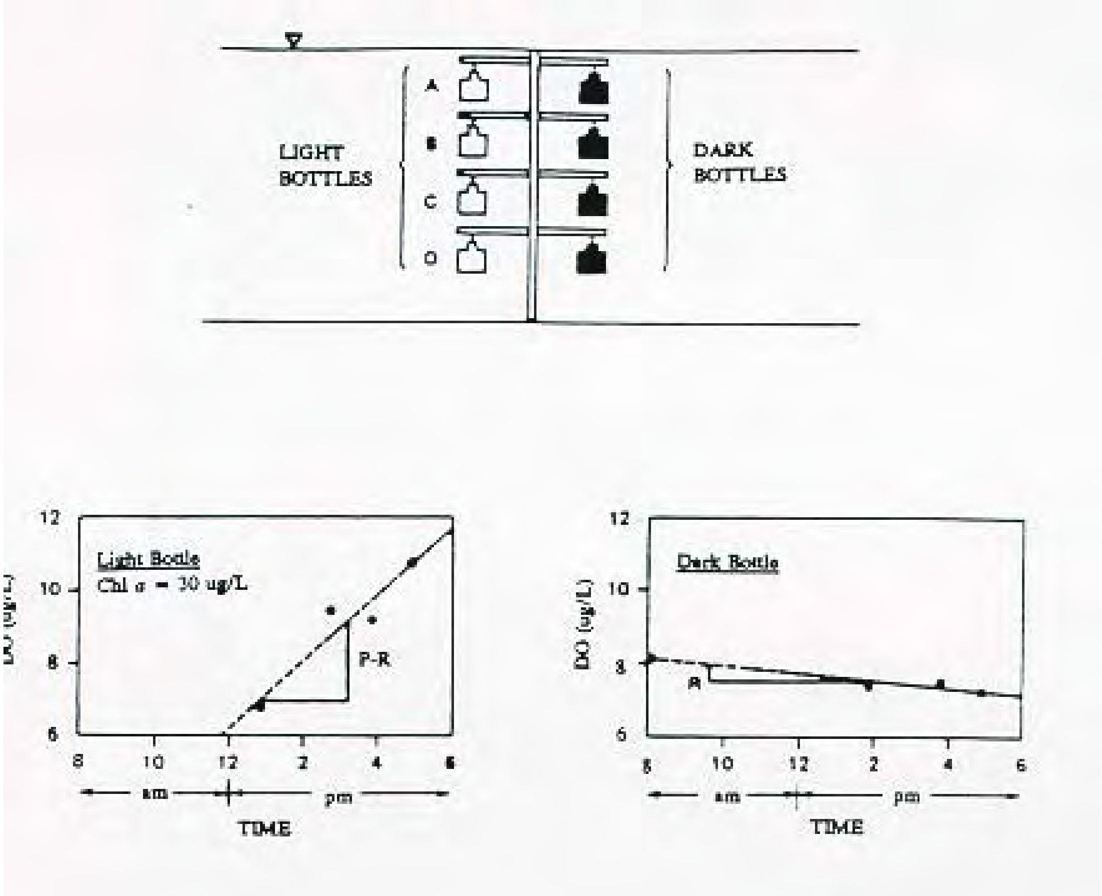
TABLE A-26. (Continued)

Location	Description	Mean g 0_2/m^2-hr	SOD Rates Range g 0_2/m^2-hr	C,V %	Mean Temp. °C
Hillsborough Bay, FL	Density-stratified bay, dynamic murk	.033	.025-.048	39.3	16.5
Hillsborough Bay, FL	Density-stratified bay, sand/silt	.046	.039-.052	20.2	16.0
Hillsborough Bay, FL	Density-stratified bay at river mouth and STP density	.094	.085-.100	8.4	18.0
Sowashee Creek, MS	Shallow creek, upstream of STP	.097	.079-.116	19.2	26.2
Sowashee Creek, MS	Shallow creek, downstream of STP	.102	.090-.124	18.7	29.5
Hillsborough Bay, FL	Shallow bay, dark sand & silt	.131	.074-.176	33.2	30.2
Tampa Bay, FL	Open Bay, sand	.074	.061-.087	24.8	
Hillsborough Bay, FL	Shallow bay, dark sand & silt	.131	.074-.176	33.2	30.2
Tampa Bay, FL	Open bay, sand	.195	.161-.276	21.5	31.0
Old Tampa Bay, FL	Shallow bay, nearshore	.111	.107-.114	3.2	31.0
Tampa Bay, FL	Shallow bay, nearshore	.203	.137-.284	26.5	30.5
Manatee River, FL	Tidal stratified river	.077	.064-.107	25.9	31.5
Manatee River, FL	Tidal stratified river	.181	.160-.193	10.2	32.0
Manatee River, FL	Tidal stratified river	.075	.062-.087	13.7	31.0
Boone Lake	TVA lake, sludge bank	.346	.171-.465	44.7	
Boone Lake	TVA lake, upstream of sludge bank	.072	.071-.073	1.4	
Boone Lake	TVA lake, downstream of sludge bank	.041	.037-0.44	8.8	
Boone Lake	TVA lake, HRM 19.7	.031	.018-.036	11.9	14.0
Boone Lake	TVA lake, HRM 26	.064	.055-.077	12.1	10.5
Boone Lake	TVA lake, HRM 29	.078	.062-.10	30.7	10.5
Boone Lake	TVA lake, HRM 31	.109	.066-.14	30.7	10.5
Boone Lake	TVA lake, WRM 3	.050	.040-.060	17.3	11.8
Boone Lake	TVA lake, WRM 7	.023	.020-.027	13.3	14.3
Boone Lake	TVA lake, WRM 11.25	.072	.64-.81	8.4	24.8
Calcasieu River	Stratified river/estuary	.027	.019-.031	20.1	24.1
Calcasieu River	Stratified river/estuary	.027	.019-.039	32.8	24.0
Calcasieu River	Stratified river/estuary	.027	.019-.039	22.0	24.2
Calcasieu River	Stratified river/estuary	.067	.061-.083	13.9	24.2
Calcasieu River	Stratified river/estuary	.055	.038-.073	27.3	26.0
Calcasieu River	Stratified river/estuary	.027	.024-.038	22.3	24.5
Calcasieu River	Stratified river/estuary	.035	.029-.043	14.6	23.0
Charlotte Harbor	Estuary; seasonal stratification	.062	.054-.070	10.4	17.0
Pine Island Sound	Estuary; shallow	.044	.038-.048	11.8	19.5
Ft. Loudoun Res.	TVA Lake, TRM-608	.046	.045-.049	5.0	24.2
Ft. Loudoun Res.	TVA Lake, TRM-608	.044	.033-.050	17.2	24.8
Ft. Loudoun Res.	TVA Lake, TRM-638	.043	.038-.051	13.9	24.2
Tellico Reservoir	TVA Lake, LTRM-16.5	.048	.042-.054	12.5	
Tellico Reservoir	TVA Lake, LTRM-21	.047	.028-.065	33.3	18.1

TABLE A-27. PHOTOSYNTHETIC OXYGEN PRODUCTION AND RESPIRATION RATES IN RIVERS (After Bowie et al., 1985)

River	T °C	Pm g/m^2-day	Pav g/m^2-day	R g/m^2-day	Reference
Grand, Michigan	28	12.7-37.6	4.4-13.0	9.3-12.7[a]	O'Connor and Di Toro (1970)
Clinton, Michigan	21	13.2-22.9	4.2-7.3	9.3[a]	O'Connor and Di Toro (1970)
Truckee, Nevada	28	12.9-26.0	4.8-9.6	3.6-6.2[a]	O'Connor and Di Toro (1970)
Ivel, Great Britain	16	24	9.0	4.6[a]	O'Connor and Di Toro (1970)
Flint, Michigan	28	4.0-40.0	1.3-18.	4.-20[a]	O'Connor and Di Toro (1970)
North Carolina Streams	-	-	9.8	21.5[b]	Thomas and O'Connell (1966)
Laboratory Streams	-	-	3.4-4.0	2.4-2.9[b]	Thomas and O'Connell (1977)
Charles, Massachusetts	19-25	-	0.0-12.0	0.0-36.[b]	Erdmann (1979a, b)
Shenandoah, Virginia	23	4.8-17.4	-	0.9-5.9[a]	Deb and Bowers (1983)
Baker, Virginia	-	-	0.45	1.9[b]	Kelly et al. (1975)
Rappahannock, Virginia	-	-	6.1	7.3[b]	Kelly et al. (1975)
S. Fork Rivanna, Virginia	-	-	2.1	3.4[b]	Kelly et al. (1975)
Rivanna Virginia	-	-	2.3	5.4[b]	Kelly et al. (1975)
South, Virginia	-	-	2.0	5.3[b]	Kelly et al. (1975)
Mechums, Virginia	-	-	1.3	2.6[b]	Kelly et al. (1975)
Havelse, Denmark	-	-	0.2-25.9[c]	4.8-22.9[b]	Simonsen & Harremoes (1978)
Experimental Channels	9-24	5-45	1.5-14.8	2.6-10.7[b]	Gulliver et al. (1982)

[a] Algal respiration only.
[b] Total community respiration.
[c] Measurements were made over the period of 1 year, and solar radiation varied by more than a factor of 10.



EXAMPLE CALCULATION (for each bottle)

(1) Slope of light DO data

$$\frac{(11.5-6.0)}{6\text{ hr}} \times \frac{(24\text{ hr})}{1\text{ day}} = 22\text{ mg/L-day}$$

(2) Slope of dark bottle DO data

$$\frac{(6.1-7.5)}{10\text{ hr}} \times \frac{(24\text{ hr})}{1\text{ day}} = 1.4\text{ mg/L-day}$$

(3) R = Slope of dark bottle data = 1.4 mg/L-day

(4) P = Slope of light bottle + Slope of dark bottle
P = 22.0 + 1.4 = 23.4 mg/L/day

(5) Calculate P for each depth sampled
(A, B, C, D) and plot as shown

(6) Equate area under curve A_1 to A_2 to determine depth averaged production rate

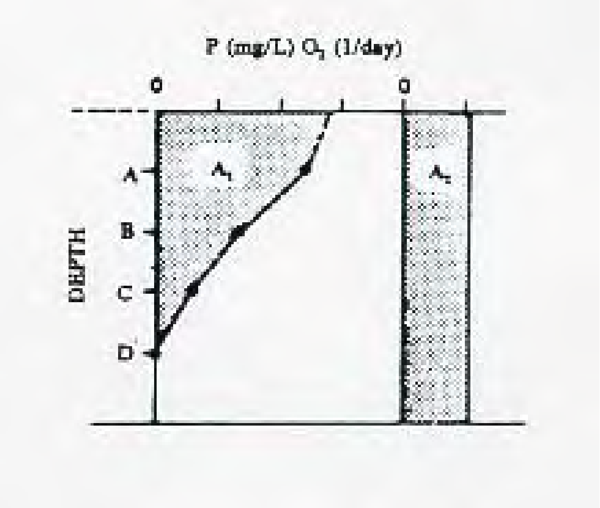


FIGURE A-9. LIGHT AND DARK BOTTLE STUDIES (After Thomann and Mueller, 1987)

R measurements derived from light and dark bottle tests.

The productivity vs. depth relationship developed from the light and dark bottle test data, shown in Figure A-9, provides a determination of the depth-averaged primary productivity rate. The extent to which it is time-averaged depends on the period of the day covered by the measurements. Because of the significant variations in P with depth and time, care must be taken that light and dark bottle test results are interpreted correctly.

When conducting the light and dark bottle work, it is essential that the light bottles not be allowed to progress to the point where saturation is exceeded. Losses of dissolved oxygen during sample handling and performing the analytical measurements would introduce errors into the test results. The maximum hourly increase in dissolved oxygen in the light bottle can be computed as follows:

$$\Delta C = \frac{a_{oc}\, a_c\, G_{\max}\, 1.066^{(T-20)}}{(1000)\ (24)} Chl \qquad \text{(A-15)}$$

where

ΔC = maximum hourly increase in dissolved oxygen (mg/L-hr)

a_{oc} = stoichiometric ratio of oxygen to carbon = 2.67 (mg O_2/mg C)

a_c = stoichiometric ratio of carbon to chlorophyll *a* (mg C/mg Chl)

G_{max} = maximum algal growth rate (day-1)

T = water temperature (°C)

Chl = instantaneous chlorophyll *a* concentration (μg/L)

Equation A-15 can be used to estimate appropriate sampling intervals and maximum duration of light bottle measurements.

A.8.2 BOD Deoxygenation in Bottles

The following example illustrates how to calculate the algal respiration from the light and dark bottle results with significant BOD: The initial DO in a light and dark bottle test is 7.0 mg/L. After 1 day, the DO in the dark bottle is 2.0 mg/L and the DO in the light bottle is 9.0 mg/L. The BOD_5 of the water sample without algae is 10 mg/L and K_1 is 0.3 day^{-1}. Using Equation 2-5,

$$10 = BOD_u\,[1 - e^{-(0.3)(5)}]$$

yielding BOD_u equal to 12.9 mg/L. Thus, BOD_1 can be calculated as

$$BOD_1 = 12.9\,[1 - e^{-(0.3)(1)}] = 3.34$$

That is, the amount of oxygen consumed by bacteria for BOD decay is 3.34 mg/L. The algal respiration is then equal to 7.0 - 2.0 - 3.34 or 1.66 mg/L.

A.9 PHYTOPLANKTON KINETIC RATES

A.9.1 Growth Rate

Eppley (1972) summarized algal growth data from a variety of sources as a function of temperature and developed the following equation:

$$G_T = G_{\max}\Theta^{T-20} \qquad \text{(A-16)}$$

where

G_T = temperature-adjusted growth rate (day^{-1})

$G_{\max}$ = maximum growth rate at 20°C (day^{-1})

Θ = constant for temperature adjustment

T = temperature (°C)

Both G_T and $G_{\max}$ are specific growth rates under optimum light and nutrient conditions. Reported ranges for $G_{\max}$ and Θ are:

$$G_{\max} = 1 \text{ to } 3\ day^{-1} \text{ at } 20°C$$
$$\Theta = 1.01 \text{ to } 1.18$$

As a first approximation, $G_{\max}$ = 1.8 day^{-1} and Θ = 1.066 may be used (Thomann and Meuller, 1987). Thus,

$$G_T = 1.8\,(1.066)^{(T-20)} \qquad \text{(A-17)}$$

Equation A-17 is shown in Figure 2-5. This relationship (Eppley, 1972) can be viewed as an envelope representing the maximum growth rate at any temperature, under optimum light and nutrient conditions.

A.9.2 Light Effect on Phytoplankton Growth

A depth- and time-averaged effect of available light energy on phytoplankton growth rate can be obtained (Di Toro et al., 1971), by integrating the light intensity relationships over depth and time. This reduces to

$$r_L = \frac{2.718f}{K_e\,HT}\ (e^{-\alpha_1} - e^{-\alpha_2}) \qquad \text{(A-18)}$$

where

$$\alpha_1 = -\frac{I_T}{I_s}\,e^{-K_e H}$$

$$\alpha_2 = -\frac{I_T}{I_s}$$

r_L = light limitation factor
f = photoperiod - daylight fraction of averaging period (day)
T = averaging period (day)
K_e = light extinction coefficient (m^{-1})
H = average depth of segment (m)
I_a = average of incident light on water surface over a 24-hour day (ly/day)
I_T = average of incident light over photoperiod (=I_a/f)(ly)
I_s = saturated light intensity (ly/day) (see Figure 2-6)

The full expression for algal growth can be synthesized from Equations A-16, A-17, and A-18 as follows:

$$G_p = G_{\max} 1.066^{(T-20)} \left[\frac{2.718 f}{K_e\, HT} \left(e^{-\alpha_1} - e^{-\alpha_2} \right) \right]$$

$$\min \left(\frac{DIN}{K_{mn} + DIN} ; \frac{DIP}{K_{mp} + DIP} ; \frac{Si}{K_{si} + si} \right) \qquad \text{(A-19)}$$

Solar radiation is measured routinely at selected weather stations in the United States. It is usually reported as langleys (ly), which is a measure of the total radiation of all wavelengths that reach the surface of the earth. One ly is equal to 1 g cal/cm^2. Algae and other photosynthetic plants respond to solar radiation in the visible range of the spectrum. Visible light energy was historically measured in terms of intensity as footcandles. A common conversion used in calculations is 2000 ft-candles = 350 ly/day. Contemporary primary production studies generally report incident light intensity with units of μEm^{-2}- sec (micro einstein) where the appropriate conversion factors are:

$$1\ ly\ day^{-1} = 0.485\ watt\ m^{-2}\ (w\ m^{-2})$$

$$1\ w\ \ m^{-2} \approx\ 4.6\ \mu\ Em^{-2}\ sec^{-1}$$

The light reduction factor, rL, interpreted as the percentage of the optimum growth rate, is sensitive to the product of KeH, which appears as the denominator in Equation A-18. It can be seen that shallow and clear waters yield high rL values and offer a favorable condition for algal growth when compared with turbid and deep waters. KeH, a dimensionless number, is also referred to as the light extinction factor.

TABLE A-28. TYPICAL LIGHT EXTINCTION COEFFICIENTS

Ke = 1.5/SD (m^{-1})	Secchi Depth (m)	Types of Waterbodies
0.02 - 0.06	30 - 80	Clear, mid-ocean oligotrophic waters
0.2	8 -10	Clear lake waters
0.2 - 0.5	3 - 8	Coastal zone marine waters
0.5-5.0	0.3-3.0	Rivers and estuaries

Typical K_e values vary widely with type of waterbody, principally as a function of the amount of suspended solids and phytoplankton biomass present in the water column. Table A-28 summarizes typical ranges of K_e for different types of waterbodies.

Extinction coefficients can be determined directly using light intensity measurements from the field. Light attenuation with depth is approximated by the following equation:

$$I = I_o\, e^{-K_e\, z} \qquad \text{(A-20)}$$

That is, the slope of *ln (I/I$_o$)* vs. depth, *z*, provides an estimate of K_e. In lieu of the direct measurements of light intensity at various depths, K_e may be determined by the following empirical equation:

$$K_e = \frac{1.7}{Secchi\ depth} \qquad \text{(A-21)}$$

It should be pointed out that the correlation between Secchi depth and light extinction coefficient is notoriously poor in many waters; factors may range from 1.5 to 5.0 (see Holmes, 1970.) Thus, field determination of K_e is recommended.

Di Toro (1978) has provided a theoretical and empirical basis for estimating the extinction coefficient as a function of nonvolatile suspended solids, detritus, and phytoplankton chlorophyll:

$$K_e = 0.052\ NVSS + 0.174\ VSS + 0.031\ Chl \qquad \text{(A-22)}$$

where
$NVSS$ = nonvolatile suspended solids concentration (mg/L)
VSS = detritus concentration (mg/L)
Chl = chlorophyll concentration (μg/L)

The nonvolatile suspended solids (the inorganic particulates) both absorb and scatter the light, whereas the organic detritus and phytoplankton chlorophyll mainly absorb the light. Di Toro has shown that Equation A-22 applies to K_e values of approximately less than 5.0 m^{-1} and phytoplankton biomass up to approximately 15 μg/L.

A.9.3 Death Rate

The endogenous respiration rate, D_{p1} (Equation 2-16) is given approximately by

$$D_{p1} = \mu_R (1.08)^{T-20} \tag{A-23}$$

where μ_R varies from 0.05 to 0.25 day^{-1} (Thomann and Mueller, 1987). A value of 0.15 day^{-1} is usually used as a first approximation.

A.9.4 Settling Rate

Phytoplankton settling rate is estimated by dividing the settling velocity by the stream depth. Phytoplankton settling velocities are presented in Table A-29. Additional data are available in a review by Smayda (1970). Some phytoplankton such as blue-green algae develop gas vacuoles, which result in buoyancy and subsequent aggregation at the water surface. The proliferation of such species is a particular problem because the settling velocity may be zero or even negative and phytoplankton tend to remain in the water column or at the surface (e.g., 1983 *Microcystis* bloom in Potomac estuary).

A.9.5 Biomass Stoichiometry

Dry weight biomass is related to the major nutrients (carbon, nitrogen, and phosphorus) and chlorophyll *a* through stoichiometric ratios that give the ratios of each nutrient to the total biomass. Typical algal nutrient compositions are summarized in Table A-30. Ratios for different algal groups or species (e.g., blue-green, diatom, etc.) can be found in the literature (O'Connor et al., 1973; Bowie et al., 1985) but are not included in Table A-29 as this manual addresses the algal modeling only on a total population basis.

A.9.6 Half Saturation Constants

Half saturation constants are required to describe the nutrient dependence of the phytoplankton growth rate (Equation A-19). Table A-31 summarizes an extensive compilation of phytoplankton half saturation constants for populations of diatoms, flagellates, chlorophytes and chrysophytes.

A.10 NUTRIENT RECYCLING RATES

A.10.1 Phosphorus Mineralization Rate

The rate of transformation from particulate phosphorus to orthophosphate in the water column ranges from 0.02 to 0.10 day^{-1} (Bowie et al., 1985). As a first approximation, a value of 0.03 day^{-1} may be used.

A.10.2 Organic Nitrogen Hydrolysis Rate and Ammonia Nitrification Rate

Table A-32 presents the rate coefficients for nitrogen transformations reported in a number of modeling studies.

A.11 SEDIMENT NUTRIENT RELEASE RATE

Sediment nutrient releases measured in the field are usually reported in mg/m^2-day. Table A-33 (from Thomann and Mueller, 1987) shows some reported nutrient fluxes from the sediments under both aerobic and anaerobic conditions. When the overlying water is anaerobic, the flux of phosphorus from the sediment increases significantly as a result of increased diffusion between the sediment and the water. Such increased diffusion results from changes in the iron complexes at the water-sediment interface. Table A-34 presents data for two stations in the Peconic Bay in Long Island (Garber, 1990).

Since data are usually not available to characterize sediment nutrient processes for many streams and rivers, aerobic sediment flux rates of ammonia and phosphate can be estimated as the stoichiometric equivalent of the *biochemical component of SOD* using Redfield ratios (by weight) for O:C:N:P (109:41: 7.2: 1) (Redfield et al., 1963). Using the O:N ratio of 109 mg O_2: 7.2 mg N benthic regeneration of ammonia can be estimated as:

$$jNH_3 = SOD\ [1000\ mg\ O_2\ /g\ O_2]\ \ [1\ mg\ N/15.14\ mg\ O_2]$$

where SOD is in units of g O_2/m^2-day and jNH_3 has units of mg N/m^{-2}-day. Di Toro (1986) has summarized paired measurements of SOD and jHN_3 flux to substantiate the relationship of SOD and jHN_3 under aerobic conditions.

Although the sediment-water interactions for phophorus recycling are complex, Redfield stoichiometry is appropriate for a preliminary estimate of phosphate flux from the sediments under aerobic conditions.

TABLE A-29. TOTAL PHYTOPLANKTON SETTLING VELOCITIES
(After Bowie et al., 1985)

Settling Velocity (m/day)	References
0.05 - 0.5	Chen and Orlob (1975) Tetra Tech (1976) Chen (1970) Chen and Wells (1975, 1976)
0.05 - 0.2	O'Connor et al. (1975, 1981) Thomann et al. (1975, 1979) Di Toro and Matystik (1980) Di Toro and Connolly (1980) Thomann and Fitzpatrick (1982)
0.02 - 0.05	Canale et al. (1976)
0.4	Lombardo (1972)
0.03 - 0.05	Scavia (1980)
0.04	Bierman et al. (1980)
0.2 - 0.25	Youngberg (1977)
0.04 - 0.6	Jorgensen (1976) Jorgensen et al. (1978, 1981)
0.01 - 4.0	Baca and Arnett (1976)
0.0 - 2.0	Chen and Orlob (1975) Smith (1978a)
0.15 - 2.0	Duke and Masch (1973) Roesner et al. (1977)
0.0 - 0.2	Brandes (1976)
0.0 - 30.0	Jorgensen (1979)

TABLE A-30. NUTRIENT COMPOSITION OF ALGAL CELLS - RATIO TO CHLOROPHYLL*a* (as μg/μg) (After Bowie et al., 1985)

Algal Type	$\frac{\text{C}}{\text{Chl } a}$	$\frac{\text{N}}{\text{Chl } a}$	$\frac{\text{P}}{\text{Ch l}a}$	$\frac{\text{Si}}{\text{Chl } a}$	References
Total Phytoplankton	50 - 100	7 - 15	0.5 - 1.0		Thomann et al. (1975, 1979) O'Connor et al. (1981) Di Toro & Matystik (1980) Di Toro & Connolly (1980) Salas and Thomann (1978)
		0.5			Salisbury et al. (1983)
	7.2	0.63			Larsen et al. (1973)
	25 - 112[a]	7 - 29[a]	1.0[a]		Jorgensen (1979)
	10 - 100[a]	2.7 - 9.1[a]			O'Connor et al. (1981)
Diatoms	100	1. - 15	0.5 - 1.0	40 - 50	Di Toro & Connolly (1980)
					Di Toro & Matystik (1980) Thomann et al. (1979)
		0.5			Salisbury et al. (1983)
	5.-200[b]				Baca & Arnett (1976)
	18-500[a]	2.2-74.6[a]	0.27-19.2[a]	2.4-50.7[a]	Di Toro et al. (1971)
Green Algae	20-100[b]				Baca & Arnett (1976)
Blue-green Algae	14-67[b]				Baca & Arnett (1976)
Dino flagellates	275	19.3			O'Connor et al. (1981)

Dry Weight (mg/mg DW) percentage of five phytoplankton function groups. (O'Connor et al., 1973)

Total Phytoplankton	**% Carbon**	**% Nitrogen**	**% Phosphorus**
Average % SD(N=5)	39 % 4.4	6.1 % 1.9	1.6% 0.8
Range	19!50	2.7!9.1	0.4!3.3

[a] Literature values.
[b] Model documentation values.

N = available inorganic nitrogen concentration, mass/volume
C = available inorganic carbon concentration, mass/volume
Si= available inorganic siliconcentration mass/volume.

TABLE A-31. LITERATURE SUMMARY OF PHYTOPLANKTON HALF-SATURATION CONSTANTS FOR NITROGEN, PHOSPHORUS, AND SILICA
(After Tetra Tech, Inc., 1992)

Species	Taxa	NO_3 µgN/L	NH_4-N µgN/L	$Si(OH)_4$ µgSi/L	PO_4-P µgP/L
mixed/natural populations	DIATOM:AVG	15.25	22.46	55.00	27.46
mixed/natural populations	DIATOM:MIN	1.40	0.28	2.46	10.00
mixed/natural populations	DIATOM:MAX	71.40	130.20	158.20	53.32
mixed/natural populations	DIATOM:OBS	43	30	39	5
Skeletonema costatum	DIATOM:AVG	6.30	19.74	24.98	
Skeletonema costatum	DIATOM:MIN	5.60	6.16	11.76	
Skeletonema costatum	DIATOM:MAX	7.00	50.40	50.68	
Skeletonema costatum	DIATOM:OBS	2	4	9	ND
mixed/natural populations	FLAGELL:AVG	64.12	56.47		
mixed/natural populations	FLAGELL:MIN	1.40	15.40		
mixed/natural populations	FLAGELL:MAX	144.20	79.80		
mixed/natural populations	FLAGELL:OBS	5	3	ND	ND
Dunalliella tertiolecta	CHLORO:AVG	7.14	2.17		
Dunalliella tertiolecta	CHLORO:MIN	1.82	0.28		
Dunalliella tertiolecta	CHLORO:MAX	19.60	4.90		
Dunalliella tertiolecta	CHLORO:OBS	4	6	ND	ND
Monchrysis lutheri	CHRYSO:AVG	7.14	4.37		
Monchrysis lutheri	CHRYSO:MIN	5.88	0.42		
Monchrysis lutheri	CHRYSO:MAX	8.40	7.42		
Monchrysis lutheri	CHRYSO:OBS	2	9	ND	ND

TABLE A-32. NITROGEN TRANSFORMATION RATES IN WATER COLUMN (day-1).
(After Bowie et al., 1985)

PON[a] to NH_3	NH_3 to NO_2	NH_3 to NO_3	NO_2 to NO_3	Reference
0.035		0.04		Thomann et al. (1975)
0.03				Thomann et al. (1979)
0.03		0.12		Di Toro and Connolly (1980)
0.03		0.20		Di Toro and Matystik (1980)
0.075		0.09-0.13		Thomann and Fitzpatrick (1982)
0.1-0.15		0.05-0.15		Lung (1986a)
0.1	0.05			Lung and Paerl (1988)
0.003	0.02		0.25	Tetra Tech, Inc. (1980)
0.1	0.02		0.25	Porcella et al. (1983)
0.1-0.4	0.1-0.5		5.0-10.0	Baca et al. (1973)
0.02-0.04	0.1-0.5		3.0-10.0	Baca and Arnett (1976)

[a] PON = particulate organic nitrogen.

TABLE A-33. SEDIMENT NUTRIENT RELEASE RATES
(After Thomann and Mueller, 1987)

	Flux-Aerobic Conditions (mg/m^2 -day)		Flux-Anaerobic Conditions (mg/m^2 -day)			
Location	**Total Dissolved Phosphorus**	**NH_3-N**	**Total Dissolved Phosphorus**	**NH_3-N**	**Si-silicon**	**Reference**
Muddy River, Boston, MA	9.6		(96 Max)			Fillos and Swanson (1975)
Lake Wamer, Amherst, MA	1.2		(26 Max)			Bannerman et al. (1975)
Lake Ontario	0.2					
Lake Erie						
Western Basin	6.0	44				DiToro and Connolly (1980)
Central Basin	3.0	30				
Eastern Basin	2.0	22				
White Lake, Muskegon Co., MI			34	32	297	Freedman and Canale (1977)
Cape Lookout Bight, NC		40 (winter) 325 (summer)				Martens et al. (1980)
LaJolla Bight	2.4 (-13 to 16)					Hartwig (1975)
Potomac Estuary	1-10					Calendar and Hammond (1982)

TABLE A-34. SELECTED MEASUREMENTS OF SOD AND BENTHIC AMMONIA FLUX IN PECONIC BAY, JULY 1989 (after Garber, 1990)

Region	Station	SOD (g 0_2/m^2 -day)	NH3 flux (gN/m2 -day	Temp (°C)
Gardiner Bay	BT-6A	0.76	14.6	23.7
Little Peconic Bay	BT-3	0.8	33.3	24.8

Using the N:P ratio of 7.2 mg N:1.0 mg P, benthic regeneration of phosphate can be estimated as:

$$j\,PO_4 = jHN_3 \ \ (1mg\ P/7.2\ mg\ N)$$

where jPO_4 is in units of mg P/m^2-day and jNH_3 has units of mg N/m^2-day. In addition to benthic release of nitrogen, benthic uptake of ammonia and nitrate can be a potentially significant component of the overall nitrogen balance in a stream or river (e.g., Williams and Lewis, 1986). Reported measured rates of benthic nitrification and benthic denitrification are summarized in Table A-35.

TABLE A-35. REPORTED RATES OF BENTHIC NITRIFICATION AND BENTHIC NITRATE LOSS (as mg N/m2-day)(Negative values indicate water column loss to the sediments; positive values indicate sediment source to the water column)

Study Site	Range	Reference
Benthic Nitrification		
Lake Mendota	-540 to - 900	Kaushik et al. (1981)
Sewage-enriched stream	0 to - 150	Kaushik et al. (1981)
Laboratory stream	-29[a] to - 69[b]	Kaushik et al. (1981)
Benthic Nitrate Loss		
Sewage effluent/canals	-913c	Kaushik et al. (1981)
Swifts Brook/Ontario	-480	Kaushik et al. (1981)
Duffin Creek/Ontario	-40 to -300	Kaushik et al. (1981)
Laboratory columns	-61 to -166	Kaushik et al. (1981)
Silt-enriched columns	-100 to -251	Kaushik et al. (1981)
Sand/gravel columns	-20 to -60	Kaushik et al. (1981)
Streams	-50[a] to -90[b]	Kaushik et al. (1981)
Upper Potomac estuary	-266 to +23	MWCOG (1987)
Gunston Cove, Potomac	-36	Seitzinger (1988)

Notes:

[a] Absence of tubificid worms.

[b] Presence of tubificid worms enhances nitrification and denitrification/nitrate loss rates.

[c] Reported units are mg NO_3-N/m^2-day.

TABLE A-36. REPORTED VALUES FOR ATMOSPHERIC REAERATION TEMPERATURE COEFFICIENT (After Bowie et al., 1985)

Temperature Coefficient, Θ	Reference
1.047	Streeter (1926)
1.0241	Elmore and West (1961)
1.0226	Elmore and West (1961)
1.020	Downing and Truesdale (1955)
1.024	Downing and Truesdale (1955)
1.016	Downing and Truesdale (1955)
1.016	Streeter (1926)
1.018	Truesdale and Van Dyke (1958)
1.015	Truesdale and Van Dyke (1958)
1.008	Truesdale and Van Dyke (1958)
1.024	Churchill et al. (1962)
1.022	Tsivoglou (1967)
1.024	Committee on Sanitary Engineering Research (1960)

A.12 TEMPERATURE EFFECTS ON REACTION RATE COEFFICIENTS

Water temperature plays an important role in affecting many first-order kinetic coefficients because the reactions are temperature-dependent following the usual van't Hoff-Arrhenius temperature correlation:

$$K_T = K_{20}\,\Theta^{T-20} \qquad \text{(A-24)}$$

in which K_T and K_{20} are values of the kinetic coefficient at temperatures T and 20 °C, respectively. Tables A-36 through A-41 present ranges of Θ values for a variety of kinetic coefficients.

TABLE A-37. VALUES OF THE TEMPERATURE COEFFICIENT FOR CARBONACEOUS BOD DECAY
(After Bowie et al, 1985)

Temperature coefficient, Θ	Temperature Limits (°C)	Reference
1.047		Chen (1970) Harleman et al. (1977) Medina (1979) Genet et al. (1974) Bauer et al. (1979) JRB Associates (1983) Bedford et al. (1983) Thomann and Fitzpatrick (1982) Velz (1984) Roesner et al. (1981)
1.05		Crim and Lovelace (1973) Rich (1973)
1.03-1.06	(0-5)-(30-35)	Smith (1978b)
1.075		Imhoff et al. (1981)
1.024		MWCOG (1982)
1.021-1.06		Baca and Arnett (1976) Baca et al. (1973)
1.04		Di Toro and Connolly (1980)
1.05-1.15	5-30	Fair et al. (1968)

TABLE A-38. TEMPERATURE COEFFICIENT, Θ, FOR NITRIFICATION
(After Bowie et al. 1985)

Reference	Ammonia Oxidation	Nitrite Oxidation
Stratton (1966)	1.0876	1.0576
Knowles et al. (1965)	1.0997	1.0608
Buswell et al. (1957)	1.0757	-
Wild et al. (1971)	1.0548	-
Bridle et al. (1979)	1.1030	-
Sharma and Ahlert (1977)	1.0689	1.0470
Laudelout and Van Tichelen (1960)	-	1.0689
Mean	1.0850	1.0586

TABLE A-39. TYPICAL VALUES OF THE TEMPERATURE COEFFICIENT FOR SOD USED IN WATER QUALITY MODELS
(After Bowie et al., 1985)

Model	Θ	Q_{10}[a]	Reference
DOSAG-3	1.047	1.58	Duke and Masch (1973)
QUAL-II	1.047	1.58	Roesner et al. (1977)
Vermont QUAL-II	1.047	1.58	JRB Associates (1983)
Lake Erie Model	1.08	2.16	Di Toro and Connolly (1980)
WASP	1.08	2.16	Thomann and Fitzpatrick (1982)
WASP	1.1	2.59	O'Connor et al. (1981)
LAKECO	1.02	1.22	Chen and Orlob (1972, 1975)
WQRRS	1.02-1.04	1.22-1.48	Smith (1978)
ESTECO	1.02-1.04	1.22-1.48	Brandes (1976)
DEM	1.04	1.48	Genet et al. (1974)
EAM	1.02	1.22	Bowie et al. (1980)
EAM	1.047	1.58	Tetra Tech (1980); Porcella et al. (1983)
USGS - Steady	1.065	1.88	Bauer et al. (1979)
AQUA-IV	1.02-1.09	1.22	Baca and Arnett (1976)
EXPLORE-I	1.05	1.63	Baca et al. (1973)
Laboratory/Field Studies	1.040-1.130	1.5-3.4	Zison et al. (1978); Whittemore (1984)

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[a] Q_{10} = ratio of K (20°C)/K(10°C) = Θ^{10} from $K(T) = K_{20}\,\Theta^{T-20}$ for T = 10°C

TABLE A-40. TYPICAL EXPERIMENTAL VALUES OF THE EFFECT OF TEMPERATURE ON SOD
(After Whittemore, 1986)

Reference	Range, °C	Coefficient, Θ
Edberg and Hofsten (1973)	5-15	1.130
	10-20	1.180
	15-25	1.040
Edwards and Rolley (1965)	10-20	1.077
Karlgren (1968)	2-22	1.090
McDonnell and Hall (1969)	5-25	1.067
Pamatmat (1971)	5-10	1.088
Thomann (1972)	10-30	1.065
Fair et al. (1941)	10-25	(1.07)[a]
Baity (1938)	22-29	(1.05)[a]
Masch et al. (1971)	-	1.0159

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[a] Estimated based upon author's conclusion.

TABLE A-41. TEMPERATURE DEPENDENCE OF BENTHIC AMMONIA REGENERATION IN ESTUARINE WATERS

Location	Q_{10}	Θ	Reference
Narragansett Bay	4.48	1.16	Kremer and Nixon (1978)
Peconic Bay			
mean	7.0	1.215	Garber (1990)
range (N = 8)	0.8-11.6	0.98-1.28	
Chesapeake Bay (Anoxic mainstem)	3.0	1.116	Garber (1990)

Note: Relationship of Q_{10} to Θ and reaction rates at 10°C (K_{10}) and 20°C (K_{20})

$Q_{10} = K_{20}/K_{10}$

$\Theta = 10^{0.1 \log Q^{10}}$

$\Theta = \exp(0.1 \ln Q_{10})$

APPENDIX B: SAMPLE TOTAL MAXIMUM DAILY LOAD ANALYSES

B.1 INTRODUCTION

Two studies, the Rivanna River study and the Willamette River study, are included in this appendix to illustrate the application of WASP5 and QUAL2E—two EPA supported stream water quality models. Additionally, an analytical solution approach is included in the Rivanna River study to demonstrate how different water quality parameters can be directly calculated, without the use of a complex simulation model, for a relatively simple BOD-DO and eutrophication problem. The Willamette River Basin modeling study highlights the use of the QUAL2E model in assessing dissolved oxygen (DO), nutrients, and phytoplankton biomass for a large river system in Oregon.

Readers are cautioned that site-specific data must be used when performing TMDL analyses, and that values presented in this (or any other) example must not be substituted for site-specific data. When such data are not available, applicable values should be developed by following the procedures detailed in the text.

B.2 RIVANNA RIVER STUDY

B.2.1 Problem Setting

This example is based on the earlier example problem presented in the *Technical Guidance Manual for Performing Waste Load Allocations* (USEPA, 1983) in calculating simple biochemical oxygen demand and dissolved oxygen conditions in streams. This example extends further to include a eutrophication problem and assessment of variations in channel geometry. Addressing the algal growth as a component of this example provides a more comprehensive analysis of production and respiration processes and their effect on dissolved oxygen in rivers.

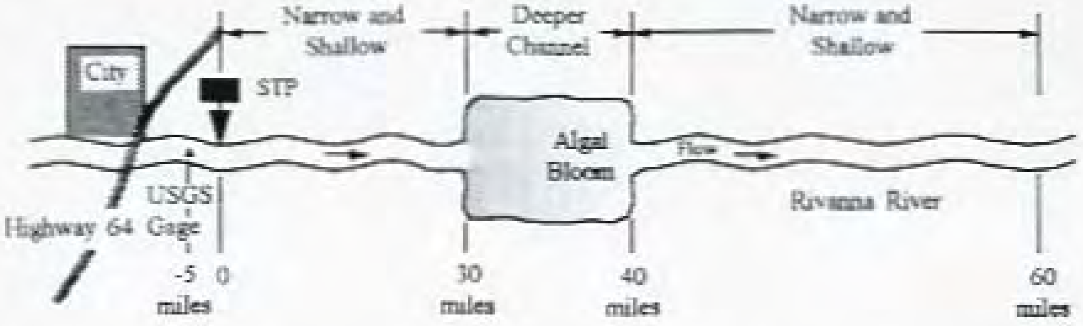
A city of approximately 60,000 people discharges its wastewater into a relatively small river, Rivanna River, with an average annual flow of about 250 cfs. The city's wastewater is presently treated by a trickling filter plant that provides about 85 percent BOD removal and has reached its design capacity of 7.5 mgd. The population is projected to increase by more than 50 percent to 92,000 people (with a range of 75,000 to 120,000 people) by the year 2000. Expansion of the treatment plant to a capacity of 11.5 mgd and provision of an activated sludge system for secondary treatment has been proposed.

Rivanna River, for 60 miles downstream of the treatment plant outfall, is classified as B1, which has a designated water use of fish and wildlife propagation. The pertinent State water quality standards for this example are a minimum DO level of 5.0 mg/L, a maximum un-ionized ammonia concentration as specified by *EPA Ambient Water Quality Criteria for Ammonia* (USEPA, 1984), and a narrative nutrient criteria as specified in the State water quality standards. The river is used locally for fishing and is bordered by several campgrounds and a State park. Approximately 30 miles downstream of the treatment plant outfall is a wider, slow moving section of the river, which under low flow conditions has experienced algal blooms, with chlorophyll *a* proliferation in the range of 30-40 ug/L. Occasional violations of DO and un-ionized ammonia have been observed downstream of the wastewater treatment plant. The watershed consists of approximately 60 percent agricultural, 35 percent forested, and 5 percent urban areas under existing conditions. The size of the urban portion of the watershed is 9.4 square miles which is projected to increase to 14.4 square miles under future conditions (2000 A.D.). Future land use conditions are expected to show a conversion of a portion of agricultural land to urban areas. This more developed watershed is estimated to consist of 58 percent agricultural, 35 percent forested, and 7 percent urban areas. A summary of the problem setting and treatment plant data is presented in Figures B-1 and B-2.

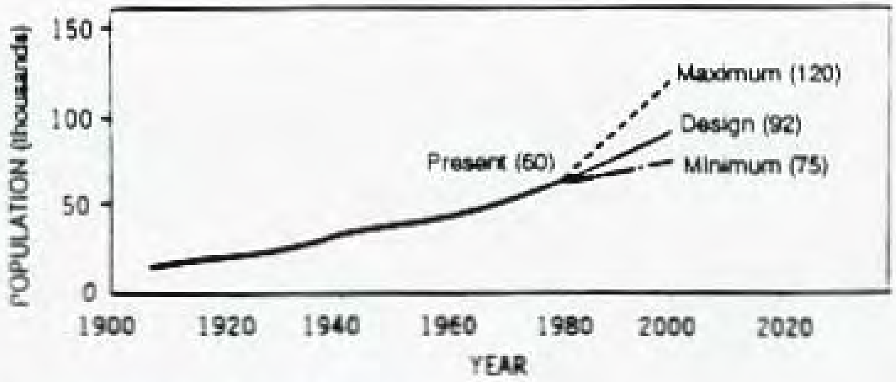
B.2.2 River Characteristics

The river flow is gaged by the United States Geological Service (USGS) 5 miles upstream of the treatment plant discharge. The average monthly flows for a 30- year period are summarized in Figure B-3(A). The

A. STUDY AREA



B. POPULATION PROJECTION: Present - 60,000 Projected - 92,000



C. RIVER CLASSIFICATION AND USE

1. State Classification - B1

2. Designated Use - Fish and Wildlife Propagation

3. Water Quality Standards (partial)
   a. DO Concentration - Greater than 5.0 mg/L
   b. Un-ionized Ammonia - Less than 0.02 mg/L
   c. Chlorophyll *a* - Less than 20 µg/L

4. Activities and Use
   a. Active and locally popular fishery
   b. Several campgrounds and State parks have river as an attraction

5. Problems
   a. Limited water quality data show occasional violations of DO and un-ionized ammonia standards under low flow conditions; periodic algal blooms occur in the wide deep section of the stream.

FIGURE B-1. PROBLEM SETTING.

A.TREATMENT FACILITIES

Present: Trickling filter plant constructed in 1958
Plant now at design capacity of 7.5 MGD
Effluent does not meet NPDES permit for
secondary treatment

Proposed: Activated sludge system to provide secondary treatment

B. EFFLUENT CHARACTERISTICS

	Present	Design (1)
Flow MGD	7.5	11.5
BOD_5 mg/l kg/day	40 1136	30 1306
$CBOD_u$ (2) mg/l kg/day	80 2271	60 2612
NH3-N (3) mg/l kg/day	15 426	15 653
NBOD mg/l kg/day	68 1930	68 2960

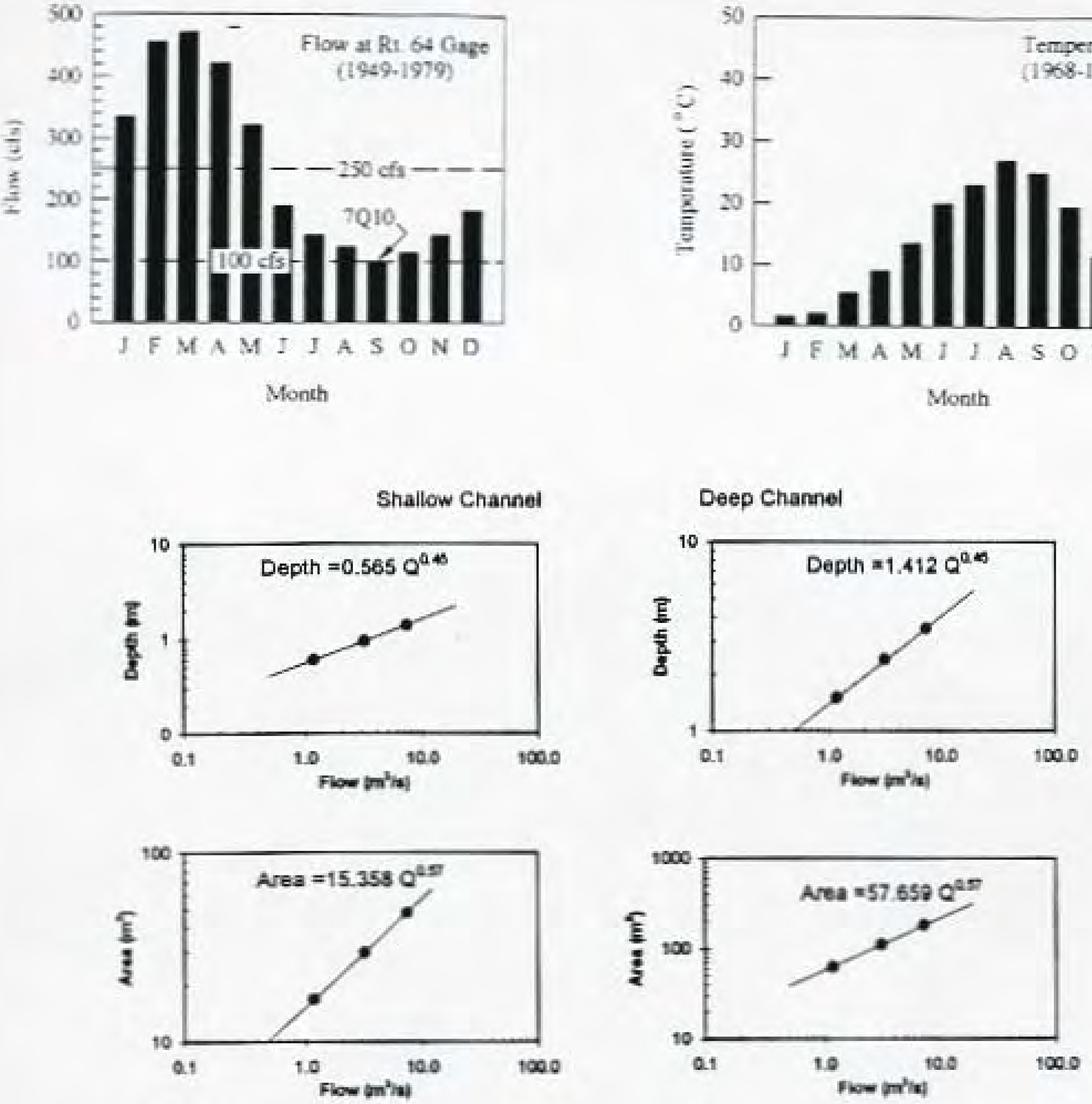
(1) **Preliminary basis - standard secondary treatment.**

Note that the 30-mg/L BOD_5 used in this example represents the secondary treatment effluent standard, and that better performance may occur during warm weather. Therefore, effluent characteristics used in modeling real-life situations should reflect the performance expected of the proposed facility during the critical period.

(2) **Long-term BOD tests indicate ratio of $CBOD_u/BOD_5 = 2.0$.**

(3) **NBOD = Stoichiometric oxygen requirements for oxidation of reduced nitrogen forms = 4.57 x NH_3-N (effluent oxidizable organic nitrogen is negligible)**

FIGURE B-2. TREATMENT FACILITIES AND EFFLUENT CHARACTERISTICS.



C. River Velocity

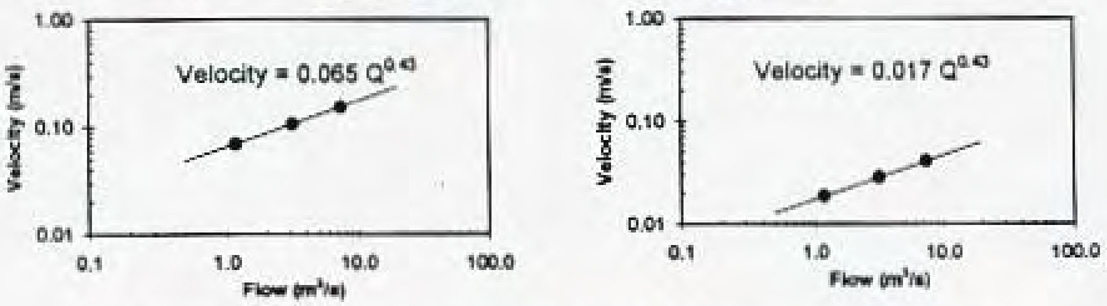


FIGURE B-3. RIVER FLOW, TEMPERATURE, GEOMETRY, AND VELOCITY.

average annual flow is about 250 cfs with a minimum monthly average low flow of 100 cfs, which occurs in September. However, the State requires that minimum DO standards must be met for the minimum 7 consecutive-day flow with a return period of once every 10 years (7Q10). (As discussed in *Book VI, Design Conditions*, other design flows may apply under summer conditions.) From a statistical analysis of the flow records, the 7Q10 is determined to be 30 cfs and occurs between August and October. (For further discussion of critical conditions, refer to Section 3.3 of this document.)

Critical conditions of DO, un-ionized ammonia concentration and nutrient enrichment in the river occur during the summer when the flow is low and the river's water temperature is high. From 11 years of water temperature data collected as part of a limited river monitoring program, the maximum average monthly river temperature is calculated to be 27^{o}C and occurs in August. The river condition selected to represent critical conditions assume a river flow of 30 cfs and a river temperature of 27^{o}C.

Note that for this example, both the critical low flow (7Q10) and the maximum average monthly temperature have been used in the projection, even though historical records (summarized in Figure B-3A) show minimum average monthly flow and temperature to occur in different months. This tacitly assumes that although the minimum average monthly flow occurs in September, the critical 7Q10 could occur in August, the month of maximum average temperatures. In areas where it can be shown that the 7Q10 will occur in a month with lower temperature, then the appropriate combination should be used rather than each of the extreme values. For example, critical low flows frequently occur during October in the northeast. An appropriate approach in such cases would be to define the 7Q10 and temperature conditions for each of the critical months (e.g. June-October), determine which month is most critical, and use that month in allocation calculations.

For this example, assume that three surveys were conducted to measure stream cross-sectional area under different flow conditions. Cross-sections were measured at 20 locations within the 65-mile long study area. From analysis of cross-sectional area measurements, it can be concluded that the study area may be divided into three relatively homogeneous reaches as shown in Figure B-1. The first and the third reaches have almost the same channel geometry, while the second reach has a wider and deeper channel. Throughout the length of each of these reaches the river is assumed to have uniform geometry. The first reach stretches from the STP outfall to 30 miles downstream. The second reach is 10 miles long and begins 30 miles downstream of the STP outfall. The final reach is 20 miles long. Two representative cross-sectional areas are used to characterize the reaches for each set of flow conditions; one cross-sectional area at a given flow represents the first and the third reaches, and another represents the second reach.

The average river velocity during each of the cross-sectional area survey periods was computed by application of the equation VELOCITY = FLOW/AREA. The average flow for each survey period is obtained from USGS records. Alternatively, dye study techniques could be used to more accurately determine average velocity for a given river section.

River cross-sectional area, depth, and velocity generally form linear relationships with flow when the data are plotted on log-log scales. Figure B-3B shows two sets of log-log plots derived from stream cross-sectional data. One set represents the relationships between the channel geometry and flow for the narrow and shallow sections between 0-30 miles and 40-60 miles downstream of the discharge. Another set represents the wide and deep channel between 30-40 miles. Figure B-3C shows the relationship between stream velocity and flow. Interpolations and extrapolations of river geometry and velocity at specific flows can be made directly from the log-log plots or can be computed from the equation of the line of best fit. The equation for the line of best fit has the form $Y = IQ^{s}$ where I is the intercept at Q = 1 cfs and s is the slope scaled directly from the plot. A summary of the mathematical expressions of the graphs presented in Figure B-3 are as follows:

For the narrow and shallow sections

$$\text{AREA (m}^2) = 15.358\ [\text{Q (m}^3/\text{s})]^{0.57} \qquad \text{(B-1)}$$

$$\text{DEPTH (m)} = 0.565\ [\text{Q (m}^3/\text{s})]^{0.45} \qquad \text{(B-2)}$$

$$\text{VELOCITY (m/s)} = 0.065\ [\text{Q (m}^3/\text{s})]^{0.43} \qquad \text{(B-3)}$$

For the wide and deep sections

AREA (m2) = 57.659 [Q (m3/s)]0.57 (B-4)

DEPTH (m) = 01.413 [Q (m3/s)]0.45 (B-5)

VELOCITY (m/s) = 0.017 [Q (m3/s)]0.43 (B-6)

River area, depth, and velocity can be computed for any flow in the appropriate section of the river by using the equations listed above. If river geometry data are available for only one flow condition, the relationship presented in Section A.3.1 (Equations A-1 through A-4) can be used to calculate river depth, area, and velocity at other flows.

B.2.3 Review of Water Quality Data

Historic river water quality data within the study area are limited. As part of the State environmental department's overall monitoring program for this river basin, water samples are periodically collected at stations located at river miles 25 and 55. These data represent approximately one grab sample per month during the summer over a 5 year period. A review of these data reveal occasional water quality problems with regard to dissolved oxygen and un-ionized ammonia. Further downstream, periodic algal blooms violate the State's narrative nutrient criteria. Problems appear to occur only under extreme low flow conditions. Since there are indications of occasional violations of water quality criteria, a TMDL is needed to assess load allocations under future conditions. The TMDL should address the occasional DO, un-ionized ammonia problems, and the eutrophication in the downstream recreational area. The TMDL should consider both upstream nonpoint source loadings and the local point source discharge.

Considering the conditions under which problems occur, an appropriate level of effort for a TMDL study initially can be limited to the analysis of a single river water quality data set collected during summer low-flow conditions. Accordingly, a survey was conducted during two days in August when the river flow averaged 100 cfs and the river water temperature was 25oC. The results of this survey and the State environmental data are presented in Figure B-4.

The DO data in Figure B-4, both August 1979 data and historical data, show stream DO levels above the standard of 5.0 mg/L at a flow condition of 100 cfs. The increase in river BOD5 and the ammonia concentrations at zero mile point show the impact of the treatment plant discharge. The gradually decreasing ammonia profile and increasing nitrite and nitrate profile suggest that nitrification is occurring in the river. There is evidence that a natural nitrification process, in which nitrate and some oxygen-demanding material are removed from the water, may occur in some streams.

Un-ionized ammonia has been demonstrated to be the principal form of ammonia toxic to biological life. Temperature and pH have been shown to affect ammonia toxicity. The EPA Ambient Water Quality Criteria for Ammonia (USEPA, 1984) requires two conditions to be met—a 4-day average for chronic toxicity and an 1-hour average concentration for acute toxicity. For a river temperature of 25^{o}C and pH of 7.75, with salmonids or other sensitive coldwater species absent, the 4-day average standards are 0.043 mg/L un-ionized ammonia (0.0353 mg/L un-ionized NH3-N) and 1.39 mg/L total ammonia (1.142 mg/L NH3-N). For a 1-hour averaging period standards are 0.32 mg/L un-ionized ammonia (0.263 mg/L un-ionized NH3-N) and 10.384 mg/L total ammonia (8.384 mg/L NH3-N). During the August 1979 survey, the ambient total ammonia concentration was less than the standard of 8.384 mg/L total NH3-N for the 1-hour average condition. The 1-hour average values are used for calibration so that the worst case scenario can be portrayed. Historical river water quality data collected near the USGS gage provides concentrations under 7Q10 flow conditions. Table B-1 shows estimated data for upstream boundary conditions used in modeling. The boundary conditions take into account the proportionally highly nonpoint source loads under the 100 and 250 cfs flow conditions. Monitored concentrations of water quality constituents in the wastewater treatment plant effluent are listed in Table B-2. Table B-2 also shows estimated effluent concentrations for management alternatives used in modeling (e.g., activated sludge treatment and advanced water treatment). These values were obtained through a literature search of typical loadings from various levels of treatment.

B.2.4 The Simplified Analytical Solution Approach

The Simplified Analytical Solution Approach uses the exact solutions to differential equations presented in this document to analyze the Rivanna River example. This approach provides a better insight into the fundamentals of modeling DO and eutrophication problems in rivers. The analytical solution approach is similar to the methods described in the *Technical Guidance Manual for Performing Waste Load Allocation, Simplified Analytical Method for Determine NPDES Effluent Limitation for POTWs Discharging into Low Flow Streams* (USEPA, 1980) in calculating ammonia toxicity and DO concentrations. The analytical solution presented for DO presented here expands on the modified Streeter-Phelps equation to account for phytoplankton production and respiration

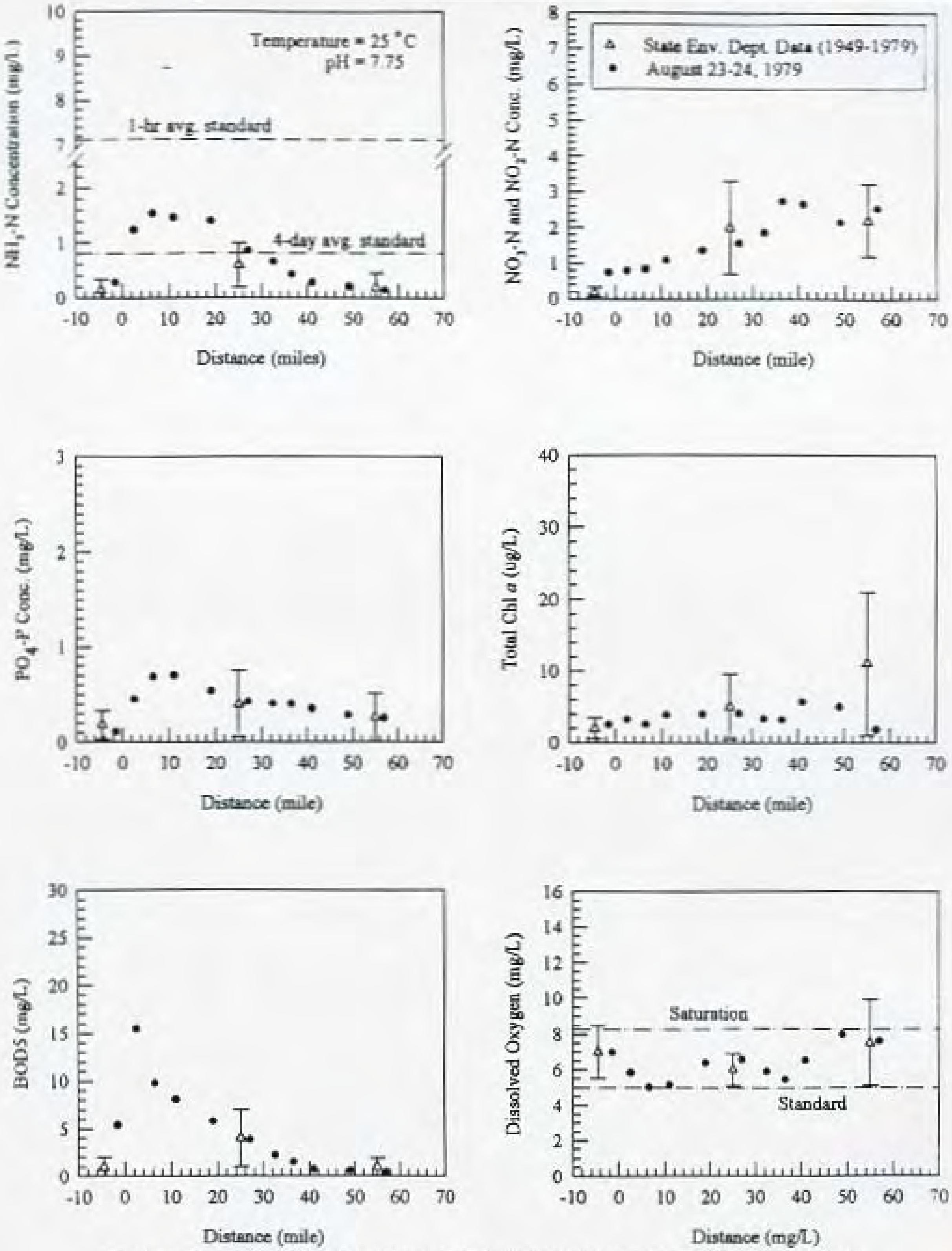


FIGURE B-4. INORGANIC NITROGEN AND PHOSPHORUS, CHLOROPHYLL *A*, BOD AND DO DATA (August 23-24, 1979).

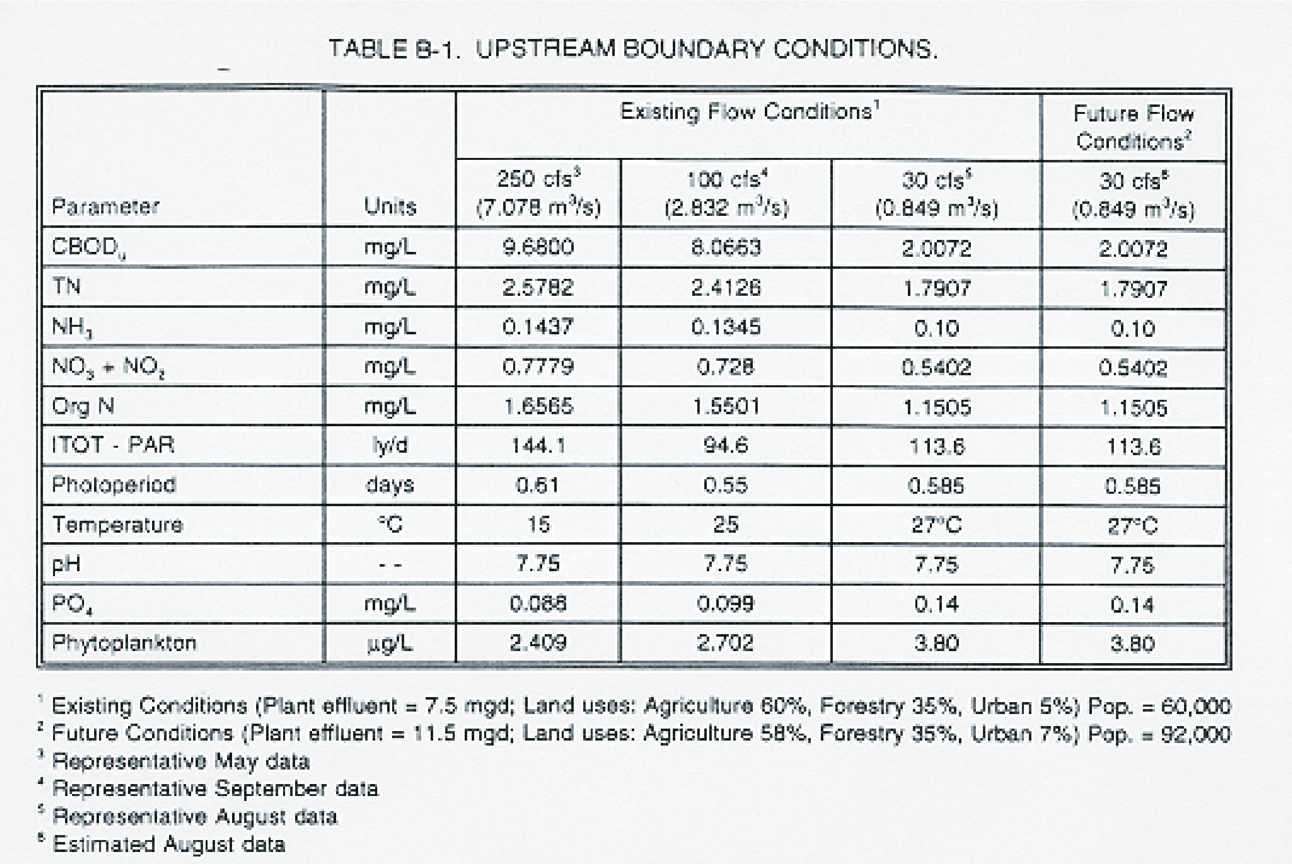


TABLE B-1. UPSTREAM BOUNDARY CONDITIONS.

Parameter	Units	Existing Flow Conditions[1]			Future Flow Conditions[2]
		250 cfs[3] ($7.078\ m^3/s$)	100 cfs[4] ($2.832\ m^3/s$)	30 cfs[5] ($0.849\ m^3/s$)	30 cfs[6] ($0.849\ m^3/s$)
$CBOD_u$	mg/L	9.6800	8.0663	2.0072	2.0072
TN	mg/L	2.5782	2.4126	1.7907	1.7907
NH_3	mg/L	0.1437	0.1345	0.10	0.10
$NO_3 + NO_2$	mg/L	0.7779	0.728	0.5402	0.5402
Org N	mg/L	1.6565	1.5501	1.1505	1.1505
ITOT - PAR	ly/d	144.1	94.6	113.6	113.6
Photoperiod	days	0.61	0.55	0.585	0.585
Temperature	°C	15	25	27°C	27°C
pH	- -	7.75	7.75	7.75	7.75
PO_4	mg/L	0.086	0.099	0.14	0.14
Phytoplankton	µg/L	2.409	2.702	3.80	3.80

[1] Existing Conditions (Plant effluent = 7.5 mgd; Land uses: Agriculture 60%, Forestry 35%, Urban 5%) Pop. = 60,000

[2] Future Conditions (Plant effluent = 11.5 mgd; Land uses: Agriculture 58%, Forestry 35%, Urban 7%) Pop. = 92,000

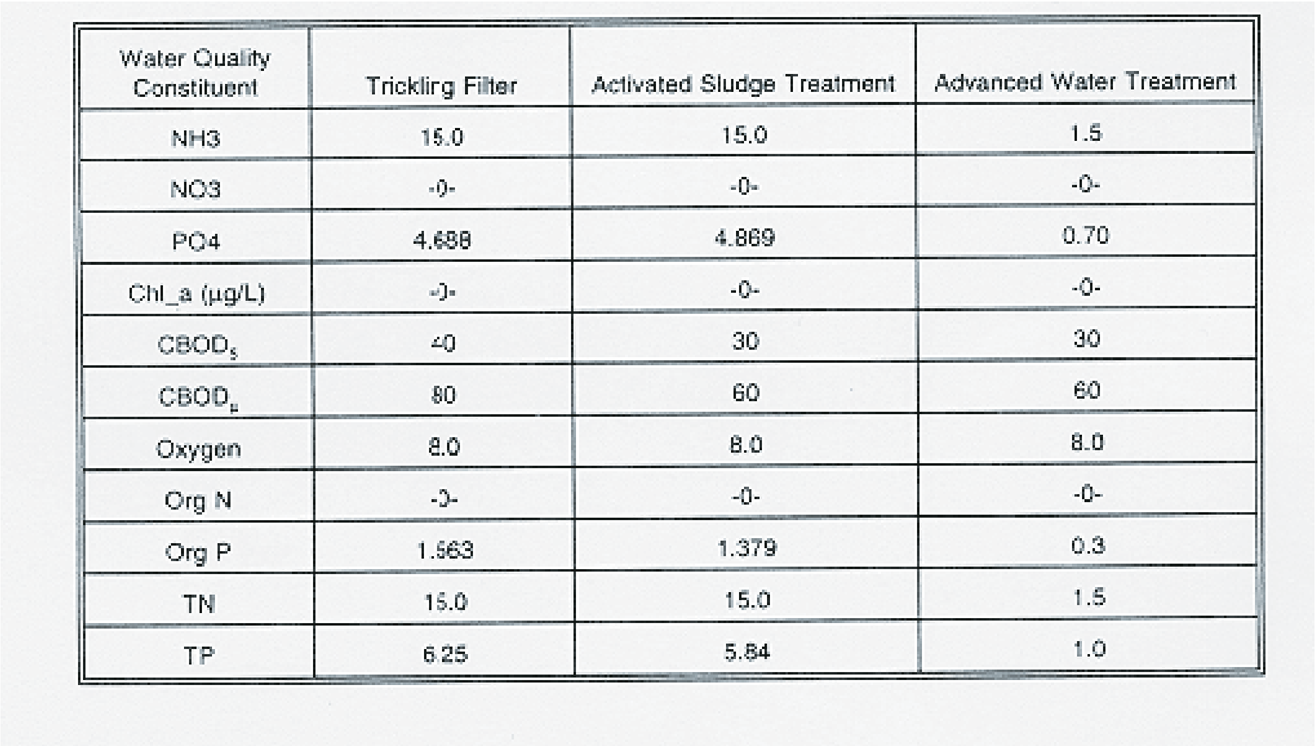
[3] Representative May data

[4] Representative September data

[5] Representative August data

[6] Estimated August data

TABLE B-2. CHARACTERISTICS OF EFFLUENT TREATMENT FOR DIFFERENT PROCESSES.



Water Quality Constituent	Trickling Filter	Activated Sludge Treatment	Advanced Water Treatment
NH3	15.0	15.0	1.5
NO3	-0-	-0-	-0-
PO4	4.688	4.869	0.70
Chl_a (µg/L)	-0-	-0-	-0-
$CBOD_5$	40	30	30
$CBOD_u$	80	60	60
Oxygen	8.0	8.0	8.0
Org N	-0-	-0-	-0-
Org P	1.563	1.379	0.3
TN	15.0	15.0	1.5
TP	6.25	5.84	1.0

as well as, BOD, SOD, and reaeration (Thomann and Mueller, 1987). Although ammonia toxicity is likely to occur near the vicinity of the wastewater treatment plant discharge, this approach also shows how to calculate the ammonia concentrations at downstream locations. The allowable instream total ammonia concentration is based on the un-ionized ammonia concentration as a function of pH and temperature. The *EPA Ambient Water Quality Criteria for Ammonia—1984* (USEPA, 1984) describes how to calculate allowable chronic and acute toxicity levels for un-ionized and total ammonia at a given pH and temperature. This approach also shows clearly the effect of changing channel geometry on DO sag. A simplified analysis calculates the location of DO sag due to a wastewater discharge for BOD decay only. If the problem is further compounded by conditions associated with lower reaeration coefficients and phytoplankton growth, a second DO sag may occur. Which of the two sags produces the minimum DO depends on overall conditions. The analytical solution approach provided here includes a more elaborate process for dealing with nonuniform channels and eutrophication issues. In the absence of phytoplankton growth and any significant variation in channel geometry, the analytical approach reduces to the method presented in the above mentioned document.

The analytical solution approach also addresses some of the limitations of reaeration models. The method presented here uses a discrete segment approach similar to that employed by more sophisticated computer simulation programs. To avoid the tedious work of repetitive calculations, a spreadsheet or a short computer program can be set up to solve the appropriate mathematical equations. Repeated solutions of the equations presented here were used to generate solutions at desired locations of the stream. Application of this approach is, however, limited to steady state conditions. A set of sample calculations are shown in Table B-3. The first step in the analytical model application is to divide the system under study into reaches with relatively uniform physical characteristics. The stream is divided into four reaches based on wastewater treatment plant discharge location and channel geometry. The upstream boundary conditions were used as initial conditions for the first reach. Concentrations of water quality constituents at the downstream end of each reach are used as initial conditions for the next reach. Therefore, for the second reach, initial conditions are the concentrations resulting from mixing of the treatment plant effluent with instream concentrations at the end of the first reach. Effluent from the wastewater treatment plant is assumed to mix completely with stream water immediately after discharge. Since the second reach is 30 miles long, a more detailed assessment could be performed by calculating concentrations at various locations along the stream, possibly more densely near the discharge. The calculations shown in Table B-3 can be grouped into six steps, 1) calculating physical parameters, 2) calculating net phytoplankton growth rate, 3) checking nutrient limits for phytoplankton growth, 4) estimating reaction rates, 5) calculating DO components and DO concentration, and 6) calculating concentrations of nutrients, BOD, and chlorophyll a. This simplified approach does not include analysis of organic nitrogen, organic phosphorus, and exchange of nutrients between water and sediment. Additional equations can be used to incorporate these factors in the analysis of phytoplankton growth. Alternatively, more detailed water quality models, such as WASP5, can explicitly consider a wider range of nutrient species, interactions, sources, and sinks.

As stream water quality data are available for 100 cfs flow conditions, it is used for calibration. For an analytical solution approach, calibration consists of the determination of the reaction rate coefficients (presented in Chapter 2 and Appendix A) that describe the spatial distribution of CBOD, ammonia, nitrite and nitrate, phytoplankton growth, and DO. The first set of calculations must be made based on an educated guess of reaction rates. Then a comparison between the calculated values and observed data will allow the modeler to make a better estimate of reaction rates. The overall loss rate of NBOD can be estimated solely by matching observed ammonia and nitrate data. The overall loss rate of CBOD and the effective deoxygenation rate can be estimated by matching observed CBOD and DO data, respectively. Calibration of the analytical solution model, as shown in Figure B-5, provides 0.15 day-1, 0.30 day-1, 0.30 day-1 and 0.20 day-1 for the overall loss rate of NBOD (Kn), the effective deoxygenation rate (Kd), the overall loss rate of CBOD (Kr) and endogenous respiration of phytoplankton, respectively. Loss rates are determined by finding a value that provides the best fit with the August 1979 CBOD data. The CBOD removal rate by settling (Ks) is assumed to be zero for the secondary effluent, and the CBOD oxidation rate (Kd) equals the total removal rate (Kr). These values can be adjusted for simulating water quality constituents under different management conditions. The atmospheric reaeration rate (Ka) is determined in accordance with Table A-24.

TABLE B-3. SAMPLE CALCULATIONS FOR REACH 1.

Let us assume that a site-specific investigation provided following additional information.

Background Light extinction coefficient, $K_e = 1.50\ m^{-1}$

Endogenous respiration of phytoplankton, $\mu_R = 0.20\ day^{-1}$

Phytoplankton settling rate, $V_s = 0.5\ m/day$

Carbon to chlorophyll a ratio, $a_C = 33\ mg\ C/mg\ Chl\ a$

Nitrogen to chlorophyll a ratio, $a_N = 5.8\ \mu g\ N/\mu g\ Chl\ a$

Phosphorus to chlorophyll a ratio, $a_P = 0.79\ \mu g\ P/\mu g\ Chl\ a$

Phytoplankton maximum growth rate, $G_{max} = 1.8\ day^{-1}$ at 20 °C

At x = 5.0 miles, which is just upstream of the wastewater treatment plant discharge:

i) Depth, velocity and travel time

Depth, $H = 0.565 \cdot Q^{0.45} = 0.565 \cdot 2.832^{0.45} = 0.902\ m$

Velocity, $u = 0.065 \cdot Q^{0.45} = 0.065 \cdot 2.832^{0.45} = 0.1017\ m/s$

Travel time, $t^* = \frac{x}{u} = \dfrac{5.0\ (mile) \cdot 1609.3 \left(\dfrac{m}{mile}\right)}{0.1017 \left(\dfrac{m}{\sec}\right) \cdot 86400 \left(\dfrac{\sec}{day}\right)} = 0.9157\ day$

ii) Net algal growth rate

Phytoplankton growth with temperature adjustment,

$$G_T = G_{max}(1.066)^{T-20} = 1.8 \cdot 1.066^{(25-20)} = 2.478\ day^{-1} \qquad \text{(2-14a)}$$

Light extinction coefficient considering the self shading of algae (Riley, 1956),

$$K_e = K_{e1} + 0.0088 \cdot A_0 + 0.054 \cdot A_0^{2/3}, \qquad \text{where } A_0 \text{ is the Chl a conc. in } \mu g/L$$

$$= 1.5 + 0.0088 \cdot 2.702 + 0.054 \cdot 2.702^{2/3} = 1.63\ m^{-1}$$

Light dependent growth reduction factor,

$$\alpha_1 = \frac{I_T}{I_S f} e^{-K_e H} = \frac{113.6}{200 \cdot 0.585} \cdot e^{-(1.63)(0.902)} = 0.223 \qquad \text{(2-15a)}$$

$$\alpha_2 = \frac{I_T}{I_S f} = 0.971 \qquad \text{(2-15b)}$$

$$r_L = \frac{2.718\ f}{K_e H}(e^{-\alpha_1} - e^{-\alpha_2}) = \frac{2.718 \cdot 0.585}{1.63 \cdot 0.902} \cdot (e^{-0.223} - e^{-0.971}) = 0.456 \qquad \text{(2-15)}$$

Nutrient dependent growth reduction factor,

$$r_n = \min\left(\frac{DIN}{K_{sN} + DIN}, \frac{DIP}{K_{sP} + DIP}\right) \qquad \text{(2-17)}$$

$$= \min\left(\frac{135 + 728}{25 + (135 + 728)}, \frac{99.2}{1 + 99.2}\right) = \min(0.971, 0.99) = 0.971$$

Phytoplankton growth rate after adjusting for temperature, light and nutrient limit

$$G_p = G_T r_L r_n = 2.478 \cdot 0.456 \cdot 0.971 = 1.097\ day^{-1} \qquad \text{(2-14)}$$

Temperature adjusted endogenous respiration rate

$$D_{P_1} = \mu_R \cdot 1.08^{25-20} = 0.2 \cdot 1.08^5 = 0.294\ day^{-1} \qquad \text{(A-23)}$$

Phytoplankton death rate,

$$D_p = D_{P_1}(T) + D_z = 0.294 + 0.0 = 0.294\ day^{-1} \qquad \text{(2-18)}$$

Net phytoplankton growth rate,

$$G_n = G_p - D_p - \frac{v_s}{H} = 1.097 - 0.294 - \frac{0.5}{0.902} = 0.246\ day^{-1}$$

iii) Check nutrient limit

Phosphorus Limit

$$A_0' = \frac{a_p G_p A_0}{G_n}$$

$$= \frac{0.79\left(\frac{mg\ P}{mg\ Chl}\right) \cdot 1.097\left(\frac{1}{day}\right) \cdot 2.702\left(\frac{\mu g\ Chl}{L}\right) \cdot \left(\frac{mg}{1000\ \mu g}\right)}{0.246\left(\frac{1}{day}\right)} = 0.00952\ mg\ P/L$$

$$t^*_P = \frac{1}{G_n} \ln\left[\frac{A_0' + P_0 - 0.025}{A_0'}\right]$$

$$= \frac{1}{0.246\left(\frac{1}{day}\right)} \ln\left[\frac{0.00952\left(\frac{mg\ P}{L}\right) + 0.099\left(\frac{mg\ P}{L}\right) - 0.025\left(\frac{mg\ P}{L}\right)}{0.00952\left(\frac{mg\ P}{L}\right)}\right] \qquad \text{(4-7)}$$

$$= 8.83\ day$$

Nitrogen Limit

$$A_0'' = \frac{a_N G_p A_0}{G_n}$$

$$= \frac{5.8\left(\frac{mg\ N}{mg\ Chl}\right) \cdot 1.097\left(\frac{1}{day}\right) \cdot 2.702\left(\frac{\mu g\ Chl}{L}\right) \cdot \left(\frac{mg}{1000\ \mu g}\right)}{0.246\left(\frac{1}{day}\right)} = 0.07\ mg\ N/L$$

TABLE B-3. (CONTINUED)

$$t'_N = \frac{1}{G_n} \ln\left[\frac{A_0'' + N_0 - 0.125}{A_0''}\right]$$

$$= \frac{1}{0.246\left(\frac{1}{day}\right)} \ln\left[\frac{0.07\left(\frac{mg\ N}{L}\right) + [0.135 + 0.728]\left(\frac{mg\ N}{L}\right) - 0.125\left(\frac{mg\ N}{L}\right)}{0.07\left(\frac{mg\ N}{L}\right)}\right] \qquad (4\text{-}8)$$

$$= 9.94\ day$$

As $t'_P > t'$ and $t'_N > t'$ no nutrient limit occurs in the first reach.

iv) Reaction rates

Atmospheric reaeration rate using Bennett and Rathbun (1972) formula :

$$K_{a,20°C} = \frac{20.2\, u^{0.607}}{H^{1.689}}$$

$$= \frac{20.2 \cdot \left[0.1017\left(\frac{m}{\sec}\right) \cdot 3.281\left(\frac{ft}{m}\right)\right]^{0.607}}{\left[0.902\ m \cdot 3.281\left(\frac{ft}{m}\right)\right]^{1.689}} = 1.66\ day^{-1} \qquad \text{(Table A-24)}$$

Temperature adjustment reaeration rate,

$$K_{a,25°C} = K_{a,20°C} \cdot \theta^{25-20} = 1.66 \cdot 1.028^5 = 1.906\ day^{-1}$$

It is important to check the value of reaeration rate at each reach. Hydroscience (1971) suggests a minimum value of the oxygen transfer coefficient (oxygen transfer coefficient, $K_L = K_a \cdot H$) of 0.6 to 1.0 m/day. If K_L is less than 0.6 m/day at any reach, K_a should be adjusted accordingly.

<u>The following values of K_N, K_r and K_d were determined through calibration (25°C).</u>

$$K_N = 0.22\ day^{-1}$$

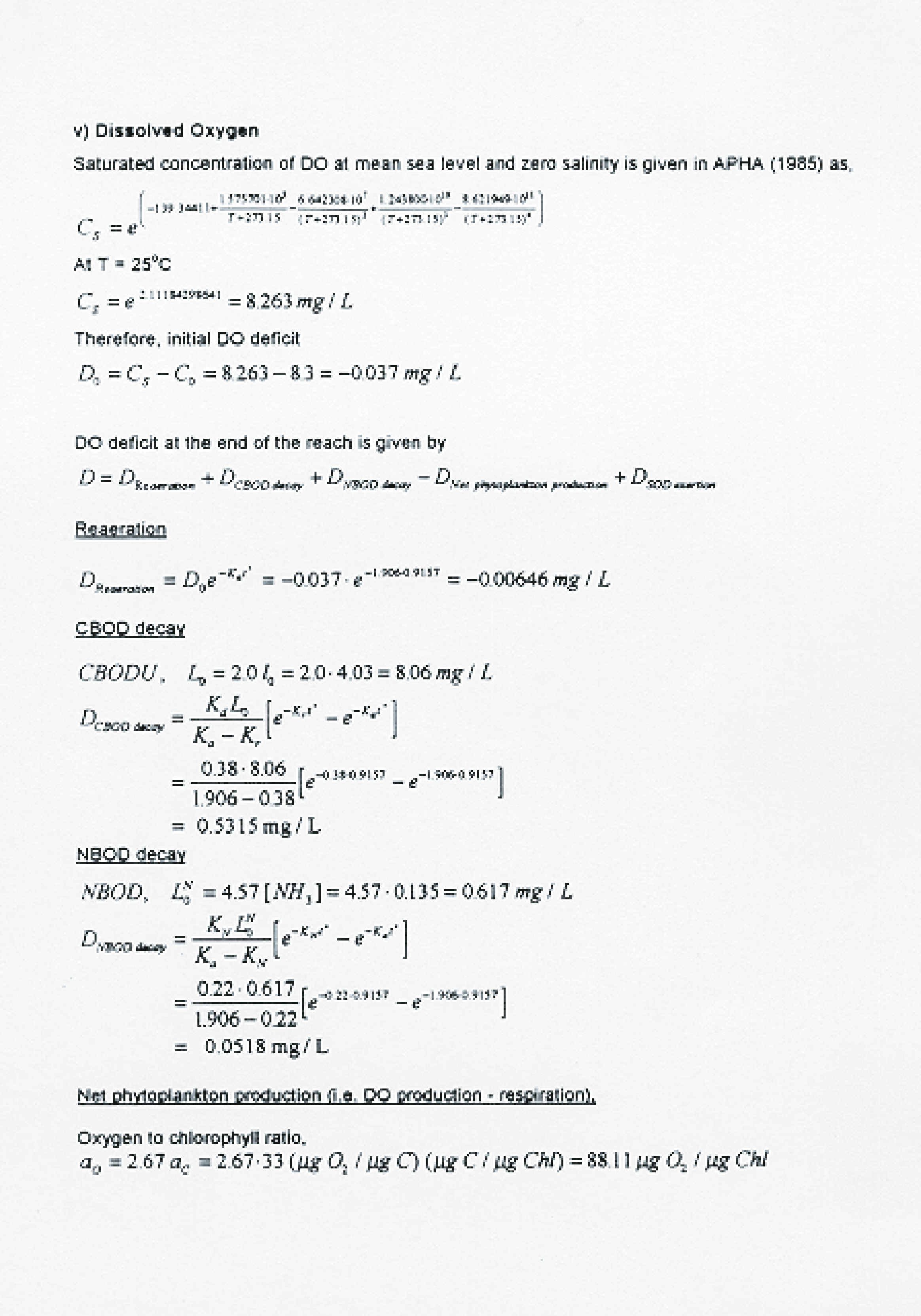
$$K_r = 0.38\ day^{-1}$$

$$K_d = 0.38\ day^{-1}$$

Therefore, at 20°C

$$K_N = 0.22 \cdot 1.08^{20-25} = 0.15\ day^{-1}$$
$$K_r = 0.38 \cdot 1.047^{20-25} = 0.30\ day^{-1}$$
$$K_D = 0.38 \cdot 1.047^{20-25} = 0.30\ day^{-1}$$

TABLE B-3. (CONTINUED)



v) Dissolved Oxygen

Saturated concentration of DO at mean sea level and zero salinity is given in APHA (1985) as,

$$C_S = e^{\left(-139.34411 + \frac{1.575701 \cdot 10^5}{T+273.15} - \frac{6.642308 \cdot 10^7}{(T+273.15)^2} + \frac{1.243800 \cdot 10^{10}}{(T+273.15)^3} - \frac{8.621949 \cdot 10^{11}}{(T+273.15)^4}\right)}$$

At T = 25°C

$$C_S = e^{2.11184298541} = 8.263\ mg / L$$

Therefore, initial DO deficit

$$D_0 = C_S - C_0 = 8.263 - 8.3 = -0.037\ mg / L$$

DO deficit at the end of the reach is given by

$$D = D_{Reaeration} + D_{CBOD\ decay} + D_{NBOD\ decay} - D_{Net\ phytoplankton\ production} + D_{SOD\ exertion}$$

Reaeration

$$D_{Reaeration} = D_0 e^{-K_a t'} = -0.037 \cdot e^{-1.906 \cdot 0.9157} = -0.00646\ mg / L$$

CBOD decay

$$CBODU, \quad L_0 = 2.0\ l_0 = 2.0 \cdot 4.03 = 8.06\ mg / L$$

$$D_{CBOD\ decay} = \frac{K_d L_0}{K_a - K_r}\left[e^{-K_r t'} - e^{-K_a t'}\right]$$

$$= \frac{0.38 \cdot 8.06}{1.906 - 0.38}\left[e^{-0.38 \cdot 0.9157} - e^{-1.906 \cdot 0.9157}\right]$$

$$= 0.5315\ \mathrm{mg / L}$$

NBOD decay

$$NBOD, \quad L_0^N = 4.57\,[NH_3] = 4.57 \cdot 0.135 = 0.617\ mg / L$$

$$D_{NBOD\ decay} = \frac{K_N L_0^N}{K_a - K_N}\left[e^{-K_N t'} - e^{-K_a t'}\right]$$

$$= \frac{0.22 \cdot 0.617}{1.906 - 0.22}\left[e^{-0.22 \cdot 0.9157} - e^{-1.906 \cdot 0.9157}\right]$$

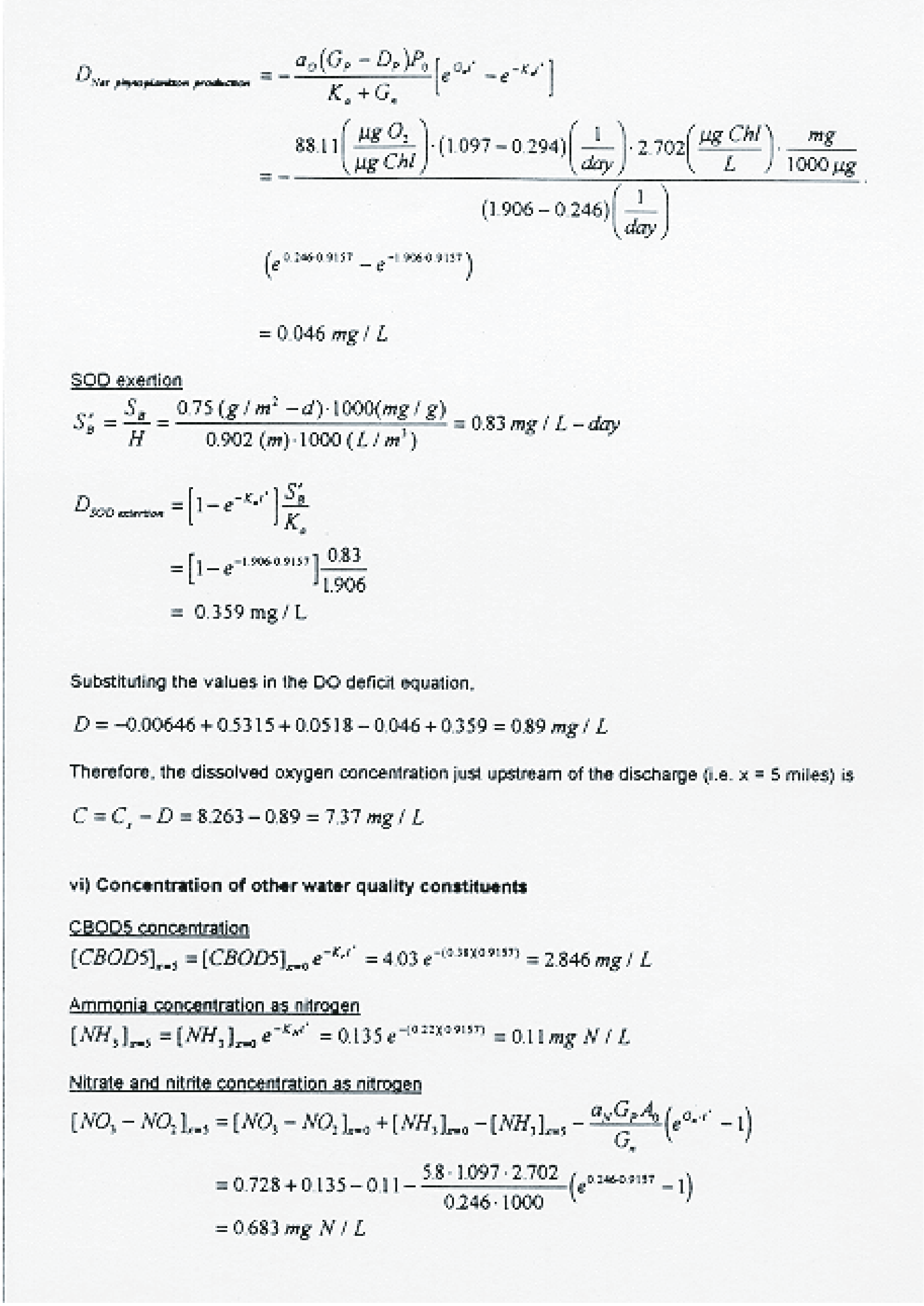
$$= 0.0518\ \mathrm{mg / L}$$

Net phytoplankton production (i.e. DO production - respiration),

Oxygen to chlorophyll ratio,

$$a_O = 2.67\ a_C = 2.67 \cdot 33\ (\mu g\ O_2 / \mu g\ C)\ (\mu g\ C / \mu g\ Chl) = 88.11\ \mu g\ O_2 / \mu g\ Chl$$

TABLE B-3. (CONTINUED)



$$D_{Net\ phytoplankton\ production} = -\frac{a_O(G_p - D_p)P_0}{K_a + G_n}\left[e^{G_n t'} - e^{-K_a t'}\right]$$

$$= -\frac{88.11\left(\frac{\mu g\ O_2}{\mu g\ Chl}\right)\cdot(1.097 - 0.294)\left(\frac{1}{day}\right)\cdot 2.702\left(\frac{\mu g\ Chl}{L}\right)\cdot\frac{mg}{1000\ \mu g}}{(1.906 - 0.246)\left(\frac{1}{day}\right)}\cdot$$

$$\left(e^{0.246\cdot 0.9157} - e^{-1.906\cdot 0.9157}\right)$$

$$= 0.046\ mg\ /\ L$$

SOD exertion

$$S_B' = \frac{S_B}{H} = \frac{0.75\ (g\ /\ m^2 - d)\cdot 1000(mg\ /\ g)}{0.902\ (m)\cdot 1000\ (L\ /\ m^3)} = 0.83\ mg\ /\ L - day$$

$$D_{SOD\ exertion} = \left[1 - e^{-K_a t'}\right]\frac{S_B'}{K_a}$$

$$= \left[1 - e^{-1.906\cdot 0.9157}\right]\frac{0.83}{1.906}$$

$$= 0.359\ \mathrm{mg\ /\ L}$$

Substituting the values in the DO deficit equation,

$$D = -0.00646 + 0.5315 + 0.0518 - 0.046 + 0.359 = 0.89\ mg\ /\ L$$

Therefore, the dissolved oxygen concentration just upstream of the discharge (i.e. x = 5 miles) is

$$C = C_s - D = 8.263 - 0.89 = 7.37\ mg\ /\ L$$

vi) Concentration of other water quality constituents

CBOD5 concentration

$$[CBOD5]_{x=5} = [CBOD5]_{x=0}\, e^{-K_r t'} = 4.03\ e^{-(0.38)(0.9157)} = 2.846\ mg\ /\ L$$

Ammonia concentration as nitrogen

$$[NH_3]_{x=5} = [NH_3]_{x=0}\, e^{-K_N t'} = 0.135\ e^{-(0.22)(0.9157)} = 0.11\ mg\ N\ /\ L$$

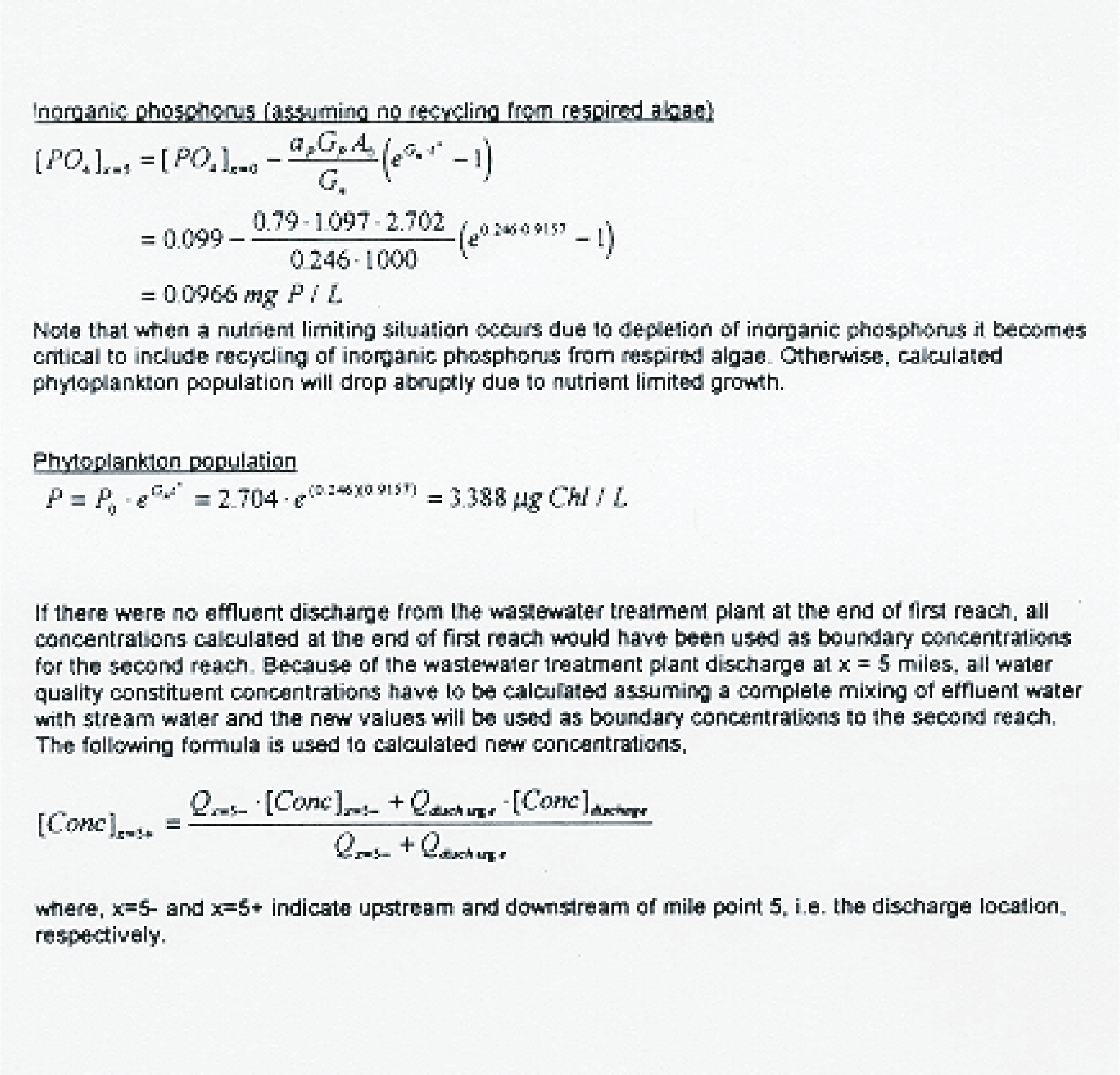
Nitrate and nitrite concentration as nitrogen

$$[NO_3 - NO_2]_{x=5} = [NO_3 - NO_2]_{x=0} + [NH_3]_{x=0} - [NH_3]_{x=5} - \frac{a_N G_p A_0}{G_n}\left(e^{G_n t'} - 1\right)$$

$$= 0.728 + 0.135 - 0.11 - \frac{5.8\cdot 1.097\cdot 2.702}{0.246\cdot 1000}\left(e^{0.246\cdot 0.9157} - 1\right)$$

$$= 0.683\ mg\ N\ /\ L$$

TABLE B-3. (CONTINUED)



Inorganic phosphorus (assuming no recycling from respired algae)

$$[PO_4]_{x=5} = [PO_4]_{x=0} - \frac{a_p G_p A_0}{G_n}\left(e^{G_n \cdot t^*} - 1\right)$$

$$= 0.099 - \frac{0.79 \cdot 1097 \cdot 2.702}{0.246 \cdot 1000}\left(e^{0.246 \cdot 0.9157} - 1\right)$$

$$= 0.0966 \; mg\ P / L$$

Note that when a nutrient limiting situation occurs due to depletion of inorganic phosphorus it becomes critical to include recycling of inorganic phosphorus from respired algae. Otherwise, calculated phytoplankton population will drop abruptly due to nutrient limited growth.

Phytoplankton population

$$P = P_0 \cdot e^{G_n t^*} = 2.704 \cdot e^{(0.246)(0.9157)} = 3.388 \; \mu g\ Chl / L$$

If there were no effluent discharge from the wastewater treatment plant at the end of first reach, all concentrations calculated at the end of first reach would have been used as boundary concentrations for the second reach. Because of the wastewater treatment plant discharge at x = 5 miles, all water quality constituent concentrations have to be calculated assuming a complete mixing of effluent water with stream water and the new values will be used as boundary concentrations to the second reach. The following formula is used to calculated new concentrations,

$$[Conc]_{x=5+} = \frac{Q_{x=5-} \cdot [Conc]_{x=5-} + Q_{discharge} \cdot [Conc]_{discharge}}{Q_{x=5-} + Q_{discharge}}$$

where, x=5- and x=5+ indicate upstream and downstream of mile point 5, i.e. the discharge location, respectively.

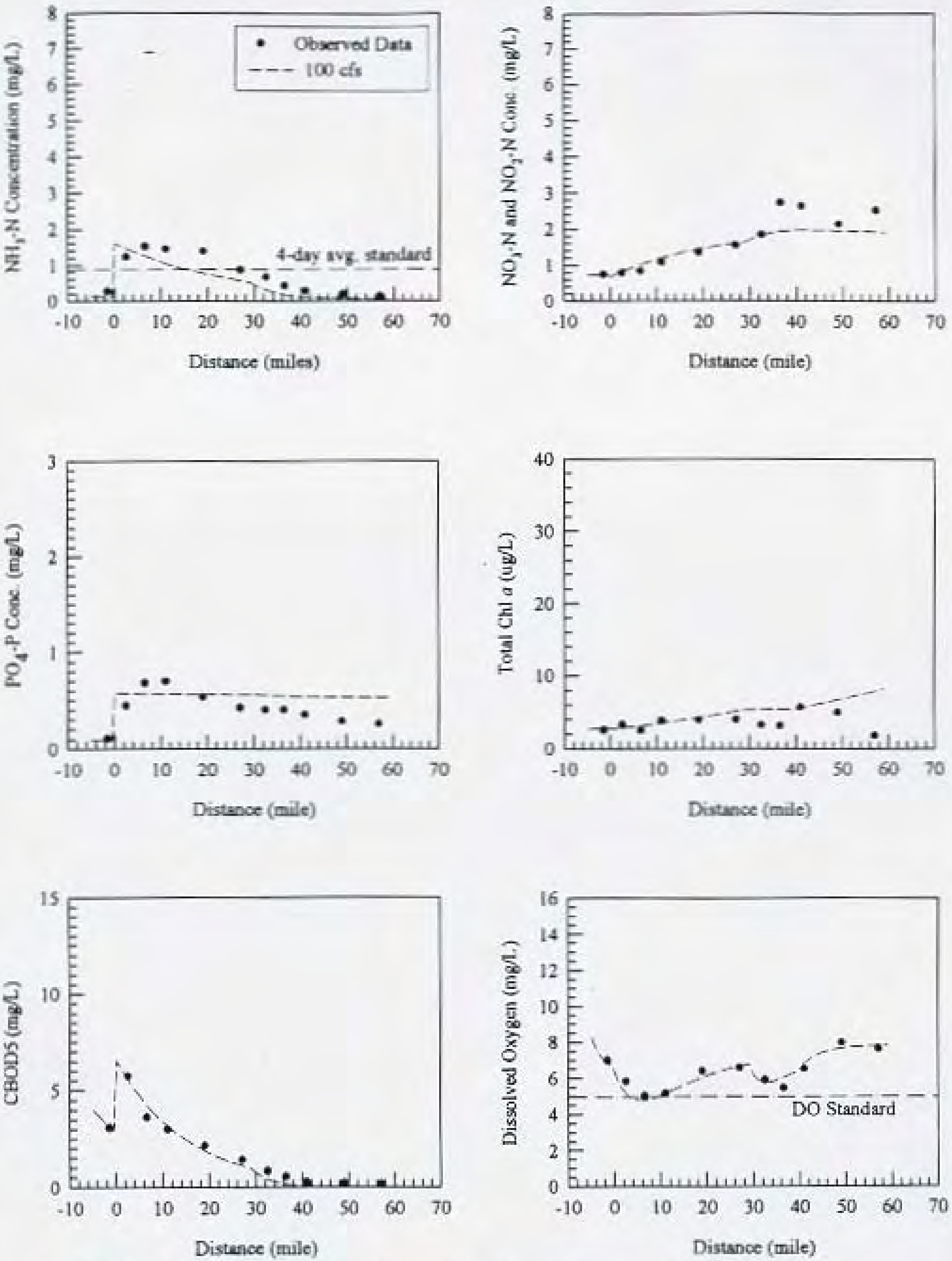


FIGURE B-5. CALIBRATION OF ANALYTICAL SOLUTION MODEL (August 23-24, 1979).

(Ka) is determined in accordance with Table A-24. In the wider section of the river, Ka drops below 0.5 day-1. Hydroscience (1971) suggests that the minimum value for oxygen transfer coefficient is 0.6 to 1.0 m/d. The minimum Ka is then determined by dividing the minimum oxygen transfer coefficient by the corresponding river depth.

In this example, the calculation of the DO profile agrees with the measured data quite favorably without any adjustments. In some cases, the calculated DO profile does not initially agree with the data because of sources and sinks of oxygen which may not be accounted for, such as SOD. Benthic oxygen demand, phytoplankton production and respiration are included in the analytical calculations. Because no exchange is considered between the benthic layer and the water column, observed NH3 data are found to be higher than model predicted values in Figure B-5. A good agreement between observed and simulated data is found in the chlorophyll a calibration. The observed inorganic phosphate data shows a decline with distance downstream which is not matched by the analytical solution. This difference is probably due to the assumption that only dissolved phosphate is considered in the analytical solution approach. The loss of phosphate by settling is therefore omitted.

Having calibrated a model for CBOD, DO, nutrients, and chlorophyll a (i.e., having defined site-specific coefficients and accepting that some reservations on reliability exist since the model is not tested against an independent data set), an analyst may use the model to project water quality impacts that might be expected under conditions of interest. Three different flow conditions are modeled for existing loading conditions to evaluate the range of conditions under which problems may occur. For each of the three flow conditions, the upstream boundary condition is varied to account for changes in nonpoint source loading contributions. Under the 30 cfs case, upstream flow is considered to be comprised wholly of baseflow. Ammonia, nitrate, inorganic phosphate, chlorophyll a, BOD5, and DO profiles are presented in Figure B-6.

The calculated profiles in Figure B-6 show that present wastewater loads would result in DO water quality standard violations over approximately 8 miles of the river under design 30 cfs drought flow conditions (7Q10 flow and a river temperature of 27oC). The lowest DO concentration is about 3.4 mg/L under 7Q10 flow conditions. Total ammonia violates the standard for approximately 25 miles downstream of discharge. For a river temperature of 27oC and pH of 7.75, the total ammonia standard that corresponds to an acceptable un-ionized ammonia level is 1.14 mg/L NH3-N. The highest predicted total ammonia concentration is more than three times the standard. It is important to note here that if the observed effluent ammonia concentration and the 100 cfs flow condition persists for 4 days, violations of the standard would occur over a 25-mile section of river downstream of the discharge. The BOD5 profile shows a significantly higher concentration immediately downstream of discharge, but it decreases rapidly within first 30 miles downstream. Another major concern is the growth of phytoplankton. The total chlorophyll a profile shows that under 7Q10 flow eutrophic conditions exist in the downstream reaches of the river. Accelerated growth of algal is likely to result in a reduction of the recreational value of the river.

The analysis of the three flow conditions shows that the initial selection of 30 cfs as the critical condition is justified. Developing the TMDL for the 30 cfs flow condition should be protective of other flow conditions and result in a conservative estimate of required load reduction.

The next step in the analysis is to consider three alternative levels of treatment for the wastewater treatment plant. The three different treatment scenarios are simulated using future population and land use conditions for the low flow critical condition (e.g., 30 cfs). For each management alternative, boundary conditions were defined by baseflow concentrations (equivalent to 30 cfs case under existing conditions). Each treatment alternative is then compared with the water quality criteria for DO and un-ionized ammonia. The narrative nutrient standard must be equated to a numeric measure for comparison with model results. Based on a review of similar rivers in the State a chlorophyll a threshold of 20 g/l is selected as a target goal. Continued monitoring should be used to reevaluate the target in future years.

Calculated ammonia, nitrate, phosphate, chlorophyll a, BOD5, and DO profiles for the projected wastewater loads are presented in Figure B-7. No significant difference exists between the effluent nutrient concentrations that result from trickling filter and activated sludge treatment processes under simulated future loading conditions. Therefore, ammonia, nitrate and chlorophyll a profiles for these two alternatives coincide. Effluent discharged from the

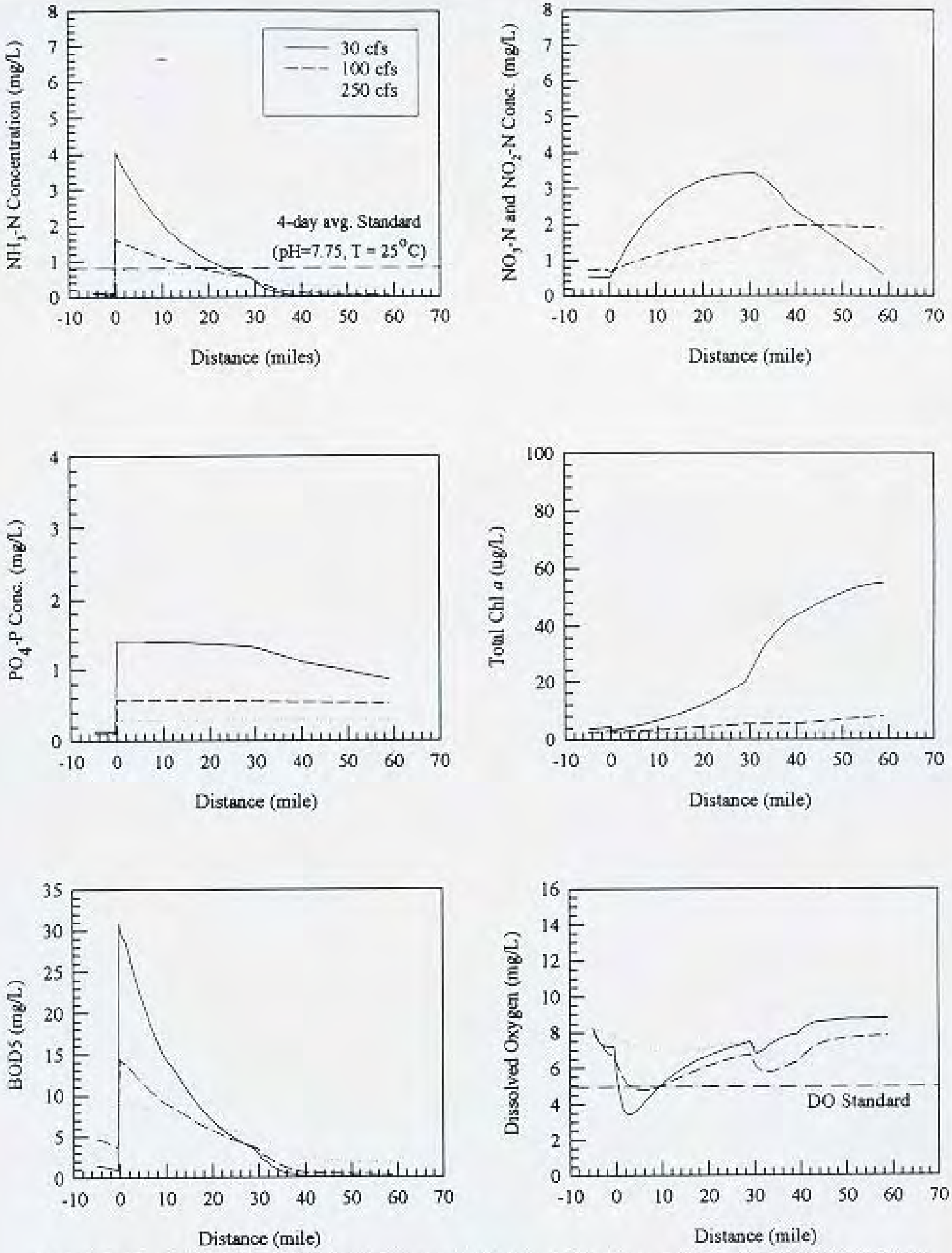


FIGURE B-6. CONCENTRATIONS OF NUTRIENTS, CHLOROPHYLL *A*, BOD$_5$ AND DO FOR DIFFERENT FLOW CONDITIONS UNDER EXISTING LOADS.

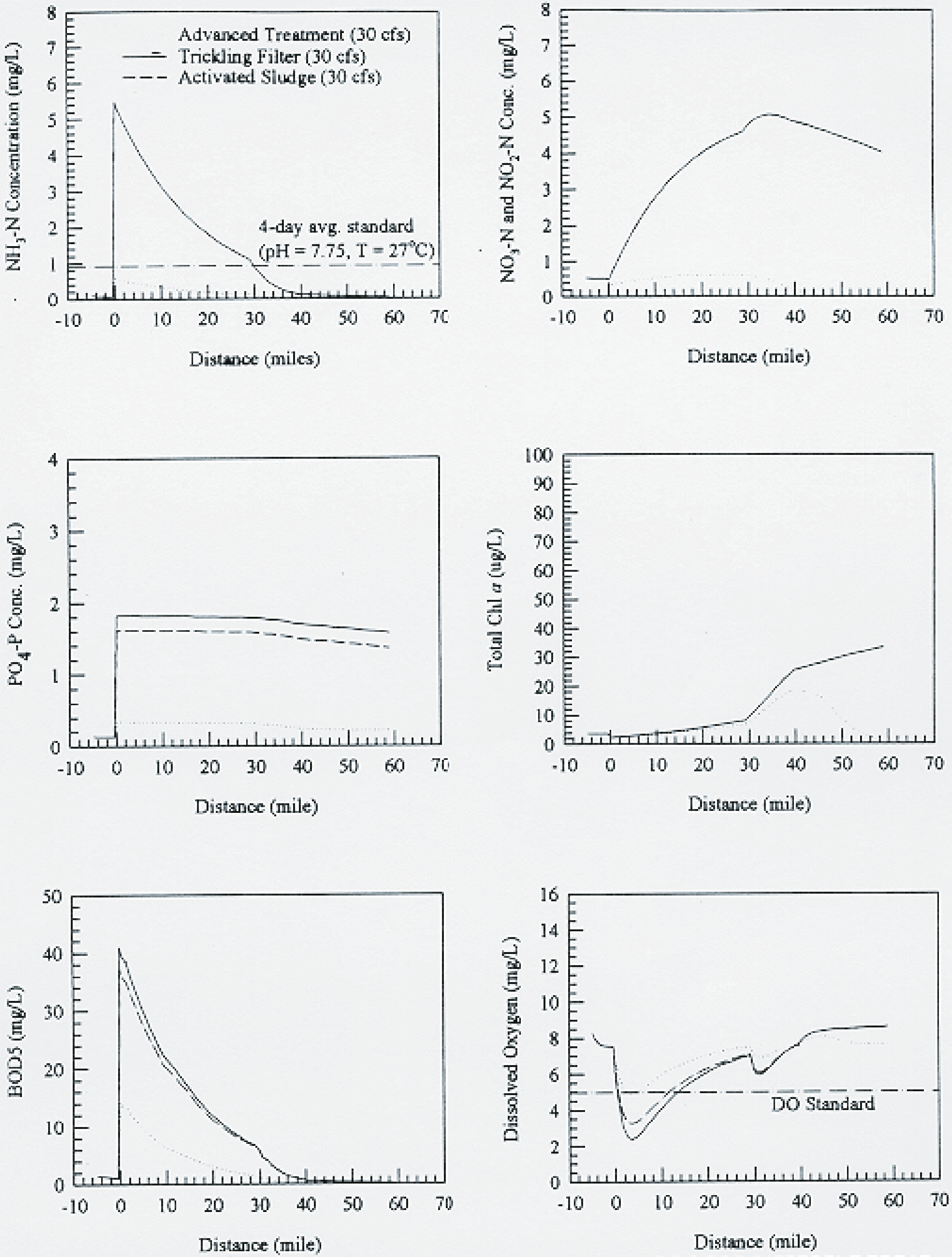


FIGURE B-7. PROJECTED NUTRIENTS, CHLOROPHYLL *A*, BOD$_5$ AND DO FOR DIFFERENT TREATMENT OPTIONS UNDER FUTURE LOADS.

trickling filter or an activated sludge treatment process is estimated to violate the total ammonia and the DO standards under the low flow critical condition. The minimum DO simulated for trickling filter and activated sludge processes are 2.4 and 3.5 mg/L, respectively. An algal bloom is predicted to continue up to the end of 65-mile study area and reaches 30 ug/L of chlorophyll a. Advanced water treatment is the only option which allows the river to meet total ammonia and DO standards at 30 cfs. It also controls the algal growth to a maximum of 18.6 ug/L. The lowest daily average DO concentration of about 5.0 mg/L occurs at 3 miles downstream of discharge location. A sensitivity analysis is recommended when differences between different options are small. Sensitivity analysis can also be used in the determination of Margin of Safety (MOS). More refined analysis can be used to reduce the MOS and in some cases increase the allowable discharge.

B.2.5WASP5 EXAMPLE

This example shows an alternative approach to analysis of the Rivanna River example and considers in greater detail the in-stream impacts of changes in the contributions from nonpoint source runoff, baseflow, and effluent concentrations to the river reaches. WASP5 includes more detailed transformation and exchange processes than an analytical approach. For example WASP5 can account for the settling of inorganic particulate material, recycling of organic nutrients to the inorganic pool, and nutrient flux from the bottom sediment layer to the water column. This application illustrates the capabilities of a steady state WASP5 model application, similar to the previous example. In addition, the development of a WASP5 model allows the user the flexibility to examine continuous simulation results should further examination of dynamic nonpoint sources loadings prove necessary.

The WASP5 model is developed based on the data provided in Problem Setting and River Characterization (Sections B.2.1 and B.2.2). The WASP5 model is configured as 45 segments. The first 35 miles are represented by 15 one-mile segments followed by 10 two-mile segments. The wider portion of the river is divided into 10 one-mile segments followed by 20 miles divided into 10 two-mile segments. Additional data required by the WASP5 model includes downstream boundary conditions, geometric data for each segment, and air temperature. Downstream boundary conditions and initial conditions are estimated based on observed stream data. Depth and width for each segment are derived as a function of flow and hydraulic coefficients.

The model is executed using a timestep of 0.05 days (1.2 hours) for the three flow conditions and three treatment scenarios considered earlier. A 30-day simulation period is used to allow sufficient time for the model to reach steady-state conditions. The modeling results are based on the final 5 days of the simulation period.

The first step in any model application is to calibrate the model with existing data. For this example, the WASP5 model is calibrated to the observed data gathered from August 23-24, 1979. Figure B-8 shows the model calibration analyses for WASP5 with the observed data. The analytical solution, described in the previous section, provides an additional check for model calibration. Analytical solutions are generally recommended for evaluation of model performance. This comparison demonstrates the ability of each model to mimic the water quality responses of the river and shows the similarity between the results of each model for a steady-state application. Although the results agree with the measured data, the WASP5 run has significantly higher input requirements and demand a greater level of effort for application. Table B-4 shows a listing of the WASP5 input file.

Similar to the analytical solution, once calibrated, WASP5 was run for three different flow scenarios (30 cfs, 100 cfs, and 250 cfs) and three treatment scenarios for future point source loading (30 cfs flow only) for a total of 6 runs. Figures B-9 and B-10 show the resulting (based on the last 5 days of the simulation period) in-stream concentrations of oxygen, BOD5, ammonia, nitrate, total organic nitrogen, and total nitrogen for existing and future conditions, respectively. The worst case, with DO below the recommended limit of 5 mg/l, occurs under the 30 cfs low flow condition (Figure B-9). Under low flow conditions, the baseflow contribution is assumed to be constant and changes are attributed solely to the increase in load from the treatment plant. WASP5 simulation results show problems very similar to those of analytical solution approach for ammonia, nitrate, chlorophyll a and DO. Interestingly, with a 20 percent particulate phosphate and a 0.5 m/d settling rate, WASP5 is well calibrated for instream phosphate concentrations. This is important because it causes nutrient inhibition for phytoplankton growth and a resulting decline of chlorophyll *a*

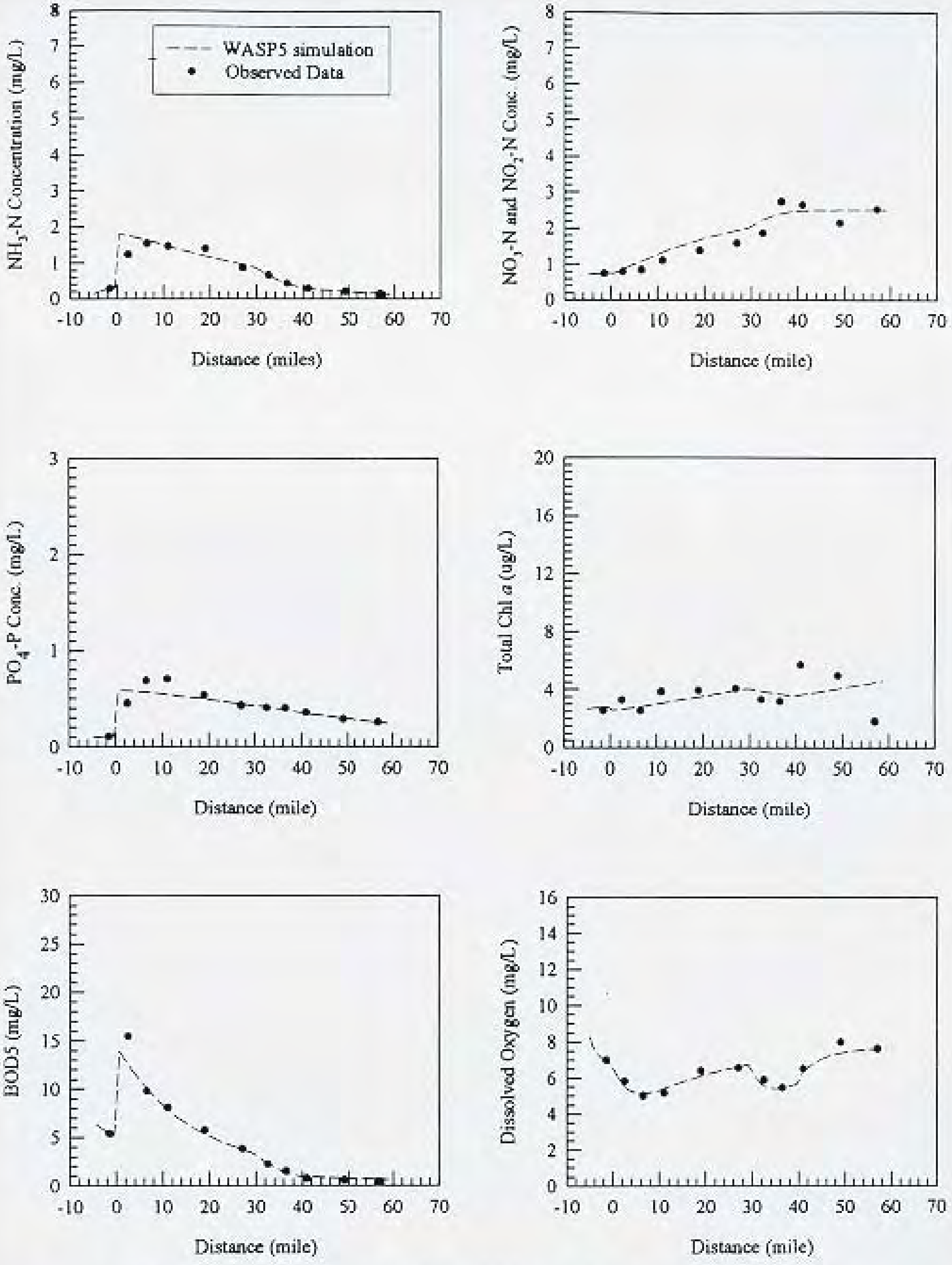
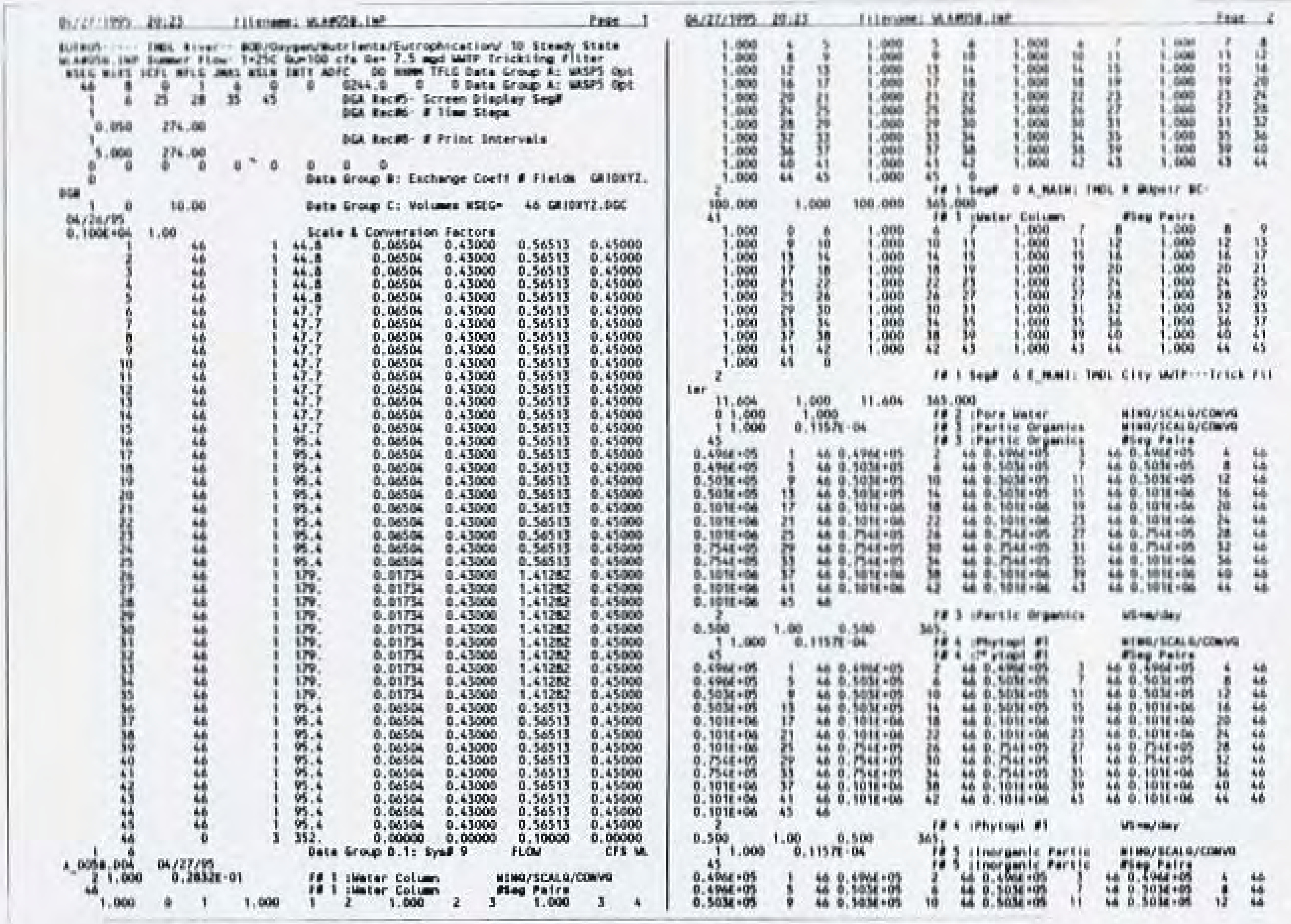


FIGURE B-8. CALIBRATION OF WASP5 MODEL USING AUGUST 23-24, 1979 DATA.

TABLE B-4. SAMPLE WASP INPUT FILE



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 NSEG NSYS ICFL MFLG JMAS NSLN INTY ADFC   DD HHMM TFLG Data Group A: WASP5 Opt
   46    8    0    1    0    0    0    0244.0    0    0 Data Group A: WASP5 Opt
    1    6   25   28   35   45           DGA Rec#5- Screen Display Seg#
    1                                    DGA Rec#6- # Time Steps
    0.050    274.00
    1                                    DGA Rec#8- # Print Intervals
    5.000    274.00
    0    0    0    0    0    0    0    0    0
    0                               Data Group B: Exchange Coeff # Fields  GRIDXYZ.
DGB
    1    0     10.00                Data Group C: Volumes NSEG=   46 GRIDXYZ.DGC
 04/26/95
 0.100E+04   1.00                   Scale & Conversion Factors
        1         46        1  44.8      0.06504   0.43000   0.56513   0.45000
        2         46        1  44.8      0.06504   0.43000   0.56513   0.45000
        3         46        1  44.8      0.06504   0.43000   0.56513   0.45000
        4         46        1  44.8      0.06504   0.43000   0.56513   0.45000
        5         46        1  44.8      0.06504   0.43000   0.56513   0.45000
        6         46        1  47.7      0.06504   0.43000   0.56513   0.45000
        7         46        1  47.7      0.06504   0.43000   0.56513   0.45000
        8         46        1  47.7      0.06504   0.43000   0.56513   0.45000
        9         46        1  47.7      0.06504   0.43000   0.56513   0.45000
       10         46        1  47.7      0.06504   0.43000   0.56513   0.45000
       11         46        1  47.7      0.06504   0.43000   0.56513   0.45000
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       15         46        1  47.7      0.06504   0.43000   0.56513   0.45000
       16         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       17         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       18         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       19         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       20         46        1  95.4      0.06504   0.43000   0.56513   0.45000
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       22         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       23         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       24         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       25         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       26         46        1  179.      0.01734   0.43000   1.41282   0.45000
       27         46        1  179.      0.01734   0.43000   1.41282   0.45000
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       30         46        1  179.      0.01734   0.43000   1.41282   0.45000
       31         46        1  179.      0.01734   0.43000   1.41282   0.45000
       32         46        1  179.      0.01734   0.43000   1.41282   0.45000
       33         46        1  179.      0.01734   0.43000   1.41282   0.45000
       34         46        1  179.      0.01734   0.43000   1.41282   0.45000
       35         46        1  179.      0.01734   0.43000   1.41282   0.45000
       36         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       37         46        1  95.4      0.06504   0.43000   0.56513   0.45000
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       41         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       42         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       43         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       44         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       45         46        1  95.4      0.06504   0.43000   0.56513   0.45000
       46          0        3  352.      0.00000   0.00000   0.10000   0.00000
        1
        0                           Data Group D.1: Sys# 9       FLOW       CFS WL
A_005B.DOA   04/27/95
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   44                               F# 1 :Water Column      #Seg Pairs
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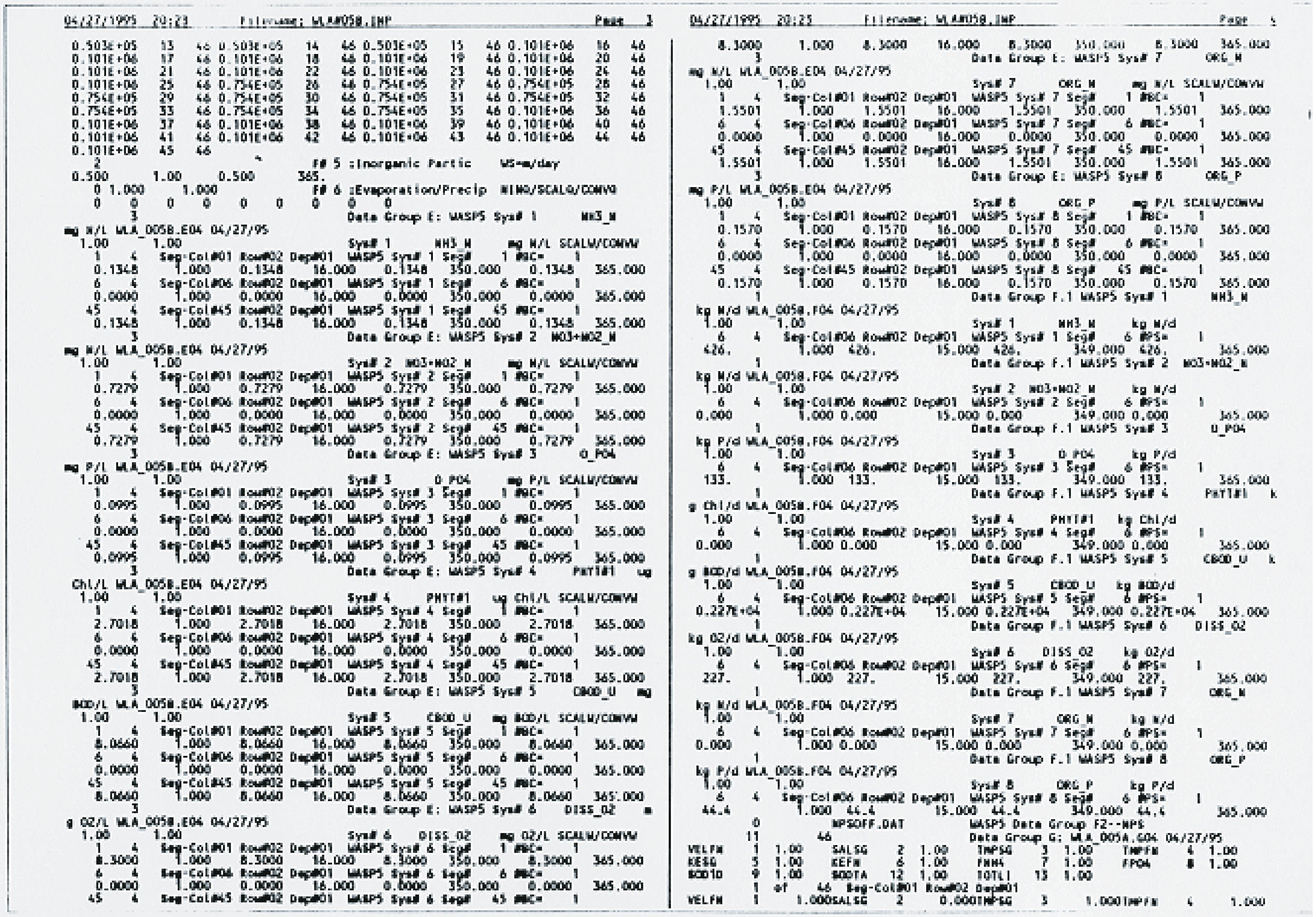
     1.000   12   13     1.000   13   14     1.000   14   15     1.000   15   16
     1.000   16   17     1.000   17   18     1.000   18   19     1.000   19   20
     1.000   20   21     1.000   21   22     1.000   22   23     1.000   23   24
     1.000   24   25     1.000   25   26     1.000   26   27     1.000   27   28
     1.000   28   29     1.000   29   30     1.000   30   31     1.000   31   32
     1.000   32   33     1.000   33   34     1.000   34   35     1.000   35   36
     1.000   36   37     1.000   37   38     1.000   38   39     1.000   39   40
     1.000   40   41     1.000   41   42     1.000   42   43     1.000   43   44
     1.000   44   45     1.000   45    0
    2                               F# 1 Seg#  0 A_MAIN: TMDL R @Upstr BC-
  100.000      1.000   100.000   365.000
   41                               F# 1 :Water Column      #Seg Pairs
     1.000    0    6     1.000    6    7     1.000    7    8     1.000    8    9
     1.000    9   10     1.000   10   11     1.000   11   12     1.000   12   13
     1.000   13   14     1.000   14   15     1.000   15   16     1.000   16   17
     1.000   17   18     1.000   18   19     1.000   19   20     1.000   20   21
     1.000   21   22     1.000   22   23     1.000   23   24     1.000   24   25
     1.000   25   26     1.000   26   27     1.000   27   28     1.000   28   29
     1.000   29   30     1.000   30   31     1.000   31   32     1.000   32   33
     1.000   33   34     1.000   34   35     1.000   35   36     1.000   36   37
     1.000   37   38     1.000   38   39     1.000   39   40     1.000   40   41
     1.000   41   42     1.000   42   43     1.000   43   44     1.000   44   45
     1.000   45    0
    2                               F# 1 Seg#  6 E_MAIN: TMDL City WWTP···Trick Fil
ter
   11.604      1.000    11.604   365.000
    0 1.000      1.000              F# 2 :Pore Water        NINQ/SCALQ/CONVQ
    1 1.000    0.1157E-04           F# 3 :Partic Organics   NINQ/SCALQ/CONVQ
   45                               F# 3 :Partic Organics   #Seg Pairs
 0.496E+05    1   46 0.496E+05    2   46 0.496E+05    3   46 0.496E+05    4   46
 0.496E+05    5   46 0.503E+05    6   46 0.503E+05    7   46 0.503E+05    8   46
 0.503E+05    9   46 0.503E+05   10   46 0.503E+05   11   46 0.503E+05   12   46
 0.503E+05   13   46 0.503E+05   14   46 0.503E+05   15   46 0.101E+06   16   46
 0.101E+06   17   46 0.101E+06   18   46 0.101E+06   19   46 0.101E+06   20   46
 0.101E+06   21   46 0.101E+06   22   46 0.101E+06   23   46 0.101E+06   24   46
 0.101E+06   25   46 0.754E+05   26   46 0.754E+05   27   46 0.754E+05   28   46
 0.754E+05   29   46 0.754E+05   30   46 0.754E+05   31   46 0.754E+05   32   46
 0.754E+05   33   46 0.754E+05   34   46 0.754E+05   35   46 0.101E+06   36   46
 0.101E+06   37   46 0.101E+06   38   46 0.101E+06   39   46 0.101E+06   40   46
 0.101E+06   41   46 0.101E+06   42   46 0.101E+06   43   46 0.101E+06   44   46
 0.101E+06   45   46
    2                               F# 3 :Partic Organics   WS-m/day
 0.500       1.00      0.500       365.
    1 1.000    0.1157E-04           F# 4 :Phytopl #1        NINQ/SCALQ/CONVQ
   45                               F# 4 :Phytopl #1        #Seg Pairs
 0.496E+05    1   46 0.496E+05    2   46 0.496E+05    3   46 0.496E+05    4   46
 0.496E+05    5   46 0.503E+05    6   46 0.503E+05    7   46 0.503E+05    8   46
 0.503E+05    9   46 0.503E+05   10   46 0.503E+05   11   46 0.503E+05   12   46
 0.503E+05   13   46 0.503E+05   14   46 0.503E+05   15   46 0.101E+06   16   46
 0.101E+06   17   46 0.101E+06   18   46 0.101E+06   19   46 0.101E+06   20   46
 0.101E+06   21   46 0.101E+06   22   46 0.101E+06   23   46 0.101E+06   24   46
 0.101E+06   25   46 0.754E+05   26   46 0.754E+05   27   46 0.754E+05   28   46
 0.754E+05   29   46 0.754E+05   30   46 0.754E+05   31   46 0.754E+05   32   46
 0.754E+05   33   46 0.754E+05   34   46 0.754E+05   35   46 0.101E+06   36   46
 0.101E+06   37   46 0.101E+06   38   46 0.101E+06   39   46 0.101E+06   40   46
 0.101E+06   41   46 0.101E+06   42   46 0.101E+06   43   46 0.101E+06   44   46
 0.101E+06   45   46
    2                               F# 4 :Phytopl #1        WS-m/day
 0.500       1.00      0.500       365.
    1 1.000    0.1157E-04           F# 5 :Inorganic Partic  NINQ/SCALQ/CONVQ
   45                               F# 5 :Inorganic Partic  #Seg Pairs
 0.496E+05    1   46 0.496E+05    2   46 0.496E+05    3   46 0.496E+05    4   46
 0.496E+05    5   46 0.503E+05    6   46 0.503E+05    7   46 0.503E+05    8   46
 0.503E+05    9   46 0.503E+05   10   46 0.503E+05   11   46 0.503E+05   12   46
```

TABLE B-4. (CONTINUED)



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04/27/1995  20:23        Filename: WLA#05B.INP                          Page   3

  0.503E+05   13    46 0.503E+05   14    46 0.503E+05   15    46 0.101E+06   16    46
  0.101E+06   17    46 0.101E+06   18    46 0.101E+06   19    46 0.101E+06   20    46
  0.101E+06   21    46 0.101E+06   22    46 0.101E+06   23    46 0.101E+06   24    46
  0.101E+06   25    46 0.754E+05   26    46 0.754E+05   27    46 0.754E+05   28    46
  0.754E+05   29    46 0.754E+05   30    46 0.754E+05   31    46 0.754E+05   32    46
  0.754E+05   33    46 0.754E+05   34    46 0.754E+05   35    46 0.101E+06   36    46
  0.101E+06   37    46 0.101E+06   38    46 0.101E+06   39    46 0.101E+06   40    46
  0.101E+06   41    46 0.101E+06   42    46 0.101E+06   43    46 0.101E+06   44    46
  0.101E+06   45    46
     2                                F# 5 :Inorganic Partic    WS=m/day
  0.500      1.00      0.500      365.
     0 1.000      1.000           F# 6 :Evaporation/Precip  NINQ/SCALQ/CONVQ
     0     0     0     0     0     0     0     0     0
          3                            Data Group E: WASP5 Sys# 1      NH3_N
mg N/L WLA_005B.E04 04/27/95
   1.00        1.00                    Sys# 1       NH3_N      mg N/L SCALW/CONVW
     1     4    Seg-Col#01 Row#02 Dep#01  WASP5 Sys# 1 Seg#     1 #BC=     1
     0.1348      1.000    0.1348     16.000    0.1348   350.000     0.1348   365.000
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 1 Seg#     6 #BC=     1
     0.0000      1.000    0.0000     16.000    0.0000   350.000     0.0000   365.000
    45     4    Seg-Col#45 Row#02 Dep#01  WASP5 Sys# 1 Seg#    45 #BC=     1
     0.1348      1.000    0.1348     16.000    0.1348   350.000     0.1348   365.000
          3                            Data Group E: WASP5 Sys# 2  NO3+NO2_N
mg N/L WLA_005B.E04 04/27/95
   1.00        1.00                    Sys# 2  NO3+NO2_N     mg N/L SCALW/CONVW
     1     4    Seg-Col#01 Row#02 Dep#01  WASP5 Sys# 2 Seg#     1 #BC=     1
     0.7279      1.000    0.7279     16.000    0.7279   350.000     0.7279   365.000
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 2 Seg#     6 #BC=     1
     0.0000      1.000    0.0000     16.000    0.0000   350.000     0.0000   365.000
    45     4    Seg-Col#45 Row#02 Dep#01  WASP5 Sys# 2 Seg#    45 #BC=     1
     0.7279      1.000    0.7279     16.000    0.7279   350.000     0.7279   365.000
          3                            Data Group E: WASP5 Sys# 3      O_PO4
mg P/L WLA_005B.E04 04/27/95
   1.00        1.00                    Sys# 3       O PO4      mg P/L SCALW/CONVW
     1     4    Seg-Col#01 Row#02 Dep#01  WASP5 Sys# 3 Seg#     1 #BC=     1
     0.0995      1.000    0.0995     16.000    0.0995   350.000     0.0995   365.000
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 3 Seg#     6 #BC=     1
     0.0000      1.000    0.0000     16.000    0.0000   350.000     0.0000   365.000
    45     4    Seg-Col#45 Row#02 Dep#01  WASP5 Sys# 3 Seg#    45 #BC=     1
     0.0995      1.000    0.0995     16.000    0.0995   350.000     0.0995   365.000
          3                            Data Group E: WASP5 Sys# 4      PHYT#1    ug
 Chl/L WLA_005B.E04 04/27/95
   1.00        1.00                    Sys# 4       PHYT#1    ug Chl/L SCALW/CONVW
     1     4    Seg-Col#01 Row#02 Dep#01  WASP5 Sys# 4 Seg#     1 #BC=     1
     2.7018      1.000    2.7018     16.000    2.7018   350.000     2.7018   365.000
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 4 Seg#     6 #BC=     1
     0.0000      1.000    0.0000     16.000    0.0000   350.000     0.0000   365.000
    45     4    Seg-Col#45 Row#02 Dep#01  WASP5 Sys# 4 Seg#    45 #BC=     1
     2.7018      1.000    2.7018     16.000    2.7018   350.000     2.7018   365.000
          3                            Data Group E: WASP5 Sys# 5      CBOD_U    mg
 BOD/L WLA_005B.E04 04/27/95
   1.00        1.00                    Sys# 5       CBOD_U    mg BOD/L SCALW/CONVW
     1     4    Seg-Col#01 Row#02 Dep#01  WASP5 Sys# 5 Seg#     1 #BC=     1
     8.0660      1.000    8.0660     16.000    8.0660   350.000     8.0660   365.000
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 5 Seg#     6 #BC=     1
     0.0000      1.000    0.0000     16.000    0.0000   350.000     0.0000   365.000
    45     4    Seg-Col#45 Row#02 Dep#01  WASP5 Sys# 5 Seg#    45 #BC=     1
     8.0660      1.000    8.0660     16.000    8.0660   350.000     8.0660   365.000
          3                            Data Group E: WASP5 Sys# 6    DISS_O2      m
g O2/L WLA_005B.E04 04/27/95
   1.00        1.00                    Sys# 6    DISS_O2      mg O2/L SCALW/CONVW
     1     4    Seg-Col#01 Row#02 Dep#01  WASP5 Sys# 6 Seg#     1 #BC=     1
     8.3000      1.000    8.3000     16.000    8.3000   350.000     8.3000   365.000
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 6 Seg#     6 #BC=     1
     0.0000      1.000    0.0000     16.000    0.0000   350.000     0.0000   365.000
    45     4    Seg-Col#45 Row#02 Dep#01  WASP5 Sys# 6 Seg#    45 #BC=     1
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04/27/1995  20:23        Filename: WLA#05B.INP                          Page   5

     8.3000      1.000    8.3000     16.000    8.3000   350.000     8.3000   365.000
          3                            Data Group E: WASP5 Sys# 7      ORG_N
mg N/L WLA_005B.E04 04/27/95
   1.00        1.00                    Sys# 7       ORG_N      mg N/L SCALW/CONVW
     1     4    Seg-Col#01 Row#02 Dep#01  WASP5 Sys# 7 Seg#     1 #BC=     1
     1.5501      1.000    1.5501     16.000    1.5501   350.000     1.5501   365.000
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 7 Seg#     6 #BC=     1
     0.0000      1.000    0.0000     16.000    0.0000   350.000     0.0000   365.000
    45     4    Seg-Col#45 Row#02 Dep#01  WASP5 Sys# 7 Seg#    45 #BC=     1
     1.5501      1.000    1.5501     16.000    1.5501   350.000     1.5501   365.000
          3                            Data Group E: WASP5 Sys# 8      ORG_P
mg P/L WLA_005B.E04 04/27/95
   1.00        1.00                    Sys# 8       ORG_P      mg P/L SCALW/CONVW
     1     4    Seg-Col#01 Row#02 Dep#01  WASP5 Sys# 8 Seg#     1 #BC=     1
     0.1570      1.000    0.1570     16.000    0.1570   350.000     0.1570   365.000
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 8 Seg#     6 #BC=     1
     0.0000      1.000    0.0000     16.000    0.0000   350.000     0.0000   365.000
    45     4    Seg-Col#45 Row#02 Dep#01  WASP5 Sys# 8 Seg#    45 #BC=     1
     0.1570      1.000    0.1570     16.000    0.1570   350.000     0.1570   365.000
          1                            Data Group F.1 WASP5 Sys# 1      NH3_N
 kg N/d WLA_005B.F04 04/27/95
   1.00        1.00                    Sys# 1       NH3_N      kg N/d
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 1 Seg#     6 #PS=     1
  426.           1.000  426.          15.000  426.          349.000  426.          365.000
          1                            Data Group F.1 WASP5 Sys# 2  NO3+NO2_N
 kg N/d WLA_005B.F04 04/27/95
   1.00        1.00                    Sys# 2  NO3+NO2_N     kg N/d
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 2 Seg#     6 #PS=     1
  0.000          1.000 0.000          15.000 0.000          349.000 0.000          365.000
          1                            Data Group F.1 WASP5 Sys# 3      O_PO4
 kg P/d WLA_005B.F04 04/27/95
   1.00        1.00                    Sys# 3       O PO4      kg P/d
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 3 Seg#     6 #PS=     1
  133.           1.000  133.          15.000  133.          349.000  133.          365.000
          1                            Data Group F.1 WASP5 Sys# 4      PHYT#1    k
g Chl/d WLA_005B.F04 04/27/95
   1.00        1.00                    Sys# 4       PHYT#1    kg Chl/d
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 4 Seg#     6 #PS=     1
  0.000          1.000 0.000          15.000 0.000          349.000 0.000          365.000
          1                            Data Group F.1 WASP5 Sys# 5      CBOD_U    k
g BOD/d WLA_005B.F04 04/27/95
   1.00        1.00                    Sys# 5       CBOD_U    kg BOD/d
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 5 Seg#     6 #PS=     1
  0.227E+04      1.000 0.227E+04      15.000 0.227E+04      349.000 0.227E+04      365.000
          1                            Data Group F.1 WASP5 Sys# 6    DISS_O2
kg O2/d WLA_005B.F04 04/27/95
   1.00        1.00                    Sys# 6    DISS_O2      kg O2/d
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 6 Seg#     6 #PS=     1
  227.           1.000  227.          15.000  227.          349.000  227.          365.000
          1                            Data Group F.1 WASP5 Sys# 7      ORG_N
 kg N/d WLA_005B.F04 04/27/95
   1.00        1.00                    Sys# 7       ORG_N      kg N/d
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 7 Seg#     6 #PS=     1
  0.000          1.000 0.000          15.000 0.000          349.000 0.000          365.000
          1                            Data Group F.1 WASP5 Sys# 8      ORG_P
 kg P/d WLA_005B.F04 04/27/95
   1.00        1.00                    Sys# 8       ORG_P      kg P/d
     6     4    Seg-Col#06 Row#02 Dep#01  WASP5 Sys# 8 Seg#     6 #PS=     1
  44.4           1.000  44.4          15.000  44.4          349.000  44.4          365.000
          0          NPSOFF.DAT        WASP5 Data Group F2--NPS
         11       46                   Data Group G: WLA_005A.G04 04/27/95
VELFN     1  1.00     SALSG     2  1.00     TMPSG     3  1.00     TMPFN     4  1.00
KESG      5  1.00     KEFN      6  1.00     FNH4      7  1.00     FPO4      8  1.00
BOD10     9  1.00     BODTA    12  1.00     TOTLI    13  1.00
          1  #f     46  Seg-Col#01 Row#02 Dep#01
VELFN     1      1.000SALSG     2      0.000TMPSG     3      1.000TMPFN     4      1.000
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TABLE B-4. (CONTINUED)

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04/27/1995  20:23        Filename: WLA#05B.INP                                      Page  5
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
         2  of     46  Seg-Col#02 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
         3  of     46  Seg-Col#03 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA  ~12     1.065TOTLI   13     1.000
         4  of     46  Seg-Col#04 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
         5  of     46  Seg-Col#05 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
         6  of     46  Seg-Col#06 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
         7  of     46  Seg-Col#07 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
         8  of     46  Seg-Col#08 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
         9  of     46  Seg-Col#09 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        10  of     46  Seg-Col#10 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        11  of     46  Seg-Col#11 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        12  of     46  Seg-Col#12 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        13  of     46  Seg-Col#13 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        14  of     46  Seg-Col#14 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        15  of     46  Seg-Col#15 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        16  of     46  Seg-Col#16 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        17  of     46  Seg-Col#17 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
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04/27/1995  20:23        Filename: WLA#05B.INP                                      Page  6
        18  of     46  Seg-Col#18 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        19  of     46  Seg-Col#19 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        20  of     46  Seg-Col#20 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        21  of     46  Seg-Col#21 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        22  of     46  Seg-Col#22 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        23  of     46  Seg-Col#23 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        24  of     46  Seg-Col#24 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        25  of     46  Seg-Col#25 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     1.000
KESG     5     1.500KEFN     6     1.000FNH4     7    49.538FPO4     8     6.880
SOD1D    9     0.750SODTA   12     1.065TOTLI   13     1.000
        26  of     46  Seg-Col#26 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
KESG     5     1.500KEFN     6     2.000FNH4     7    82.563FPO4     8    11.467
SOD1D    9     1.250SODTA   12     1.065TOTLI   13     1.000
        27  of     46  Seg-Col#27 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
KESG     5     1.500KEFN     6     2.000FNH4     7    82.563FPO4     8    11.467
SOD1D    9     1.250SODTA   12     1.065TOTLI   13     1.000
        28  of     46  Seg-Col#28 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
KESG     5     1.500KEFN     6     2.000FNH4     7    82.563FPO4     8    11.467
SOD1D    9     1.250SODTA   12     1.065TOTLI   13     1.000
        29  of     46  Seg-Col#29 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
KESG     5     1.500KEFN     6     2.000FNH4     7    82.563FPO4     8    11.467
SOD1D    9     1.250SODTA   12     1.065TOTLI   13     1.000
        30  of     46  Seg-Col#30 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
KESG     5     1.500KEFN     6     2.000FNH4     7    82.563FPO4     8    11.467
SOD1D    9     1.250SODTA   12     1.065TOTLI   13     1.000
        31  of     46  Seg-Col#31 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
KESG     5     1.500KEFN     6     2.000FNH4     7    82.563FPO4     8    11.467
SOD1D    9     1.250SODTA   12     1.065TOTLI   13     1.000
        32  of     46  Seg-Col#32 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
KESG     5     1.500KEFN     6     2.000FNH4     7    82.563FPO4     8    11.467
SOD1D    9     1.250SODTA   12     1.065TOTLI   13     1.000
        33  of     46  Seg-Col#33 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
KESG     5     1.500KEFN     6     2.000FNH4     7    82.563FPO4     8    11.467
SOD1D    9     1.250SODTA   12     1.065TOTLI   13     1.000
        34  of     46  Seg-Col#34 Row#02 Dep#01
VELFN    1     1.000SALSG    2     0.000TMPSG    3     1.000TMPFN     4     2.000
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04/27/1995  20:23        Filename: WLA#05B.INP                                          Page  7
KESG      5      1.500KEFN      6        2.000FNH4      7     82.563FPO4       8      11.467
SOD1D     9      1.250SODTA    12        1.065TOTLI    13      1.000
         35   of      46  Seg-Col#35 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       2.000
KESG      5      1.500KEFN      6        2.000FNH4      7     82.563FPO4       8      11.467
SOD1D     9      1.250SODTA    12        1.065TOTLI    13      1.000
         36   of      46  Seg-Col#36 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA  ^ 12        1.065TOTLI    13      1.000
         37   of      46  Seg-Col#37 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA    12        1.065TOTLI    13      1.000
         38   of      46  Seg-Col#38 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA    12        1.065TOTLI    13      1.000
         39   of      46  Seg-Col#39 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.75[illegible]TA    12        1.065TOTLI    13      1.000
         40   of      [illegible]  [illegible]g-Col#40 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA    12        1.065TOTLI    13      1.000
         41   of      46  Seg-Col#41 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA    12        1.065TOTLI    13      1.000
         42   of      46  Seg-Col#42 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA    12        1.065TOTLI    13      1.000
         43   of      46  Seg-Col#43 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA    12        1.065TOTLI    13      1.000
         44   of      46  Seg-Col#44 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA    12        1.065TOTLI    13      1.000
         45   of      46  Seg-Col#45 Row#02 Dep#01
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      1.500KEFN      6        1.000FNH4      7     49.538FPO4       8       6.880
SOD1D     9      0.750SODTA    12        1.065TOTLI    13      1.000
         46   of      46  Seg-Col#01 Row#01 Dep#02
VELFN     1      1.000SALSG     2        0.000TMPSG     3      1.000TMPFN      4       1.000
KESG      5      0.000KEFN      6        1.000FNH4      7      0.000FPO4       8       0.000
SOD1D     9      0.000SODTA    12        0.000TOTLI    13      0.000
WLA#05A.DGH                              Data Group H: #Sys=09 04/27/95
GLOBAL              1
 All Const         43
       WTYPE        1  2.00          K12C        11 0.150
        K12T       12  1.08          KNIT        13  0.50
        K20C       21 0.15           K20T        22  1.045
        KNO3       23 0.500          K1C         41  1.80
         K1T       42  1.066         LGHTS       43  1.00
       PHIMX       44  720.          XKC         45 0.170E-01
        CCHL       46  33.0          IS1         47  200.
       KMNG1       48 0.025          KMPG1       49 0.005
        K1RC       50 0.200          K1RT        51  1.08
         K1D       52 0.000          K1G         53  0.00
      NUTLIM       54 0.000          KPZDC       55 0.000
       KPZDT       56  1.08          PCRB1       57 0.240E-01
       NCRB1       58 0.176          KMPHYT      59 0.500E-02
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04/27/1995  20:23        Filename: WLA#05B.INP                                          Page  8
         KDC          71 0.300         KDT        72  1.047
        KDSC          73 0.000         KDST       74  1.08
        KBOD          75 0.500         OCRB       81  2.67
          K2          82 0.000         K71C       91 0.150
        K71T          92  1.08         KONDC      93 0.000
       KONDT          94  1.08         FON        95 0.500
        K83C         100 0.150         K83T      101  1.08
       KOPDC         102 0.000         KOPDT     103  1.08
         FOP         104 0.500
NH3-N                  0
NO3+NO2-N              0
O-PO4                  0
Phyt#1                 0
CBOD                   0
Diss O2                0
Org-N                  0
Org-P                  0
        15                              Data Group I: WLA#05B Qw=100 cfs Qe=7.5 mg
d
TEMP1     4     1   FT#01-Temp #1               WLA#05B.TEM     4 11 FF time # 1 Temp#
1
   25.0000      1.00   25.0000      16.00   25.0000    350.00   25.0000     365.00
TEMP2     4     2   FT#02-Temp #2               WLA#05B.TEM     5 11 FF time # 2 Temp#
2
   25.0000      1.00   25.0000      16.00   25.0000    350.00   25.0000     365.00
SOLAR     4     5   FT#05-Solar Radiation       WLA#05B.sol     6 10 FF time # 5 Solar
_Tot
   94.6000      1.00   94.6000      16.00   94.6000    350.00   94.6000     365.00
PHOTO     4     6   FT#06-Photoperiod           WLA#05B.sol     7 10 FF time # 6 Photo
Per
    0.5500      1.00    0.5500      16.00    0.5500    350.00    0.5500     365.00
 WIND     4     7   FT#07-Wind Velocity         WLA#05B.sol     9 10 FF time # 7 Wind
vel
    4.5000      1.00    4.5000      16.00    4.5000    350.00    4.5000     365.00
KE#01     4     8   FT#08-KE#01                 WLA#05B.EXT     4  8 FF time # 8 Ke #1

    1.0000      1.00    1.0000      16.00    1.0000    350.00    1.0000     365.00
KE#02     4     9   FT#09-KE#01                 WLA#05B.EXT     5  8 FF time # 9 Ke #2

    1.0000      1.00    1.0000      16.00    1.0000    350.00    1.0000     365.00
TFNH4     4    13   FT#13-NH4 Flux Th=1.079   WLA#05B.sed     5  9 FF time #13 NH4 f
lux
    1.4625      1.00    1.4625      16.00    1.4625    350.00    1.4625     365.00
TFPO4     4    14   FT#14-PO4 Flux Th=1.079   WLA#05B.sed     6  9 FF time #14 PO4 f
lux
    1.4625      1.00    1.4625      16.00    1.4625    350.00    1.4625     365.00
VEL#1     4    15   FT#15-VELN(1)               WLA#05B.VEL     4  7 FF time #15 VELN(
1)
    0.0000      1.00    0.0000      16.00    0.0000    350.00    0.0000     365.00
VEL#2     4    16   FT#16-VELN(2)               WLA#05B.VEL     5  7 FF time #16 VELN(
2)
    0.0000      1.00    0.0000      16.00    0.0000    350.00    0.0000     365.00
VEL#3     4    17   FT#17-VELN(3)               WLA#05B.VEL     6  7 FF time #17 VELN(
3)
    0.0000      1.00    0.0000      16.00    0.0000    350.00    0.0000     365.00
VEL#4     4    18   FT#18-VELN(4)               WLA#05B.VEL     7  7 FF time #18 VELN(
4)
    0.0000      1.00    0.0000      16.00    0.0000    350.00    0.0000     365.00
 SALT     4    20   FT#20-Salinity    ppt       WLA#05B.SAL     9 11 FF time #20 SALT

    0.0000      1.00    0.0000      16.00    0.0000    350.00    0.0000     365.00
 AIRT     4    21   FT#21-Air Temperature       WLA#05B.TEM     9 11 FF time #21 AIRT

   17.5000      1.00   17.5000      16.00   17.5000    350.00   17.5000     365.00
NH3_N        mg/L       Sys#01  GW05Amm3.DGJ      3 1.00 0.100E+08 04/13/95
BG001     0.2500     1.0000SG002     0.2500     1.0000SG003     0.2500     1.0000
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TABLE B-4. (CONTINUED)

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07/28/1995  18:41        Filename: WLA#05B.INP                           Page  9

SG004     0.2500     1.0000SG005     0.2500     1.0000SG006     0.2500     1.0000
SG007     0.2500     1.0000SG008     0.2500     1.0000SG009     0.2500     1.0000
SG010     0.2500     1.0000SG011     0.2500     1.0000SG012     0.2500     1.0000
SG013     0.2500     1.0000SG014     0.2500     1.0000SG015     0.2500     1.0000
SG016     0.2500     1.0000SG017     0.2500     1.0000SG018     0.2500     1.0000
SG019     0.2500     1.0000SG020     0.2500     1.0000SG021     0.2500     1.0000
SG022     0.2500     1.0000SG023     0.2500     1.0000SG024     0.2500     1.0000
SG025     0.2500     1.0000SG026     0.2500     1.0000SG027     0.2500     1.0000
SG028     0.2500     1.0000SG029     0.2500     1.0000SG030     0.2500     1.0000
SG031     0.2500     1.0000SG032     0.2500     1.0000SG033     0.2500     1.0000
SG034     0.2500     1.0000SG035     0.2500     1.0000SG036     0.2500     1.0000
SG037     0.2500     1.0000SG038     0.2500     1.0000SG039     0.2500     1.0000
SG040     0.2500     1.0000SG041     0.2500     1.0000SG042     0.2500     1.0000
SG043     0.2500     1.0000SG044     0.2500     1.0000SG045     0.2500     1.0000
SG046     0.0000     1.0000

NO3+NO2_N mg/L     Sys#02  G#05ANO3.DGJ     5 2.69 0.100E+08 04/13/95
SG001     0.5000     1.0000SG002     0.5000     1.0000SG003     0.5000     1.0000
SG004     0.5000     1.0000SG005     0.5000     1.0000SG006     0.5000     1.0000
SG007     0.5000     1.0000SG008     0.5000     1.0000SG009     0.5000     1.0000
SG010     0.5000     1.0000SG011     0.5000     1.0000SG012     0.5000     1.0000
SG013     0.5000     1.0000SG014     0.5000     1.0000SG015     0.5000     1.0000
SG016     0.5000     1.0000SG017     0.5000     1.0000SG018     0.5000     1.0000
SG019     0.5000     1.0000SG020     0.5000     1.0000SG021     0.5000     1.0000
SG022     0.5000     1.0000SG023     0.5000     1.0000SG024     0.5000     1.0000
SG025     0.5000     1.0000SG026     0.5000     1.0000SG027     0.5000     1.0000
SG028     0.5000     1.0000SG029     0.5000     1.0000SG030     0.5000     1.0000
SG031     0.5000     1.0000SG032     0.5000     1.0000SG033     0.5000     1.0000
SG034     0.5000     1.0000SG035     0.5000     1.0000SG036     0.5000     1.0000
SG037     0.5000     1.0000SG038     0.5000     1.0000SG039     0.5000     1.0000
SG040     0.5000     1.0000SG041     0.5000     1.0000SG042     0.5000     1.0000
SG043     0.5000     1.0000SG044     0.5000     1.0000SG045     0.5000     1.0000
SG046     0.0000     1.0000

PO4 Ortho mg/L     Sys#03  G#05APO4.DGJ     5 2.69 0.100E+08 04/13/95
SG001     0.1000     0.8000SG002     0.1000     0.8000SG003     0.1000     0.8000
SG004     0.1000     0.8000SG005     0.1000     0.8000SG006     0.1000     0.8000
SG007     0.1000     0.8000SG008     0.1000     0.8000SG009     0.1000     0.8000
SG010     0.1000     0.8000SG011     0.1000     0.8000SG012     0.1000     0.8000
SG013     0.1000     0.8000SG014     0.1000     0.8000SG015     0.1000     0.8000
SG016     0.1000     0.8000SG017     0.1000     0.8000SG018     0.1000     0.8000
SG019     0.1000     0.8000SG020     0.1000     0.8000SG021     0.1000     0.8000
SG022     0.1000     0.8000SG023     0.1000     0.8000SG024     0.1000     0.8000
SG025     0.1000     0.8000SG026     0.1000     0.8000SG027     0.1000     0.8000
SG028     0.1000     0.8000SG029     0.1000     0.8000SG030     0.1000     0.8000
SG031     0.1000     0.8000SG032     0.1000     0.8000SG033     0.1000     0.8000
SG034     0.1000     0.8000SG035     0.1000     0.8000SG036     0.1000     0.8000
SG037     0.1000     0.8000SG038     0.1000     0.8000SG039     0.1000     0.8000
SG040     0.1000     0.8000SG041     0.1000     0.8000SG042     0.1000     0.8000
SG043     0.1000     0.8000SG044     0.1000     0.8000SG045     0.1000     0.8000
SG046     0.0000     0.8000

PHYT#1      ug chl/L  Sys#04  G#05ACHL.DGJ     4 2.69 0.100E+08 04/13/95
SG001     2.0000     0.0000SG002     2.0000     0.0000SG003     2.0000     0.0000
SG004     2.0000     0.0000SG005     2.0000     0.0000SG006     2.0000     0.0000
SG007     2.0000     0.0000SG008     2.0000     0.0000SG009     2.0000     0.0000
SG010     2.0000     0.0000SG011     2.0000     0.0000SG012     2.0000     0.0000
SG013     2.0000     0.0000SG014     2.0000     0.0000SG015     2.0000     0.0000
SG016     2.0000     0.0000SG017     2.0000     0.0000SG018     2.0000     0.0000
SG019     2.0000     0.0000SG020     2.0000     0.0000SG021     2.0000     0.0000
SG022     2.0000     0.0000SG023     2.0000     0.0000SG024     2.0000     0.0000
SG025     2.0000     0.0000SG026     2.0000     0.0000SG027     2.0000     0.0000
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07/28/1995  18:41        Filename: WLA#05B.INP                           Page 10

SG028     2.0000     0.0000SG029     2.0000     0.0000SG030     2.0000     0.0000
SG031     2.0000     0.0000SG032     2.0000     0.0000SG033     2.0000     0.0000
SG034     2.0000     0.0000SG035     2.0000     0.0000SG036     2.0000     0.0000
SG037     2.0000     0.0000SG038     2.0000     0.0000SG039     2.0000     0.0000
SG040     2.0000     0.0000SG041     2.0000     0.0000SG042     2.0000     0.0000
SG043     2.0000     0.0000SG044     2.0000     0.0000SG045     2.0000     0.0000
SG046     0.0000     0.0000

CBODU       mg/L      Sys#05  G#05ABOD.DGJ     3 2.69 0.100E+08 04/13/95
SG001     2.0000     0.8000SG002     2.0000     0.8000SG003     2.0000     0.8000
SG004     2.0000     0.8000SG005     2.0000     0.8000SG006     2.0000     0.8000
SG007     2.0000     0.8000SG008     2.0000     0.8000SG009     2.0000     0.8000
SG010     2.0000     0.8000SG011     2.0000     0.8000SG012     2.0000     0.8000
SG013     2.0000     0.8000SG014     2.0000     0.8000SG015     2.0000     0.8000
SG016     2.0000     0.8000SG017     2.0000     0.8000SG018     2.0000     0.8000
SG019     2.0000     0.8000SG020     2.0000     0.8000SG021     2.0000     0.8000
SG022     2.0000     0.8000SG023     2.0000     0.8000SG024     2.0000     0.8000
SG025     2.0000     0.8000SG026     2.0000     0.8000SG027     2.0000     0.8000
SG028     2.0000     0.8000SG029     2.0000     0.8000SG030     2.0000     0.8000
SG031     2.0000     0.8000SG032     2.0000     0.8000SG033     2.0000     0.8000
SG034     2.0000     0.8000SG035     2.0000     0.8000SG036     2.0000     0.8000
SG037     2.0000     0.8000SG038     2.0000     0.8000SG039     2.0000     0.8000
SG040     2.0000     0.8000SG041     2.0000     0.8000SG042     2.0000     0.8000
SG043     2.0000     0.8000SG044     2.0000     0.8000SG045     2.0000     0.8000
SG046     0.0000     0.8000

OXYGEN      mg/L      Sys#06  G#05AOXY.DGJ     5 1.00 0.100E+08 04/13/95
SG001     8.0000     1.0000SG002     8.0000     1.0000SG003     8.0000     1.0000
SG004     8.0000     1.0000SG005     8.0000     1.0000SG006     8.0000     1.0000
SG007     8.0000     1.0000SG008     8.0000     1.0000SG009     8.0000     1.0000
SG010     8.0000     1.0000SG011     8.0000     1.0000SG012     8.0000     1.0000
SG013     8.0000     1.0000SG014     8.0000     1.0000SG015     8.0000     1.0000
SG016     8.0000     1.0000SG017     8.0000     1.0000SG018     8.0000     1.0000
SG019     8.0000     1.0000SG020     8.0000     1.0000SG021     8.0000     1.0000
SG022     8.0000     1.0000SG023     8.0000     1.0000SG024     8.0000     1.0000
SG025     8.0000     1.0000SG026     8.0000     1.0000SG027     8.0000     1.0000
SG028     8.0000     1.0000SG029     8.0000     1.0000SG030     8.0000     1.0000
SG031     8.0000     1.0000SG032     8.0000     1.0000SG033     8.0000     1.0000
SG034     8.0000     1.0000SG035     8.0000     1.0000SG036     8.0000     1.0000
SG037     8.0000     1.0000SG038     8.0000     1.0000SG039     8.0000     1.0000
SG040     8.0000     1.0000SG041     8.0000     1.0000SG042     8.0000     1.0000
SG043     8.0000     1.0000SG044     8.0000     1.0000SG045     8.0000     1.0000
SG046     0.0000     1.0000

TON_N       mg/L      Sys#07  G#05ATON.DGJ     5 2.69 0.100E+08 04/13/95
SG001     0.2500     0.8000SG002     0.2500     0.8000SG003     0.2500     0.8000
SG004     0.2500     0.8000SG005     0.2500     0.8000SG006     0.2500     0.8000
SG007     0.2500     0.8000SG008     0.2500     0.8000SG009     0.2500     0.8000
SG010     0.2500     0.8000SG011     0.2500     0.8000SG012     0.2500     0.8000
SG013     0.2500     0.8000SG014     0.2500     0.8000SG015     0.2500     0.8000
SG016     0.2500     0.8000SG017     0.2500     0.8000SG018     0.2500     0.8000
SG019     0.2500     0.8000SG020     0.2500     0.8000SG021     0.2500     0.8000
SG022     0.2500     0.8000SG023     0.2500     0.8000SG024     0.2500     0.8000
SG025     0.2500     0.8000SG026     0.2500     0.8000SG027     0.2500     0.8000
SG028     0.2500     0.8000SG029     0.2500     0.8000SG030     0.2500     0.8000
SG031     0.2500     0.8000SG032     0.2500     0.8000SG033     0.2500     0.8000
SG034     0.2500     0.8000SG035     0.2500     0.8000SG036     0.2500     0.8000
SG037     0.2500     0.8000SG038     0.2500     0.8000SG039     0.2500     0.8000
SG040     0.2500     0.8000SG041     0.2500     0.8000SG042     0.2500     0.8000
SG043     0.2500     0.8000SG044     0.2500     0.8000SG045     0.2500     0.8000
SG046     0.0000     0.8000
```

TABLE B-4. (CONTINUED)

```
07/28/1995  18:41        Filename: WLAP058.INP                        Page  11

INP F     mg/l      SysmR08  GAP05A10P.DAJ    3 2.49 0.100E+08 06/13/95
SG001     0.0250    0.8000SG002    0.0250     0.8000SG003    0.0250    0.8000
SG004     0.0250    0.8000SG005    0.0250     0.8000SG006    0.0250    0.8000
SG007     0.0250    0.8000SG008    0.0250     0.8000SG009    0.0250    0.8000
SG010     0.0250    0.8000SG011    0.0250     0.8000SG012    0.0250    0.8000
SG013     0.0250    0.8000SG014    0.0250     0.8000SG015    0.0250    0.8000
SG016     0.0250    0.8000SG017    0.0250     0.8000SG018    0.0250    0.8000
SG019     0.0250    0.8000SG020    0.0250     0.8000SG021    0.0250    0.8000
SG022     0.0250    0.8000SG023    0.0250     0.8000SG024    0.0250    0.8000
```

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07/28/1995  18:41        Filename: WLAP058.INP                        Page  12

SG025     0.0250    0.8000SG026    0.0250     0.8000SG027    0.0250    0.8000
SG028     0.0250    0.8000SG029    0.0250     0.8000SG030    0.0250    0.8000
SG031     0.0250    0.8000SG032    0.0250     0.8000SG033    0.0250    0.8000
SG034     0.0250    0.8000SG035    0.0250     0.8000SG036    0.0250    0.8000
SG037     0.0250    0.8000SG038    0.0250     0.8000SG039    0.0250    0.8000
SG040     0.0250    0.8000SG041    0.0250     0.8000SG042    0.0250    0.8000
SG043     0.0250    0.8000SG044    0.0250     0.8000SG045    0.0250    0.8000
```

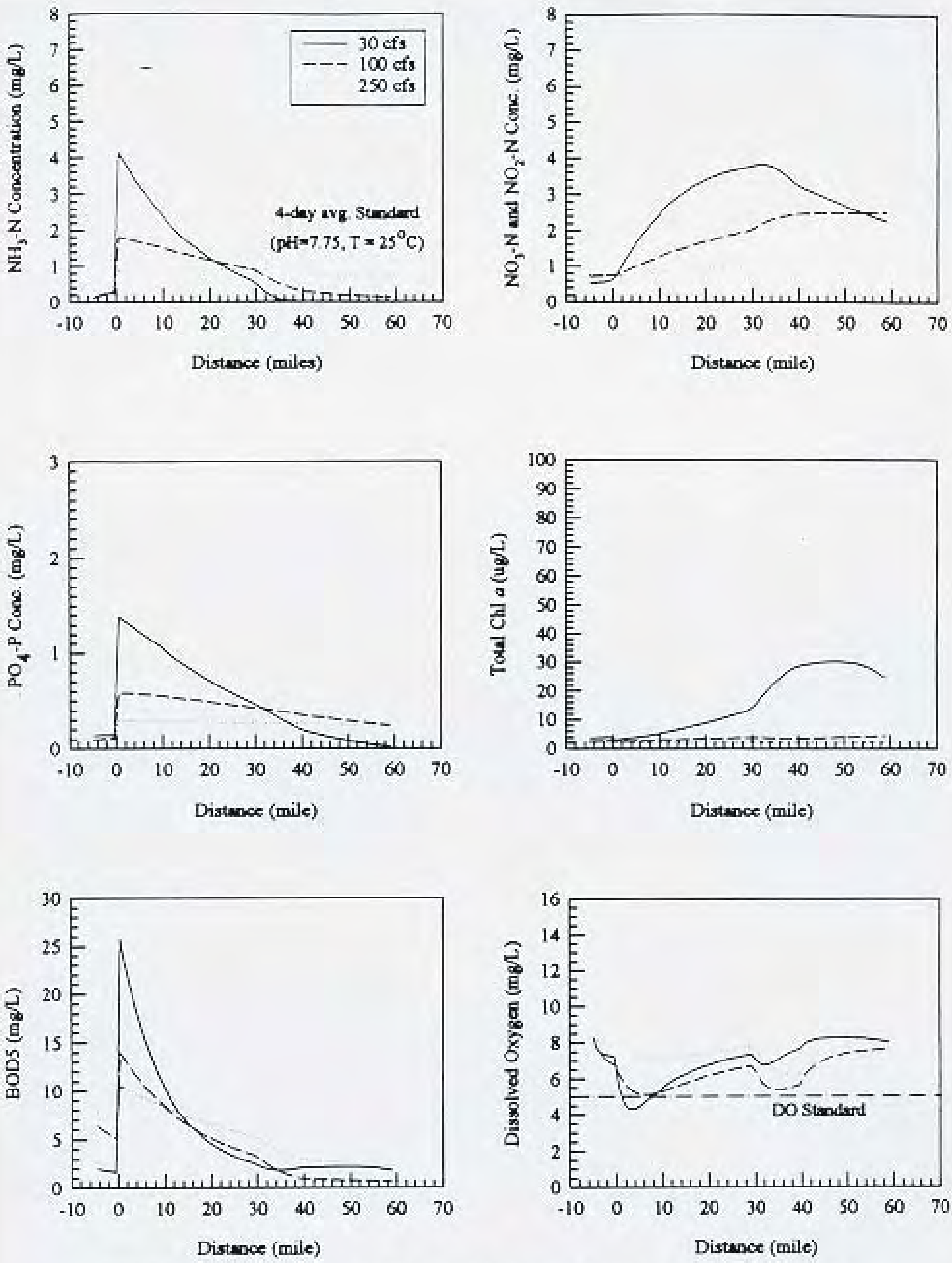

FIGURE B-9. WASP5 PREDICTION OF INORGANIC NITROGEN, PHOSPHORUS, CHLOROPHYLL *A*, BOD AND DISSOLVED OXYGEN FOR DIFFERENT FLOW CONDITIONS UNDER EXISTING LOADS.

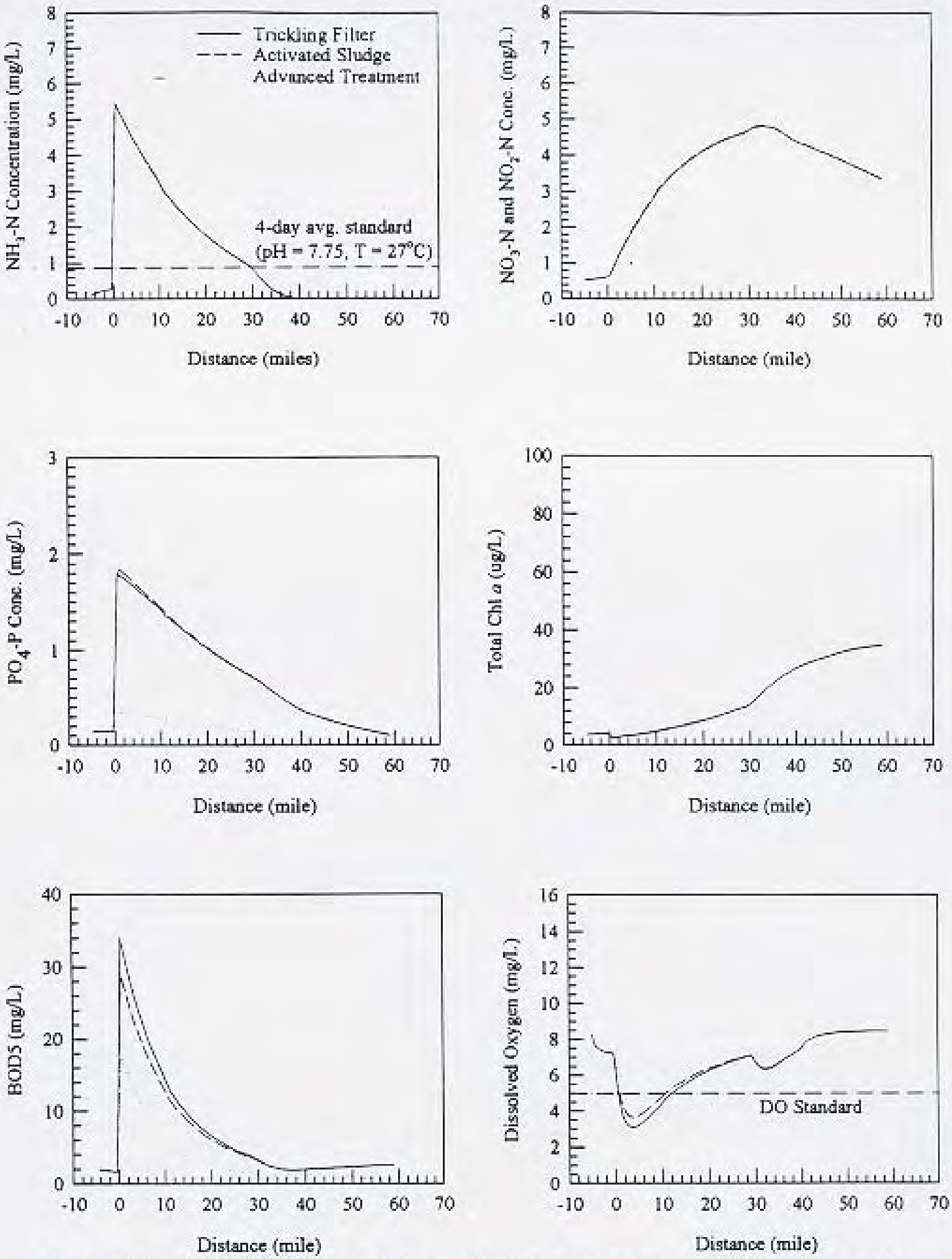

FIGURE B-10. PROJECTED INORGANIC NITROGEN, PHOSPHORUS, CHLOROPHYLL *A*, BOD_5, AND DO UNDER DIFFERENT FUTURE LOADINGS UNDER LOW FLOW CONDITIONS.

concentration at approximately 40 miles downstream for existing 30 cfs flow conditions.

For the low flow condition, the WASP5 run results in approximately the same oxygen and ammonia concentrations as that of the analytical example under both existing and future conditions. Oxygen concentrations drop to approximately 4 mg/L and ammonia concentrations reach approximately 4.2 mg/L, resulting in predicted violations of both oxygen and ammonia standards. The BOD5 plot shown in Figure B-10 includes the demand exerted by decaying algae which was not considered in the analytical solution approach. In the WASP5 model, benthic denitrification is not accounted for since it is assumed to be negligible.

B.2.6 Conclusions

For this example, the results showed that low flow conditions (7Q10) represent a critical condition for maintenance of DO and NH3 standards. Critical conditions occurred primarily under low flow (30 cfs) when the system was dominated by point source loads. Recommended management for implementation of a TMDL is to pursue a load reduction from point sources. As the flow decreases, an increasing percentage of the CBOD and nutrient load can be attributed to the sewage treatment plant. Low flow conditions are also critical for augmented algal growth. For other types of pollutants or other site specific conditions, analyses may result in different conclusions. In some cases a continuous simulation of storm inputs and receiving water response may be required to determine the frequency and duration of stream impacts. As shown here, steady state examination of several flow conditions using a analytical solution or WASP5 can assist in screening for the range of flow conditions where problems may occur.

For development of an actual TMDL, some additional investigations are recommended. Calibration and validation of a nonpoint source loading model and river response model may need to be conducted and additional data collected if possible. Future development that may cause an additional expansion of the treatment plant should be considered. Any model uncertainty and future conditions should be built into a margin of safety for the TMDL. A final TMDL should not be assigned until all of these factors are considered carefully.

B.3 WILLAMETTE RIVER EXAMPLE QUAL2E MODEL

B.3.1 Introduction

The Willamette River basin modeling study was selected to show a recent example of a QUAL2E application. This example highlights the use of the QUAL2E model in assessing DO (DO), nutrients, and phytoplankton biomass for a large river system in Oregon. For additional discussion on the use of QUAL2E in the development of TMDLs, see Sections 2.3 and 3.4. The use of QUAL2E for uncertainty analysis is shown in Appendix D.

The Willamette River Basin Water Quality Study (WRBWQS) is an ongoing interdisciplinary study that includes investigations of river hydrology, sediment transport, toxic organic compounds and trace elements, point and nonpoint pollution sources, and aquatic ecosystems. The development of predictive computer models under low flow conditions of late summer was one goal of the Phase I study. The low-flow period coincides with the critical period for DO and is suitable for steady-state DO modeling. A model was needed to more fully assess the interactions among nutrients, phytoplankton, and DO, which had not previously been undertaken. A summary of the selection, calibration, and preliminary evaluation of the Phase I DO model is presented below. A water quality management case scenario involving the effect of variation of the river flow regime on DO and phytoplankton biomass is also presented. A more complete description of the development and evaluation of the WRBWQS Phase I DO model has been provided in a number of technical reports (Tetra Tech, 1993a, b, c).

B.3.2 Problem Setting

The Willamette River drainage basin area is approximately 30,000 km2 in area and is bounded by the Coast and Cascade mountain ranges (Figure B-11). The mainstem of the river meanders in a northerly direction through an alluvial valley approximately 300 km to the Columbia River. The problem setting and essential characteristics of the river system are summarized in Figure B-12.

Although the drainage basin contains the majority of the State’s inhabitants, approximately half of the basin is forested. However, significant changes have occurred in the drainage basin since the arrival of European immigrants beginning in the early 1890s

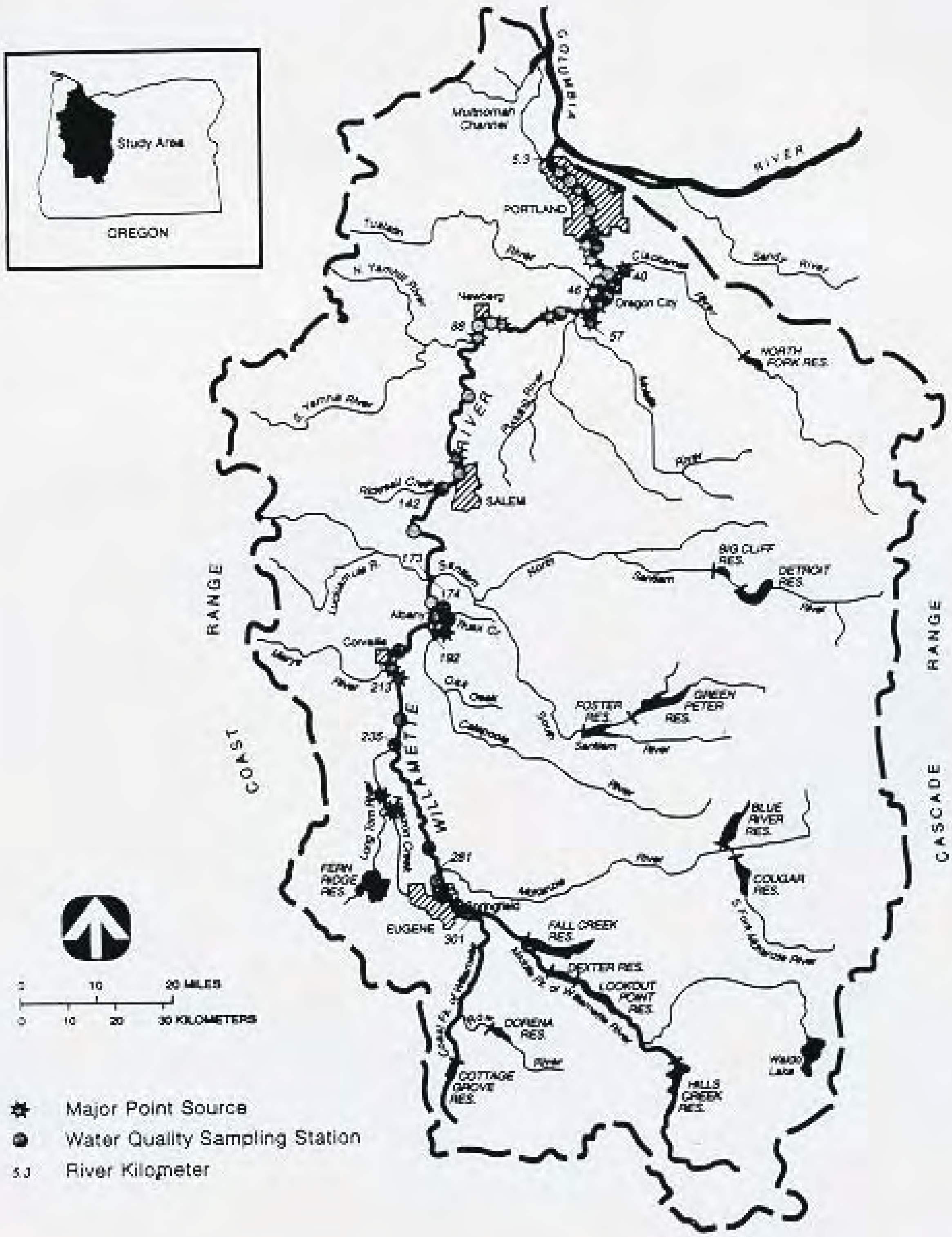

FIGURE B-11. THE WILLAMETTE RIVER BASIN STUDY AREA, LOCATIONS OF MAJOR POINT SOURCE DISCHARGES ALONG THE MAINSTEM OF THE RIVER, AND LOCATIONS OF THE AUGUST 1992 SYNOPTIC SURVEY STATIONS.

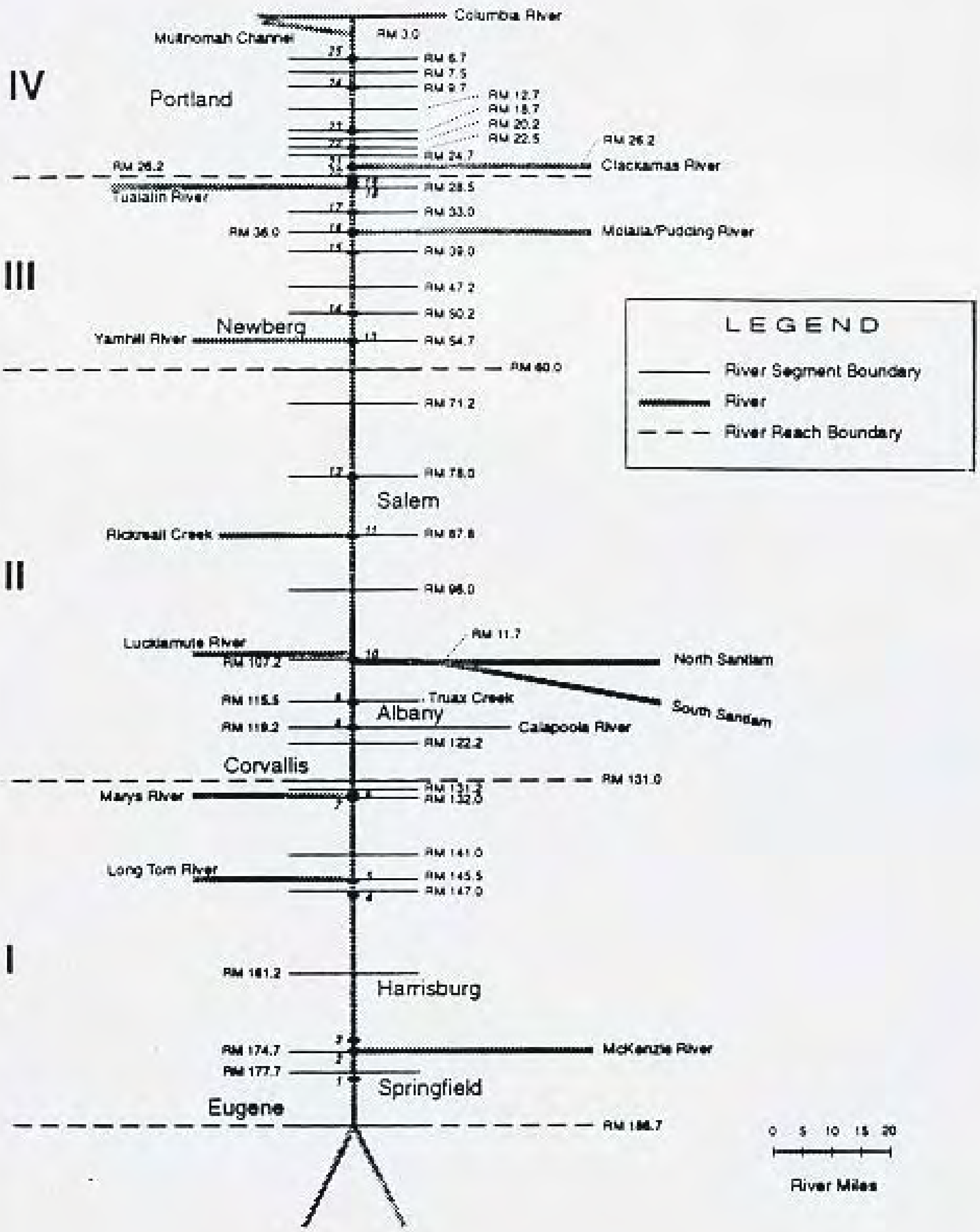

FIGURE B-12. RIVER SEGMENTATION, POINT SOURCE AND TRIBUTARY LOCATION SCHEMATIC

Gleeson, 1972; Sedell and Frogatt, 1984). Approximately one-third of the basin is currently used for agriculture, and the forests have been exploited for timber production. About 10 percent of the basin has been urbanized or is in residential use. The river receives direct inputs of treated municipal wastes and industrial effluents—primarily from pulp and paper processing facilities. Although nonpoint source inputs are significant during winter rainfall runoff, this source is considered minor relative to point sources during the dry critical period for DO.

Oregon water quality standards relevant to the modeling study include the state standards for DO and an “action level” for chlorophyll a (Oregon Administrative Rules, Chapter 340, Division 41). The state DO standards vary for each reach of the river:

- The Tidal Reach [river kilometer (RK) 0-43] - 5 mg/L
- Newberg Pool (RK 43-80) - 6 mg/L
- Newberg Pool to Salem (RK 80-137) - 7 mg/L

Above Salem, the DO standard is a minimum of 90 percent of the saturation concentration for DO.

The state action level for chlorophyll a is 15 g/L. This action level applies to natural lakes that do not thermally stratify, and to reservoirs, rivers, and estuaries. The action level is intended to identify water bodies where phytoplankton might impair beneficial uses. If it is determined that the action level is exceeded in a particular water body, additional studies might be conducted to determine the causes of the exceedances and impacts on beneficial uses. Control strategies, including additional standards or pollutant load limitations, could then be developed.

The need for predictive water quality models was underscored by the continued industrial and agricultural development and population growth within the basin. As of 1990, the population within the nine counties that cover the basin had almost reached 2 million. Highest population growth rates have occurred in the counties that encompass the large urban centers of Eugene, Salem, and Portland. These urban centers and other smaller towns and industrial facilities are found along the banks of the mainstem of the river.

Currently there are 21 major dischargers to the mainstem of the Willamette River. Twelve additional facilities discharge to tributaries of the Willamette. Discharger name, type, receiving water, and river mile location are summarized in Table B-5. Pollutant loading information for the various facilities is shown in Table B-6.

B.3.3 River Characteristics

At its mouth, the Willamette is the 10th largest river in the continental United States, in terms of total discharge (Sedell and Frogatt, 1984), and the discharge per unit area is the highest of the large rivers in the Nation due to the heavy winter rainfall at lower elevations in the basin during the winter months (Rickert and Hines, 1978). At higher elevations the winter precipitation occurs as snow, which contributes to extended high flow as spring snowmelt runoff. The climate is temperate and characterized by wet, mild winters and dry, moderately warm summers. Most of the rainfall occurs in the fall, winter, and spring, with little rainfall during June, July, and August. The period of low river flow during the late summer coincides with the period of low rainfall and highest air temperatures.

River discharge is managed for flood control, irrigation, and navigation purposes by impoundments located on a number of the large tributaries. Nonetheless, river discharge varies seasonally, with greatest runoff occurring during the winter months (December-February) (ca. 1,800 m3/sec) and lowest flows occurring in summer (July-September) (ca. 283 m3/sec), with a mean annual flow of approximately 943 m3/sec (Moffatt et al., 1990). Low summer flows are augmented by controlled releases from tributary impoundments to provide for commercial navigation. A natural flow control occurs at RK 42—Willamette Falls—although a lock, powerhouse, and fish ladder have also been constructed at this location. Below the falls, the river is tidally influenced via the confluence with the Columbia River which flows to the Pacific Ocean approximately 160 km to the west. Due to the great distance to the ocean, flow reversals in this 42-km reach cause intrusion of only fresh Columbia River water into the Willamette. Flow in the reach below the falls is further complicated by the presence of two channels—the main channel, which enters the Columbia River at RK 162, and the Multnomah Channel, which passes between Willamette RK 5 and Columbia RK 140.

Based on hydraulic and physical characteristics, the mainstem of the river may be divided into three distinct reaches (Rickert et al., 1976). The upstream reach is 217 km long and is characterized by fast-moving currents flowing over a shallow, meandering

TABLE B-5. MAJOR NPDES PERMITTEES IN THE WILLAMETTE RIVER BASIN

RIVER REACH	PERMETTEE	DISCHARGE TYPE	RECEIVING WATER	RIVER MILE
I	Metropolitan Wastewater Management Commission	Municipal	Willamette River	178.0
I	James River Paper Company, Inc.	Industrial	Willamette River	147.4
I	Pope & Talbot, Inc.	Industrial	Willamette River	147.4
I	Evanite Fiber Corporation	Industrial	Willamette River	132.2
I	Cottage Grove, City of	Municipal	Coast Fork of Willamette River	21.5
I	Weyerhaeuser Company	Industrial	McKenzie River	14.7
II	Corvallis, City of	Municipal	Willamette River	131.0
II	Albany, City of	Municipal	Willamette River	119.0
II	Willamette Industries, Inc.	Industrial	Willamette River	116.5
II	Salem, City of	Municipal	Willamette River	78.2
II	Oregon Metallurgical Corporation	Industrial	Oak Creek	2.0
II	Teledyne Industries, Inc.	Industrial	Truax Creek	0.4
II	Sweet Home, City of	Municipal	South Fork of Santiam River	33.6
II	Lebanon, City of	Municipal	South Fork of Santiam River	17.4
II	Dallas, City of	Municipal	Rickreal Creek	8.5
III	Newberg, City of	Municipal	Willamette River	50.3
III	Smurfit Newsprint Corporation - Newberg	Industrial	Willamette River	50.0
III	Wilsonville, City of	Municipal	Willamette River	39.0
III	Canby, City of	Municipal	Willamette River	
III	Smurfit Newsprint Corporation - Oregon City	Industrial	Willamette River	27.5
III	McMinnville, City of	Municipal	Yamhill River	4.0
III	Woodburn, City of	Municipal	Pudding River	21.5
III	Unified Sewerage Agency of Washington County, Forest Grove	Municipal	Tualatin River	56.7
III	Unified Sewerage Agency of Washington County, Hillsboro	Municipal	Tualatin Rive	44.0
III	Unified Sewerage Agency of Washington County, Rock Creek	Municipal	Tualatin River	38.0
III	Unified Sewerage Agency of Washington County, Durham	Municipal	Tualatin River	9.6
IV	Simpson Paper Company	Industrial	Willamette River	26.4
IV	Tri-City Service District	Municipal	Willamette River	25.2
IV	Portland, City of	Municipal	Willamette River	20.3
IV	Oak Lodge Sanitary District	Municipal	Willamette River	20.1
IV	Clackamas County Service District # 1	Municipal	Willamette River	18.5
IV	Elf Atochem North America, Inc.	Industrial	Willamette River	7.4
IV	Rhone-Poulenc AG Company	Industrial	Willamette River	7.0

TABLE B-6. AUGUST 1992 POINT SOURCE AND TRIBUTARY DATA FOR THE WILLAMETTE RIVER QUAL2E MODEL

Point Source Name	Flow (cfs)	DO (mg/L)	CBOD$_u$ (mg/L)	Chl a (ug/L)	Org. N (mg/L)	NH$_3$-N (mg/L)	NO$_2$-N (mg/L)	NO$_3$-N (mg/L)	Org. P (mg/L)	Ortho P (mg/L)
Metropolitan Wastewater Management Commission	35.35	5.9	14	0	3.7	9.97	0.06	1.41	0.2	0.31
James River Paper Company, Inc.	16.87	10.5	2.7	2.3	0.18	0.02	0.006	0.024	0.02	0.02
Pope & Talbot, Inc.	19.49	5	44	0	2.8	3.65	0.03	0.5	0.18	1.79
Evanite Fiber Corporation	6.1	5	8	0	0.23	0.3	0.23	5	0.26	1.95
Cottage Grove, City of	28	19.4	3.2	1.8	0.16	0.04	0.006	0.534	0.054	0.016
Weyerhaeuser Company	18	9.4	2.1	1.7	0.18	0.02	0.006	0.154	0.025	0.025
Corvallis, City of	1.35	5	254.2	0	16.2	21	0.31	2.67	1.84	2.76
Albany, City of	7.91	4.5	20	0	1.6	5.9	0.39	9.27	3.42	5.35
Willamette Industries, Inc.	8.6	7.3	6.6	0	1.36	3.49	0.24	5.45	0.00	3.20
Salem, City of	22	7.7	4.2	1.3	0.16	0.04	0.006	0.594	0.03	0.02
Oregon Metallurgical Corporation	12.22	5	66.4	0	0.6	0.82	0.13	2.67	0.2	0.3
Teledyne Industries, Inc.	2.43	5	4.2	0	2.0	10.79	1.36	32.64	0.9	3.2
Sweet Home, City of	14.30	10.9	3.4	1.5	0.18	0.02	0.006	0.104	0.008	0.008
Lebanon, City of	15	10	3.4	1	0.2	0.02	0.006	0.1	0.008	0.008
Dallas, City of	6	8.6	3.2	2	0.37	0.03	0.006	0.394	0.29	0.150
Newberg, City of	43.93	6	9.5	0	3.5	9.5	0.18	0.68	1.52	1.18
Smurfit Newsprint Corporation - Newberg	65	8.9	4.4	9.2	0.330	0.025	0.026	0.704	0.048	0.093
Wilsonville, City of	1.98	6	11	0	3.2	3	0.2	6.1	0.9	3.2
Canby, City of	20.6	5	127.1	0	7.6	2.6	0.001	0.02	8.8	13.2
Smurfit Newsprint Corporation - Oregon City	1.27	6	22.5	0	0.71	1.91	0.2	6.1	0.9	3.2
McMinnville, City of	15	9.6	2.1	1.1	0.26	0.04	0.006	0.274	0.014	0.006
Woodburn, City of	72	8	2.1	0.7	0.5	0.075	0.006	2.49	0.06	0.48
Unified Sewerage Agency of Washington County, Forest Grove	1.35	6	29.3	0	3.2	3	0.2	6.1	0.9	3.2
Unified Sewerage Agency of Washington County, Hillsboro	103	5	9.5	11.3	0.23	0.17	0.08	2.72	0.07	0.06
Unified Sewerage Agency of Washington County, Rock Creek	16.43	1.9	184.5	0	1.52	1.98	0.13	2.67	0.2	1.9
Unified Sewerage Agency of Washington County, Durham	11.6	5	56.3	0	1.52	1.98	0.13	2.67	0.2	1.9
Simpson Paper Company	7.38	6	18	0	12.2	0.51	0.02	4.92	1	1.5
Tri-City Service District	591	10.5	3.8	1.9	0.18	0.02	0.002	0.014	0.02	0.01
Portland, City of	9.8	6	26.3	0	3.2	3	0.2	6.1	0.9	3.2
Oak Lodge Sanitary District	4.16	6	39.2	0	3.2	3	0.2	6.1	0.9	3.2
Clackamas County Service District # 1	10.23	6	13.9	0	3.41	9.27	0.12	2.88	0.9	3.2
Elf Atochem North America, Inc.	39.82	5	0.0	0	3.2	3	0.2	6.1	0.9	3.2
Rhone-Poulenc AG Company	0.005	5	29	0	3.2	3	0.2	6.1	0.9	3.2

riverbed composed of cobbles and gravel. The middle reach (Newberg Pool) is a 54-km-long, deep and slow-moving portion of the river formed by the natural impoundment behind Willamette Falls. The tidally influenced reach below the falls (Tidal Reach) is also relatively deep and has the longest estimated travel time—241 hours during critical flow conditions (Rickert et al., 1975).

B.3.4 Model Application

The objective of the WRBWQS was to develop and calibrate a predictive DO model for the Willamette River to evaluate river basin management alternatives and meet regulatory mandates. To first identify appropriate predictive models, several DO models of varying complexity were identified and evaluated using a number of selection criteria: 1) *Dimensionality*—a one-dimensional model was considered adequate; 2) *Temporal characteristics*—a steady-state model was considered appropriate for the summer low-flow period of interest; 3) *Consideration of relevant processes*—these processes included the capability to model phytoplankton growth and nutrient interactions; 4) *Suitability for a range of applications*—temperature or bacteria modeling is an example; *5) Data requirements*—the data required for model calibration had to be within the resources of the study; and *6) Ease of use*—the selected model needed to be sufficiently easy to use so water quality managers could practically apply the model as a decision-making tool.

Based on these selection criteria, the model QUAL2E (Version 3.14) was selected. This one-dimensional, steady-state model incorporates all of the relevant processes and has a menu-driven input and output system that facilitates use of the model. Additionally, the model includes applications for component, sensitivity, first-order error analysis, and Monte Carlo simulations.

B.3.4.1 Database Development and Model Calibration

Historical water quality data and previous DO modeling efforts were reviewed to identify relevant data and modeling approaches that could be incorporated into the QUAL2E model calibration effort (Tetra Tech, 1992a, b). The historical data review also identified data gaps to support the design of a synoptic field sampling effort to provide a data set for calibration of the model.

The field sampling effort was conducted in August 1992 and included diurnal DO and temperature measurements at 15 stations and measurements of nutrients and CBOD at 24 stations along the mainstem (Tetra Tech, 1992c). ODEQ collected single grab samples from 10 locations, and Tetra Tech collected samples at approximately 6-hr intervals over a 24-hr period at 15 stations. Data were also collected by the USGS at RK 20.6 as part of its National Stream Quality Assessment Network (NASQUAN) on August 17, 1992. USGS data were also incorporated into the model calibration effort. Point source loading data were compiled for the 21 major municipal and industrial effluent discharges to the mainstem of the Willamette River using the permit-required monitoring reports submitted to ODEQ and additional data collected during the synoptic field study (Tetra Tech, 1992c, d) (Table B-4).

The QUAL2E model was first discretized based on river hydraulic information provided by USGS (M. Fretwell, May 20, 1992, personal communication). The river was divided into 35 model segments with similar physical characteristics, resulting in a model consisting of 35 reaches divided into elements 1.2 km long for a total of 249 model elements. The tributaries and major municipal and industrial point sources were modeled as point sources inputs to the mainstem of the river.

Since river depths, velocities, and cross-sectional areas in each model segment vary under different hydrologic conditions, discharge coefficients and exponents were estimated for the model calculation of these variables as a function of discharge. (Note: It was assumed that the effective channel width would not change under low flow conditions.) To estimate the coefficients and exponents for each model segment velocity- and depth-discharge relationship, low- and high-flow stream channel hydraulic information provided by USGS was used (M. Fretwell, May 20, 1992, personal communication). The resulting QUAL2E model output for discharge, velocity, depth, and cross-sectional area are compared to the channel hydraulics data provided by USGS for the 2-yr, 60-day recurrence interval flow in Figure B-13. The model-predicted discharge, velocity, depth, and cross-sectional area are also shown in Figure B-14 for the August 1992 sampling period. A sample of the model input file is shown in Table B-7.

The model was then calibrated to the 1992 synoptic water quality survey data using a combination of visual best-fit and error minimization techniques. A preliminary calibration was conducted first using best

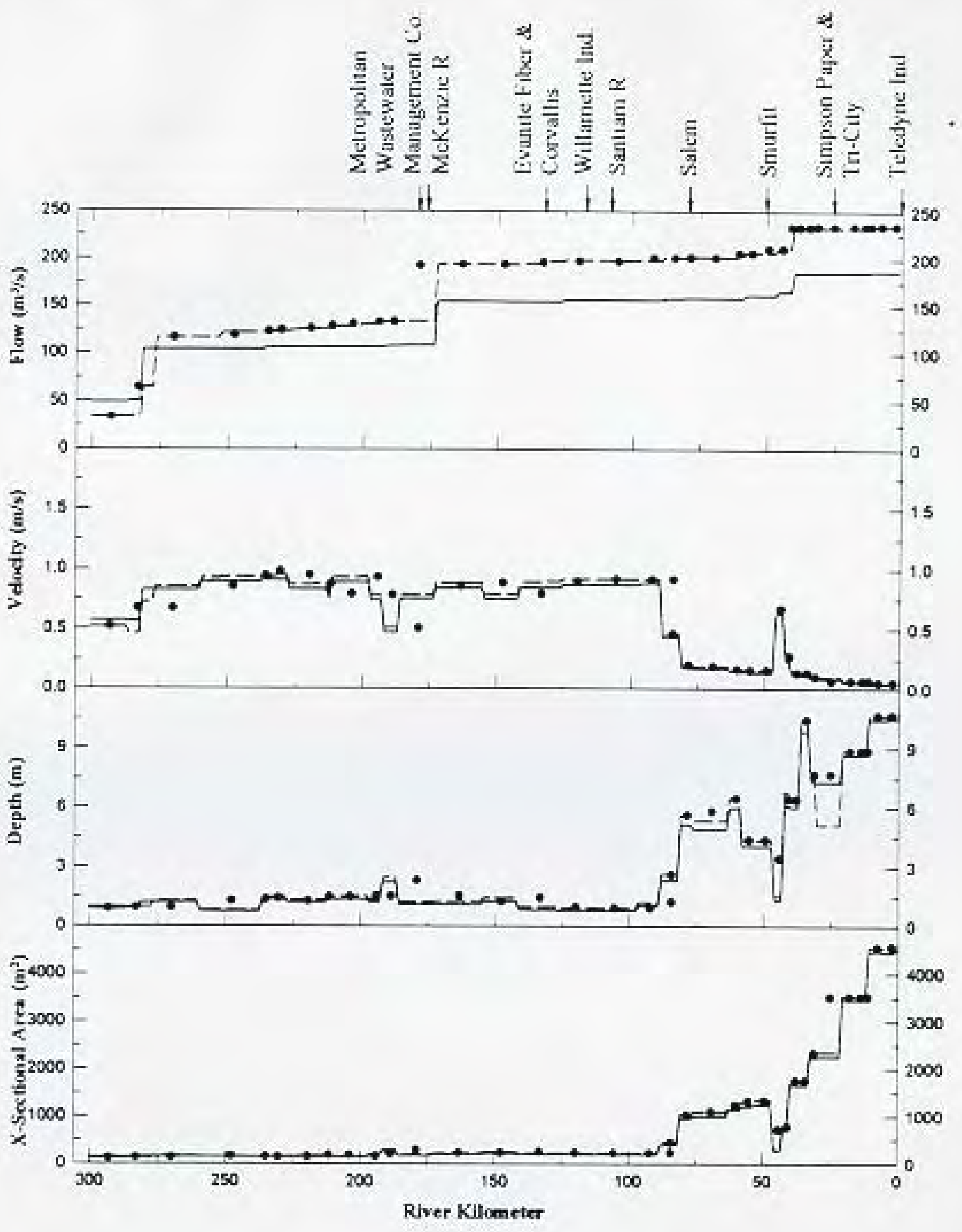

FIGURE B-13. WILLAMETTE RIVER FLOW, VELOCITY, DEPTH, AND CROSS-SECTION AREA RELATIONSHIPS

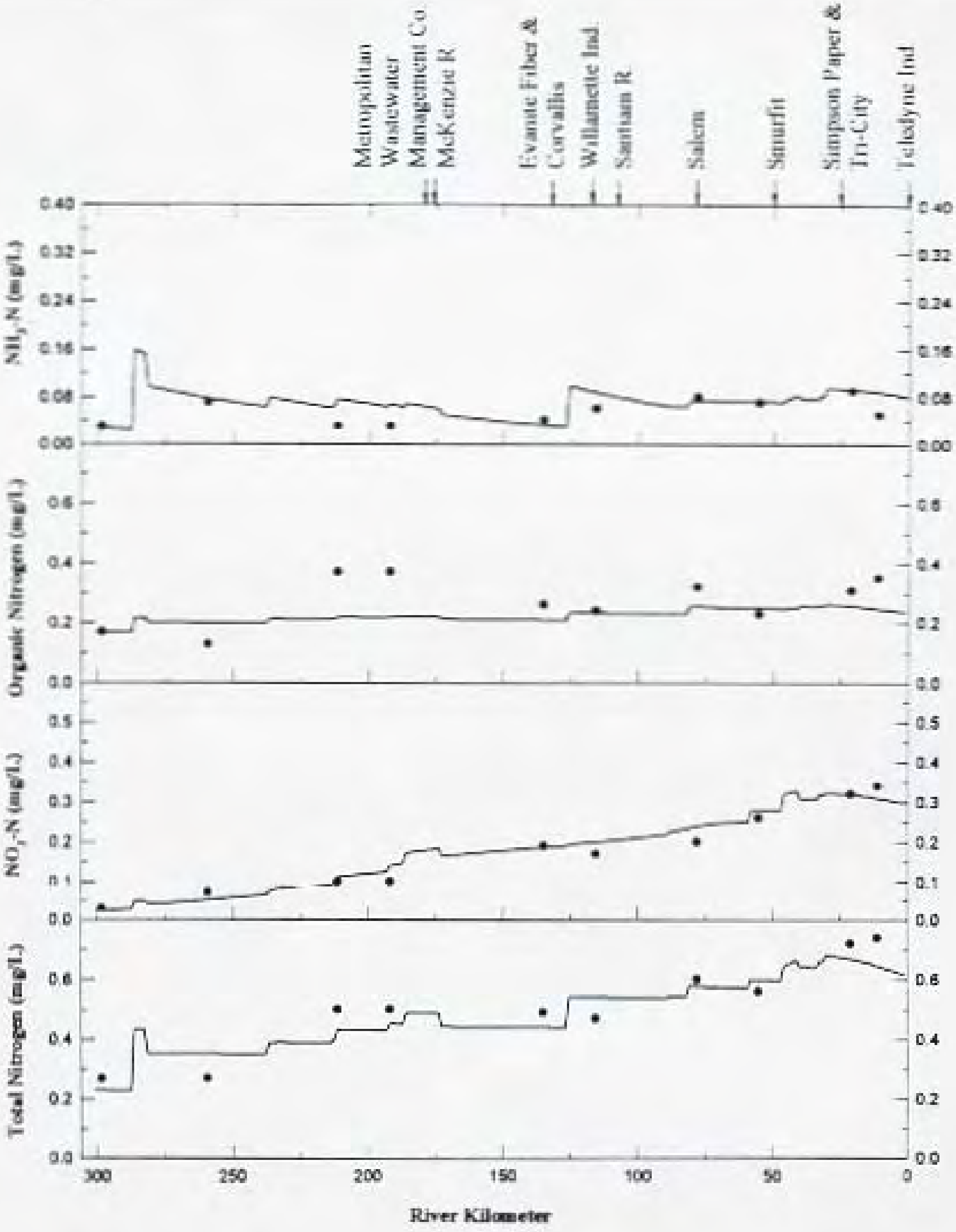

FIGURE B-14. COMPARISON OF THE CALIBRATED QUAL2E MODEL OUTPUT TO THE AUGUST 1992 FIELD DATA FOR NITROGEN COMPOUNDS.

TABLE B-7. SAMPLE QUAL2E INPUT FILE FOR THE WILLAMETTE RIVER.

```
11/29/1992  16:55        Filename: CURT2.INP                              Page  1

TITLE01            component analysis, case 1 full eutro
TITLE02            WILLAMETTE RIVER
TITLE03   NO       CONSERVATIVE MINERAL I          IN
TITLE04   NO       CONSERVATIVE MINERAL II         IN
TITLE05   NO       CONSERVATIVE MINERAL III        IN
TITLE06   NO       TEMPERATURE
TITLE07  YES       BIOCHEMICAL OXYGEN DEMAND
TITLE08  YES       ALGAE AS CHL-A IN UG/L
TITLE09  YES       PHOSPHORUS CYCLE AS P IN MG/L
TITLE10              (ORGANIC-P, DISSOLVED-P)
TITLE11  YES       NITROGEN CYCLE AS N IN MG/L
TITLE12              (ORGANIC-N, AMMONIA-N, NITRITE-N, NITRATE-N)
TITLE13  YES       DISSOLVED OXYGEN IN MG/L
TITLE14  YES       FECAL COLIFORMS IN NO./100 ML
TITLE15   NO       ARBITRARY NON-CONSERVATIVE
ENDTITLE
LIST DATA INPUT
WRITE OPTIONAL SUMMARY
NO FLOW AUGMENTATION
STEADY STATE
NO TRAPEZOIDAL X-SECTIONS
NO PRINT LCD/SOLAR DATA
PLOT DO AND BOD
FIXED DNSTM COND (YES=1)=      0.0         5D-ULT BOD CONV K COEF  =      0.23
INPUT METRIC (YES=1)    =      0.0         OUTPUT METRIC (YES=1)   =       0.0
NUMBER OF REACHES       =     35.0         NUMBER OF JUNCTIONS     =       0.0
NUM OF HEADWATERS       =      1.0         NUMBER OF POINT LOADS   =      25.0
TIME STEP (HOURS)       =      0.0         LNTH  COMP  ELEMENT (DX)=      0.75
MAXIMUM ITERATIONS      =     30.0         TIME INC. FOR RPT2 (HRS)=       0.0
LATITUDE OF BASIN (DEG) =     45.0         LONGITUDE OF BASIN (DEG)=     123.0
STANDARD MERIDIAN (DEG) =     75.0         DAY OF YEAR START TIME  =     180.0
EVAP. COEFF. (AE)       =  0.00068         EVAP. COEFF. (BE)       =   0.00027
ELEV. OF BASIN (ELEV)   =    200.0         DUST ATTENUATION COEF.  =      0.13
ENDATA1
O UPTAKE BY NH3 OXID(MG O/MG N)=   3.43  O UPTAKE BY NO2 OXID(MG O/MG N)=   1.14
O PROD  BY ALGAE (MG O/MG A)   =    1.6  O UPTAKE BY ALGAE (MG O/MG A)  =    2.0
N CONTENT OF ALGAE (MG N/MG A) =   0.09  P CONTENT OF ALGAE (MG P/MG A) =  0.014
ALG MAX SPEC GROWTH RATE(1/DAY)=    2.6  ALGAE RESPIRATION RATE (1/DAY) =   0.05
N HALF SATURATION CONST (MG/L) =    0.2  P HALF SATURATION CONST  (MG/L)=   0.04
LIN ALG SHADE CO (1/H-UGCHA/L) = 0.0015  NLIN SHADE (1/H-(UGCHA/L)**2/3)=    0.0
LIGHT FUNCTION OPTION (LFNOPT) =    2.0  LIGHT SATURATION COEF (INT/MIN)=   0.11
DAILY AVERAGING OPTION (LAVOPT)=    2.0  TOTAL DAILY SOLAR RADTN (INT)  =   0.92
NUMBER OF DAYLIGHT HOURS (DLH) =   14.0  TOTAL DAILY SOLAR RADTN (INT)  = 1400.0
ALGY GROWTH CALC OPTION(LGROPT)=    1.0  ALGAL PREF FOR NH3-N (PREFN)   =    0.5
ALG/TEMP SOLR RAD FACTOR(TFACT)=   0.45  NITRIFICATION INHIBITION COEF  =   10.0
ENDATA1A
ENDATA1B
STREAM REACH    1.RCH=COAST + MID FORK   FROM      186.75    10      177.75
STREAM REACH    2.RCH=EUGENE             FROM      177.75    10      174.75
STREAM REACH    3.RCH=MCKENZIE R         FROM      174.75    10      161.25
STREAM REACH    4.RCH=HARRISBURG         FROM      161.25    10       147.0
STREAM REACH    5.RCH=JAMES R            FROM       147.0    10       145.5
STREAM REACH    6.RCH=LONG TOM R         FROM       145.5    10       141.0
STREAM REACH    7.RCH=PEORIA             FROM       141.0    10       132.0
STREAM REACH    8.RCH=MARYS R            FROM       132.0    10      131.25
STREAM REACH    9.RCH=CORVALLIS          FROM      131.25    10      122.25
STREAM REACH   10.RCH=BENTON CO          FROM      122.25    10      119.25
STREAM REACH   11.RCH=ALBANY             FROM      119.25    10       115.5
STREAM REACH   12.RCH=TRAUX CR           FROM       115.5    10      107.25
STREAM REACH   13.RCH=SANTIAM R          FROM      107.25    10        96.0
STREAM REACH   14.RCH=INDEPENDENCE       FROM        96.0    10       87.75
STREAM REACH   15.RCH=RICKREAL CR        FROM       87.75    10        78.0
STREAM REACH   16.RCH=SALEM              FROM        78.0    10       71.25
STREAM REACH   17.RCH=                   FROM       71.25    10        60.0
STREAM REACH   18.RCH=                   FROM        60.0    10       54.75
STREAM REACH   19.RCH=YAMHILL R          FROM       54.75    10       50.25
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11/29/1992  16:55        Filename: CURT2.INP                              Page  2

STREAM REACH   20.RCH=NEWBERG            FROM       50.25    TO       47.25
STREAM REACH   21.RCH=                   FROM       47.25    TO        39.0
STREAM REACH   22.RCH=WILSONVILLE        FROM        39.0    TO        36.0
STREAM REACH   23.RCH=MOLLALA R          FROM        36.0    TO        33.0
STREAM REACH   24.RCH=CANBY              FROM        33.0    TO        28.5
STREAM REACH   25.RCH=TUALATIN R         FROM        28.5    TO       26.25
STREAM REACH   26.RCH=WILLAM FALLS       FROM       26.25    TO       24.75
STREAM REACH   27.RCH=CLACKAMAS R        FROM       24.75    TO        22.5
STREAM REACH   28.RCH=                   FROM        22.5    TO       20.25
STREAM REACH   29.RCH=PORTLAND           FROM       20.25    TO       18.75
STREAM REACH   30.RCH=JOHNSON CR         FROM       18.75    TO       12.75
STREAM REACH   31.RCH=                   FROM       12.75    10        9.75
STREAM REACH   32.RCH=                   FROM        9.75    10         7.5
STREAM REACH   33.RCH=                   FROM         7.5    10        6.75
STREAM REACH   34.RCH=                   FROM        6.75    10         3.0
STREAM REACH   35.RCH=DOWNSTREAM         FROM         3.0    10         0.0
ENDATA2
ENDATA3
FLAG FIELD RCH=  1.0       12          1 2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH=  2.0        4          6 2 2 2
FLAG FIELD RCH=  3.0       18          6 2 2 2 6 2 2 2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH=  4.0       19          2 2 2 2 2 2 2 2 2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH=  5.0        2          6 2
FLAG FIELD RCH=  6.0        6          6 2 2 2 2 2
FLAG FIELD RCH=  7.0       12          2 2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH=  8.0        1          6
FLAG FIELD RCH=  9.0       12          6 2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH= 10.0        4          2 2 2 2
FLAG FIELD RCH= 11.0        5          6 2 2 2 2
FLAG FIELD RCH= 12.0       11          6 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH= 13.0       15          6 2 2 2 2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH= 14.0       11          2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH= 15.0       13          6 2 2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH= 16.0        9          6 2 2 2 2 2 2 2 2
FLAG FIELD RCH= 17.0       15          2 2 2 2 2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH= 18.0        7          2 2 2 2 2 2 2
FLAG FIELD RCH= 19.0        6          6 2 2 2 2 2
FLAG FIELD RCH= 20.0        4          6 2 2 2
FLAG FIELD RCH= 21.0       11          2 2 2 2 2 2 2 2 2 2 2
FLAG FIELD RCH= 22.0        4          6 2 2 2
FLAG FIELD RCH= 23.0        4          6 2 2 2
FLAG FIELD RCH= 24.0        6          6 2 2 2 2 2
FLAG FIELD RCH= 25.0        3          6 6 2
FLAG FIELD RCH= 26.0        2          6 2
FLAG FIELD RCH= 27.0        3          6 2 2
FLAG FIELD RCH= 28.0        3          2 2 2
FLAG FIELD RCH= 29.0        2          6 2
FLAG FIELD RCH= 30.0        8          6 2 2 2 2 2 2 2
FLAG FIELD RCH= 31.0        4          2 2 2 2
FLAG FIELD RCH= 32.0        3          2 2 2
FLAG FIELD RCH= 33.0        1          6
FLAG FIELD RCH= 34.0        5          6 2 2 2 2
FLAG FIELD RCH= 35.0        4          2 2 2 5
ENDATA4
HYDRAULICS RCH=  1.0       60.0    0.0526    0.4556    0.0447    0.5444       0.02
HYDRAULICS RCH=  2.0       60.0     0.001 0.9575007    2.1559    0.0425       0.02
HYDRAULICS RCH=  3.0       60.0     0.114    0.3793    0.0219    0.6207       0.02
HYDRAULICS RCH=  4.0       60.0    0.1427     0.362    0.0117     0.638       0.02
HYDRAULICS RCH=  5.0       60.0    0.2919    0.2822    0.0105    0.7178       0.02
HYDRAULICS RCH=  6.0       60.0    0.1919    0.3276     0.016    0.6724       0.02
HYDRAULICS RCH=  7.0       60.0    0.1305     0.363    0.0192     0.637       0.02
HYDRAULICS RCH=  8.0       60.0    0.1374    0.3451    0.0182    0.6549       0.02
HYDRAULICS RCH=  9.0       60.0    0.2606    0.2883    0.0115    0.7117       0.02
HYDRAULICS RCH= 10.0       60.0    0.1775    0.3122    0.0125    0.6878       0.02
HYDRAULICS RCH= 11.0       60.0    0.0189    0.4385    0.0624 0.5675002       0.02
HYDRAULICS RCH= 12.0       60.0    0.2105     0.293    0.0095     0.707       0.02
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TABLE B-7. (CONTINUED)

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11/29/1992  16:55        Filename: CURI2.INP                                   Page  3
HYDRAULICS RCH= 13.0        60.0   0.7514    0.2853    0.0072    0.7147        0.02
HYDRAULICS RCH= 14.0        60.0   0.1556    0.3177    0.0112    0.6823        0.02
HYDRAULICS RCH= 15.0        60.0   0.2523    0.2857    0.0056    0.7143        0.02
HYDRAULICS RCH= 16.0        60.0   0.2507    0.2796     0.005    0.7204        0.02
HYDRAULICS RCH= 17.0        60.0   0.2507    0.2796     0.005    0.7204        0.02
HYDRAULICS RCH= 18.0        60.0   0.6836    0.1665    0.0024    0.8335        0.02
HYDRAULICS RCH= 19.0        60.0   0.0651    0.3528    0.0273    0.6472        0.02
HYDRAULICS RCH= 20.0        60.0    0.004    0.5727    0.4115    0.4273        0.02
HYDRAULICS RCH= 21.0        60.0   0.0033    0.5896    0.4525    0.4104        0.02
HYDRAULICS RCH= 22.0        60.0   0.0022    0.6217    0.7314    0.3783        0.02
HYDRAULICS RCH= 23.0        60.0   0.0019    0.6316    0.5368    0.3684        0.02
HYDRAULICS RCH= 24.0        60.0   0.0021 0.6171002     0.472    0.3829        0.02
HYDRAULICS RCH= 25.0        60.0   0.1186    0.3269    0.0122 0.6731002        0.02
HYDRAULICS RCH= 26.0        60.0   0.0102    0.4954     0.245    0.5046        0.02
HYDRAULICS RCH= 27.0        60.00.00077952     0.702    1.4254     0.298        0.02
HYDRAULICS RCH= 28.0        60.00.00077952     0.702    2.3324     0.298        0.02
HYDRAULICS RCH= 29.0        60.00.00029711     0.777    3.3657     0.223        0.02
HYDRAULICS RCH= 30.0        60.0 0.0002953     0.777      3.36    0.2225        0.02
HYDRAULICS RCH= 31.0        60.0 0.0000753    0.8833   10.1541    0.1167        0.02
HYDRAULICS RCH= 32.0        60.0 0.0000753    0.8833   10.1541    0.1167        0.02
HYDRAULICS RCH= 33.0        60.0 0.0000753    0.8833   10.1541    0.1167        0.02
HYDRAULICS RCH= 34.0        60.0 0.0000494    0.9022   14.47130.09779999        0.02
HYDRAULICS RCH= 35.0        60.0 0.0000494    0.9027   14.57680.09729999        0.02
ENDATA5
REACT COEF RCH=  1.0   0.138      0.0       0.0 1.0    0.878      0.0      0.0
REACT COEF RCH=  2.0   0.138      0.0       0.0 1.0    0.971      0.0      0.0
REACT COEF RCH=  3.0   0.138      0.0       0.0 1.0    0.618      0.0      0.0
REACT COEF RCH=  4.0   0.138      0.0       0.0 1.0    1.694      0.0      0.0
REACT COEF RCH=  5.0   0.138      0.0     0.024 1.0    0.544      0.0      0.0
REACT COEF RCH=  6.0   0.138      0.0     0.024 1.0    0.507      0.0      0.0
REACT COEF RCH=  7.0   0.138      0.0     0.024 1.0    0.606      0.0      0.0
REACT COEF RCH=  8.0   0.138      0.0     0.024 1.0    0.518      0.0      0.0
REACT COEF RCH=  9.0   0.138      0.0     0.024 1.0    0.504      0.0      0.0
REACT COEF RCH= 10.0   0.138      0.0     0.024 1.0    0.606      0.0      0.0
REACT COEF RCH= 11.0   0.138      0.0     0.024 1.0    0.354      0.0      0.0
REACT COEF RCH= 12.0   0.138      0.0     0.024 1.0    0.718      0.0      0.0
REACT COEF RCH= 13.0   0.138      0.0     0.024 1.0    0.732      0.0      0.0
REACT COEF RCH= 14.0   0.138      0.0     0.024 1.0    0.563      0.0      0.0
REACT COEF RCH= 15.0   0.138      0.0     0.024 1.0    1.455      0.0      0.0
REACT COEF RCH= 16.0   0.138      0.0     0.024 1.0    1.558      0.0      0.0
REACT COEF RCH= 17.0   0.138      0.0     0.024 1.0    1.558      0.0      0.0
REACT COEF RCH= 18.0   0.138      0.0     0.024 1.0    0.813      0.0      0.0
REACT COEF RCH= 19.0   0.069      0.0     0.024 1.0    0.239      0.0      0.0
REACT COEF RCH= 20.0   0.069      0.1     0.105 1.0    0.086      0.0      0.0
REACT COEF RCH= 21.0   0.069      0.1     0.105 1.0    0.092      0.0      0.0
REACT COEF RCH= 22.0   0.069      0.1     0.105 1.0    0.072      0.0      0.0
REACT COEF RCH= 23.0   0.069      0.1     0.105 1.0    0.116      0.0      0.0
REACT COEF RCH= 24.0   0.069      0.1     0.105 1.0    0.116      0.0      0.0
REACT COEF RCH= 25.0   0.069      0.1     0.029 1.0    0.525      0.0      0.0
REACT COEF RCH= 26.0   0.069      0.1     0.029 1.0    0.081      0.0      0.0
REACT COEF RCH= 27.0   0.069      0.1     0.029 1.0    0.082      0.0      0.0
REACT COEF RCH= 28.0   0.069      0.1     0.029 1.0    0.046      0.0      0.0
REACT COEF RCH= 29.0   0.069      0.1     0.029 1.0    0.065      0.0      0.0
REACT COEF RCH= 30.0   0.069      0.1     0.029 1.0    0.065      0.0      0.0
REACT COEF RCH= 31.0   0.069      0.1     0.029 1.0    0.053      0.0      0.0
REACT COEF RCH= 32.0   0.069      0.1     0.029 1.0    0.053      0.0      0.0
REACT COEF RCH= 33.0   0.069      0.1     0.029 1.0    0.053      0.0      0.0
REACT COEF RCH= 34.0   0.069      0.1     0.029 1.0    0.043      0.0      0.0
REACT COEF RCH= 35.0   0.069      0.1     0.029 1.0    0.042      0.0      0.0
ENDATA6
N AND P COEF    RCH=  1.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH=  2.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH=  3.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH=  4.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH=  5.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH=  6.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
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11/29/1992  16:55        Filename: CURI2.INP                                   Page  4
N AND P COEF    RCH=  7.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH=  8.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH=  9.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 10.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 11.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 12.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 13.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 14.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 15.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 16.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 17.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 18.0     0.0   0.02  0.743     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 19.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 20.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 21.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 22.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 23.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 24.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 25.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 26.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 27.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 28.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 29.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 30.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 31.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 32.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 33.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 34.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
N AND P COEF    RCH= 35.0     0.0   0.02  0.006     0.0     2.0     0.0    0.02     0.0
ENDATA6A
ALG/OTHER COEF RCH=  1.0    55.0     1.0  0.164     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH=  2.0    55.0     1.0  0.131     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH=  3.0    55.0     1.0  0.131     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH=  4.0    55.0     1.0  0.131     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH=  5.0    55.0     1.0  0.226     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH=  6.0    55.0     1.0  0.171     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH=  7.0    55.0     1.0  0.357     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH=  8.0    55.0     1.0  0.357     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH=  9.0    55.0     1.0  0.357     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 10.0    55.0     1.0  0.357     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 11.0    55.0     1.0  0.357     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 12.0    55.0     1.0   0.43     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 13.0    55.0     1.0   0.43     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 14.0    55.0     1.0  0.165     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 15.0    55.0     1.0  0.165     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 16.0    55.0     1.0  0.165     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 17.0    55.0     1.0  0.165     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 18.0    55.0     1.0  0.323     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 19.0    55.0     1.0  0.323     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 20.0    55.0     1.0  0.323     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 21.0    55.0     1.0  0.323     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 22.0    55.0     1.0  0.268     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 23.0    55.0     1.0  0.268     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 24.0    55.0     1.0  0.296     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 25.0    55.0     1.0  0.278     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 26.0    55.0     1.0  0.253     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 27.0    55.0     1.0  0.241     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 28.0    55.0     1.0  0.241     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 29.0    55.0     1.0  0.287     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 30.0    55.0     1.0  0.287     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 31.0    55.0     1.0  0.287     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 32.0    55.0     1.0  0.287     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 33.0    55.0     1.0  0.287     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 34.0    55.0     1.0  0.287     1.0     0.0     0.0     0.0
ALG/OTHER COEF RCH= 35.0    55.0     1.0  0.287     1.0     0.0     0.0     0.0
ENDATA6B
```

TABLE B-7. (CONTINUED)

```
11/29/1992  16:55        Filename: CURT2.INP                                  Page  5

INITIAL  COND-1 RCH=  1.0  65.12    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH=  2.0  67.23    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH=  3.0  68.27    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH=  4.0  68.79    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH=  5.0  69.31    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH=  6.0  69.85    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH=  7.0  69.88    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH=  8.0  69.92    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH=  9.0  69.95    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 10.0  69.99    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 11.0  70.02    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 12.0  70.52    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 13.0  70.41    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 14.0  70.29    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 15.0  70.97    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 16.0  71.41    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 17.0  71.86    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 18.0  72.31    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 19.0  72.75    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 20.0   73.2    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 21.0   73.2    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 22.0  73.65    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 23.0  73.91    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 24.0  74.16    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 25.0   74.8    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 26.0  72.32    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 27.0  71.85    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 28.0  72.08    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 29.0   72.3    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 30.0   75.2    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 31.0   75.2    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 32.0  75.65    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 33.0   76.1    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 34.0   76.1    10.0      2.0     0.0     0.0     0.0      0.0  100.0
INITIAL  COND-1 RCH= 35.0   76.1    10.0      2.0     0.0     0.0     0.0      0.0  100.0
ENDATA7
INITIAL  COND-2 RCH=  1.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH=  2.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH=  3.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH=  4.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH=  5.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH=  6.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH=  7.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH=  8.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH=  9.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 10.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 11.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 12.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 13.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 14.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 15.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 16.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 17.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 18.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 19.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 20.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 21.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 22.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 23.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 24.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 25.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 26.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 27.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 28.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 29.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
INITIAL  COND-2 RCH= 30.0    3.0    0.15     0.03     0.0    0.15    0.02     0.05
```

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11/29/1992  16:55        Filename: CURT2.INP                                  Page  6

INITIAL  COND-2 RCH= 31.0    3.0   0.15   0.03    0.0   0.15   0.02   0.05
INITIAL  COND-2 RCH= 32.0    3.0   0.15   0.03    0.0   0.15   0.02   0.05
INITIAL  COND-2 RCH= 33.0    3.0   0.15   0.03    0.0   0.15   0.02   0.05
INITIAL  COND-2 RCH= 34.0    3.0   0.15   0.03    0.0   0.15   0.02   0.05
INITIAL  COND-2 RCH= 35.0    3.0   0.15   0.03    0.0   0.15   0.02   0.05
ENDATA7A
INCR  INFLOW-1  RCH=  1.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH=  2.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH=  3.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH=  4.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH=  5.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH=  6.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH=  7.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH=  8.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH=  9.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 10.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 11.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 12.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 13.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 14.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 15.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 16.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 17.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 18.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 19.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 20.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 21.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 22.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 23.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 24.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 25.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 26.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 27.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 28.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 29.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 30.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 31.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 32.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 33.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 34.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
INCR  INFLOW-1  RCH= 35.0    0.0   70.0   0.0   0.0   0.0   0.0   0.0   0.0   0.0
ENDATA8
INCR  INFLOW-2  RCH=  1.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH=  2.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH=  3.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH=  4.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH=  5.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH=  6.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH=  7.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH=  8.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH=  9.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 10.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 11.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 12.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 13.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 14.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 15.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 16.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 17.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 18.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 19.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 20.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 21.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 22.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 23.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
INCR  INFLOW-2  RCH= 24.0    0.0    0.0    0.0    0.0    0.0    0.0    0.0
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TABLE B-7. (CONTINUED)

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11/29/1992  16:55        Filename: CURTZ.INP                               Page  7

INCR  INFLOW-2   RCH= 25.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 26.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 27.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 28.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 29.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 30.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 31.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 32.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 33.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 34.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
INCR  INFLOW-2   RCH= 35.0     0.0     0.0     0.0     0.0     0.0     0.0     0.0
ENDATA8A
ENDATA9
HEADWTR-1 HDW=   1.0UPSTREAM               2528.0  56.9   9.1   1.3   0.0   0.0   0.0
ENDATA10
HEADWTR-2 HDW=   1.0     0.0 100.0    1.9  0.17  0.03 0.006 0.024 0.013 0.019
ENDATA10A
POINTLD-1 PTL=   1.0EUGENE         0.0   35.35  72.4   5.9  14.0   0.0   0.0   0.0
POINTLD-1 PTL=   2.0MCKENZIE 1     0.0  1687.0  59.9   9.1   2.7   0.0   0.0   0.0
POINTLD-1 PTL=   3.0MCKENZIE 2     0.0     0.0  59.9   0.0   0.0   0.0   0.0   0.0
POINTLD-1 PTL=   4.0P+1/JRPC       0.0    25.6  79.9   5.0  35.4   0.0   0.0   0.0
POINTLD-1 PTL=   5.0LONG TOM R     0.0    28.0  75.2   9.4   3.2   0.0   0.0   0.0
POINTLD-1 PTL=   6.0MARYS/EVANIT   0.0    19.4  71.7   9.1  19.7   0.0   0.0   0.0
POINTLD-1 PTL=   7.0CORVALLIS      0.0     7.9  71.4   4.5  20.0   0.0   0.0   0.0
POINTLD-1 PTL=   8.0ALBANY/CALAP   0.0    30.6  72.0   6.3   4.9   0.0   0.0   0.0
POINTLD-1 PTL=   9.0WILLIND/TELE   0.0    14.7  83.9   5.0  56.1   0.0   0.0   0.0
POINTLD-1 PTL=  10.0SANTIAM/LUCK   0.0  1445.0  65.8   9.1   3.4   0.0   0.0   0.0
POINTLD-1 PTL=  11.0RICKREAL CR    0.0     6.0  55.4   8.6   3.2   0.0   0.0   0.0
POINTLD-1 PTL=  12.0SALEM          0.0    43.9  76.3   6.0   9.5   0.0   0.0   0.0
POINTLD-1 PTL=  13.0YAMHILL R      0.0    65.0  72.5   8.9   4.4   0.0   0.0   0.0
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POINTLD-2 PTL=  14.0     0.0   1.0    0.0  7.21 2.635 0.018 0.553   2.0  6.16
POINTLD-2 PTL=  15.0     0.0   5.0    0.0  0.71  1.91   0.2   6.1   0.9   3.2
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POINTLD-2 PTL=  21.0     0.0  99.0   1.68  0.33  0.03 0.003  0.09  0.03  0.03
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ENDATA11A
DAM DATA          DAM=  1.0  26.0   2.0   1.0   1.0   1.0   1.0
ENDATA12
DOWNSTREAM BOUNDARY-1      70.0    6.0    0.0   10.0    0.0    0.0    0.0    0.0
ENDATA13
ENDATA13A
LOCAL CLIMATOLOGY                                  0.0    60.0    60.0  30.0      0.0
BEGIN   RCH     1
PLOT    RCH     1     2     3     4     5     6     7     8     9    10    11    12    13    14
PLOT    RCH    15    16    17    18    19    20    21    22    23    24    25    26    27    28
PLOT    RCH    29    30    31    32    33    34    35
```

professional judgment and a visual best-fit technique to arrive at reasonable values for the rate constants in the following steps:

- Nitrogen balance: Adjustment of the ammonia oxidation nitrification and organic nitrogen hydrolysis rates to fit the model predictions to the nitrate, ammonia, and organic nitrogen data.
- Phytoplankton growth:
  1) Variation of the model options for algal growth limitation and photosynthesis.
  2) Adjustment of the specific maximum algal growth rate, algal respiration rate, and the phytoplankton settling rate to fit the model predictions to the chlorophyll *a* and nutrient data.
- Phosphorus balance: Adjustment of the organic phosphorus decay rate to fit the organic phosphorus and soluble phosphorus data.
- DO balance: Fixing of the instream ultimate CBOD decay rate and the atmospheric reaeration coefficient based on previous studies of the Willamette River (reported by McKenzie et al., 1979) and adjustment of the SOD to fit the DO field data.

Final model calibration was achieved by minimizing the cumulative absolute relative error (CARE) between model output and field data using the ammonia oxidation rate, the organic phosphorus decay rate, the maximum specific algal growth rate, the algal settling rate, and the sediment oxygen demand rate.

The calibrated model's fit to the synoptic survey data are shown in Figures B-14, B-15, and B-16. The location and concentration of the minimum DO measured during the synoptic survey at RK 43 was matched by the model (7.3 mg/L) (Figure B-16). The model-predicted DO concentrations ranged up to 8.6 percent of the 24-hour average DO concentrations measured at 15 stations, with a mean and median relative difference of 2.5 and 1.7 percent, respectively. The model-predicted DO concentrations did not fit the concentrations measured using single grab samples collected by ODEQ. In general, single grab samples for DO were considered inadequate for the calibration of a steady-state model, especially for the upper river reach where large diurnal fluctuations in DO occur.

The maximum chlorophyll *a* concentrations measured in the lower river were also predicted well by the model, although the model prediction increased exponentially to much higher levels below RK 11 (Figure B-17). The model did not predict the relatively high chlorophyll *a* levels measured in the upper river. Suspended algal biomass in the upper river reach is considered to be derived from sloughing of periphyton in this relatively shallow stretch of river. Because the model does not consider the influence of periphyton growth, the model DO predictions for the upper river reach reflect only variation in the steady-state DO concentration due to point source inputs and reaeration.

B.3.4.2 Model Validation

Validation of a calibrated model with an independent data set is meant to substantiate the model's predictive power under environmental conditions similar to those under which the model was calibrated. With this goal in mind, the calibrated model was applied to August 1990 conditions using point source data provided by HydroQual (1990) and water quality data available as part of ODEQ's Ambient Monitoring Program. Although model-predicted and measured conditions were in relatively good agreement (Figure B-17 shows the model's fit to the DO and chlorophyll *a* data), the model was not considered to be fully validated because the DO concentrations reported by ODEQ are for single grab samples. These types of samples were not considered adequate for the calibration or validation of a steady-state model. The model was considered sufficiently tested for management analysis. Full validation will be achieved upon completion of additional monitoring and future updates of the model.

B.3.5 Conclusions

Figure B-18 shows the effect of variation in the Willamette River flow regime on Willamette River DO and chlorophyll a concentrations. The relative effect of various flow regimes, ranging from 135 to 218 m^3/sec measured at Salem, on the calibrated-model prediction of DO at RK 43 (using the August 1992 model inputs) and chlorophyll *a* at RK 16, are presented. In general, variation in river flow had a noticeable effect on DO throughout the river and on chlorophyll *a* below RK 80 where the river enters the Newberg Pool reach. This analysis substantiates the assumption that flow augmentation of the Willamette River during the low-flow period of July through September can be an effective means of water quality

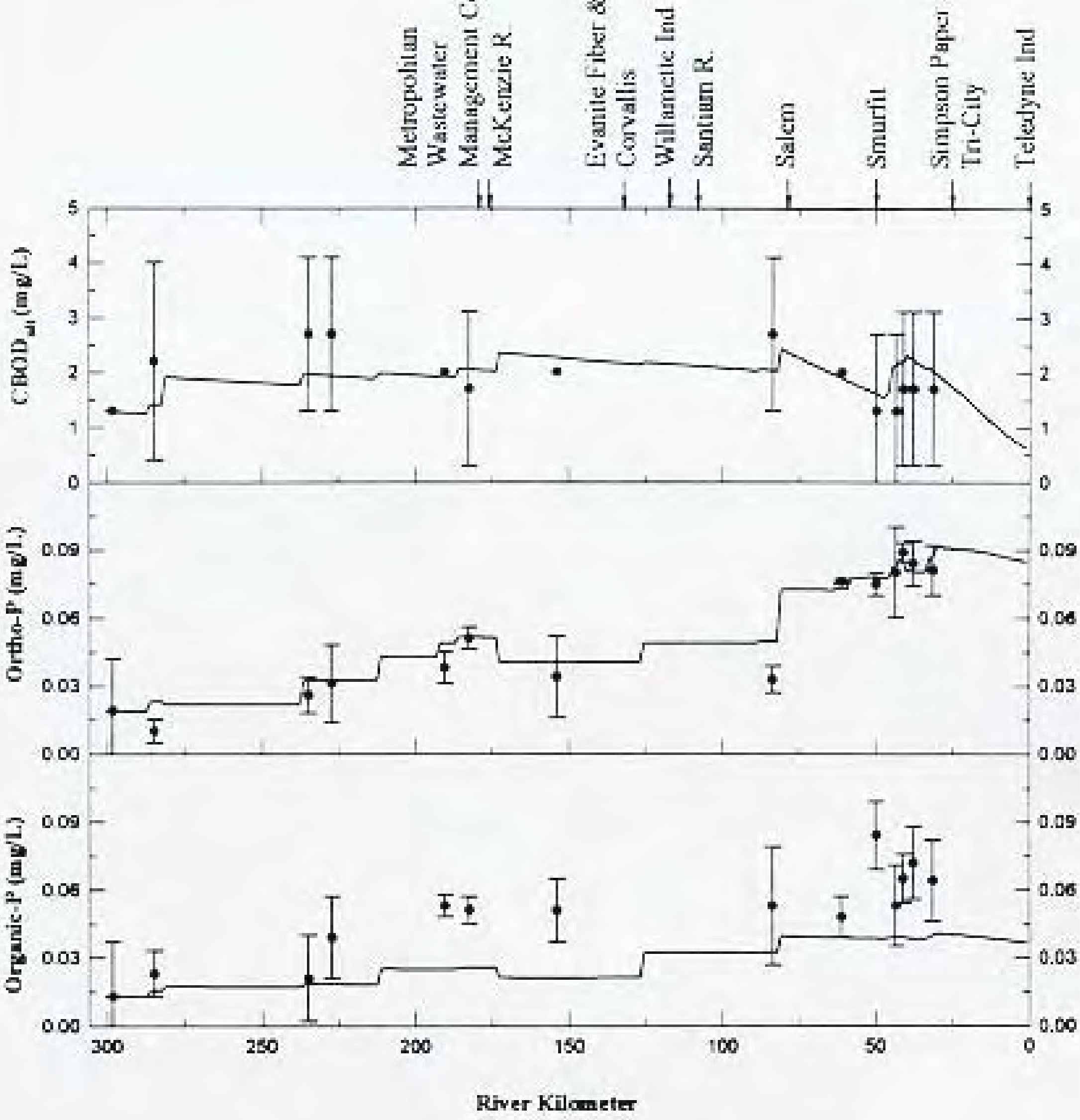

FIGURE B-15. COMPARISON OF CALIBRATED QUAL2E MODEL OUTPUT TO THE AUGUST 1992 FILE DATA FOR ULTIMATE CBOD AND PHOSPORUS COMPOUNDS

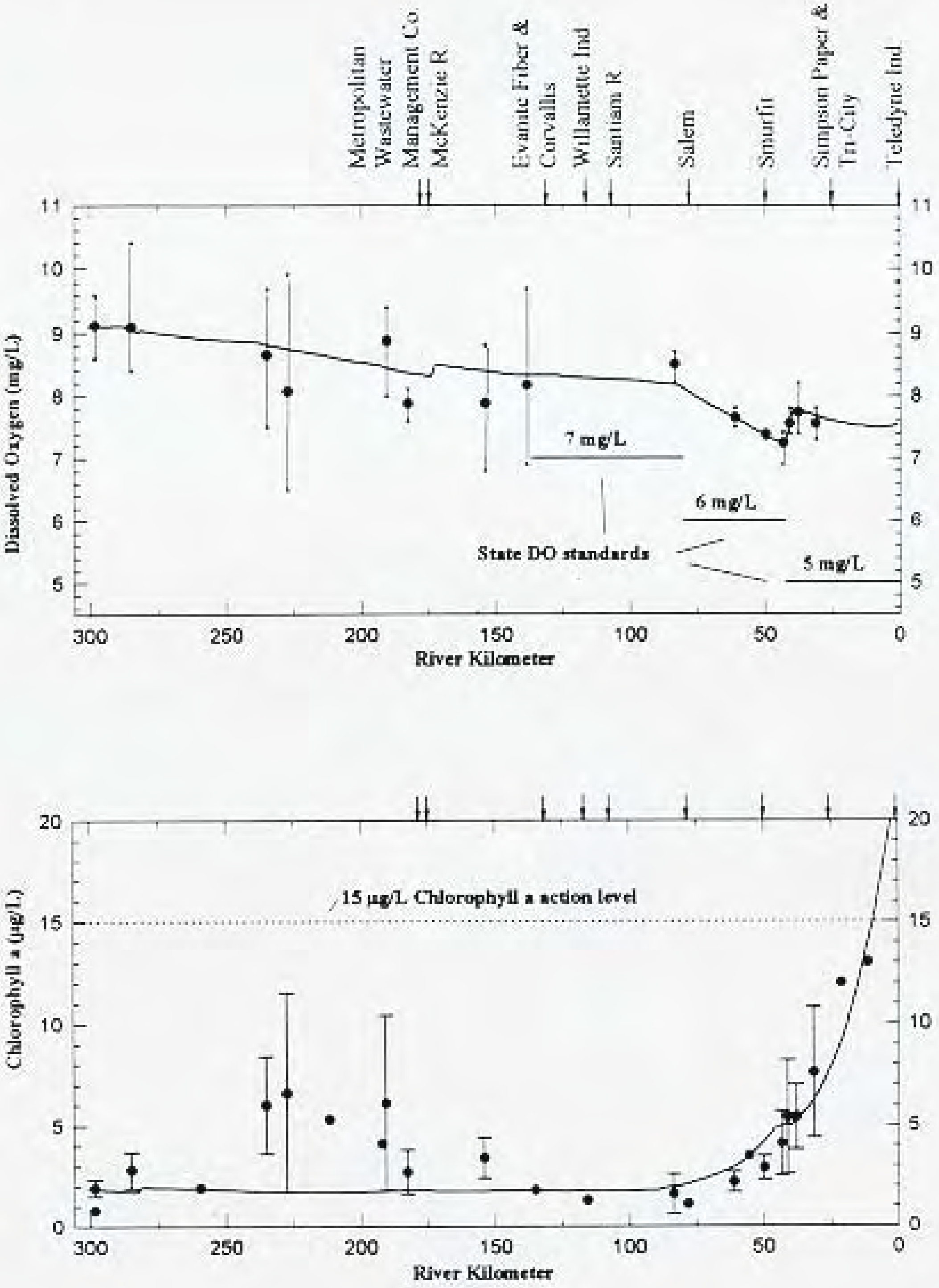

FIGURE B-16. COMPARISON OF THE CALIBRATED QUAL2E MODEL OUTPUT TO THE AUGUST 1992 FIELD DATA FOR CHLOROPHYLL *A* AND DISSOLVED OXYGEN.

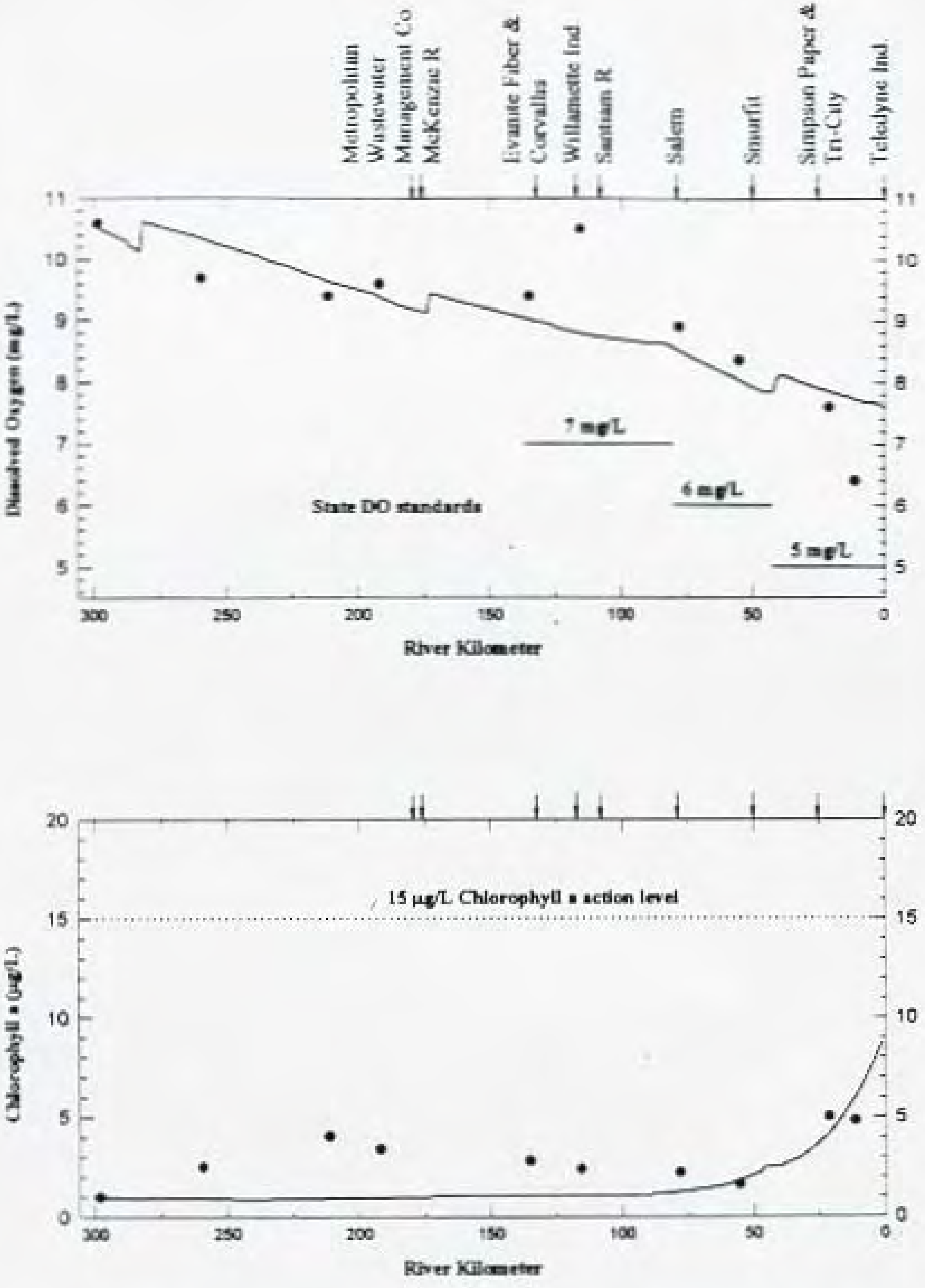

FIGURE B-17. VERIFICATION OF THE CALIBRATED QUAL2E MODEL AGAINST AUGUST 1990 ODEQ AMBIENT MONITORING PROGRAM DATA FOR DISSOLVED OXYGEN AND CHLOROPHYLL *A*

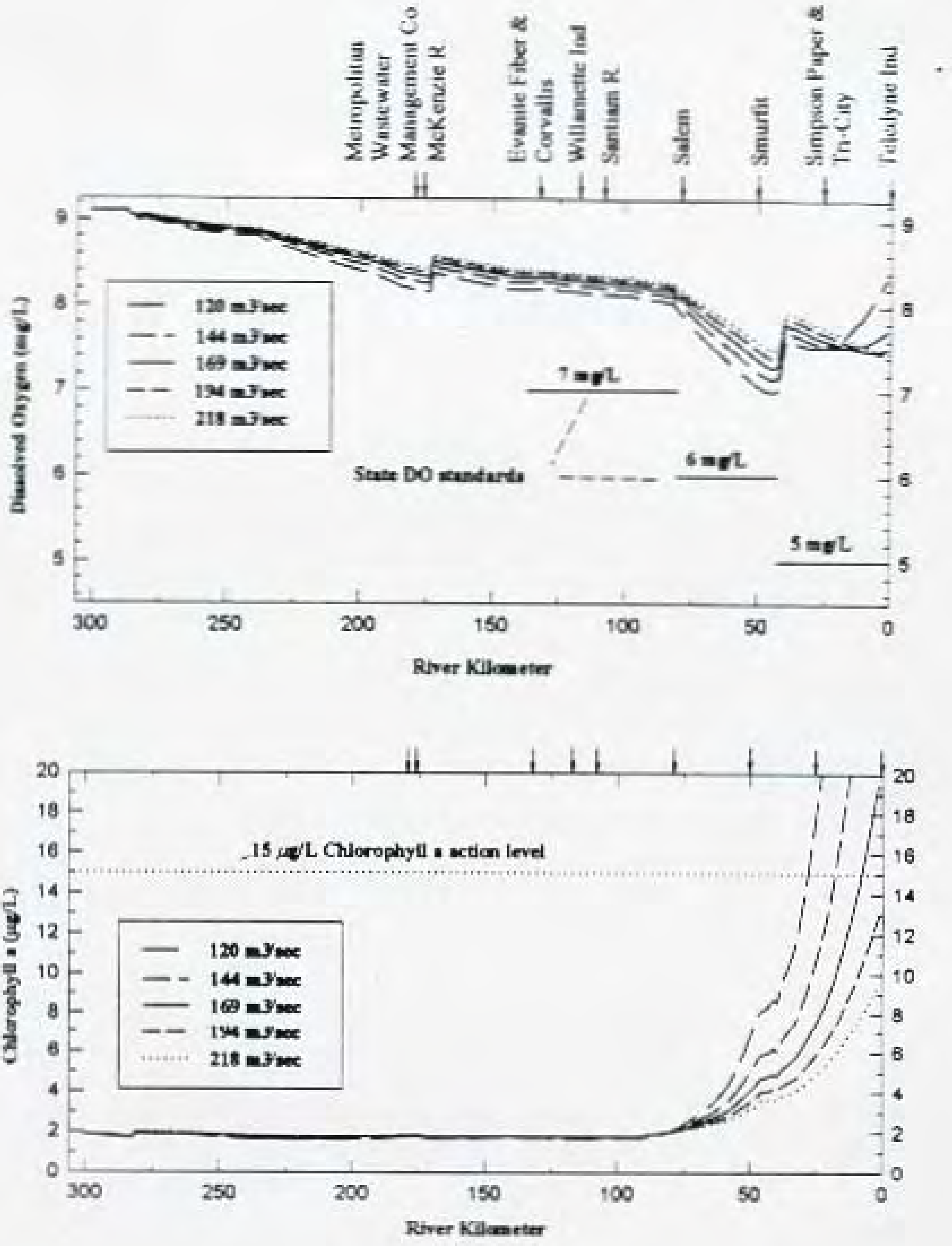

FIGURE B-18. THE EFFECT OF VARIATION IN THE FLOW REGIME OF THE WILAMETTE RIVER ON THE QUAL2E-PREDICTED AUGUST 1992 DISSOLVED OXYGEN AND CHLOROPHYLL *A* CONCENTRATIONS management. A minimum flow of 170 m^3/sec at the Salem gauge (RK 134) is maintained during these months to allow for navigation, and also, to maintain adequate DO levels in the river (Rickert et al. 1980). The model-predicted DO concentrations at RK 43 vary almost linearly from 7.0 to 7.5 mg/L over the range of flow regimes evaluated. The model-predicted effect of flow on chlorophyll *a* concentration was not linear. The chlorophyll *a* concentration is predicted by the model to increase rapidly when river flow at Salem falls much below 150 m^3/sec. These results support the hypothesis of Rickert et al. (1977) that phytoplankton biomass in the lower river is most strongly controlled by variation in the flow (i.e., the water residence time) and that management of the flow regime is not only an effective means to control DO levels, but should also be effective in the control of phytoplankton biomass.

There are several assumptions inherent to the modeling analysis that should be considered in interpretation of the calibrated model results.

1) The model does not incorporate the effect of periphyton production on DO. The effect of periphyton production on DO might be significant in the upper reach of the river above RK 80.

2) The model does not account for tidal mixing with the Columbia River. Therefore, the model output below RK 16 should be interpreted with caution.

3) The model does not consider high-flow events or dynamic conditions. The model assumes steady-state conditions that are appropriate for representing the low-flow conditions present when problems in the Willamette River typically occur.

4) The model does not explicitly consider minor point sources of nutrients or oxygen-demanding substances. Preliminary estimates suggest that minor point sources could contribute as much as an additional 10 percent to the estimated CBOD demand load to the mainstem of the river (Tetra Tech, 1992d).

5) The model-predicted DO concentration in the lower river was very sensitive to the model-specified rate of SOD. However, no field data were collected to establish the reliability of the rates of SOD specified in the model.

None of the assumptions associated with the model development precluded its application and use as a predictive tool.

Future work includes a field study conducted during the summer of 1994 by ODEQ and USGS, including measurements of SOD that will allow further refinement of the QUAL2E-UNCAS DO model. Planned Phase II model improvements include updating the model to QUAL2E Version 3.20, which will allow for the incorporation of minor point sources and the evaluation of the model-specified SOD rates. Depending on the results of this evaluation, the model will be recalibrated and validated using the more recently collected field data. The update of the model will refine estimation and predictive capabilities.

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APPENDIX C: QUALITY ASSURANCE FOR FIELD MONITORING PROGRAMS

C.1 OVERVIEW

As used here, quality assurance (QA) is a system of activities used to provide documented assurance that a data product of known and acceptable quality is produced.

The importance of QA should be evident. However, because of the additional effort required to provide QA (advance planning, management, supervision, and resources) it is often neglected or overlooked. This manual has addressed, at some length, guidelines for the analysis of data that will lead to the performance of technically sound, defensible TMDL studies. This is particularly important where decisions derived from TMDL studies have serious economic and environmental impacts.

A properly planned and implemented QA program will enable the substantiation of data accuracy and precision by an outside, impartial review and forestall any attempts to discredit or impeach the data produced. This section outlines the minimum QA effort required to ensure a reliable TMDL study. Its aim is to assist the user in developing a reliable and effective quality assurance program that will meet data user requirements for completeness, precision, accuracy, and comparability of data. Note that the QA requirements given herein are the minimum requirements; they are to serve as a foundation on which the user can build a viable QA program.

C.2 ACCURACY AND PRECISION

Accuracy refers to agreement between the measurement and the true value of the measurand, with the discrepancy normally referred to as error. Precision refers to the reproducibility (repeatability) of the measurement, when repeated on a homogenous, time-stationary measurand, regardless of the displacement of the observed value from the true value.

The statistical measures of location or central tendency (e.g., the various averages, mean, median, and mode) are related to accuracy. The statistical measures of dispersion or variability (e.g., variance, standard deviation, coefficient of variation, and other measures derived from central moments of the probability density function) are related to precision.

Discrepancies between the results of repeated observations, or errors, are inherent in any measurement process since it is recognized that the true value of an object of measurement can never be exactly established. These errors are customarily classified into two main groups: systematic and random (or accidental) errors. Systematic errors usually enter into records with the same sign and frequently with either the same magnitude (e.g., a zero offset) or an establishable relationship between the magnitude of the measurement and the error. The methods of symmetry and substitution are frequently used to detect and quantify systematic errors. In the method of symmetry, the test is repeated in a symmetrical or reversed manner with respect to the particular condition that is suspect. In the method of substitution, the object of measurement is replaced by one of known magnitude (a calibration standard); an instrument with a known calibration curve is substituted for the measuring instrument in question, and so on. Thus, systematic errors bear heavily on the accuracy of the measurement.

Random errors, on the other hand, are due to irregular causes, too many in number and too complex in nature to allow their origin to be determined. One of the chief characteristics of random errors is that they are normally as likely to be positive as negative and, therefore, are not likely to have a great effect on the mean of a set of measurements. The chief aim of a data quality assurance effort is to account for systematic errors and thereby reduce errors to the random class, which can be treated by simple probability theory, in order to determine the most probable value of the object of observation and a measure of the confidence placed in this determination.

C.3 ELEMENTS OF A QA PROGRAM

The basic elements of any quality assurance program include the following:

- Management's commitment to provide the resources necessary to implement quality assurance activities (approximately 10 to 20 percent of total water monitoring resources).

- Designation of a quality assurance coordinator responsible for coordinating and implementing necessary quality assurance activities.
- Documentation of a quality assurance plan outlining the specifics of and responsibilities for the development and implementation of internal and external quality assurance checks.

A complete QA program for water quality measurements would incorporate a variety of specific elements. These can be depicted on a quality assurance wheel, as shown in Figure C-1. The wheel arrangement illustrates the nature of a quality assurance system that addresses all elements and at the same time allows program managers the flexibility to emphasize those elements which are most applicable to their particular program. Quality assurance elements are grouped on the wheel according to the organizational level to which responsibility is normally assigned. These organizational levels are the quality assurance coordinator (normally a staff function), supervisor (a line function), and operator. Together the supervisor and quality assurance coordinator must see that all these elements form a complete and integrated system and achieve the desired program objectives.

The following specific elements are suggested as minimal requirements for structuring a QA program for a TMDL study. Any proposed program should be compared against these criteria to determine its acceptability.

- A written quality assurance plan should be prepared. It should define the oversight role of management; identify personnel responsible for the quality assurance program; and specify proper sample collection, use of approved measurement techniques, calibration standards and their verification, internal quality control practices, and appropriate data management controls.
- An estimate of costs associated with the quality assurance program in terms of percentage of overall project cost should be developed. Normally, a minimum of 10 percent of the estimated sample collection and analysis costs will be necessary for adequate quality control.
- A program for demonstration of acceptable performance through the use of audit samples should be established and used throughout the duration of the study.
- Provision should be made for performing on-site field and laboratory audits at the option of and on a schedule established by the project officer. Such audits would evaluate performance and document the availability of all equipment and supplies necessary for successful execution of the study.
- Documentation of quality control performance should be submitted with the final report and otherwise as directed by the project officer.

C.4 ASPECTS OF A QA PROGRAM

A number of aspects of a QA program must be addressed by the QA plan if the minimal requirements are to be met. These aspects can be aggregated into three general categories: water chemistry (laboratory), field data collection, and data handling and reporting.

C.4.1 Water Chemistry

The minimum QA requirements for water chemistry are as follows:

- Quality control management manual
  - Outline of quality assurance program objectives
  - Outline of the administrative structure of the laboratory (including an organizational chart)
  - Clear identification of the responsibilities for implementing the specific quality control activities
  - Commitment of resources by management to implement the necessary quality control activities
  - Description of laboratory training program
  - Designation of a laboratory quality assurance coordinator, including a statement addressing coordination responsibilities and duties
- Laboratory operations manual
  - Description of analytical methodologies and procedures
  - Description of laboratory quality control activities

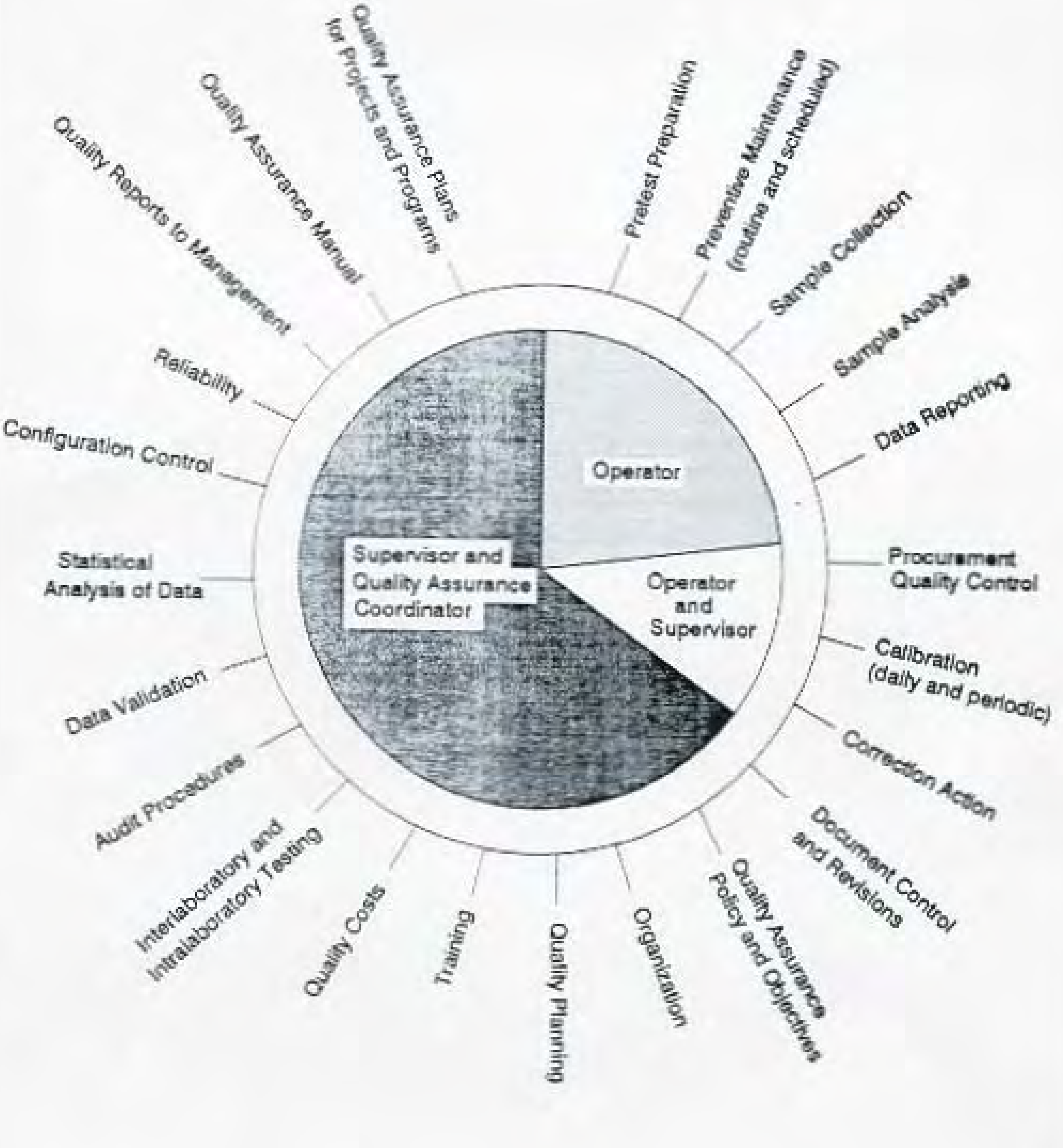

FIGURE C-1. QUALITY ASSURANCE ELEMENTS AND RESPONSIBILITIES (THE QUALITY ASSURANCE WHEEL)

  - Description of the laboratory's internal chain-of-custody procedures
  - Description of general laboratory requirements
  - Description of laboratory communication and coordination
- Sample log manual
- Quality control records manual
- Blind duplicate and spiked field samples
  - Sample audits
  - Parameters included in the program
- Audit sample preparation procedures
- Data evaluation
  - Audit follow-up and corrective action
- Estimation of limits for laboratory accuracy checks

C.4.2 Data Collection

The minimum QA requirements for field data collection are as follows:

- Sampling network design
- Sampling procedures
- Calibration of direct-reading field instruments and fixed continuous monitoring devices
- Record keeping
- Quality assurance checks in field sampling
- Personnel training
- Flow measurements
- Records, data storage and retrieval
- Sample handling and identification procedures (chain of custody)
- Collection of samples/field investigations

C.4.3 Handling and Reporting

The minimum QA requirements for data handling and reporting are as follows:

- Preprinted forms and labels
- Data sheets
- Data flow
- Significant figures and rounding procedures
- Calculation checks
- Data corrections
- Data reviews
- Reasonableness and consistency checks
- Data acceptance
- Data storage and retrieval

APPENDIX D: UNCERTAINTY ANALYSIS

D.1 INTRODUCTION

Uncertainty analysis should be included as an integral component of water quality modeling. One of the primary purposes is to quantify the error in predicting water quality and evaluate the effect of input parameters on model output. By quantifying this error, improved management decisions can be made. Such quantification also facilitates subsequent studies such as risk assessments to evaluate alternative waste load allocations. In addition, uncertainty analysis may provide insight into the need for additional data collection to refine the estimate of certain loads, initial conditions, or reaction rates. For example, if the model is sensitive to the reaeration rate (that is, a small change in reaeration rate results in large changes in the prediction of critical water quality parameters such as dissolved oxygen), it may be appropriate to allocate resources to more accurately estimate the reaeration rate of that stream or river.

There are three techniques for performing uncertainty analysis: sensitivity analysis, first-order error analysis, and Monte Carlo simulation. Each technique has advantages and disadvantages in terms of applicability and computational burden that will make one method more suitable than another for a particular analysis. In many instances, the modeler may need to explore the results from all three procedures. The three methods may produce discrepancies in their results since the methodologies and assumptions differ. Each of these techniques is available in QUAL2E-UNCAS, and the following discussion is limited to the features available in that model. An example uncertainty analysis using QUAL2E-UNCAS is provided at the end of this appendix.

D.2 TECHNIQUES IN UNCERTAINTY ANALYSIS

D.2.1 Sensitivity Analysis

Sensitivity analysis is the simplest of the three techniques for assessing the effect of an input variable on the output variable. This analysis technique can be used to evaluate simple alternatives and projections such as the effect of reducing all pollutant loads by 10 percent. Simple what-if scenarios are particularly useful for managers who must make decisions among alternative load reduction strategies. For the modeler, the same analysis can serve as a useful guide for model calibration.

In the single-factor approach, the modeler varies one of the input variables, *X*, and observes the effect in a particular output variable, *Y*. A sensitivity coefficient is then computed as the percentage change in *Y* divided by the percentage change in *X*. In general, a sensitivity coefficient can be estimated at all points where the output variable is predicted. However, this process can result in an enormous interpretation burden and it is generally recommended that the analysis be limited to critical points along the modeled stream. This process can then be repeated for a number of different perturbations in *X* as well as other input variables. By evaluating the relative change in the sensitivity coefficients for different input variable perturbations, the modeler can determine the model nonlinearity for that input variable.

Similarly, several input variables can be varied simultaneously. As the number of input variables and combinations is increased, the interpretation of results is complicated. Experimental design strategies can be applied in this situation to elicit main and interaction effects of input variables. Specifically, in QUAL2E-UNCAS, the modeler may specify a 2^2 or 2^3 factorial design. In other words, the modeler may be able to examine the main and interactive effects of two or three variables evaluated at two levels (e.g., perturbations). The statistical significance of the interaction and main effects are evaluated by comparing an appropriate ratio of the sum of squares to a critical F ratio.

To perform sensitivity analysis with QUAL2E-UNCAS, the user must specify the type of analysis (single/multiple variable or factorial design), the input variables to be modified, and the perturbation as a percent of the input variable.

D.2.2 First-Order Error Analysis

First-order error analysis can be used in a manner similar to that used for sensitivity analysis. In addition to estimating the change of an output variable with respect to an input variable, first-order error analysis provides an estimate of the output variance. A first-order approximation (from the Taylor series expansion) to the relationship for computing variances in multivariate situations is used. Input variables are assumed to be independent, and the model is assumed to respond linearly to the input variables. (In some instances, the assumptions may not be correct.) The linear assumption can be evaluated by computing the normalized sensitivity coefficients for several different input parameter perturbations. If the normalized sensitivity coefficients are similar or the difference is small, the model can be assumed to be linear for that input parameter. If the difference in normalized coefficients is large, it may be more accurate to use the Monte Carlo simulation approach to estimate output parameter variance.

In this analysis, the sensitivity coefficients are normalized such that

$$S_{ij} = (\Delta Y_j / Y_j) / (\Delta X_i / X_i) \tag{D-1}$$

where

S_{ij} = normalized sensitivity coefficient for output Y_j to input X_i
X_i = base value of input variable
ΔX_i = magnitude of input perturbation
Y_j = base value of output variable
ΔY_j = sensitivity of output variable

The components of variance for each output variable (Y) are the percentages of output variance attributable to each input variable (X) and are computed in the following manner.

$$Var\,(Y_j) = \sum_{i=1}^{n} (\Delta Y_j / \Delta X_i)^2 \; Var\,(X_i) \tag{D-2}$$

where

$Var(Y_j)$ = variance of output variable Y_j
$Var(X_i)$ = variance of input variable X_i

Each term in the summation of Equation D-2 is a component of the total variance of the output variable. A particular input variable may be a large (small) contributor to the output variance if it has either a large (small) input variance or a large (small) sensitivity coefficient. This analysis can be used as a guide for additional field work. To apply this analysis technique using QUAL2E-UNCAS, the user must specify the magnitude of input parameter perturbation and variance. The variance term is a measure of uncertainty caused by factors such as spatial and temporal variation, sampling and analytical error, and bias in measurement or estimation techniques. A file of typical variance estimates is provided with the model.

D.2.3 Monte Carlo Simulation

Monte Carlo simulation is a numerical procedure that can be used to evaluate linear and nonlinear systems. Each input variable is defined to have a certain probability density function (pdf). Before each model run, an input variable is randomly selected from each predefined pdf. By combining the results from numerous model runs, a pdf can be developed for the output variable. The pdf is useful in evaluating overall model predictions and in assessing the likelihood of violating a water quality standard.

In general, the linear and independence assumptions of first-order error analysis can be relaxed when using Monte Carlo simulation techniques. In QUAL2E-UNCAS, only the linear assumption is relaxed. To use QUAL2E-UNCAS, the user must specify the variance of the input variable (a file of typical variance estimates is provided), the probability distribution as either normally or lognormally distributed, and the number of simulations to perform. As one would expect, the number of model runs is relatively large as compared to the number of runs typically done for sensitivity or first-order error analysis. Preliminary experience indicates that about 2000 simulations are required to achieve estimates of output standard deviations with 95 percent confidence intervals of 5 percent.

D.3 EXAMPLE APPLICATION

This section provides an example of how the uncertainty methodologies in QUAL2E-UNCAS can be applied to a QUAL2E data set. The sole purpose of this section is to demonstrate the utility of uncertainty analysis rather than to provide a definitive analysis of the river system from which the data were obtained. This appendix is a condensation of Appendix C of the QUAL2E and QUAL2E-UNCAS User Manual (Brown and Barnwell, 1987), entitled QUAL2E-UNCAS Example Application. The reader is referred to that manual for a more detailed explanation of QUAL2E-UNCAS.

The data used to demonstrate the capabilities of QUAL2E-UNCAS were obtained from a U.S. EPA Region 4 survey of the Withlacoochee River during October 1984 (Koenig, 1986). In this study, water quality simulations were examined for portions of the river subjected to both municipal and industrial waste loads. In addition, there is a significant accretion of flow from groundwater inputs. The river has a uniform low slope but is characterized by alternating shoals and pools (often in excess of 25 feet deep). Average depths during the survey periods were 5.2 to 14.8 feet; widths were 90 to 140 feet; and flows varied from 150 cfs at the headwater to 660 cfs at the end of the system. Water quality is affected by algae activity resulting from municipal waste discharges above the section of stream studied. The addition of industrial waste at RM 24, however, dramatically reduces light penetration to the extent that the algae population diminishes in the downstream direction.

A location map of the basin is shown in Figure D-1, and a plot of observed and modeled dissolved oxygen concentrations is presented in Figure D-2. Ten state variables were simulated in this study: temperature, dissolved oxygen, carbonaceous BOD, four nitrogen forms (organic, ammonia, nitrate, and nitrite), two phosphorus forms (organic and dissolved), and algae as chlorophyll *a*. A summary of the calibrated inputs and their variance estimates for the uncertainty analysis is shown in Table D-1. The calibrated values in general were obtained by adjusting field or laboratory measurements of the specific model inputs. The variance estimates were computed from replicate data taken during the survey period and by inference from other published data (McCutcheon, 1985; Bowie et al., 1985).

D.3.1 First-Order Error Analysis (FOEA)

Table D-2 shows the first-order error analysis (FOEA) results for the output variables of CBOD and DO at three locations in the Withlacoochee system: an upstream location (RM 26), a midpoint near the dissolved oxygen sag (RM 20), and a downstream location (RM 2). For the CBOD sensitivity coefficients in Table D-2(a), it is clear that the input forcing functions such as point load, headwater flows, and CBOD dominate model sensitivity.

Table D-2(a) also presents the components of variance for the modeled CBOD output. These results show a pattern similar to the sensitivity coefficients. The headwater CBOD is the dominant contributor (99 percent) to CBOD variability in the upper reaches of the basin. The point load CBOD values are the primary variance component elsewhere in the river (84 percent at RM 20 and 79 percent at RM 2). The total variability in simulated CBOD estimated by the first-order analysis, when expressed as a standard deviation, varies from 0.35 mg/L to 0.76 mg/L to 0.27 mg/L proceeding through the basin.

The FOEA results for dissolved oxygen are presented in Table D-2(b). The only forcing functions that have large DO sensitivity coefficients are the headwater inputs, not the point load inputs. DO is also very sensitive to temperature inputs. Next in importance of DO sensitivity are the reaeration rate and velocity. Similar patterns are apparent in the components of variance for dissolved oxygen (Table D-2(b)). CBOD decay has a relatively small impact on DO variance, whereas reaeration and SOD have large impacts. Temperature inputs make a minimum contribution to DO variance. The total variability in simulated DO, when expressed as a standard deviation, increases in the downstream direction varying from 0.18 mg/L to 0.30 mg/L and averaging about 5 percent of the simulated DO.

D.3.2 Effect of Model Nonlinearity

First-order error analysis uses a linear approximation to compute an estimate of output variance. The validity of that approximation can be assessed by computing the sensitivity coefficients for both large and small values of delta x, the input perturbation. Small changes in the normalized sensitivity coefficient indicate near linearity of the state variable over the range of perturbed input values, while large changes in sensitivity reflect important nonlinear effects. Table D-3 contains values of the normalized sensitivity coefficients for the state variables DO and chlorophyll *a* for input perturbations, ranging from -20 to +20 percent. The input variables selected for analysis are those having the largest sensitivity coefficients.

For dissolved oxygen (Table D-3(a)), the reaeration and headwater temperature have the largest nonlinear effects on DO. The other variables are considered linear for the conditions of the simulation. The net effect from all model input nonlinearities is manifest in the FOEA estimate of dissolved oxygen standard deviation, which decreases by 7 percent over the range of input perturbations.

The more pronounced patterns are observed for the state variable, chlorophyll *a* (Table D-3(b)). The ratio of chlorophyll *a* to algal biomass and headwater flow exhibit large nonlinear effects. The maximum algal growth rate and the algal respiration rate show modest nonlinearities, while headwater chlorophyll *a* is essentially linear. The net FOEA estimate of stand-

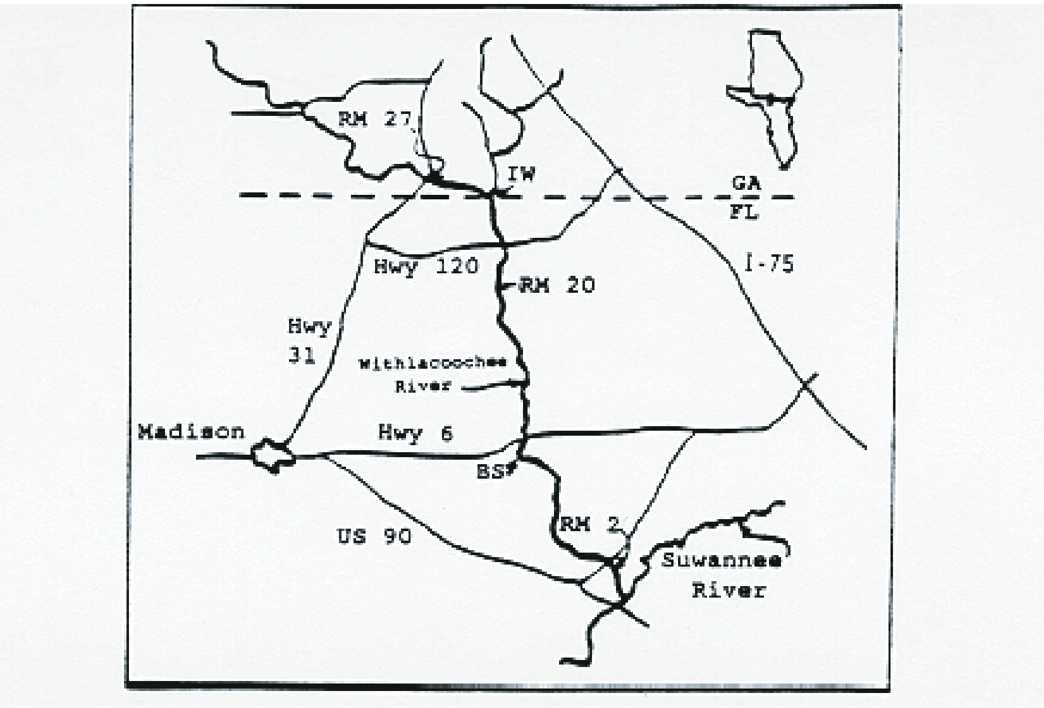

FIGURE D-1. LOCATION MAP OF THE WITHLACOOCHEE RIVER BASIN

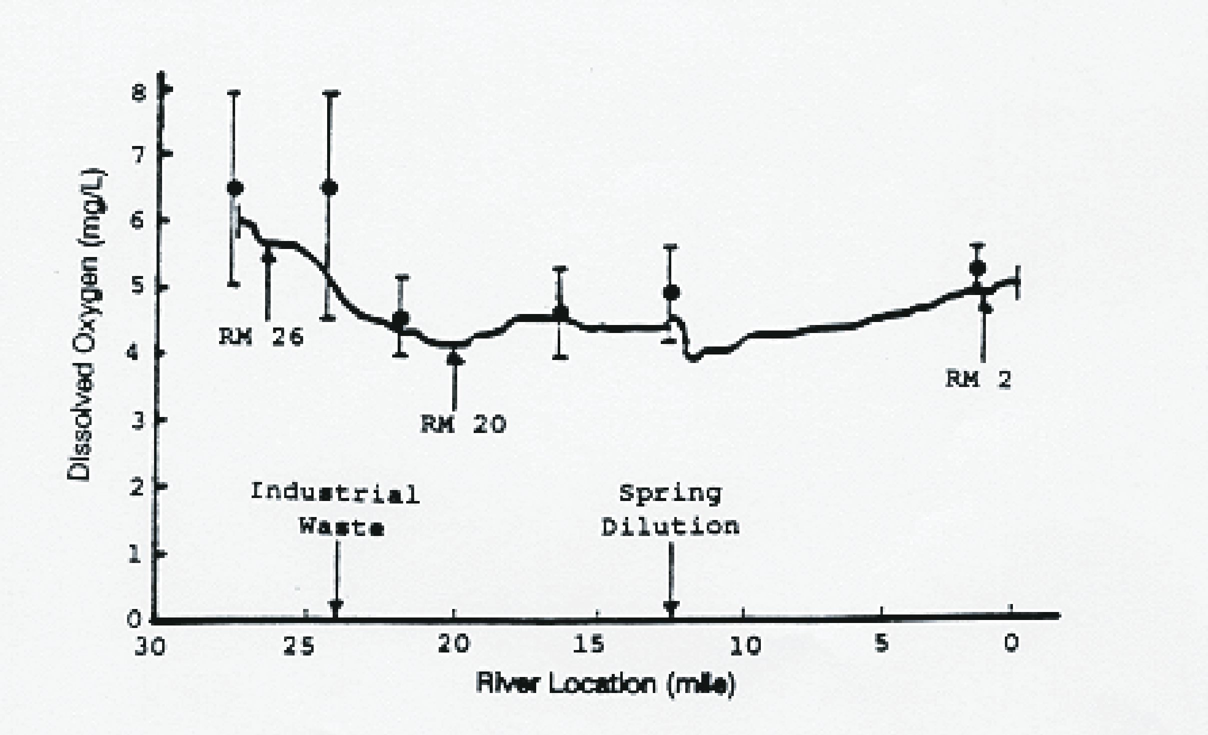

FIGURE D-2. OBSERVED AND PREDICTED DISSOLVED OXYGEN CONCENTRATIONS

TABLE D-1. SUMMARY OF INPUT DATA FOR QUAL2E-UNCAS SIMULATION– WITHLACOOCHEE RIVER SURVEY 1984

Input Parameter or Coefficient	Base Case (Mean) Values	Relative Standard Deviations (%)
Hydraulic Data (7)*		
Flow (cfs)	150-660	3%
Depths (ft)	5.2-14.8	8%
Velocities (fps)	0.12-0.78	8%
Others	a, b	10-20%
Reaction Coefficients (8)		
CBOD Decay (day^{-1})	0.04-0.10	15%
Reaeration (day^{-1})	0.08-0.08	13%
SOD (gO2/ft^2 - day)	0.04-0.13	12%
N, P, Algae	a,b	15-25%
Algae, Nutrient, Light Coefficients (17)		
Maximum Growth Rate (day^{-1})	1.3	10%
Respiration Rate (day^{-1})	0.15	10%
Others	a, b	10%
Climatology, Temperature Inputs (23)		
Wet, Dry Bulb Air Temps (^{0}F)	64.3, 74,5	2%
Temperature Coefficients	1.00-1.083	3%
Others	a, b	1-15%
Headwater, Incremental, Point Loads (27)		
DO, Temperature	a	1-3%
CBOD, N, P, Algae	a	8-25%

(a) Basin-specific values from Koenig, 1986.
(b) Typical values from Table III-3 of Koenig, 1986.

*Value in parentheses is the number of input variables of the type indicated.

TABLE D-2. SUMMARY OF FIRST ORDER SIMULATIONS FOR WITHLACOOCHEE RIVER

(a) Simulation Variable: CBOD (mg/L)

Input	Relative	Sensitivity Coefficient			Components of Variance (%)		
Variable	St Dev (%)	RM26	RM20	RM2	RM26	RM20	RM2
CBOD Decay	15	-0.06(3)[a]	-0.11	-0.22	1	2	8
Incr Flow	3	-0.05	-0.22	-0.37(3)	1	1	1
HW Flow	3	0.05	-0.44(3)	-0.05	1	1	1
HW Temp	1	-0.11(2)	-0.13	-0.16	1	1	1
HW CBOD	15	0.98(1)	0.24	0.18	99	9	6
Ptld Flow	3	0.00	0.67(2)	0.43(2)	0	3	1
Ptld CBOD	15	0.00	0.74(1)	0.69(1)	0	84	79
		Standard Deviation of Simulated (CBOD) (mg/L)			0.35	0.76	0.27
				(%)	15	12	12

(b) Simulation Variable: Dissolved Oxygen

Input Variable	Relative St Dev (%)	RM26	RM20	RM2	RM26	RM20	RM2
Velocity	8	0.03	0.05	-0.26(2)	1	2	13
CBOD Decay	15	-0.02	-0.12	-0.03	1	9	1
SOD	5	-0.05(3)	-0.23	0.09	5	5	3
Reaeration	13	0.04	0.31(3)	0.40(1)	4	45	77
Incr Temp	1	-0.01	-0.15	-0.17(3)	1	1	1
HE Temp	1	-0.25(2)	-0.70(1)	-0.13	1	1	1
HW DO	3	0.92(1)	0.55(2)	0.04	84	8	1
		Standard Deviation of Simulated DO (mg/L)			0.18	0.27	0.30
				(%)	3	6	6

[a] Value in parentheses is rank, with 1 being highest.

TABLE D-3. NORMALIZED SENSITIVITY COEFFICIENTS FOR VARIOUS SIZES OF INPUT PERTURBATIONS (WITHLACOOCHEE RM 20)

(a) Simulation Variable: Dissolved Oxygen (mg/L)

	Magnitude of Input Perturbation %				
Input Variable	**-20%**	**-1%**	**+1%**	**+20%**	**Relative Change (%)**
CBOD	-0.12	-0.12	-0.12	-0.12	0
SOD	-0.23	-0.23	-0.22	-0.23	0
Reaeration	0.33	0.31	0.31	0.30	-9
HW Temp	-0.66	-0.69	-0.69	-0.77	+16
HW DO	0.55	0.55	0.55	0.55	0
Std Dev. (mg/L)	0.28	0.27	0.27	0.26	-7
(b) Simulation Variable: Chlorophyll *a* (µg/L)					
Max Growth Rate	0.40	0.41	0.42	0.43	+7
Respiration	-0.37	-0.36	-0.35	-0.34	-8
Chl *a*/Agy-B	-1.24	-1.01	-0.98	-0.83	-33
HW Flow	0.28	0.24	0.25	0.21	-25
HW Chl-*a*	0.96	0.95	0.96	0.94	-2
Std Dev. (µg/L)	3.72	3.12	3.06	2.64	-29

TABLE D-4. DIFFERENCES IN STANDARD DEVIATION ESTIMATES FOR OUTPUT VARIABLES—WITHLACOOCHEE RIVER SURVEY 1984

Output Variables	Between FOEA Input Perturbations from -20 to +20%	Between FOEA (5%) and Monte Carlo Simulations (2000)
Temperature	5.4	1.8-4.3
Dissolved Oxygen	7.7	0.6-4.5
CBOD	0.8	1.4-2.6
Nitrogen Forms	—[a]	—[a]
Phosphorus Forms	—[a]	—[a]
Chlorophyll *a*	29	16-21
Algal Growth Rate	6.9	2-4

[a] Expected values of standard deviations are too small to compute meaningful relative differences, although values are certainly less than 10% and likely less than 5%.

ard deviation of chlorophyll *a* decreases by 29 percent over the range of input perturbations.

The results of the analysis of the other state variables (Table D-4) show changes in FOEA estimates of standard deviation of about 7 percent for algal growth rate, 5 percent for temperature, and less than 5 percent for all others, including CBOD, the nitrogen forms, and the phosphorus forms. In all cases, the FOEA estimate of standard deviation decreases as the magnitude of the input perturbation increases over the range of -20 to +20 percent.

D.3.3 Monte Carlo Simulations

The Monte Carlo simulation output in QUAL2E-UNCAS provides summary statistics and frequency distributions for the state variables at specific

TABLE D-5. SUMMARY STATISTICS FROM 2000 MONTE CARLO SIMULATIONS FOR WITHLACOOCHEE RIVER

	Dissolved Oxygen (mg/L)			**Chlorophyll *a* (μg/L)**		
Statistic	**RM26**	**RM20**	**RM2**	**RM26**	**RM20**	**RM2**
Calibrated Mean	5.83	4.48	5.06	18.1	14.4	6.6
Simulated Mean	5.82	4.47	5.05	18.9	15.0	6.6
Minimum	5.26	3.47	3.69	10.2	2.8	3.0
Maximum	6.41	5.31	5.89	53.8	41.4	22.2
Range	1.15	1.84	2.20	45.6	33.6	19.2
Std. Deviation	0.18	0.28	0.31	4.25	3.48	1.87
Coef. Variation	3.0%	6.2%	6.2%	23.5%	24.2%	28.4%
Skew Coef.	0.01	-0.15	-0.20	1.73	1.6	1.46
Std. Deviation from FOEA	0.18	0.27	0.30	3.54	2.94	1.62

locations in the basin. Table D-5 contains the summary statistics, based on 2000 Monte Carlo simulations. The same input variances employed in the first-order error analysis were used. Input probability distributions were assumed to be normal.

There is very good agreement between the calibrated mean and simulated mean for dissolved oxygen. For chlorophyll *a* the differences average 3 percent and may be attributed to the nonlinearities. For dissolved oxygen, the standard deviation grows in the downstream direction. This is the result of the fact that dissolved oxygen never recovers to approach saturation, as well as the cumulative effect of model input uncertainty. For chlorophyll *a*, the standard deviation decreases steadily in the downstream direction because the algal biomass concentration is also decreasing. This is the result of a lower algal growth rate due to reduced light penetration caused by color in the industrial waste discharge at RM 24 and due to the dilution effects from groundwater inflow. As shown in Table D-4, for the output variables of temperature, CBOD, and algae growth rate, the Monte Carlo estimate of standard deviation differs by less than 5 percent from the FOEA estimate. These differences are within the 95 percent confidence interval for the Monte Carlo estimates, thus implying negligible nonlinear effects for the conditions of this simulation. The frequency distributions for dissolved oxygen generated by the Monte Carlo analysis are shown graphically in Figure D-2. These distributions are useful in providing a visual representation of the distribution of model output at different locations in the system. In the case of dissolved oxygen shown in Figure B-3, the distributions appear nearly symmetric and the dispersion in the upper reaches of the basin is substantially smaller than that in the middle and lower reaches. Similar plots (not shown) for chlorophyll *a* data in Table D-5 clearly show the decreasing dispersion and pronounced positive skew in the simulated data.

APPENDIX E: SUPPLEMENTAL BIBLIOGRAPHY

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APPENDIX F. GLOSSARY

Activated sludge	A secondary wastewater treatment process that removes organic matter by mixing air and recycled sludge bacteria with sewage to promote decomposition.
Acute toxicity	A chemical stimulus severe enough to rapidly induce an effect; in aquatic toxicity tests, an effect observed within 96 hours or less is considered acute. When referring to aquatic toxicology or human health, an acute effect is not always measured in terms of lethality.
Adsorption-desorption	Adsorption is the process by which nutrients such as inorganic phosphorous adhere to particles via a loose chemical bond with the surface of clay particles. Desorption is the process by which inorganic nutrients are released from the surface of particles back into solution.
Advanced primary treatment	Waste treatment process that incorporates primary sedimentation of suspended solids with chemical addition and flocculation to increase the overall removal of organic solids. Advanced primary treatment typically achieves about 50% removal of suspended solids and BOD.
Advanced secondary treatment	Biological or chemical treatment processes added to a secondary treatment plant including a conventional activated sludge to increase the removal of solids and BOD. Typical removal rates for advanced secondary plants are on the order of 90% removal of solids and BOD.
Advanced waste treatment (AWT)	Wastewater treatment process that includes combinations of physical and chemical operation units designed to remove nutrients, toxic substances, or other pollutants. Advanced, or tertiary, treatment processes treat effluent from secondary treatment facilities using processes such as nutrient removal (nitrification, denitrification), filtration, or carbon adsorption. Tertiary treatment plants typically achieve about 95% removal of solids and BOD in addition to removal of nutrients or other materials.
Advection	Bulk transport of the mass of discrete chemical or biological constituents by fluid flow within a receiving water. Advection describes the mass transport due to the velocity, or flow, of the waterbody.
Aerobic	Environmental conditions characterized by the presence of dissolved oxygen; used to describe biological or chemical processes that occur in the presence of oxygen.
Algae	Any organisms of a group of chiefly aquatic microscopic nonvascular plants; most algae have chlorophyll as the primary pigment for carbon fixation. As primary producers, algae serve as the base of the aquatic food web, providing food for zooplankton and fish resources. An overabundance of algae in natural waters is known as eutrophication.
Algal bloom	Rapidly occurring growth and accumulation of algae within a body of water. It usually results from excessive nutrient loading and/or sluggish circulation regime with a long residence time. Persistent and frequent bloom can result in low oxygen conditions.
Algal growth	Algal growth is related to temperature, available light, and the available abundance of inorganic nutrients (N,P,Si). Algal species groups (e.g., diatoms, greens, etc.) are typically characterized by different maximum growth rates.

Algal respiration	Process of endogenous respiration of algae in which organic carbon biomass is oxidized to carbon dioxide.
Algal settling	Phytoplankton cells (algae) are lost from the water column by physical sedimentation of the cell particles. Algal biomass lost from the water column is then incorporated as sediment organic matter and undergoes bacterial and biochemical reactions releasing nutrients and consuming dissolved oxygen.
Ambient water quality	Natural concentration of water quality constituents prior to mixing of either point or nonpoint source load of contaminants. Reference ambient concentration is used to indicate the concentration of a chemical that will not cause adverse impact to human health.
Ammonia	Inorganic form of nitrogen; product of hydrolysis of organic nitrogen and denitrification. Ammonia is preferentially used by phytoplankton over nitrate for uptake of inorganic nitrogen.
Ammonia toxicity	Under specific conditions of temperature and pH, the un-ionized component of ammonia can be toxic to aquatic life. The un-ionized component of ammonia increases with pH and temperature.
Anaerobic	Environmental condition characterized by zero oxygen levels. Describes biological and chemical processes that occur in the absence of oxygen.
Analytical model	Exact mathematical solution of the differential equation formulation of the transport, diffusion and reactive terms of a water quality model. Analytical solutions of models are often used to check the magnitude of the system response computed using numerical model approximations.
Anoxic	Aquatic environmental conditions containing zero or little dissolved oxygen. See also anaerobic.
Anthropogenic	Pertains to the [environmental] influence of human activities.
Aquatic ecosystem	Complex of biotic and abiotic components of natural waters. The aquatic ecosystem is an ecological unit that includes the physical characteristics (such as flow or velocity and depth), the biological community of the water column and benthos, and the chemical characteristics such as dissolved solids, dissolved oxygen, and nutrients. Both living and nonliving components of the aquatic ecosystem interact and influence the properties and status of each component.
Assimilative capacity	The amount of contaminant load (expressed as mass per unit time) that can be discharged to a specific stream or river without exceeding water quality standards or criteria. Assimilative capacity is used to define the ability of a waterbody to naturally absorb and use waste matter and organic materials without impairing water quality or harming aquatic life.
Attached algae	Photosynthetic organisms that remain in a stationary location by attachment to hard rocky substrate. Attached algae, usually present in shallow hard-bottom environments, can significantly influence nutrient uptake and diurnal oxygen variability.
Autotroph	Organisms that derive cell carbon from carbon dioxide. The conversion of carbon dioxide to organic cell tissue is a reductive process that requires a net input of energy. The energy needed for cell synthesis is provided by either light or chemical oxidation. Autotroph that use light, phototroph, include photosynthetic algae and bacteria. Autotroph that use chemical energy, chemotroph, include nitrifying bacteria.

Background levels	Background levels represent the chemical, physical, and biological conditions that would result from natural geomorphological processes such as weathering or dissolution.
Bacterial decomposition	Breakdown by oxidation, or decay, of organic matter by heterotrophic bacteria. Bacteria use the organic carbon in organic matter as the energy source for cell synthesis.
Benthic	Refers to material, especially sediment, at the bottom of an aquatic ecosystem. It can be used to describe the organisms that live on, or in, the bottom of a waterbody.
Benthic ammonia flux	The decay of organic matter within the sediments of a natural water results in the release of ammonia nitrogen from the interstitial water of sediments to the overlying water column. Benthic release, or regeneration, of ammonia is an essential component of the nitrogen cycle.
Benthic denitrification	Under anaerobic, or low oxygen conditions, denitrifying bacteria synthesize cellular material by reducing nitrate to ammonia and nitrogen gas. Denitrification is a component of the overall nitrogen cycle and has been shown to account for a significant portion of the “new” nitrogen loading to freshwater and estuarine ecosystems.
Benthic nitrification	Under aerobic conditions, nitrifying bacteria synthesize cellular material by oxidizing ammonia to nitrite and nitrate. Benthic nitrification is a component of the overall nitrogen cycle and has been shown to account for a significant portion of the nitrogen budget of shallow freshwater and estuarine ecosystems.
Benthic organisms	Organisms living in, or on, bottom substrates in aquatic ecosystems.
Benthic photosynthesis	Synthesis of cellular carbon by algae attached to the bottom of a natural water system. Benthic photosynthesis typically is limited to shallow waters because of the availability of light at the bottom.
Best management practices (BMPs)	Methods, measures, or practices that are determined to be reasonable and cost-effective means for a land owner to meet certain, generally nonpoint source, pollution control needs. BMPs include structural and nonstructural controls and operation and maintenance procedures.
Biochemical oxygen demand (BOD)	The amount of oxygen per unit volume of water required to bacterially or chemically oxidize (stabilize) the oxidizable matter in water. Biochemical oxygen demand measurements are usually conducted over specific time intervals (5,10,20,30 days). The term BOD generally refers to standard 5-day BOD test.
Biological Nutrient Removal (BNR)	Waste treatment method that employs natural biological processes to reduce the quantity of nitrogen and phosphorus discharged to natural waters. Treatment processes employ the movement of primary effluent through aerobic, anoxic/anaerobic zones to facilitate bacterially mediated processes of nitrification and denitrification.
Biomass	The amount, or weight, of a species, or group of biological organisms, within a specific volume or area of an ecosystem.
Boundary conditions	Values or functions representing the state of a system at its boundary limits.
Calibration	Testing and tuning of a model to a set of field data not used in the development of the model; also includes minimization of deviations between measured field conditions and output of a model by selecting appropriate model coefficients.
Carbonaceous	Pertaining to or containing carbon derived from plant and animal residues

Channel	A natural stream that conveys water; a ditch or channel excavated for the flow of water.
Channel improvement	The improvement of the flow characteristics of a channel by clearing, excavation, realignment, lining, or other means in order to increase its capacity. Sometimes used to connote channel stabilization.
Channel stabilization	Erosion prevention and stabilization of velocity distribution in a channel using jetties, drops, revetments, vegetation, and other measures.
Chloride	An atom of chlorine in solution, bearing a single negative charge.
Chlorophyll	A group of green photosynthetic pigments that occur primarily in the chloroplast of plant cells. The amount of chlorophyll-*a*, a specific pigment, is frequently used as a measure of algal biomass in natural waters.
Chronic toxicity	Toxicity impact that lingers or continues for a relatively long period of time, often one-tenth of the life span or more. Chronic effects could include mortality, reduced growth, or reduced reproduction.
Coliform bacteria	A group of bacteria that normally live within the intestines of mammals, including humans. Coliform bacteria are used as an indicator of the presence of sewage in natural waters.
Combined sewer overflows (CSOs)	A combined sewer carries both wastewater and stormwater runoff. CSOs discharged to receiving water can result in contamination problems that may prevent the attainment of water quality standards.
Complete mixing	No significant difference in concentration of a pollutant exists across the transect of the waterbody.
Concentration	Amount of a substance or material in a given unit volume of solution. Usually measured in milligrams per liter (mg/l) or parts per million (ppm).
Conservative substance	Substance that does not undergo any chemical or biological transformation or degradation in a given ecosystem.
Contamination	Act of polluting or making impure; any indication of chemical, sediment, or biological impurities.
Conventional pollutants	As specified under the Clean Water Act, conventional contaminants include suspended solids, coliform bacteria, biochemical oxygen demand, pH, and oil and grease.
Cross-sectional area	Wet area of a waterbody normal to the longitudinal component of the flow.
Decay	Gradual decrease in the amount of a given substance in a given system due to various sink processes including chemical and biological transformation, dissipation to other environmental media, or deposition into storage areas.
Decomposition	Metabolic breakdown of organic materials; the by-products formation releases energy and simple organics and inorganic compounds. (see also respiration)
Denitrification	Describes the decomposition of ammonia compounds, nitrites, and nitrates (by bacteria) that results in the eventual release of nitrogen gas into the atmosphere.
Design stream flow	The stream flow used to conduct steady-state wasteload allocation modeling.
Designated use	Uses specified in water quality standards for each waterbody or segment regardless of actual attainment.

Detritus	Any loose material produced directly from disintegration processes. Organic detritus consists of material resulting from the decomposition of dead organic remains.
Diagenesis	Production of sediment fluxes as a result of the flux of particulate organic carbon in the sediment and its decomposition. The diagenesis reaction can be thought of as producing oxygen equivalents released by various reduced species.
Dilution	Addition of less concentrated liquid (water) that results in a decrease in the original concentration.
Discharge permits (NPDES)	A permit issued by the U.S. EPA or a State regulatory agency that sets specific limits on the type and amount of pollutants that a municipality or industry can discharge to a receiving water; it also includes a compliance schedule for achieving those limits. It is called the NPDES because the permit process was established under the National Pollutant Discharge Elimination System, under provisions of the Federal Clean Water Act.
Discharge Monitoring Report (DMR)	Report of effluent characteristics submitted by a municipal or industrial facility that has been granted an NPDES discharge permit.
Dispersion	The spreading of chemical or biological constituents, including pollutants, in various directions from a point source, at varying velocities depending on the differential instream flow characteristics.
Dissolved oxygen (DO)	The amount of oxygen that is dissolved in water. It also refers to a measure of the amount of oxygen available for biochemical activity in water body, and as indicator of the quality of that water.
Dissolved oxygen sag	Longitudinal variation of dissolved oxygen representing the oxygen depletion and recovery following a waste load discharge into a receiving water.
Diurnal	Actions or processes having a period or a cycle of approximately one tidal-day or are completed within a 24-hour period and which recur every 24 hours.
Domestic wastewater	Also called sanitary wastewater, consists of wastewater discharged from residences and from commercial, institutional, and similar facilities.
Drainage basin	A part of the land area enclosed by a topographic divide from which direct surface runoff from precipitation normally drains by gravity into a receiving water. Also referred to as watershed, river basin, or hydrologic unit.
Dye study	Use of conservative substances to assess the physical behavior of a natural system to given stimulus.
Dynamic model	A mathematical formulation describing the physical behavior of a system or a process and its temporal variability.
Dynamic simulation	Modeling of the behavior of physical, chemical, and/or biological phenomena and their variation over time.
Ecosystem	An interactive system that includes the organisms of a natural community association together with their abiotic physical, chemical, and geochemical environment.
Effluent	Municipal sewage or industrial liquid waste (untreated, partially treated, or completely treated) that flows out of a treatment plant, septic system, pipe, etc.
Effluent plume	Delineates the extent of contamination in a given medium as a result of effluent discharges (or spills). Usually shows the concentration gradient within the delineated areas or plume.
Epiphyte	A plant growing on another plant; more generally, any organism growing attached on a plant.

Estuary	Brackish-water areas influenced by the tides where the mouth of the river meets the sea.
Estuarine number	Nondimensional parameter accounting for decay, tidal dispersion, and advection velocity. Used for classification of tidal rivers and estuarine systems.
Eutrophication	Enrichment of an aquatic ecosystem with nutrients (nitrates, phosphates) that accelerate biological productivity (growth of algae and weeds) and an undesirable accumulation of algal biomass.
Eutrophication model	Mathematical formulation that describes the advection, dispersion, and biological, chemical, and geochemical reactions that influence the growth and accumulation of algae in aquatic ecosystems. Models of eutrophication typically include one or more species groups of algae, inorganic and organic nutrients (N,P), organic carbon, and dissolved oxygen.
Extinction coefficient	Measure for the reduction (absorption) of light intensity within a water column.
Factor of Safety	Coefficient used to account for uncertainties in representing, simulating, or designing a system.
Fate of pollutants	Physical, chemical, and biological transformation in the nature and changes of the amount of a pollutant in an environmental system. Transformation processes are pollutant specific. However, they have comparable kinetics so that different formulations for each pollutant are not required.
Fecal coliform bacteria	Bacteria that are present in the intestines or feces of warm-blooded animals. They are often used as indicators of the sanitary quality of water. See Coliform bacteria.
First-order kinetics	Describes a reaction in which the rate of transformation of a pollutant is proportional to the amount of that pollutant in the environmental system.
Flocculation	The process by which suspended colloidal or very fine particles are assembled into larger masses or flocules that eventually settle out of suspension.
Flux	Movement and transport of mass of any water quality constituent over a given period of time. Units of mass flux are mass per unit time.
Forcing functions	External empirical formulation used to provide input describing a number of processes. Typical forcing functions include parameters such as temperature, point and tributary sources, solar radiation, and waste loads and flow.
Geochemical	Refers to chemical reactions related to earth materials such as soil, rocks, and water.
Gradient	The rate of decrease (or increase) of one quantity with respect to another; for example, the rate of decrease of temperature with depth in a lake.
Groundwater	Phreatic water or subsurface water in the zone of saturation. Groundwater inflow describes the rate and amount of movement of water from a saturated formation.
Half-saturation constant	Nutrient concentration at which the growth rate is half the maximum rate. Half-saturation constants define the nutrient uptake characteristics of different phytoplankton species. Low half-saturation constants indicate the ability of the algal group to thrive under nutrient-depleted conditions.
Heterotroph	Organisms that use organic carbon for the formation of cell tissue. Bacteria are examples of heterotroph.
Hydrodynamic model	Mathematical formulation used in describing circulation, transport, and deposition processes in receiving water.

Hydrograph	A graph showing variation of in stage (depth) or discharge of water in a stream over a period of time.
Hydrologic cycle	The circuit of water movement from the atmosphere to the earth and return to the atmosphere through various stages or processes, such as precipitation, interception, runoff, infiltration, storage, evaporation, and transpiration.
Hydrolysis	Reactions that occur between chemicals and water molecules resulting in the cleaving of a molecular bond and the formation of new bonds with components of the water molecule.
In situ	In place; in situ measurements consist of measurement of component or processes in a full-scale system or a field rather than in a laboratory.
Initial conditions	A state of a system prior to an introduction of an induced stimulus. Describe conditions at the start-up of system simulations.
Initial mixing zone	Region immediately downstream of an outfall where effluent dilution processes occur. Because of the combined effects of the effluent buoyancy, ambient stratification, and current, the prediction of initial dilution can be involved.
Interstitial water	Water contained in the interstices, which are the pore spaces or voids in soils and rocks.
Kinetic processes	Description of the rate and mode of change in the transformation or degradation of a substance in an ecosystem.
Light saturation	Optimal light level for algae and macrophyte growth and photosynthesis.
Loading, Load, Loading rate	The total amount of material (pollutants) entering the system from one or multiple sources; measured as a rate in weight per unit time.
Load allocation (LA)	The portion of a receiving water's total maximum daily load that is attributed either to one of its existing or future nonpoint sources of pollution or to natural background sources.
Long stream	A receiving water where nutrients are in excess of growth limiting conditions, and where the travel time allows growth and physical accumulation of algal biomass.
Longitudinal dispersion	The spreading of chemical or biological constituents, including pollutants, downstream from a point source at varying velocities due to the differential instream flow characteristics.
Low-flow (7Q10)	Low-flow (7Q10) is the 7-day average low flow occurring once in 10 years; this probability-based statistic is used in determining stream design flow conditions and for evaluating the water quality impact of effluent discharge limits.
Macrophyte	Large vascular rooted aquatic plants.
Margin of Safety (MOS)	A required component of the TMDL that accounts of the uncertainty about the relationship between the pollutant load and the quality of the receiving waterbody.
Mass balance	An equation that accounts for the flux of mass going into a defined area and the flux of mass leaving the defined area. The flux in must equal the flux out.
Mathematical model	A system of mathematical expressions that describe the spatial and temporal distribution of water quality constituents resulting from fluid transport and the one, or more, individual processes and interactions within some prototype aquatic ecosystem. A mathematical water quality model is used as the basis for waste load allocation evaluations.
Mineralization	The transformation of organic matter into a mineral or an inorganic compound.

Mixing characteristics	Refers to the tendency for natural waters to blend; i.e. for dissolved and particulate substances to disperse into adjacent waters.
Monte Carlo simulation	A stochastic modeling technique that involves the random selection of sets of input data for use in repetitive model runs. Probability distributions of receiving water quality concentrations are generated as the output of a Monte Carlo simulation.
N/P ratio	The ratio of nitrogen to phosphorus in an aquatic system. The ratio is used as an indicator of the nutrient limiting conditions for algal growth; also used as indicator for the analysis of trophic levels of receiving waters.
Natural waters	Flowing water within a physical system that has developed without human intervention, in which natural processes continue to take place.
Nitrate (NO_3) and Nitrite (NO_2)	Oxidized nitrogen species. Nitrate is the form of nitrogen preferred by aquatic plants.
Nitrification	The oxidation of ammonium salts to nitrites (via *Nitrosomonas* bacteria) and the further oxidation of nitrite to nitrate via *Nitrobacter* bacteria.
Nitrifier organisms	Bacterial organisms that mediate the biochemical oxidative processes of nitrification.
Nitrobacter	Type of bacteria responsible for the conversion of nitrite to nitrate.
Nitrogenous BOD (NBOD)	Refers to the oxygen demand associated with the oxidation of nitrate.
Nitrosomonas	Type of bacteria responsible for the oxidation of ammonia to the intermediate product nitrite.
Nonconservative substance	Substances that undergo chemical or biological transformation in a given environment.
Nonpoint source	Pollution that is not released through pipes but rather originates from multiple sources over a relatively a large area. Nonpoint source can be divided into source activities related to either land or water use including failing septic tanks, improper animal-keeping practices, forest practices, and urban and rural runoff.
Numerical model	Models that approximate a solution of governing partial differential equations which describe a natural process. The approximation uses a numerical discretization of the space and time components of the system or process.
Nutrient	A primary element necessary for the growth of living organisms. Carbon dioxide, nitrogen, and phosphorus, for example, are required nutrients for phytoplankton growth.
Nutrient limitation	Deficit of nutrient (e.g., nitrogen and phosphorus) required by microorganisms in order to metabolize organic substrates.
One-dimensional model (1-D)	A mathematical model defined along one spatial coordinate of a natural water system. Typically 1-D models are used to describe the longitudinal variation of water quality constituents along the downstream direction of a stream or river. In writing the model, it is assumed that the cross-channel (lateral) and vertical variability is relatively homogenous and can, therefore, be averaged over those spatial coordinates.
Organic matter	The organic fraction that includes plant and animal residue at various stages of decomposition, cells and tissues of soil organisms, and substance synthesized by the soil population. Commonly determined as the amount of organic material contained in a soil or water sample.
Organic nitrogen	Form of nitrogen bound to an organic compound.

Orthophosphate (O_PO4_P)	Form of phosphate available for biological metabolism without further breakdown.
Outfall	Point where water flows from a conduit, stream, or drain.
Oxidation	The chemical union of oxygen with metals or organic compounds accompanied by a removal of hydrogen or another atom. It is an important factor for soil formation and permits the release of energy from cellular fuels.
Oxygen demand	Measure of the dissolved oxygen used by a system (microorganisms) in the oxidation of organic matter. See also biochemical oxygen demand.
Oxygen depletion	Deficit of dissolved oxygen in a water system due to oxidation of organic matter.
Oxygen saturation	Natural or artificial reaeration or oxygenation of a water system (water sample) to bring the level of dissolved oxygen to saturation. Oxygen saturation is greatly influence by temperature and other water characteristics.
Partition coefficients	Chemicals in solution are partitioned into dissolved and particulate adsorbed phase based on their corresponding sediment-to-water partitioning coefficient.
Peak runoff	The highest value of the stage or discharge attained by a flood or storm event, also referred to as flood peak or peak discharge.
Periphyton	Attached benthic algae.
Photoperiod	Time period of the seasonal response by organisms to change in the length of the daylight period (e.g., flowering, germination of seeds, reproduction, migration, and diapause are frequently under photoperiod control).
Photosynthesis	The biochemical synthesis of carbohydrate based organic compounds from water and carbon dioxide using light energy in the presence of chlorophyll. Photosynthesis occurs in all plants, including aquatic organisms such as algae and macrophyte. Photosynthesis also occurs in primitive bacteria such as blue-green algae.
Phyla	Species groups of same family of organisms. Phyla of phytoplankton include diatoms, blue-green algae, dinoflagellates, and green algae.
Phytoplankton	A group of generally unicellular microscopic plants characterized by passive drifting within the water column. See Algae.
Plankton	Group of generally microscopic plants and animals passively floating, drifting or swimming weakly. Plankton include the phytoplankton (plants) and zooplankton (animals).
Point source	Pollutant loads discharged at a specific location from pipes, outfalls, and conveyance channels from either municipal wastewater treatment plants or industrial waste treatment facilities. Point sources can also include pollutant loads contributed by tributaries to the main receiving water stream or river.
Pollutant	A contaminant in a concentration or amount that adversely alters the physical, chemical, or biological properties of a natural environment. The term include pathogens, toxic metals, carcinogens, oxygen demanding substances, or other harmful substances. Examples of pollutant sources include dredged spoil, solid waste, incinerator residue, sewage, garbage, sewage sludge, munitions, chemical waste, biological material, radioactive materials, heat, wrecked or discharged equipment, sediment, cellar dirt, hydrocarbons, oil, and municipal, industrial, and agricultural waste discharged into surface water or groundwater.
Postaudit	A subsequent examination and verification of model predictive performance following implementation of an environmental control program.

Pretreatment	The treatment of wastewater to remove or reduce contaminants prior to discharge into another treatment system or a receiving water.
Primary productivity	A measure of the rate at which new organic matter is formed and accumulated through photosynthesis and chemosynthesis activity of producer organisms (chiefly, green plants). The rate of primary production is estimated by measuring the amount of oxygen released (oxygen method) or the amount of carbon assimilated by the plant (carbon method)
Primary treatment plant	Wastewater treatment process where solids are removed from raw sewage primarily by physical settling. The process typically removes about 25-35% of solids and related organic matter (BOD_5).
Priority pollutant	Substances listed by the U.S. EPA under the Federal Clean Water Act as harmful substances and having priority for regulatory controls. The list includes metals (13), inorganic compounds (2), and a broad range of naturally occurring or artificial organic compounds (111).
Publicly Owned Treatment Works (POTW)	Municipal wastewater treatment plant owned and operated by a public governmental entity such as a town or city.
Raw sewage	Untreated municipal sewage.
Reaction rate coefficient	Coefficient describing the rate of transformation of a substance in an environmental medium characterized by a set of physical, chemical, and biological conditions such as temperature and dissolved oxygen level.
Reaeration	Describe the net flux of oxygen occurring from the atmosphere to a body of water with a free surface.
Receiving waters	Creeks, streams, rivers, lakes, estuaries, groundwater formations, or other bodies of water into which surface water and/or treated or untreated waste are discharged, either naturally or in man-made systems.
Refractory organics	A broad lumping of man-made organic chemicals that resist chemical or bacterial decomposition, including many pesticides, herbicides, household and industrial cleaners and solvents, photofinishing chemicals, and dry-cleaning fluids.
Reserve capacity	Pollutant loading rate set aside in determining stream waste load allocation accounting for uncertainty and future growth.
Residence time	Length of time that a pollutant remains within a section of a stream or river. The residence time is determined by the streamflow and the volume of the river reach or the average stream velocity and the length of the river reach.
Respiration	Biochemical process by means of which cellular fuels are oxidized with the aid of oxygen to permit the release of the energy required to sustain life; during respiration oxygen is consumed and carbon dioxide is released.
Rotating biological contactors (RBCs)	A wastewater treatment process consisting of a series of closely spaced rotating circular disks of polystyrene or polyvinyl chloride. Attached biological growth is promoted on the surface of the disks. The rotation of the disks allows contact with the wastewater and the atmosphere to enhance oxygenation.
Roughness Coefficient	A factor in velocity and discharge formulas representing the effects of channel roughness on energy losses in flowing water. Manning’s “n” is a commonly used roughness coefficient.
Scour	To abrade and wear away. Used to describe the weathering away of a terrace or diversion channel or streambed. The clearing and digging action of flowing water,

	especially the downward erosion by stream water in sweeping away mud and silt on the outside of a meander or during flood events.
Secchi depth	A measure of the light penetration into the water column. Light penetration is influenced by turbidity.
Secondary treatment plant	Waste treatment process where oxygen-demanding organic materials (BOD) are removed by bacterial oxidation of the waste to carbon dioxide and water. Bacterial synthesis of wastewater is enhanced by injection of oxygen.
Sediment	Particulate organic and inorganic matter that accumulates in a loose, unconsolidated form on the bottom of natural waters.
Sediment oxygen demand (SOD)	The solids discharged to a receiving water are partly organics, and upon settling to the bottom, they decompose anaerobically as well as aerobically, depending on conditions. The oxygen consumed in aerobic decomposition represents another dissolved oxygen sink for the waterbody.
Sedimentation	Process of deposition of waterborne or windborne sediment or other material; also refers to the infilling of bottom substrate in a waterbody by sediment (siltation).
Short stream	A receiving water where nutrients are in excess of growth-limiting conditions and where the time of travel within the stream reach is not sufficient to allow growth and physical accumulation of algal biomass.
Simulation	Refers to the use of mathematical models to approximate the observed behavior of a natural water system in response to a specific known set of input and forcing conditions. Models that have been validated, or verified, are then used to predict the response of a natural water system to changes in the input or forcing conditions.
Sorption	The adherence of ions or molecules in a gas or liquid to the surface of a solid particle with which they are in contact.
Spatial segmentation	A numerical discretization of the spatial component of a system into one or more dimensions; forms the basis for application of numerical simulation models.
Stabilization pond	Large earthen basins that are used for the treatment of wastewater by natural processes involving the use of both algae and bacteria.
Steady-state model	Mathematical model of fate and transport that uses constant values of input variables to predict constant values of receiving water quality concentrations.
Stoichiometric ratio	Mass-balance-based ratio for nutrients, organic carbon and algae (e.g., nitrogen-to-carbon ratio).
STORET	U.S. Environmental Protection Agency (EPA) national water quality database for STORage and RETrieval (STORET). Mainframe water quality database that includes physical, chemical, and biological data measured in waterbodies throughout the United States.
Storm runoff	Rainfall that does not evaporate or infiltrate the ground because of impervious land surfaces or a soil infiltration rate lower than rainfall intensity, but instead flows onto adjacent land or waterbodies or is routed into a drain or sewer system.
Stratification (of water body)	Formation of water layers each with specific physical, chemical, and biological characteristics. As the density of water decreases due to surface heating, a stable situation develops with lighter water overlaying heavier and denser water.
Streamflow	Discharge that occurs in a natural channel. Although the term “discharge” can be applied to the flow of a canal, the word “streamflow” uniquely describes the discharge in a surface stream course. The term streamflow is more general than “runoff” as

	streamflow may be applied to discharge whether or not it is affected by diversion or regulation.
Substrate	Refers to bottom sediment material in a natural water system.
Surface waters	Water that is present above the substrate or soil surface. Usually refers to natural waterbodies such as rivers, lakes and impoundments, and estuaries.
Suspended solids or load	Organic and inorganic particles (sediment) suspended in and carried by a fluid (water). The suspension is governed by the upward components of turbulence, currents, or colloidal suspension.
Temperature coefficient	Rate of increase in an activity or process over a 10 degree Celsius increase in temperature. Also referred to as the Q_{10}.
Tertiary treatment	Waste treatment processes designed to remove or alter the forms of nitrogen or phosphorus compounds contained in domestic sewage.
Three-dimensional model (3-D)	Mathematical model defined along three spatial coordinates where the water quality constituents are considered to vary over all three spatial coordinates of length, width, and depth.
Total Kjeldahl Nitrogen (TKN)	The total of organic and ammonia nitrogen in a sample, determined by the Kjeldahl method.
Total Maximum Daily Load (TMDL)	The sum of the individual wasteload allocations and load allocations. A margin of safety is included with the two types of allocations so that any additional loading, regardless of source, would not produce a violation of water quality standards.
Total coliform bacteria	A particular group of bacteria that are used as indicators of possible sewage pollution. They are characterized as aerobic or facultative anaerobic, gram-negative, nonspore-forming, rod-shaded bacteria which ferment lactose with gas formation within 48 hours at 35 degrees Celsius. (See also fecal coliform bacteria)
Toxic substances	Those chemical substances, such as pesticides, plastics, heavy metals, detergent, solvent, or any other material that are poisonous, carcinogenic, or otherwise directly harmful to human health and the environment.
Transit time	In nutrient cycles, average time that a substance remains in a particular form; ratio of biomass to productivity.
Transport of pollutants (in water)	Transport of pollutants in water involves two main process: (1) advection, resulting from the flow of water, and (2) diffusion, or transport due to turbulence in the water.
Travel time	Time period required by a particle to cross a transport route such as a watershed, river system, or stream reach.
Tributary	A lower order stream compared to a receiving waterbody. “Tributary to” indicates the largest stream into which the reported stream or tributary flows.
Trickling filter	A wastewater treatment process consisting of a bed of highly permeable medium to which microorganisms are attached and through which wastewater is percolated or trickled.
Turbidity	Measure of the amount of suspended material in water.
Turbulent flow	A flow characterized by irregular, random-velocity fluctuations.
Turbulence	A type of flow in which any particle may move in any direction with respect to any other particle and in a regular or fixed path. Turbulent water is agitated by cross current and eddies. Turbulent velocity is that velocity above which turbulent flow will always exist and below which the flow may be either turbulent or laminar.

Two-dimensional model (2-D)	Mathematical model defined along two spatial coordinates where the water quality constituents are considered averaged over the third remaining spatial coordinate. Examples of 2-D models include descriptions of the variability of water quality properties along: (a) the length and width of a river that incorporates vertical averaging or (b) length and depth of a river that incorporates lateral averaging across the width of the waterbody.
Ultimate Biochemical Oxygen Demand (UBOD or BOD$_U$)	Longterm oxygen demand required to completely stabilize organic carbon in wastewater or natural waters.
Uncertainty factors	Factors used in the adjustment of toxicity data to account for unknown variations. Where toxicity is measured on only one test species, other species may exhibit more sensitivity to that effluent. An uncertainty factor would adjust measured toxicity upward and downward to cover the sensitivity range of other, potentially more or less sensitive species.
Unstratified	Indicates a vertically uniform or well-mixed condition in a waterbody. See also stratified.
Verification (of a model)	Subsequent testing of a precalibrated model to additional field data usually under different external conditions to further examine model validity (also called validation).
Volatilization	Process by which chemical compounds are vaporized (evaporated) at given temperature and pressure conditions by gas transfer reactions. Volatile compounds have a tendency to partition into the gas phase.
Waste load allocation (WLA)	The portion of a receiving water's total maximum daily load that is allocated to one of its existing or future point sources of pollution.
Wastewater	Usually refers to effluent from a sewage treatment plant. See also domestic wastewater.
Wastewater treatment	Chemical, biological, and mechanical procedures applied to an industrial or municipal discharge or to any other sources of contaminated water in order to remove, reduce, or neutralize contaminants.
Water quality	The biological, chemical, and physical conditions of a water body. It is a measure of a water body to support beneficial uses.
Water quality criteria (WQC)	Water quality criteria comprised numeric and narrative criteria. Numeric criteria are scientifically derived ambient concentrations developed by EPA or States for various pollutants of concern to protect human health and aquatic life. Narrative criteria are statements that describe the desired water quality goal.
Water quality standard (WQS)	A water quality standard is a law or regulation that consists of the beneficial designated use or uses of a waterbody, the numeric and narrative water quality criteria that are necessary to protect the use or uses of that particular waterbody, and an antidegradation statement.
Wind mixing	Refers to a physical process occurring when wind over a free water surface influences the atmospheric reaeration rate.
Zero-order kinetics	Describe the rate of transformation or degradation of a substance; the reaction rate of change is independent of the concentrations in solution.
Zooplankton	Very small animals (protozoans, crustaceans, fish embryos, insect larvae) that live in a waterbody and are moved passively by water currents and wave action.

APPENDIX G. ABBREVIATIONS

AGP	algal growth potential
ARM	agricultural runoff model
ASCII	American Standard Code for Information Interchange
AT	advanced treatment
AWT	advanced water treatment
BOD	biochemical oxygen demand
BOD_5	5-day biochemical oxygen demand
BOD_u	ultimate biochemical oxygen demand
BMPs	best management practices
BNR	biological nutrient removal
CBOD	carbonaceous biochemical oxygen demand
$CBOD_5$	5-day carbonaceous biochemical oxygen demand
CEAM	Center for Exposure Assessment Modeling (EPA)
CE-QUAL-RIV1	fully dynamic one-dimensional riverine water quality model
Chl	chlorophyll concentration
COD	chemical oxygen demand
COE	U.S. Army Corps of Engineers
CSO	combined sewer overflow
DIN	inorganic nitrogen concentration (sum of ammonia, nitrate, and nitrite)
DIP	dissolved inorganic phosphorus concentration
DMR	discharge monitoring report
DO	dissolved oxygen
DYNHYD5	hydrodynamic model; a submodel of WASP5
EUTRO5	eutrophication/dissolved oxygen kinetics; a submodel of WASP5
EPA	Environmental Protection Agency
EPA STORET	U.S. Environmental Protection Agency (EPA) national water quality data base for STORage and RETrieval (STORET). Mainframe water quality data base that includes physical, chemical, and biological data measured in waterbodies throughout the United States
FOIA	Freedom of Information Act
FOEA	first-order error analysis
FORTRAN	FORmula TRANslation; revised high-level programming language for solving problems in science and engineering
FORTRAN77	FORmula TRANslation ANSI Standard of 1977; computer language often used in writing scientific equations and models as source code for water quality and hydrodynamic models.
H	surface-to-bottom depth of the water column; units of length
HSPF	Hydrologic Simulation Program - FORTRAN
HRAS	high-rate activated sludge
LA	load allocation
MBAS	methyl benzene alkyl sulfonate
MGD	million gallons per day
Multi-SMP	Simplified Method Program for multiple dischargers
NBOD	nitrogenous biochemical oxygen demand
NPDES	National Pollutant Discharge Elimination System
NTIS	National Technical Information Service
NTU	nephelometry turbidity units
NVSS	nonvolatile suspended solids concentration
O-PO4-P	orthophosphate
P	average gross photosynthesis production
PC	personal computer; usually refers to IBM DOS-compatible machines
PCS	Permit Compliance System
PDF	probability density function
POTW	publicly owned treatment works
P/R	production/respiration ratio
P-R	photosynthesis and respiration
Q	streamflow; units of volume/time
QA/QC	quality assurance/quality control
QUAL2E	stream water quality model
QUAL2E-UNCAS	stream water quality model
R	average respiration
RIVMOD	numerical, hydrodynamic, and sediment transport riverine model
RIV1H	hydrodynamic model; a submodel of CE-QUAL-RIV1
RIV1Q	water quality model; a submodel of CE-QUAL-RIV1
SOD	sediment oxygen demand
STP	sewage treatment plant
TBOD	total biochemical oxygen demand
TDS	total dissolved solids
TKN	total Kjedahl nitrogen
TMDL	total maximum daily load
TOC	total organic carbon
TOXI5	toxic chemical-sediment dynamics; a submodel of WASP5
TP	total phosphorus; sum of all forms of phosphorus: dissolved, particulate, inorganic, and organic phosphorus
TSS	total suspended solids
USGS	U.S. Geological Survey
VSS	detritus concentration
W	width across a stream channel; units of length
WASP5	Water Quality Analysis Simulation Program
WLA	waste load allocation
1-D	one-dimensional water quality model
7Q10	7-day average low flow that occurs once in 10 years

APPENDIX H: CONVERSION FACTORS

For readers who prefer to use U.S. equivalents of metric units, conversion factors for terms used in this report are listed below:

Multiply	By	To Obtain
	Length	
milimeter (mm)	0.03937	inch (in)
Meter (m)	3.281 1.094	foot (ft) yard (yd)
kilometer (km)	0.6214 0.5400	mile (mi) nautical mile (nmi)
meter per second (m/s)	3.281	foot per second (ft/s)
nanometer (nm)	3.937 x 10^{-8}	inch (in)
centimeter (cm)	0.3937	inch (in)
	Area	
square meter (m^2)	10.76 1.196	square foot (ft^2) square yard (yd^2)
	Volume	
cubic meter (m^3)	35.31 1.308	cubic foot (ft^3) cubic yard (yd^3)
liter (L)	1.057	quart (qt)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
	Mass	
millligram (mg)	0.00003527	ounce (oz)
gram (g)	0.03527 0.002205	ounce (oz)
kilogram (kg)	2,205	pound (lb)
metric ton (Mt) (1000 kg)	1.102	ton (short)
gram per square meter (g/m^2)	8.922	pound per acre (lb/acre)
	Temperature	
degree Celsius (^{o}C)	1.8 [^{o}C] +32	degree Fahrenheit (^{o}F)
	Concentration	
milligram per liter (g/L)	1.0	parts per million (ppm)
grams per liter (mg/L)	1.0	parts per thousand (ppt)
	Energy	
langley (ly)	1.0	calorie/square centimeter (cal/cm^2)
calorie/square centimeter day	3.6867	British thermal units/square foor/day (Btu/ft^2/day)

Symbol	Meaning	Conversion
PAR	Photosythetically active radiation (400-700 nanometer waveband). Measured in microeinsteins per square meter per second [µE/m^2/s]	1 watt m^{-2}~4.6µE$m^{-2}s^{-1}$ 1 ly day^{-1}~0.485 watt m^{-2}
µmho	Conductance in micromhos. A measure of the amount of dissolved ions present in water	1 part per thousand is approximately 1,500 µmho at 25^{o}C

APPENDIX I. SYMBOLS

SYMBOL	DESCRIPTION (dimension)
a_N	nitrogen:chlorophyll ratio (MM^{-1})
a_P	phosphorus:chlorophyll ratio (MM^{-1})
A	chlorophyll *a* concentration (ML^{-3})
A_g	algal biomass concentration (ML^{-3})
b_n	periodic coefficient
C	dissolved oxygen concentration in stream (ML^{-3})
C_s	saturation concentration of dissolved oxygen (ML^{-3})
C(t)	time varying oxygen value (ML^{-3})
dV	volume of the segment and is equal to $A_{\Delta x}$ (L^{-3})
Dp	phytoplankton death rate (T^{-1})
D_x	longitudinal dispersion coefficient (L^2T^{-1})
D_z	death rate (T^{-1})
f	photoperiod (T)
F	fraction of algal nitrogen uptake from ammonia pool
g	gravitational constant ($L^2 T^{-1}$)
G_{max}	maximum growth rate (T^{-1})
G_n	phytoplankton net growth rate (T^{-1})
G_p	phytoplankton growth rate (T^{-1})
G_T	temperature effect (T^{-1})
H	average depth (L)
K	first-order reaction rate (T^{-1})
K_a	stream reaeration rate coefficient (T^{-1}); atmospheric reaeration rate: reflects first-order reaction whereby fraction of oxygen deficit is satisfied
K_d	BOD oxidation rate where oxidation accounts for all CBOD removal (T^{-1})
K_e	extinction coefficient (L^{-1})
K_m	half saturation (Michaelis) constant (ML^{-3})
K_{mn}	Michaelis-Menton constant for nitrogen (ML^{-3})
K_{mp}	Michaelis-Menton constant for phosphorus (ML^{-3})
K_n	nitrification reaction rate (T^{-1})
K_r	$CBOD_5$ removal rate in the stream (T^{-1})
K_s	effective loss rate due to settling (T^{-1})
K_{si}	Michaelis-Menton constant for silica (ML^{-3})
K_1	BOD reaction rate (T^{-1})
I_a	average of incident light on water surface over a 24-hour period (ly/day)
I_s	saturating light intensity (ly/day)
I_T	total daily radiation (ly)
L	oxygen equivalence of the organic matter remaining CBOD concentration (ML^{-3}); length; liters
L_o	total oxygen demand (ML^{-3})
ly	langley (incident light intensity)
n	estuary number (dimensionless)
N	nitrogen concentration (ML^{-3})
Nut	nutrient concentration (ML^{-3})
N_1	ammonia concentration (ML^{-3})
N_2	nitrite-nitrogen concentration (ML^{-3})
N_3	nitrate-nitrogen concentration (ML^{-3})
N_4	organic nitrogen concentration (ML^{-3})

SYMBOL	DESCRIPTION (dimension)
P	average gross photosynthesis production ($ML^{-3} T^{-1}$); phosphorus concentration (ML^{-3})
P_{av}	average daily rate of photosynthetic oxygen production ($ML^{-3} T^{-1}$)
P_M	maximum rate of photosynthetic oxygen production ($ML^{-3} T^{-1}$)
P(t)	algal gross photosynthetic production of oxygen ($ML^{-3}T^{-1}$)
P_1	organic phosphorus concentration (ML^{-3})
P_2	dissolved phosphorus concentration (ML^{-3})
Q	river flow rate (L^3T^{-1})
r_L	light effect (dimensionless)
r_n	nutrient effect (dimensionless); limiting nutrient reduction factor
R	average algal oxygen respiration ($ML^{-3}T^{-1}$)
S	net settling rate (T^{-1}); stream slope (LL^{-1})
S_b	sediment oxygen demand ($ML^{-2}T^{-1}$)
S_i	dissolved inorganic silica concentration (ML^{-3})
t	time (T)
t^*	travel time in stream ; =x/U (T)
T	temperature (°); average time period
U	average stream velocity (LT^{-1})
U^*	shear velocity (LT^{-1})
V_s	phytoplankton settling velocity (MT^{-1})
W	direct loading rate (MT^{-1}); stream width (L)
x	distance downstream of effluent (L)
y	oxygen consumed (ML^{-3})
α_1	fraction of algal biomass that is nitrogen (MM-1)
α_2	fraction of algal biomass that is phosphorus (MM^{-1})
α_3	the stoichiometric ratio of oxygen production per unit of algal photosynthesis (MM-1)
α_4	the stoichiometric ratio of oxygen uptake per unit of algae respired (MM-1)
α_5	the stoichiometric ratio of oxygen uptake per unit of ammonium (MM-1)
α_6	the stoichiometric ratio of oxygen uptake per unit of nitrite-nitrogen oxidation (MM-1)
ß$_1$	ammonia oxidation rate coefficient (T^{-1})
ß$_2$	nitrite oxidation rate coefficient (T^{-1})
ß$_3$	organic nitrogen hydrolysis rate coefficient (T^{-1})
ß$_4$	organic phosphorus decay rate (T^{-1})
σ_2	benthos source rate for dissolved phosphorus ($ML^{-2}T^{-1}$)
σ_3	benthos source rate for ammonia nitrogen ($ML^{-2}T^{-1}$)
σ_4	rate coefficient for organic nitrogen settling (T -1)
σ_5	rate coefficient for organic phosphorus settling (T^{-1})
μ	algal growth rate coefficient (T^{-1})
R	algal respiration rate coefficient (T^{-1})
θ	constant for temperature adjustment (dimensionless)

Dimension codes:

L=Length M=Mass T=Time

APPENDIX J. BOD-DO-Nutrient Guidance Input files for QUAL2E and WASP5-EUTRO5 example problems, Diskette EPA 823-C-95-004. (Attached to back cover)

DISCLAIMER