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EVALUATION OF SELECTED ONE-DIMENSIONAL STREAM WATER-QUALITY MODELS WITH FIELD DATA

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Abstract

An evaluation of the U.S. Geological Survey One-Dimensional Steady-State Stream Water-Quality Model (a modified Streeter-Phelps model), the QUAL II model (Southeast Michigan Council of Governments version), and the Water Quality for River-Reservoir Systems model indicated that the models were of comparable accuracy and performed according to the documentation for each of these readily available models. The evaluation was based on a wide range of accurate steady-state data collected on the Chattahoochee River in Georgia, Willamette River in Oregon, and Arkansas River in Colorado.

A number of differences existed between these three models. However, each model has the flexibility that makes these differences relatively unimportant for typical water-quality studies. In addition to some differences in the formulation, each model had minor coding errors which have been corrected.

Modeling capabilities were summarized in tabular form to assist with the selection of models to simulate stream water-quality downstream of reservoirs. While the modified Streeter-Phelps and QUAL II models are equally valid, different modeling options may make one model preferable depending on the specific modeling application. The Water Quality for River-Reservoir Systems model is best limited to dynamic flow and water-quality modeling because data coding is tedious and involved. However, the wide range of components in that model may be needed for steady-state modeling under special conditions.

The capabilities of the MIT Transient Water Quality Network Model are also summarized for comparison but that model could not be implemented because of program problems. Pending a review and update, that model should not be used by an inexperienced user.

The Velz rational technique was not fully evaluated to determine the usefulness of the documentation or the ease-of-use but the selection of the Chattahoochee and Willamette river data made it possible to determine that the technique was of comparable accuracy.

PREFACE

The study described in part by this report has been undertaken by the U.S. Department of the Interior, Geological Survey, at the Gulf Coast Hydroscience Center (GCHC*). The purpose of the study was to evaluate and compare one-dimensional stream water-quality models. Funding was provided by the U.S. Army Corps of Engineers (COE) Environmental and Water Quality Operational Studies (EWQOS) Program through the U.S. Army Waterways Experiment Station (WES) by Interagency transfer WESRF 80-97 dated 7 November 1979. The study is part of EWQOS Task IC.3, Improve and Verify Riverine Water Quality and Ecological Predictive Techniques. The EWQOS Program is sponsored by the Office, Chief of Engineers, and is assigned to the WES, under the purview of the Environmental Laboratory (EL).

Dr. S. C. McCutcheon served as principal investigator, with the technical and administrative support of Mr. Marshall Jennings. Dr. Robert Baker, Chief of GCHC, provided general administrative support. Doyle Frederick, Acting Director of the Geological Survey, approved the publication of this report. Technical assistance at GCHC was provided by Mr. Harry Doyle, Hydrologist; Mr. Philip Curwick, Hydrologist; Miss Kathleen Flynn, Computer Specialist; Mrs. Joy Lorens, Computer Specialist; and Miss Leslie Hallman, Mr. Kenneth Burton, Mr. Alan Guess, Miss Rebecca Breeland, Mr. James Gibson, and Miss Cynthia Faulk, co-op students. The report was written by Dr. McCutcheon.

Three Geological Survey offices, the Georgia District Office, the Oregon District Office, and the Pueblo, Colorado, Subdistrict Office provided data used in the study. Mr. Robert Faye furnished information about the Chattahoochee River, Georgia. Messrs. Frank Rinella and Stuart McKenzie provided information about the Willamette River, Oregon. Messrs. Douglas Cain, Kimball Goddard, and Ronnie Steger furnished information about the Arkansas River, Colorado.

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* For convenience, abbreviations are listed and defined in the Notation (Appendix A).

Mr. Rich Johnson, U.S. Army Corps of Engineers, Portland, Oregon, Mr. Naresh Varma, James M. Montgomery Consulting Engineers, Inc., and Mr. Glen Dearth, CH2M Hill, Inc., furnished cross-section geometry data describing the channel of the Willamette River.

Mr. R. G. Willey and Dr. Michael Gee, U.S. Army Hydrologic Engineering Center (HEC), gave advice and assistance in the use of the Water Quality for River-Reservoir Systems Model. Mr. Michael Mullen and Dr. Frank Tatom of Engineering Analysis, Inc., provided advice and information concerning the Massachusetts Institute of Technology (MIT) Transient Water-Quality Network Model.

The study was conducted under the direct WES supervision of Dr. D. E. Ford and Mr. Aaron Stein and under the general supervision of Mr. D. L. Robey, Chief, Ecosystem Research and Simulation Division, Dr. J. Harrison, Chief, EL, and Dr. J. L. Mahloch, the EWQOS Program Manager.

Commanders and Directors of WES during the conduct of this study were COL N. P. Conover, CE, and COL T. C. Creel, CE. Technical Director was Mr. F. R. Brown.

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PART I: INTRODUCTION

EWQOS and Stream Water Quality

The WES has recognized the need to predict stream water quality downstream of reservoirs in order to derive the greatest benefits from reservoirs and the river downstream of reservoirs. To address this need, a component of the EWQOS program was designed to evaluate the four most likely stream water-quality models (digital computer programs). Because the U.S. Geological Survey (USGS) has an active interest in stream water-quality modeling and data collection, the GCHC and WES agreed to cooperate in an evaluation of stream water-quality models. Data collected by the USGS in several river basin studies made it possible to evaluate stream water-quality models using a wide range of field conditions.

The prediction of stream water-quality using mathematical equations can be traced at least as far back as the work of Streeter and Phelps in the 1920's. Since that time, predictive techniques have been improved and refined. The advent of practical digital computers in the early 1960's led to a proliferation of computer models describing physical systems that included a number of stream water-quality models. Water-quality modeling has improved to the point that these models are useful tools in understanding and predicting physical, chemical, and biological interactions occurring in streams.

The existence of numerous useful models for stream water-quality analysis makes it difficult to match the appropriate model to stream conditions for the purpose of accurately modeling stream water quality. To provide some guidance in model selection, four representative models were chosen for examination in this study.

Project Goals and Scope

This study was undertaken to examine four models, briefly review the literature concerning one-dimensional water-quality models, select a data base to be used to evaluate the models, and assess the need for further study.

Models included in this evaluation and comparison were a modified Streeter-Phelps model entitled the "Steady-State Stream Water-Quality Model," the Southeast Michigan Council of Governments (SEMCOG) version of the QUAL II model, the "Water Quality for River-Reservoir Systems" (WQRRS) model, and the "MIT Transient Water-Quality Network Model." The USGS version of the Streeter-Phelps model (referred to hereafter as the Streeter-Phelps model) and the QUAL II model were designed to predict water quality under conditions of steady flow and waste loading. The QUAL II model has the capability to predict time-varying concentrations of temperature, dissolved oxygen (DO), chlorophyll a, and nutrients in response to dynamic meteorological conditions and steady flow. The WQRRS and MIT models are dynamic models. Both were designed to predict time-varying stage, flow, and water quality.

Originally, a fifth model entitled "USGS Transient Model" was considered. After a brief review indicated that the model was not widely used, this model was dropped from consideration so that more time could be devoted to the other four models.

A brief literature review was aimed at confirming that the four models mentioned above were state-of-the-art or that the models had been used frequently under a variety of conditions and a general perception existed that these models were useful and valid. Other goals were to locate the most accurate set of steady-state data and confirm that a paucity of dynamic water-quality data existed.

To assist in the model evaluation, three USGS data sets were selected from steady-state water-quality studies in which flow and water quality in the stream were essentially constant. The first set was collected during the Chattahoochee River quality assessment in Georgia. The second set was collected during the Willamette River quality assessment in Oregon. The third data set was collected during a study of the Arkansas River in Colorado by the USGS for the Pueblo County, Colorado, Council of Governments.

These data sets cover a wide range of steady-state stream water-quality conditions. The Willamette River is a large sluggish stream that has three distinctly different reaches. The Chattahoochee River is of moderate size with moderate bottom slopes. The upper Arkansas River is a small stream with a high channel slope. Each stream was studied to determine the effects of point source and nonpoint source pollution associated with urbanization.

Each stream was characterized by different critical low-flow conditions. The Willamette River typically reaches a steady low flow in late summer and maintains it for about two months. The Chattahoochee River is regulated by an upstream peaking-power dam such that periods of steady low flow are normally limited to late summer weekends. The upper Arkansas River has two periods of steady low flow: one in April before the annual snowmelt and one afterwards from August to September.

Besides choosing data to cover a wide range of conditions, the data were also chosen so that independent determinations of some model coefficients were possible. In addition, the data were checked for accuracy and precision. Questionable data were labeled in the results or removed.

Each of the three data sets consisted of at least two independent subsets. One subset of data was used in calibrating the models in which model coefficients were adjusted so that model simulations matched water-quality measurements contained in the calibration data. Since the process of adjusting coefficients was an empirical process, a second subset of data was necessary to verify the calibration. The model results were compared to the independent subset of verification data without modifying the model coefficients to determine whether or not a model would adequately simulate water quality in a given stream.

The models included in the evaluation were first examined by reviewing the documentation of each model in order to summarize the conditions the models were designed to simulate and the capabilities of each model. During the application of each model to the data, as many options were used as time permitted. The Streeter-Phelps, QUAL II, and WQRRS models were calibrated using all three data sets. The MIT model could not be applied to the data because of errors in the model or the data that was coded for the model. An indepth review and modification of the computer code was outside the scope of this project. The Streeter-Phelps and QUAL II models were also verified for all three data sets because greater priority was attached to the full evaluation of the steady-state models with steady-state data. The WQRRS model was not verified because the calibration and comparison to the Streeter-Phelps and QUAL II models indicated that this model was equally valid and of comparable accuracy. Because the additional complexity and coding requirements of the WQRRS model generally preclude the use of the model for routine steady-state simulation in favor of the simpler steady-state models, the calibration of the WQRRS model using steady-state data was deemed sufficient to confirm the validity of the model. Furthermore, the time available to work with the WQRRS model was limited by the unforeseen need to correct several errors in the program. These errors were described to the HEC for their consideration and subsequent correction.

Data required for the models can be classified as follows: initial data needed to start the solution; driving data that describe headwater, tributary, and surface fluxes of mass and heat; coefficient data; and calibration and verification data. Because steady-state applications were made, the initial data were relatively unimportant. The driving data that describe inflow quality and quantity were derived from measurements so that the same information was used in each model.

Model coefficient optimization was avoided whenever possible by using independent determinations of coefficients. In addition, coefficients were standardized for all three models to assist in determining the effect of different model formulations. This isolated the effect of model differences but on occasion led to less than perfect agreement between predictions and measurements.

The data describing instream water quality were used to determine if the model calibration and verification were reasonable. Whether or not the agreement between predictions and measurements was reasonable depended on the constituent, precision of measurements, trends of predictions and measurements, and the maximum difference between predictions and measurements.

Modeling results were obtained in the following fashion:

1. Travel time and the hydraulic conditions were specified as input data from measurements or the model was calibrated to accurately reflect the measurements available.
2. Water temperature was specified or the model calibrated to predict water temperature.
3. Each model was calibrated to predict biochemical oxygen demand (BOD), organic nitrogen, ammonia, nitrite, and nitrate in that order or the independently determined coefficients were checked for accuracy.
4. Because reaeration and benthic demand were estimated from measurements or other independent studies, what remained was to compare the DO predictions to measurements to determine if these measurements or estimates were adequate.
5. As time permitted, minor constituents were simulated.
6. Following calibration, the Streeter-Phelps and QUAL II models were verified with independent data sets.

In reviewing the literature, it became evident that the Velz (1970) rational method is also perceived as an appropriate water-quality model. The detailed examination of the model was outside the scope of this work but the choice of the Chattahoochee and Willamette river data for this study made it possible to include the results of previous studies using the Velz rational method. This made it possible to determine the accuracy and validity of the Velz method but not the efficiency, ease-of-use, or utility of the documentation.

Objectives of the Report

This report describes the study undertaken to evaluate and compare four stream water-quality models. The introduction explains the purpose and goals of the project and describes the objective of this report. The next section describes the brief literature review. The model capabilities outlined in the documentations are examined in the following section and model capabilities are summarized. The following section describes the data sets that were selected for this study and presents any water-quality data that was not available in other publications. In the next three sections, modeling of the Chattahoochee, Willamette, and Arkansas rivers is described. Finally, a summary is given, the conclusions from the model evaluation are stated, and recommendations for additional study are presented.

PART II: SELECTIVE REVIEW OF STREAM WATER-QUALITY MODELING

Stream Water-Quality Models

Models in general use

The four models selected for evaluation using field data include the USGS version of the Streeter-Phelps model; the QUAL II model-SEMCOG version; the WQRRS model; and the MIT model. The Streeter-Phelps model and the QUAL II model are limited to streams with steady flow while the WQRRS model and the MIT model are dynamic models that simulate unsteady flow and water quality.

The following sections tend to confirm that except for the MIT model, these models are generally accepted by water-quality modelers. Each model has a standard documented computer code that can be easily obtained from U.S. Government agencies. The Streeter-Phelps, QUAL II, and WQRRS models are periodically reviewed and updated as needed. The USGS, U.S. Environmental Protection Agency (EPA), and COE resolve questions and provide assistance for the use of these three models. The MIT model does not receive the same level of support.

Previous reviews of water-quality modeling techniques such as Ambrose and others (1981), Harper (1971), and Lombardo (1973) and texts by Rich (1973) and Thomann (1974) tend to compare formulations or capabilities described by the model documentation. Harper assessed various mathematical algorithms used by several models. Ambrose and others (1981) offer an extensive list of stream water-quality models but their examination focused on water-quality models for upland streams that enter estuaries. Lombardo reviewed models for streams, lakes, and estuaries, listing model capabilities along with limited details on each model. The works of Harper and Lombardo were published prior to the creation of model versions used in this evaluation.

Two previous inter-model comparisons using field data were located in the literature. Bauer, Steele, and Anderson (1978) made a rigorous comparison of the Streeter-Phelps and Pioneer I models using data collected on the Yampa River in Colorado. Both models were equally accurate in predicting DO and BOD. Different model formulations led to minor differences in nutrient concentrations. Willey and Huff (1978, pp. H-1 to H-6) compared the WQRRS model to the DOSAG II model under conditions of steady flow and waste loading for the Chattahoochee River in Georgia. Despite differences in stream velocity, reaeration coefficients, and BOD loading and decay, it was claimed that the modeling results of the WQRRS model and the DOSAG II model "compare adequately."

USGS version of the Streeter-Phelps model

The USGS version of the Streeter-Phelps model titled "One-Dimensional Steady-State Stream Water-Quality Model," (Bauer, Jennings, and Miller, 1979) has been used by USGS district offices working with state and local government agencies (Bryant, Morris, and Terry, 1979; and Wilber and others, 1979). In addition, the model has also been used as a research tool (Miller, 1981). Bauer, Steele, and Anderson (1978) compared the Streeter-Phelps model to the Pioneer I model with data collected during the river basin assessment of the Yampa River, Colorado, under steady conditions. The studies mentioned above using this computer code and numerous other studies using the Streeter-Phelps equation confirm that this model is perceived as generally useful for steady-state simulations.

QUAL II model

The QUAL II model receives extensive use. The EPA recommends the model based on ease of use, effectiveness, adequate documentation, and general acceptance by water-quality modelers. The modular design of the computer code also lends flexibility. A number of documented applications (Willis, Anderson, and Dracup, 1976; Barnwell, 1978; Grenney, Teuscher, and Dixon, 1978; and Roesner, Giguere, and Evenson, 1977B) on different streams confirm the utility of the QUAL II model in waste assimilative capacity studies of streams.

In a review of the QUAL II model, the National Council of the Paper Industry for Air and Stream Improvement, Inc. (NCASI, 1980) noted that the models in the QUAL series, QUAL I, QUAL II, and QUAL III, are similar. Differences are limited to the number of water-quality constituents that are simulated and the formulation used to describe particular water-quality variables. The NCASI also notes that several versions of the

QUAL II model exist besides the SEMCOG version used in this model evaluation. These include the State of Texas version that has variable-temperature correction coefficients, sensitivity analysis, and plot output; the EPA version that simulates organic nitrogen but excludes steady-state simulation of algae and temperature; and a version of W. J. Grenney of Utah State University that has modifications to the numerical solution algorithm. Of the available versions, the NCASI chose the SEMCOG version to recommend for use by the paper industry.

WQRRS model

The WQRRS model (Smith, 1978) may be the best supported water-quality model discussed in this report. The HEC provides advice on all aspects of model use and continually updates the program as new techniques become available. The WQRRS model was originally designed as a large basin model and was applied to the Trinity River Basin in Texas.

The HEC has demonstrated the utility of the program with two studies. Willey, Abbott, and Gee (1977) used the WQRRS model to evaluate storm runoff effects and sediment transport in the Oconee River in Georgia. Willey and Huff (1978) studied urban effects of Atlanta, Georgia, on the Chattahoochee River.

MIT model

The MIT model (Harleman and others, 1977) was designed to model estuaries and rivers, but a majority of applications involved estuaries. The MIT model results from a number of studies undertaken at MIT. Nutrient modeling, as described by Najarian and Harleman (1977) is sophisticated but is valid only for nitrogen-limited waters. Sedimentation and scour were not considered in formulating the model. Thatcher, Pearson, and Mayor-Mora (Ambrose and others, 1981, p. 144) applied the MIT model to the St. Lawrence River. Tatom and Mullen (1977) applied the MIT model to a freshwater stream and shallow-lake network in Louisiana.

While the studies mentioned confirm the validity of the use of the MIT model for estuary modeling, this brief literature review did not find a steady-state riverine application. Therefore, it was not possible to confirm the validity of the MIT model for simulating river water quality using the literature readily available.

Other models

Besides these four models mentioned above, the literature review indicates that there are several other models of comparable accuracy. Ambrose and others (1981) offers an extensive listing of stream water-quality models that seems to be complete except for recently published water-quality models such as Jobson (1981).

Models such as the Velz rational technique, Pioneer I, and DOSAG, among others, have been used frequently under a variety of conditions but seemed to have less potential than the models chosen for evaluation. Unlike the Streeter-Phelps, QUAL II, and WQRRS models, these models are rarely reviewed and updated. In some cases documentation is altogether lacking or lacks detail. The Velz rationl method lacks a standard general-purpose computer code. Perhaps the establishment of a steady-state data base in this study will lead to future comparisons with models that were outside the scope of this project.

Steady-State Data Base

USGS Studies

Three data sets were selected from USGS files after considering the accuracy of the data, range of conditions described by the data, and geographical location of the study sites. Based on these criteria, studies of the Chattahoochee, Willamette, and Arkansas rivers were the three best studies available to use in examining steady-state water-quality models.

In terms of accuracy, the series of USGS river-quality assessments that included the Willamette, Chattahoochee, and Yampa rivers are among the best available. Great care was taken in the planning and execution of these studies. In addition, the studies were free of any constraints normally associated with the regulation of waste discharges.

The USGS files also contained a second group of studies performed under cooperative agreements with state and local governments to determine the waste assimilative capacity of various stream segments. These studies were modeled after the river quality assessments but tended to concentrate on specific regulatory problems such as waste load allocation. In general, the studies were shorter, few constituents were measured, and the measurements were less reliable. The study of the Arkansas River in Colorado, one of the better studies under this cooperative program, is an exception to this general rule.

The Willamette, Chattahoochee, and Yampa river data describe a wide range of conditions; however, the Yampa River is part of the Colorado River basin whereas the Arkansas River is part of the Mississippi River basin. Because the Arkansas River data seems to be as reliable and covers about the same range of conditions as the Yampa River studies, this data was selected along with data from the Willamette and Chattahoochee studies to form a data base for the model evaluation.

Other data sources

The EPA and state pollution control agencies also collect comprehensive sets of stream water-quality data suitable for modeling. However, these data are not widely distributed. The accuracy and reliability of the data varies from state to state.

In the past the EPA has concentrated their efforts on model development rather than data collection. Lately, a more balanced approach has been taken. The EPA (Barnwell, 1978) recently compiled calibration and verification data from a study of the Holston River in Tennessee. In addition, the EPA is funding the University of Florida to search the literature and compile data bases describing stream, lake, and estuary water quality and urban runoff quantity and quality.

Dynamic water-quality data

A review of USGS files along with limited inquiries to other agencies confirms that a paucity of dynamic water-quality data exists. This confirms the need for a synoptic data collection effort similar to the USGS river-quality assessments for which discharge and tributary water quality varies significantly over the period of study. A reliable data base would assist in the development of dynamic water-quality models by providing a standard to which model predictions could be compared and validated.

The best available dynamic water-quality data from the USGS was collected during the Chattahoochee River study. Jobson and Keefer (1979) made frequent measurements of flow, temperature, and dye concentrations downstream of a reservoir during periods of unsteady flow. Further downstream, Faye, Jobson, and Land (1979) made frequent measurements of transient flows and temperature from Atlanta to Whitesburg, Georgia. McConnell (1979) studied the quality of urban runoff into the Chattahoochee River. Water-quality data were collected for all nonpoint sources and for three locations on the river. Point source loadings were not measured and in-stream quality was insufficiently defined to permit dynamic water-quality modeling of the receiving water.

PART III: DESCRIPTION OF EVALUATED MODELS

Streeter-Phelps Model

Modeling capability

The Streeter-Phelps model (Bauer, Jennings, and Miller, 1979) is a general water-quality management tool. The model provides a framework within which the effects of point and nonpoint pollution can be assessed. The Streeter-Phelps equation, in which dispersion is neglected, is the basis for modeling DO. In addition, BOD, organic nitrogen, ammonia, nitrite, nitrate (or nitrogenous oxygen demand), orthophosphate-phosphorus, total coliform bacteria (optional), fecal colifiorm bacteria (optional), and three arbitary conservative substances (optional) can be modeled. Furthermore, the model predicts the length of anoxic zones and the carbonaceous BOD at the downstream end of the zone when DO drops to zero (Bauer, Jennings, and Miller, 1979, pp. 2-3).

The Streeter-Phelps model requires all tributary flows and waste loads to be constant. Discharge and water quality in the stream may change in the longitudinal direction, but the discharge and water quality at a point are assumed to be constant with time. The stream is assumed to be well mixed laterally, ignoring significant differences in water quality across the stream.

Geometric representation

The Streeter-Phelps model requires a stream be segmented using three levels of detail (Figure 1): the main stem and branching stems intersecting the main stem; reaches; and computational elements. This segmenting scheme closely depicts actual stream conditions because the discretization technique is not limited to equal length computational elements.

The stream is first divided into a main stem and major tributaries. Water quality in all major tributaries is first simulated to estimate loads from major tributaries to the main stem. Afterwards, the water quality in the main stem is simulated.

The main stem and major tributaries are subdivided into reaches.

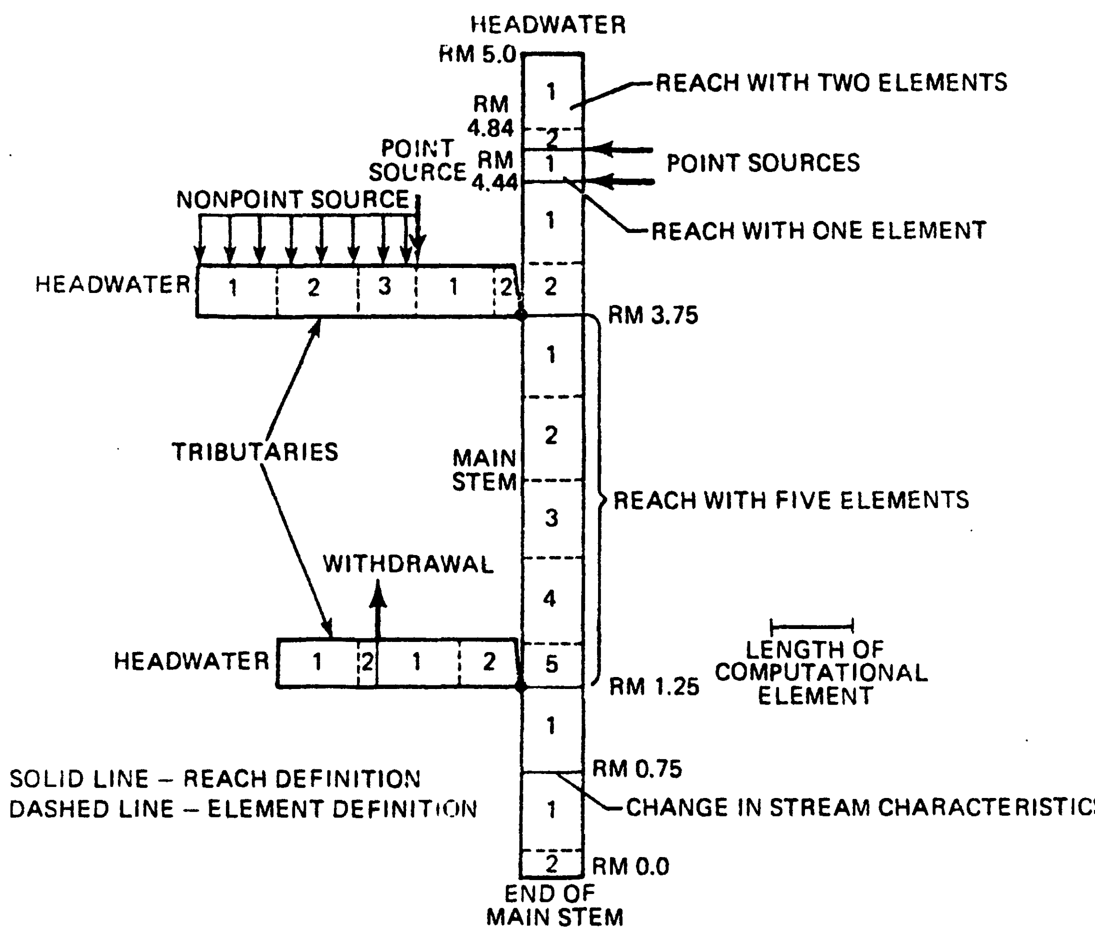


Figure 1. Illustration of the discretization method for the Streeter-Phelps model

The reaches are determined by two criteria. First, all point sources, withdrawals, and headwaters define the head of a reach that extends to the next point source. Second, when physical, chemical, or biological characteristics change, a new reach can be defined, starting at the point of change. Reaches are defined by specifying the river miles upstream of the stream mouth or some arbitrary point at or downstream of the end of the study segment (U.S. customary units are used in the model).

A computational element length is specified for the main stem and each tributary. Each reach is divided into one or more elements. Element lengths may vary from 0.08 to 8.0 kilometers (0.05-5.0 miles).

Although an element length is specified, reaches are not required to consist of an integer number of equal length elements. Very short reaches between major point sources can be defined having one short computation element. Longer reaches with lengths exceeding the specified element length contain a number of standard elements plus a short element at the end of the reach for the fraction of the element length that remains. This method does not require changes in actual reach lengths to satisfy numerical criteria. Thus, some numerical smearing of point sources and reaches can be avoided.

A stream is discretized such that point sources and withdrawals occur at the head of a reach. Nonpoint inflows or withdrawals are specified by reach and occur over the entire reach length. Limitations on the number of point sources, nonpoint sources, and major tributaries are not specified, but no more than 50 reaches or 950 computational elements can be specified.

Hydrodynamic representation

Discharge and reach-averaged depth and width are required data. The Streeter-Phelps model does not have flow routing capabilities.

Travel times are introduced by one of two options. One option allows the direct specification of travel time as input data. The second option requires the model to calculate travel times from the average velocity and length of a reach.

Water-quality representation

The Streeter-Phelps model simulates the following constituents:

1. DO.
2. BOD.
3. Organic nitrogen.
4. Ammonia.
5. Nitrite.
6. Nitrate.
7. Orthophosphate.
8. Fecal and total coliform bacteria.
9. Three conservative substances.

All reaction rates are empirically corrected for temperature, except benthic demand and net photosynthetic production and reaeration coefficients when those coefficients are specified as input data. Stream temperature is required data and is not simulated. Saturation values of DO are calculated as a function of stream temperature, barometric pressure, and salinity.

The DO simulation is controlled by several factors. Carbonaceous BOD decay and nitrification utilize DO. Reaeration adds DO to the stream. Benthic interactions and the difference between photosynthesis and respiration may add or deplete DO. Gross photosynthetic effects are specified as a mean source or sink of DO, depending on whether respiration or photosynthesis dominates.

Reactions for BOD decay, nitrification, and reaeration are assumed to be first-order processes. Benthic interactions and net photosynthetic production are treated as zero-order reactions.

Nine options are available to introduce reaeration coefficients in the program. Reaeration coefficients can be specified as input data when those data are known from previous calculations or measurements. Specification as input data is compatible with the specification of travel times as input data when tracer measurements are available. In addition, eight options allow the internal calculation of reaeration coefficients using eight different predictive equations. The nine options include:

1. Direct specification.
2. Bennett-Rathbun equation.
3. Langbein-Durum equation.
4. Padden-Gloyna equation.
5. Bansal equation.
6. Parkhurst-Pomeroy equation.

7. Tsivoglou-Wallace equation.
8. O'Connor-Dobbins equation.
9. A formulation based on the Velz iterative technique.

Simulation of nitrogen is limited to forms that are involved in the oxidation of nitrogen. Two options are available. In one case, nitrogenous BOD is treated as a first-order reaction analogous to BOD decay (an undocumented option treats nitrogenous BOD decay as a zero-order process). In the second option, organic nitrogen, ammonia, nitrite, and nitrate are simulated using a formulation developed by Thomann, O'Connor, and DiToro (Bauer, Jennings, and Miller, 1979, pp. 5-9).

The nitrification process illustrated in Figure 2 allows sinks and sources of organic nitrogen, ammonia, nitrite, and nitrate. The decay rate controls the amount of organic nitrogen, ammonia, nitrite, and nitrate remaining at the end of the travel time through an element. The forward reaction rate controls the amount of the nitrogen added as ammonia, nitrite, and nitrate. External sources and sinks of organic nitrogen, ammonia, nitrite, and nitrate, excluding tributary and waste loads, are controlled by the difference in decay and forward reaction rates. Sedimentation and scour of organic nitrogen, adsorption or desorption of ammonia, nitrite, and nitrate onto or from benthic materials, escape of ammonia gas, and the uptake or release of ammonia and uptake of nitrate by stream biota are described by the difference of two first-order reactions.

Despite this flexibility, two limitations remain. First, a source or sink of nitrogen is described by a first-order reaction. No allowance is made for zero-order processes such as the benthic release of organic nitrogen. In some cases, the uptake of ammonia and nitrate by biota may be better described by modeling biomass. Second, the model allows an abstract treatment of the nitrogen sinks and sources as the difference of two coefficients. For example, it may be possible to simulate external sources and sinks of nitrite in addition to waste loads, but it is difficult to justify this based on what is known about stream water quality.

Despite these limitations, the algorithm illustrated in Figure 2 has been adequately tested in several studies. Thomann, O'Connor, and

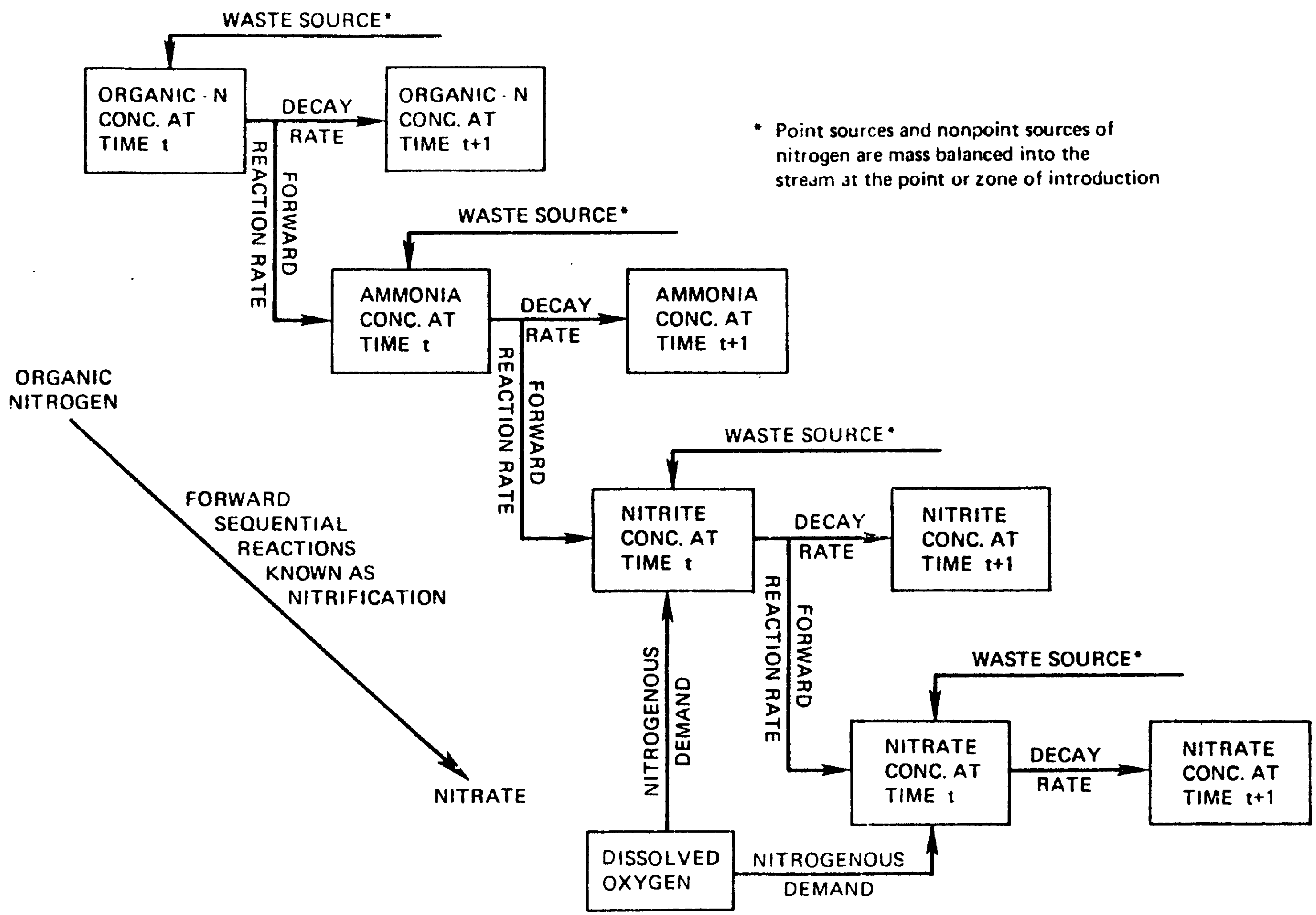


Figure 2. Nitrification cycle used by the Streeter-Phelps model (adapted from Bauer, Jennings, and Miller 1979)

DiToro (1971, pp. 37-55) originally developed the algorithms used in this Streeter-Phelps model to simulate nitrification. They also describe applications to the Delaware Estuary and Potomac Estuary. Both studies simulate ammonia and nitrate uptake by algae as first-order reactions dependent upon ammonia and nitrate concentrations, respectively. Bauer, Steele, and Anderson (1978) applied this nitrification scheme in a study of the Yampa River in Colorado with good results. In addition, good results were obtained in a study of the Chattahoochee River in Georgia by Miller and Jennings (1979).

The formulation for orthophosphate simulation uses waste inflows and the stream bottom as sources of orthophosphate. The uptake or release of orthophosphate by phytoplankton, expressed as chlorophyll a, is a sink or source of orthophosphate. Other forms of phosphorus are not simulated.

Chlorophyll a concentrations are specified as input data and are not simulated. These concentrations are used to simulate the uptake of orthophosphate-phosphorus by algae and do not affect the DO balance. Formulations for the DO balance and phosphorus balance are not coupled in the model and chlorophyll a concentrations do not modify gross photosynthetic affects (specified as input data) on the DO balance.

Finally, there are options to model coliform bacteria, conservative substances, and anaerobic zones. Fecal and total coliform bacteria die-off are modeled separately as first-order reactions. Three arbitrary conservative substances can be modeled with the results reported in milligrams per liter. Conservative substances are mass balanced at each inflow. When DO levels reach 0.1 milligrams per liter, the program estimates the length of the anaerobic zone and the carbonaceous BOD at the downstream end of the anaerobic zone. However, it is unclear whether these algorithms have been fully verified. Bauer, Jennings, and Miller (1979) do not fully explain or demonstrate this option in the appended example problems of the model documentation.

Program utility

Input data formats for the Streeter-Phelps model are inefficient. Decay coefficients and forward reaction coefficients are coded in two different places; i.e., similar coefficients are coded on different cards. In addition, internal program checks on the input data are inadequate.

Despite these drawbacks, the input data are flexible and can be easily modified when reaches are added or deleted. This flexibility is due to the IBM computer utility subroutine REREAD. When the model is adapted to other computer systems, this subroutine must be replaced.

Program output is well organized but lengthy, and the user has no option to delete parts. Tables of input data are well organized but must be checked to ensure the same data are used in internal calculations. Internal computations are reported in a manner that readily assists in tracing errors in data. Results are not summarized in a final table but instead are presented in line printer plots of each constituent versus river mile. Model calibration is greatly simplified and made easier by the plotting of observed measurements with predicted values for each constituent.

Because water-quality equations are solved analytically, computing costs are low; internal calculations are simplified and easy to understand; and the discretization scheme accurately depicts stream geometry. Measurements of travel time, average depth, and temperature can be specified as input data. However, the flexibility of the model is limited by the lack of options to simulate travel time, depth, width, and temperature.

The source code is written in Fortran IV and is about 3000 lines in length. Seventy-five thousand words of storage are needed along with some temporary file storage.

The USGS maintains an operational version of the Streeter-Phelps model on three computer systems. The most up-to-date version is maintained on two systems: AMDAHL 470V/F (RE2), USGS National Headquarters, Reston, Virginia; and IBM 3033, Applied Physics Laboratory, John Hopkins University, Silver Springs, Maryland. A version using DO deficits as DO input data is maintained on the Water and Power Resources Service (Bureau of Reclamation) CDC Cyber-70 System in Denver, Colorado.

The program source deck and documentation can be requested from the following address:

Deterministic Models Project
Gulf Coast Hydroscience Center
U.S. Geological Survey
NSTL Station, MS 39529
(601-688-3071, FTS 494-3071)

QUAL II Model-SEMCOG Version

Modeling capability

The QUAL II model-SEMCOG version (hereafter referred to as the QUAL II model) is a one-dimensional steady-state water-quality model applicable to branched streams (Roesner, Giguere, and Evenson, 1977A and 1977B). Water Resources Engineers developed the QUAL II model in 1972 for the EPA. That version was a refinement of previous work by the Texas Water Development Board and F. D. Masch and Associates in 1970. In 1976, a number of modifications and refinements resulted in the SEMCOG version.

The QUAL II model was designed as a water-quality planning tool. The model accepts multiple waste inflows, tributaries, withdrawals, and nonpoint sources. The effects of waste load magnitude, quality, and location on stream quality can be predicted for nonpoint and point source pollution. The required dilution flows to meet prescribed levels of DO can be calculated. A dynamic option allows the simulation of diurnal variations of DO, nutrients, BOD, algae, and temperature resulting from a diurnal variation of meteorological conditions. Stream discharge and tributary inflows must remain constant. In addition, the model was formulated to include longitudinal dispersion in the transport calculations.

Geometric representation

A stream system is discretized into three levels of detail (Figure 3) for the QUAL II model simulation. The QUAL II model can simulate water quality in the main stem and multiply branched tributaries (i.e., dendritic stream systems). Each branch is divided into one or more

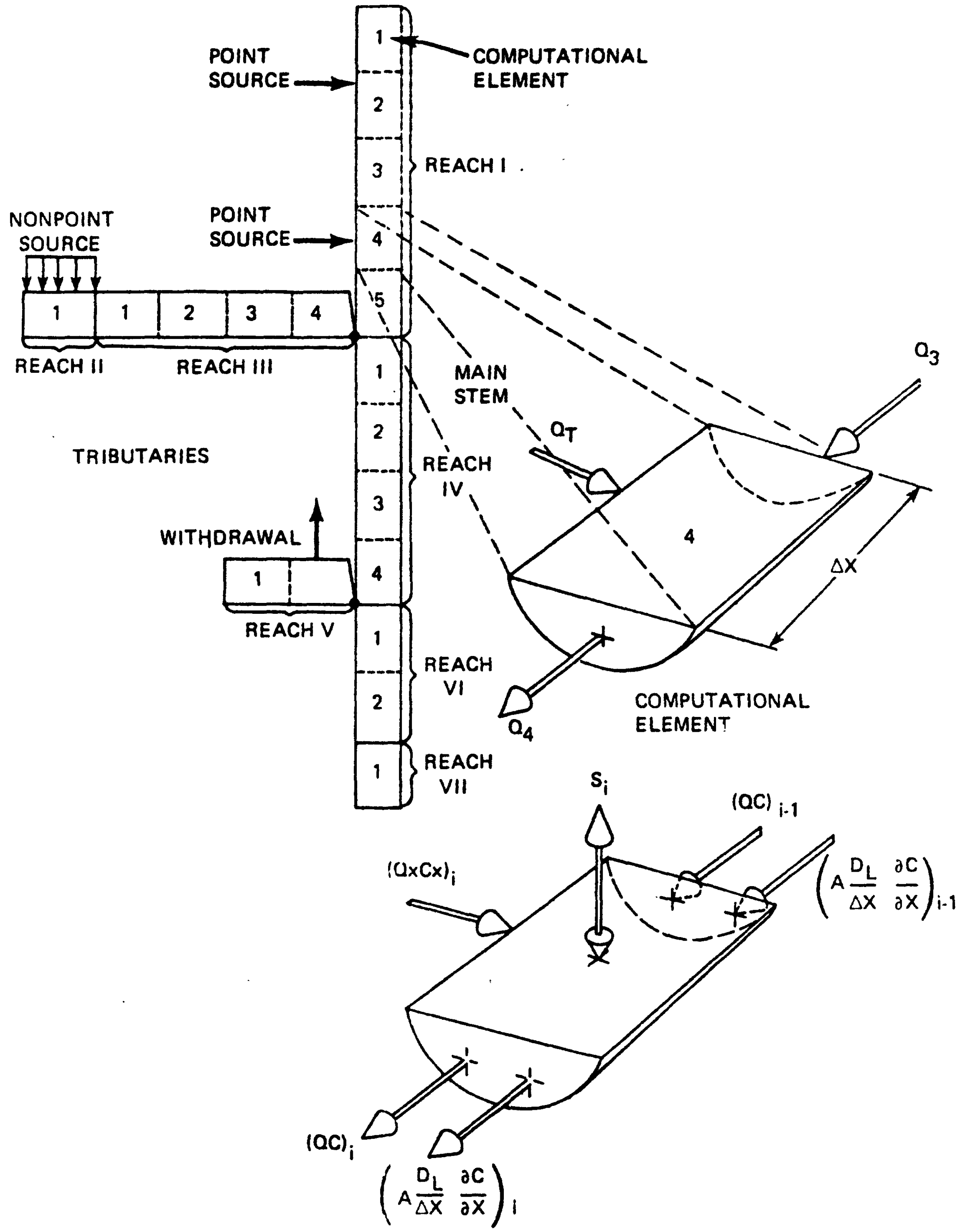


Figure 3. Discretization scheme for stream systems used by the QUAL II model and a schematic definition of the mass balance on each element (adapted from Roesner, Giguere, and Evenson, 1977A)

reaches. All reaches consist of one or more computational elements of equal length. The element length is equal for all branches in the system.

The requirement of equal-length elements can lead to discretization errors when reach lengths are not equal to an integer number of element lengths. If two or more point sources enter the same element, the sources must be combined by mass balancing.

In coding the data, the type of element is specified according to the hydraulic and geometric characteristics of the stream at that point. A headwater element begins each tributary or branch. Junction elements and elements just above junctions are specified. Elements receiving inflow from waste discharges or tributaries and elements having water withdrawn are declared as such. The final element in the stream system is specified, and all remaining elements are standard elements. Nonpoint flows may enter any element and require no change in the above specifications.

Consecutive elements having similar physical, chemical, and biological properties are grouped into reaches. Input data are specified by reach. Parameters governing the physical, chemical, and biological response of the stream system are supplied once for each reach.

The QUAL II model is general in nature, but certain limits exist. These limits include:

1. Maximum of 75 reaches.
2. Maximum of 500 elements, but no more than 20 in one reach.
3. Maximum of 15 headwater elements.
4. Maximum of 15 junctions.
5. Maximum total of 90 inflows and withdrawal elements.

A longitudinal coordinate scheme is used by the QUAL II model to label reach locations. Reaches are defined by specifying beginning and ending river kilometers (miles) from the mouth of the river or some arbitrary point at or downstream of the end of the study segment. U.S. customary or metric units can be used and the model can convert the results to either system.

Hydrodynamic representation

The hydraulic response of the stream is simulated by one of two methods. One, reach-averaged coefficients can be specified that relate average velocity to discharge:

$$\bar{u} = aQ^b \tag{1}$$

and average depth to discharge:

$$d = \alpha Q^\beta \tag{2}$$

in which a, b, α, and β are constant for a reach. The other method allows the user to approximate cross sections as trapezoids, and the program solves Manning's equation by trial and error for the average velocity and average depth.

The longitudinal dispersion coefficient (Roesner, Giguere, and Evenson, 1977A) is expressed as,

$$D_L = 22.6\ n\ \bar{u}\ d^{0.833} \tag{3}$$

in which n = Manning's coefficient. Equation 3 was derived for wide straight channels and underpredicts longitudinal dispersion in natural channels (Fischer and others, 1979, and Bansal, 1976).

Water-quality representation

The QUAL II model solves mass-balance equations for each water-quality constituent. The equations are numerically integrated over time. Advection, dispersion, dilution, constituent reactions and interactions, and sources and sinks of the material are considered.

The mass balance equations for each constituent used in the QUAL II model are cast in a forward-in-time centered-in-space finite difference formulation. A series of simultaneous linear equations result, in which the coefficient matrix is a tri-diagonal matrix that lends itself to an efficient computer solution. Initial conditions are specified to initiate the finite difference solution.

The QUAL II model focuses on the balance of DO in the stream as shown in Figure 4. The effects on the DO balance are the primary concern in modeling other constituents. For example, only chlorophyll a, ammonia, nitrite, and nitrate are simulated by the QUAL II model formulation for the nitrogen cycle.

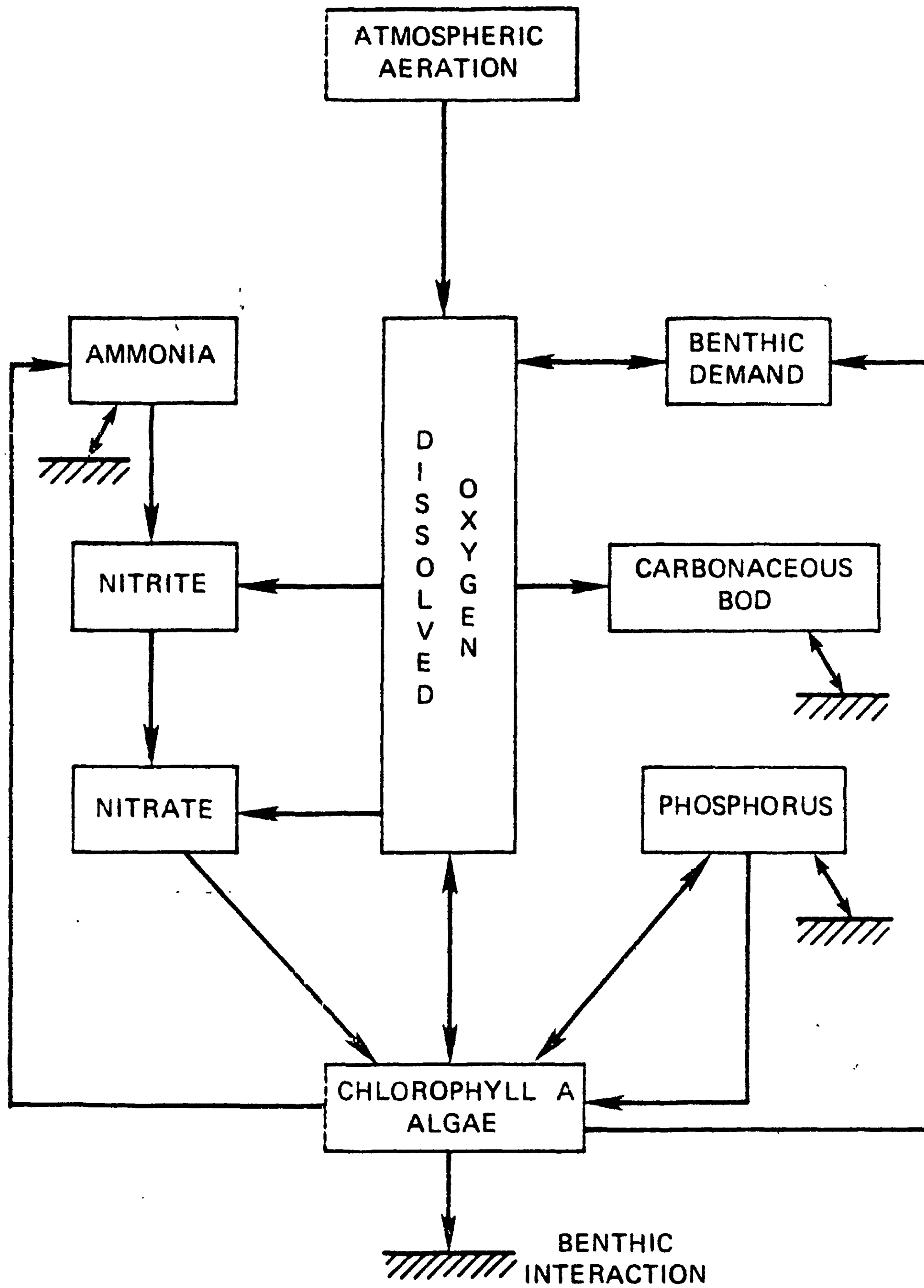


Figure 4. Water-quality constituents modeled by the QUAL II model (adapted from Roesner, Giguere, and Evenson, 1977A)

Any combination of nine water-quality constituent groups can be modeled. These groups include:

1. DO.
2. BOD.
3. Temperature.
4. Algae as measured by chlorophyll a.
5. Ammonia-nitrogen, nitrite-nitrogen, and nitrate-nitrogen.
6. Dissolved orthophosphate-phosphorus.
7. Coliform bacteria.
8. One arbitrary constituent governed by first-order kinetics.
9. Three conservative constituents.

Thus, any constituent can be modeled separately without modeling any of the other possible parameters except that ammonia-nitrogen, nitrite-nitrogen, and nitrate-nitrogen are modeled as a group. If chlorophyll a is simulated without simulating nitrogen or orthophosphates, the model assumes algae growth is not nutrient limited.

Water temperature can be either specified as initial data or simulated. The QUAL II model will simulate water temperature given wind speed, dry-bulb temperature, wet-bulb temperature, air pressure, and percent cloudiness. Temperature modeling is based on a heat budget of each element. The heat budget considers the surface flux of heat and the heat content of inflows into the stream. Tree shading and the flux of heat to or from the channel bottom are neglected.

Stream temperatures, whether specified or simulated, are used in the adjustment of reaction rates. Temperature is used to empirically adjust all reaction coefficients except BOD settling.

The formulation for the oxygen balance includes atmospheric reaeration, BOD decay, benthic oxygen demand, net photosynthetic oxygen production, and the oxidation of ammonia and nitrite. The saturation values for DO are calculated as a function of temperature, but are not corrected for pressure differences from standard barometric pressure or for chloride concentrations in the water.

Eight options are available for the calculation of the reaeration coefficient K_2:

1. Direct specification of K_2.
2. Churchill, Elmore, and Buckingham equation.
3. O'Connor and Dobbins equation.

4. Owens, Edwards, and Gibbs equation.
5. Thackston and Krenkel equation.
6. Langbien and Durum equation.
7. Tsivoglou and Wallace equation.
8. $K_2 = aQ^b$.

Methods in options 2, 4, 6, and 8 are empirical relationships relating K_2 to average velocity and depth. The Thackston and Krenkel formula is the most complex, involving the Froude number and shear velocity. The Tsivoglou and Wallace equation has two coefficients that are specified as input data and is the most flexible. The O'Connor and Dobbins equation has been the most frequently used equation because of the rational basis and simple form.

The mass balance of the carbonaceous BOD includes decay of carbonaceous material, settling, and the release of BOD from the stream sediments. Decay and settling are approximated by first-order formulations. Release of BOD from sediments can be approximated by specifying a negative settling coefficient or adding a nonpoint source of BOD.

Benthic oxygen demand or sediment oxygen demand is simulated with a constant uptake rate. This uptake rate is not coupled to the settling or release of BOD.

The QUAL II model simulates the nitrogen cycle on a simplified basis. Chorophyll a is used as a measure of phytoplanktonic algae biomass and is assumed to be a function of a local specific growth rate, a local respiration rate, and a local settling rate. The local specific growth rate is related to a specified maximum specific growth rate, availability of nitrate and orthophosphate, and light intensity. Thus phosphorus, nitrogen, or light-limited conditions can be modeled. When algae is modeled and nutrients are not, the model assumes that algal growth is not limited by nitrogen or phosphorus.

The decompositions of ammonia and nitrite into nitrate are modeled as first-order reactions. Phytoplankton take nitrate from the water and release ammonia to complete the nitrogen cycle in the QUAL II model. The settling of algae is modeled using a first-order formulation. The benthic release of ammonia is simulated with a zero-order reaction. The desorption of ammonia to the atmosphere is not explicitly considered.

However, this phenomena can be simulated by specifying a negative benthos release coefficient (Roesner, Giguere, and Evenson, 1977B, p. IV-9). Nevertheless, caution is advised where ammonia desorption is significant since Stratton (1968) demonstrates ammonia desorption is a first-order process rather than a zero-order process.

The formulation for phosphorus includes the uptake and release of orthophosphate by algae and the adsorption or release of orthophosphate from bottom sediments. Algae interactions are approximated by first-order reactions. Benthic interactions are approximated as zero-order reactions.

Coliform die-off and the single arbitrary nonconservative constituent are modeled as first-order reactions. The three arbitrary conservative constituents are mass balanced as inflows enter the stream.

Program utility

The input data formats for the QUAL II model are well organized. A separate appendix in the documentation presents coding sheets and includes detailed instructions for coding of the data. Data are coded by reach, and reaches can be added or deleted without recoding data.

The output data, consisting of three sections, are also well organized. In the first section, the input data (except meteorological data) are printed verbatim (echoed) after numerous internal program checks. Next, internal calculations are summarized illustrating the convergence of the numerical solution. Finally, water-quality and hydraulic predictions, along with reaeration and reaction coefficients, are summarized efficiently in a table. The user has the option of reducing the output to the final summary table.

Two disadvantages exist with input and output data. First, some input data for algae simulation are always required even if algae are not modeled (corrected recently by NCASI, 1980). Second, the QUAL II model-SEMCOG version makes no allowance to plot the results. However, if a graphical summary is needed, other versions of the model have this option (NCASI, 1980).

The QUAL II model computer code is quite flexible. Various functions of the model are handled by subroutines that may be easily modified or improved. The code is written in FORTRAN IV and has been executed using the UNIVAC 1108, CDC 6400, DEC-10, Xerox SIGMA 7, and IBM 3033 computer systems among others. The EPA also maintains a version of the QUAL II model on their Washington computer that supports the STORET data-management system. Approximately 51,000 words of core storage are required. Input of data is by card reader and the only required output device is a line printer.

The EPA has recently established the Center for Water Quality Modeling in Athens, Georgia. The Center will furnish a tape of the source code that can be copied and returned. In addition, the Center now formally offers consultation on the use of the QUAL II model within the EPA. The source code and consultation may be requested from:

Center for Water Quality Modeling
Environmental Research Laboratory
USEPA
College Station Road
Athens, GA 30613
(404-546-3585 or FTS 250-3585)

Water Quality for River-Reservoir Systems Model

Modeling capability

The WQRRS model is designed as a basin-scale ecological modeling system (Chen, 1970). The WQRRS model was developed by Water Resources Engineers, Inc. and the HEC. This model was developed during a study of the Trinity River Basin in Texas. The HEC supports the program with consultation, training courses, and periodic program updates.

The model consists of three separate modules: (1) reservoir module; (2) stream-hydraulics module; and (3) stream-quality module. These modules are linked with magnetic tape interfaces that extend the model capabilities. Such a link with the watershed runoff model STORM is possible. In addition, the WQRRS model can receive information from the Geometric Elements from Cross-Section Coordinates program of the HEC for stream-channel geometric properties and uses the HEC-2 format for channel properties. To analyze results, the WQRRS model can also transfer data to an HEC plotting and statistical postprocessing program.

Stream systems are broken into segments in which reservoir or stream quality is modeled. Modules transfer information about flow quantity and quality from one segment to the next. Information about reservoir outflow quantity is transferred to the stream-hydraulics module, and information describing quality is transferred to the stream-quality module. In an individual stream segment, the stream-hydraulics module generates and transfers necessary hydraulics information to the stream-quality module. In consecutive stream segments, information about amounts and quality of flow are passed from the upstream segment to the downstream segment and become headwater inflows for the downstream modeled segment.

The one-dimensional lake module adds utility to the WQRRS model that is not found with other one-dimensional stream water-quality models. The reservoir module is designed for aerobic one-dimensional impoundments. Small to moderately large lakes, with large residence times, are best suited to the model. A vertically stratified lake is described with a series of well-mixed vertically-stacked layers. Other than to mention these capabilities, the lake module will not be considered in this evaluation.

Geometric representation

The WQRRS model is a dynamic water-quality model with a wide range of flow-routing capabilities. Dynamic routing of flow and water quality lead to complex criteria for discretizing the stream system.

The stream-hydraulics module has the capability of modeling a branched stream or a network of streams as illustrated in Figure 5. In each branch, multiple reaches can be defined. Each reach is divided into nodes or grid points for the stream-hydraulics module. The volume between nodes is the computational element for the stream water-quality module.

The stream discretization scheme has several programming limitations. For a water-quality study, these limitations are as follows:

1. 41 points at which the channel cross section can be defined.
2. 100 elements.
3. 105 nodes.

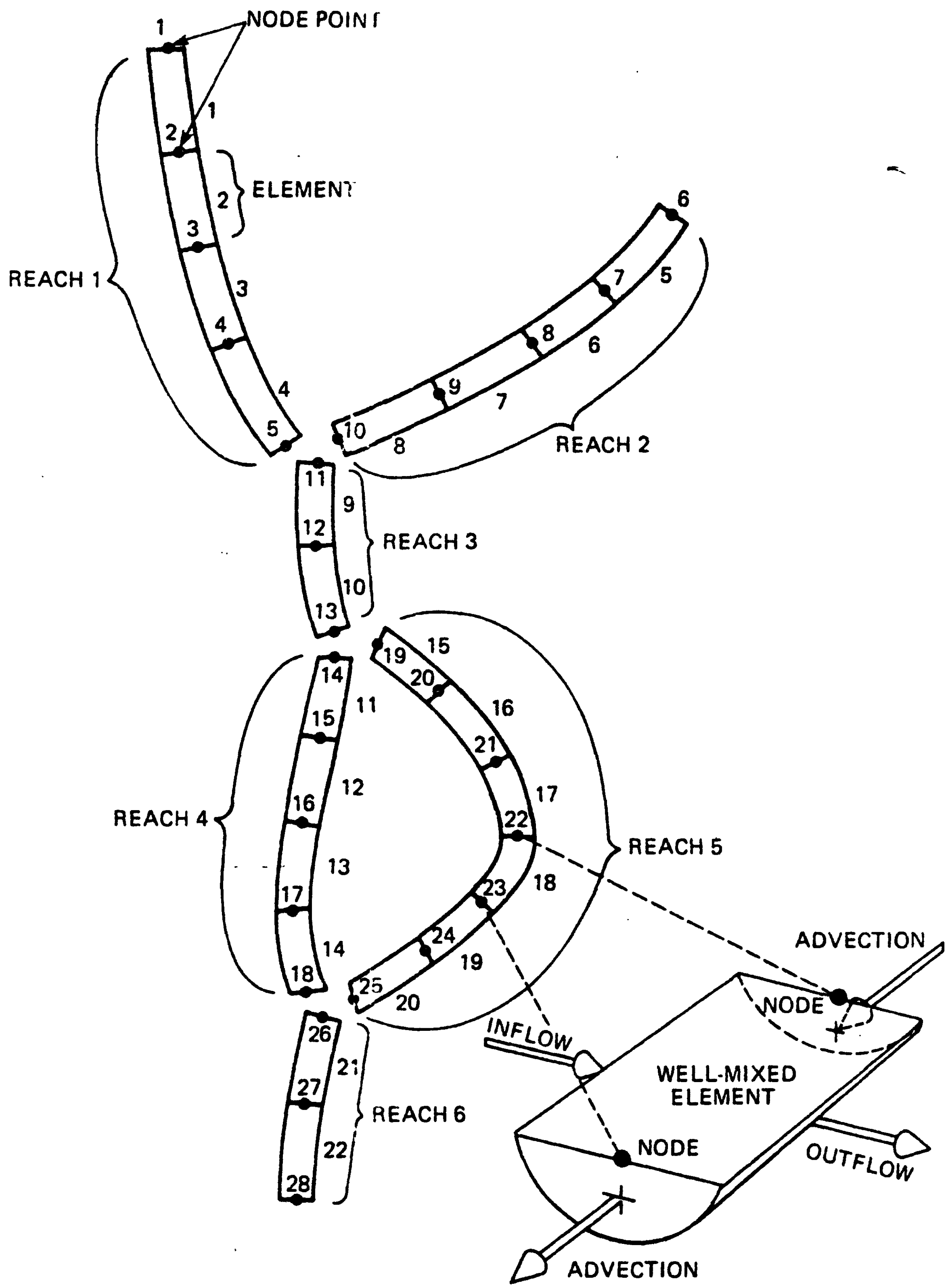


Figure 5. The WQRRS model discretization method for a stream network (adapted from Smith, 1978)

4. 10 inflows, withdrawals, and nonpoint source zones, including the headwater inflow.
5. 10 reaches.

These limitations can be overcome using the magnetic tape interfaces to transfer information for the downstream model runs.

Multiple model applications are made in a system when the number of measured channel cross-sections exceed 41; the number of inflows, withdrawals, headwaters, and nonpoint zones exceed 10; or when chemical or biological characteristics change. Reaches in a segment are defined when control structures break the study segment. Control structures are low water dams, rapids, or waterfalls at which critical depth occurs. The cross-sectional properties may vary over the length of the reaches defined for the WQRRS model.

Each reach is divided into an even number of elements (Figure 5). Recommended element lengths are 0.8-3.2 kilometers (0.5-2 miles). Three different element lengths may be specified in any reach.

Two options are available to specify channel geometry. First, lateral and vertical channel coordinates can be specified. Second, elevation versus hydraulic radius, area, and top width can be specified.

Location of point sources, the headwater, and withdrawals are specified by river kilometer or mile. Nonpoint zones are specified by beginning and ending river kilometers or miles. Point inflows cannot be specified in the last element of a reach and nonpoint source zones should begin and end at nodes.

Reaches, nodes, elements, and inflows are specified by river kilometers or miles. Measured channel cross sections are specified by river kilometers and meters between cross sections (miles and feet). All water-surface elevations and cross-section coordinates are referred to a common vertical datum, usually mean sea level. The model accepts U.S. customary or metric units and converts the results to either system if needed.

Hydrodynamic representation

The stream-hydraulics module simulates one-dimensional steady, gradually varied, or fully dynamic flows. Six options are available for these hydraulic computations:

1. Hydraulic backwater solution (steady flow).
2. Complete St. Venant equations.
3. Kinematic wave equations.
4. Direct input of a stage-flow relationship (steady flow).
5. Muskinqum hydrologic routing.
6. Modified Puls hydrologic routing.

The stream-hydraulics module solves for discharge and elevation at each node. The solution technique varies according to the routing option. An initial time step is specified, but the program checks the time step and decreases it if convergence does not occur.

Water-quality representation

The stream-quality module assumes that each control element is well mixed and that aerobic conditions are maintained. The solution of the water-quality transport equations involves a system of linear equations in a finite difference form describing water quality in each stream element. The resulting equations are then integrated numerically in time.

Source and sink terms for water-quality variables include first-order decay, settling, surface flux (reaeration or heat transfer), chemical transformations, biological uptake and release, and mortality. Groups of organisms in the food chain are simulated with sink and source terms for settling, growth, respiration, mortality, predation, and self-propulsion.

An extensive set of water-quality and biological parameters are simulated by the WQRRS model. These parameters are:

1. Temperature.
2. DO.
3. Carbonaceous BOD.
4. Coliform bacteria.
5. Organic detritus.
6. Ammonia-nitrogen.
7. Nitrate-nitrogen.
8. Nitrite-nitrogen.
9. Orthophosphate-phosphorus.
10. Total dissolved solids.
11. Phytoplankton No. 1.
12. Phytoplankton No. 2.
13. Zooplankton.
14. Total inorganic carbon.
15. Alkalinity as $CaCO_3$.
16. Organic sediment.
17. Benthic animals.
18. Fish No. 1.
19. Fish No. 2.
20. Fish No. 3.

21. Suspended solids No. 1.
22. Suspended solids No. 2.
23. Suspended solids No. 3.
24. Suspended solids No. 4.
25. Suspended solids No. 5.
26. Inorganic sediment.
27. Aquatic insects.
28. Benthic algae No. 1.
29. Benthic algae No. 2.
30. Unit toxicity.

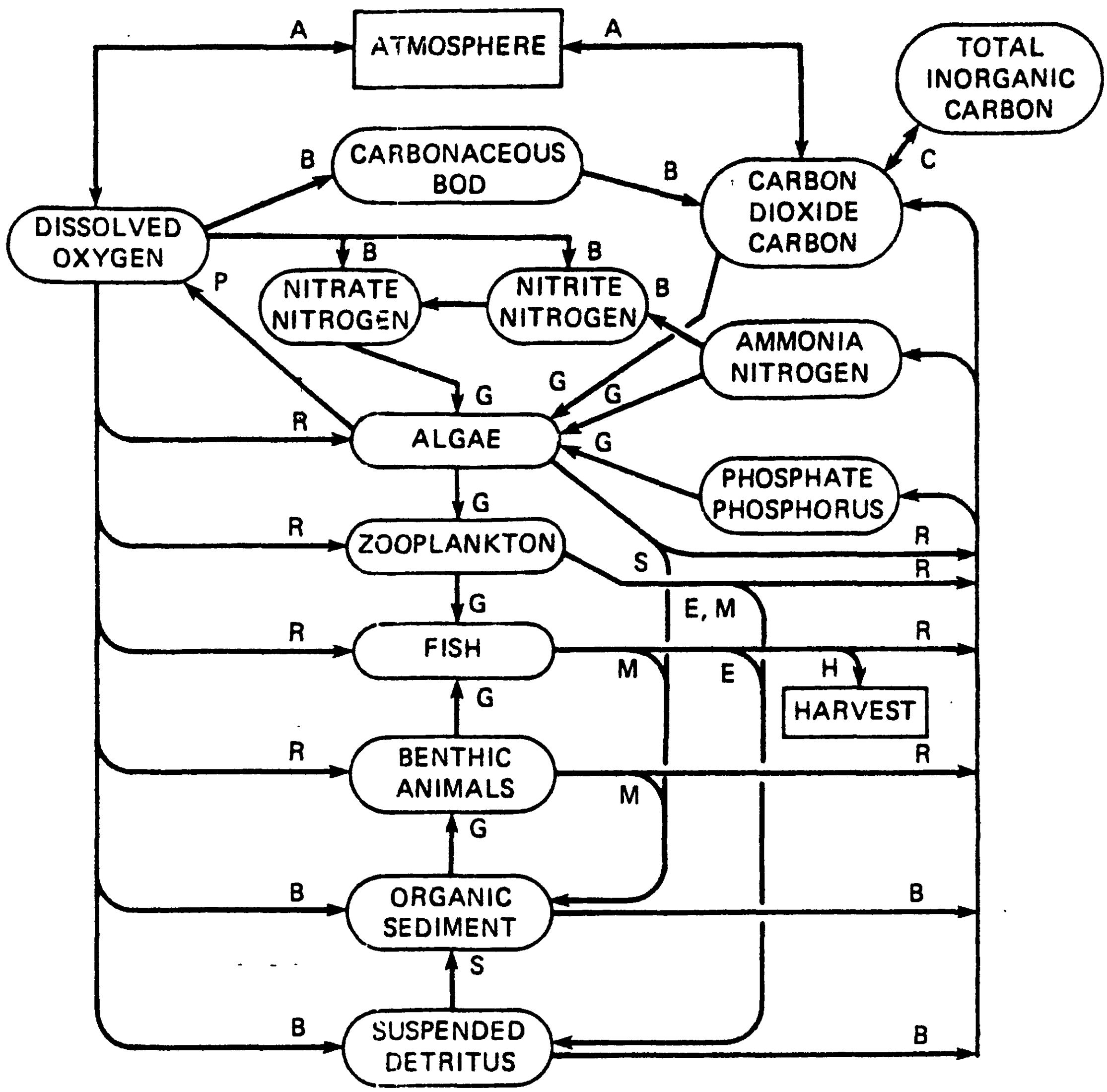
As the list above indicates, the WQRRS model simulates a DO, nitrogen, carbon, and orthophosphate balance and a food chain. Figure 6 illustrates some of the interactions between components. Because the documentation does not specifically state what combinations of these variables can be modeled, caution is advised when neglecting some components.

Temperature can be specified or simulated by two methods: the heat budget method or the equilibrium temperature method. Heat exchange with the stream bottom is considered. Short- and long-wave radiation are calculated, and bank shading is not explicitly considered.

The formulation for the DO balance includes atmospheric reaeration, carbonaceous BOD decay, nitrification, photosynthesis, respiration, detritus decay, and organic sediment decay. Reaeration, BOD decay, ammonia decay, nitrite decay, detritus and sediment decay, and photosynthesis and respiration are first-order reactions. Respiration and photosynthesis due to algae are considered separate from respiration and photosynthesis of the other biota.

Ultimate carbonaceous BOD (BOD_{ULT}) decay is approximated as a first-order process ignoring benthic sinks and sources. Oxygen demand associated with suspended organic particles and organic sediments are modeled separately as first-order processes and are included in the DO balance. Thus only dissolved BOD_{ULT} is simulated with the BOD mass balance equation. However, the results should not differ from the standard approach of modeling a combination of dissolved and suspended BOD if detritus and organic sediment decay are not simulated. The water-quality sampling program should reflect this difference when detritus and organic sediment simulation is necessary or the model will compensate for detritus decay when BOD values are specified as negative numbers.

In addition, 5-day BOD is specified as input data. Details of the conversion of 5-day BOD to ultimate BOD are lacking in Smith (1978).



A Aeration
B Bacterial Decay
C Chemical Equilibrium
E Excreta
G Growth
M Mortality
P Photosynthesis
R Respiration
S Settling
H Harvest

Figure 6. Ecologic and water-quality relationships modeled by the WQRRS model (adapted from Smith, 1978)

However, Donald Smith* notes that 5-day BOD is converted to ultimate BOD in the DO mass-balance equation using a constant factor of 1.46.

Seven reaeration coefficient options are available:

1. Direct specification of K_2.
2. Churchill, Elmore, and Buckingham equation.
3. O'Connor and Dobbins equation.
4. Owens, Edwards, and Gibbs equation.
5. Thackston and Krenkel equation.
6. Langbein and Durum equation.
7. Tsivoglou and Wallace equation.

Benthic interactions are modeled in detail. Sediments are divided into organic and inorganic sediments. The formulation describing organic sediment includes the decay of organic sediment; settling of detritus, algae, particles of excrement, and dead predators; and grazing of organic sediment by predators. The decay of detritus and organic sediments releases orthophosphate, carbon, and ammonia while removing DO from the water in the stream. Inorganic suspended solids settle to become inorganic sediment. Inorganic sediment does not interact with other water-quality parameters. Neither organic or inorganic sediments are resuspended as detritus or inorganic solids.

Three types of aquatic plants and three types of aquatic animals are simulated. Benthic algae, phytoplankton, zooplankton, fish, benthic animals, and aquatic insects comprise a food chain that is linked to the DO, nutrient, and organic sediment balances.

The formulated carbon balance is similar to the DO balance. That balance includes CO_2 exchange through the water surface; release of CO_2 by BOD, detritus, and sediment decay; and CO_2 uptake and release by biota.

The nitrogen balance includes organic nitrogen in detritus, sediments, and biota; ammonia; nitrite; and nitrate. Ammonia and nitrite decay are first-order reactions. Ammonia is consumed and released by the biota and released by the decay of detritus and organic sediment. Nitrate is consumed by the biota.

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* Written communication, June 4, 1981, Donald Smith, Resource Management Associates, Lafayette, California.

The orthophosphate balance includes release of orthophosphate by organic sediments and detritus decay. In addition, the biota release and consume orthophosphate. Inorganic sources of orthophosphate are ignored.

The WQRRS model also simulates coliform bacteria, alkalinity, total dissolved solids, and unit toxicity. Coliform bacteria die-off is simulated as a first-order process while alkalinity, total dissolved solids, and unit toxicity are treated as conservative substances.

Program utility

The input and output data are well organized for the stream-hydraulics module and the stream-quality module. Both modules print input data and provide internal checks. Nevertheless, one problem has been noted. The stream-hydraulics module is not well suited for calibration based on travel time or reach-averaged depth and velocity. To use these calibration criteria, it is necessary to execute both modules. If a reach-averaged depth and velocity or travel time summary was given in the stream hydraulics module, it would be possible to calibrate the flow model before executing the water-quality model.

A number of physical, chemical, and biological coefficients are needed to execute the program, but default values can be used for preliminary investigations. Most coefficients can be specified by changing default values (e.g., temperature coefficients that adjust biological and chemical reaction rates can be specified as input data).

Despite the usefulness of this default option, some problems occur in specifying coefficients. Coefficients are specified once for each study segment and cannot be varied over the segment even if multiple reaches are defined. The direct specification of the reaeration coefficient is an exception since K_2 is specified for each element.

The HEC supports the WQRRS model on the University of California at Berkeley CDC-7600 computer system. A source deck and consultation can be requested from:

U.S. Army Corps of Engineers
The Hydrologic Engineering Center
609 Second Street
Davis, CA 95616
(916-440-2105 or FTS 448-2105)

MIT Transient Water-Quality Network Model

Modeling capability

The MIT model simulates the hydrodynamics and water quality of an aerobic estuary or river. The model (Harleman and others, 1977) was originally developed for nitrogen limited estuaries but use has been extended to riverine conditions. The model was intended to assist in resource decisions concerning the degree of eutrophication due to distributed and point sources of nitrogen in estuaries.

The MIT model evolved through several studies at MIT and was packaged as a single program in a project funded by the EPA. These different studies resulted in a combination of subroutines that lacks the homogeneity of a model constructed in a single effort.

The MIT model solves the one-dimensional continuity and momentum equations for stream discharge and water-surface elevation as a function of time and longitudinal distance from the beginning point of the study reach. The hydraulic modeling results are used in the solution of the conservation of mass equations for water-quality variables. These equations are solved using an implicit finite element scheme. Longitudinal dispersion terms are retained in mass balance equations for water quality.

Geometric representation

The MIT model will simulate a complex network of one-dimensional stream channels. This simulation can also include flow reversals in the system. The network is represented by a number of reaches between nodes in which cross-sectional geometry may change. These reaches are broken into computational elements.

Nodes represent the junction of two or more stream branches or the beginning of a stream segment (headwater). The stream segment between nodes is a reach. A stream segment may be broken into two or more reaches by control structures (low-water dam, weir, rapids, or waterfall). Nodes are specified just upstream and downstream of the control. The MIT model is also capable of simulating water-surface elevations at controls.

Time steps and computational element lengths can be specified separately for the hydrodynamic solution and water-quality solution. Element lengths can be varied in a reach, leading to better resolution in zones where concentration may change rapidly.

Channel geometry can be specified by several options. Channel shapes may be specified as rectangular, trapezoidal, circular, or irregular. A variable or constant top width may be specified for irregular channel shapes. Elevation versus top width, wetted perimeter, and cross-sectional area are specified for the irregular channel option. Related options allow the specification of ice cover and permit the separation of conveyance and storage areas in the channel.

Limitations on the number of nodes, reaches, inflows, and measured cross sections are not specified. DIMENSION and COMMON statements must be modified to fit the size of the system being modeled using a preprocessor program. Harleman and others (1977, pp. 171-175) offer a program to change the dimensions of the eighteen variables given in Table 1. To conserve storage space, the program dimensions were originally reduced to fit the example problems given in the documentation. Normally these dimensions will need to be increased for field applications.

Inflows into the system are specified in two ways. Lateral inflows are used when the volume of the inflow is important. Injections are used to specify a flux of water-quality constituents when the volume of the inflow is insignificant. Point sources are specified as lateral inflows of zero width.

Longitudinal distances are specified in feet from the upstream end of a reach. Elevations are given in feet, usually referred to mean sea level. U.S. customary units are used by the model.

Hydrodynamic representation

The MIT model uses a finite element technique to solve the equations of momentum and continuity for discharge and water-surface elevation at each mesh point between elements. The hydrodynamic solution is coupled with salinity computations. This solution technique is limited to sub-critical one-dimensional reversing flows.

Convergence of the solution is controlled by the specified time

Table 1

Variables That Must be Given Larger Dimensions to Increase the Capabilities of the MIT Model*

No.	Variable	Definition
1	kjh	Maximum number of hydraulic mesh points in a network
2	kji	Maximum total of table entries for computational channel cross-section data
3	kjq	Maximum number of water-quality mesh points in a network
4	nk	Maximum number of reaches in a network
5	nl	Maximum number of lateral inflows in a network
6	nil	Maximum number of table entries for lateral inflows
7	nzq	Maximum number of table entries for hydraulic boundary conditions
8	ncf	Maximum number of table entries for water-quality boundary conditions
9	nj	Maximum number of injection points
10	nij	Maximum number of table entries for injection points
11	ln	Maximum number of constituents
12	njh	Maximum number of hydraulic mesh points per reach
13	njq	Maximum number of water-quality mesh points per reach
14	nn	Maximum number of nodes ($\geq$nk + 1)
15	ngra	Maximum number of time graphs and hydrodynamics or quality graphs
16	npro	Maximum number of profiles
17	ntem	Maximum number of table entries for meteorological conditions
18	matr	Maximum number of elements in banded node matrix, Maximum value (full matrix) = (2 • no. reaches + no. nodes)^2. For large systems, reduction may be worthwhile. Output will give actual size required.

* Adapted from Harleman and others (1977).

step. The choice of time steps and mesh spacing is based on competing criteria. Computing time is minimized when the time step and element length are maximized. Resolution of detail usually improves as the time step and element length are decreased. Computing cost and resolution have to be balanced.

Since the water-quality predictions are highly dependent on hydraulic conditions, the hydrodynamic model should be calibrated (Harleman and others, 1977). The MIT hydrodynamic model is calibrated by varying the Manning's roughness coefficient.

Water-quality representation

Mass balance equations for each water-quality constituent are also solved using finite-element techniques. A mass-balance equation for each water-quality constituent is written for each element between mesh points. The mass balance considers dispersion, advection, sources, and sinks.

The MIT model simulation primarily involves variables that affect the DO and nitrogen balance. The following water-quality constituent groups are modeled:

1. Salinity.
2. Temperature.
3. Carbonaceous BOD.
4. DO.
5. Fecal coliform bacteria.
6. Nitrogen cycle consisting of:
    a. Ammonia-nitrogen.
    b. Nitrite-nitrogen.
    c. Nitrate-nitrogen.
    d. Phytoplankton-nitrogen.
    e. Zooplankton-nitrogen.
    f. Particulate organic-nitrogen.
    g. Dissolved organic-nitrogen.

Reaction rates for BOD, nitrogen, fecal coliform bacteria, and reaeration are corrected for temperature. BOD and the nitrogen cycle are coupled to the solution of the DO equation.

Temperature can be modeled with the one-dimensional heat conservation equation or can be directly specified as input data. The temperature simulation includes the heat flux through the water surface and the heat content of tributaries and non-point sources. Heat exchange with the channel bottom and bank shading are assumed to be negligible. Required meteorological data include a record of ambient air temperature, relative humidity, wind velocity at two meters (6.56 feet) above the water surface, net flux of solar radiation, net flux of atmospheric radiation, atmospheric pressure, and temperature and volume of inflows.

Oxygen concentrations of inflows are specified as input data and converted to oxygen deficits by the model. The formulation for the oxygen deficit includes dispersion; oxidation of BOD, ammonia, and nitrite; atmospheric reaeration; and tributary oxygen deficit. Carbonaceous BOD decay is simulated with a first-order reaction.

Calculation of the reaeration coefficient is one of the model limitations. The calculation of the reaeration coefficient is limited to the following form,

$$K_2 = C(\theta^{(T-20)}) \frac{V^{0.6}}{H^{1.4}} H \frac{B}{A} \tag{4}$$

in which T = temperature, degrees Centigrade

V = average velocity, feet per second

H = average depth, feet

B = total top width, feet

A = total cross-sectional area, square feet

C = constant, default value = 10.86

θ = temperature correction coefficient, default value = 1.016

The nitrogen cycle is simulated using the seven forms of nitrogen illustrated in Figure 7. These forms include ammonia, nitrite, nitrate, phytoplankton, zooplankton, particulate organic nitrogen, and dissolved organic nitrogen. Seven transformations of these nitrogen forms are modeled. Ammonia is converted to nitrate, through nitrite, by nitrifying bacteria that utilize DO. Ammonia and nitrate are utilized by phytoplankton. Zooplankton grazing converts phytoplankton-nitrogen to zooplankton-nitrogen. Organic nitrogen is released by two processes:

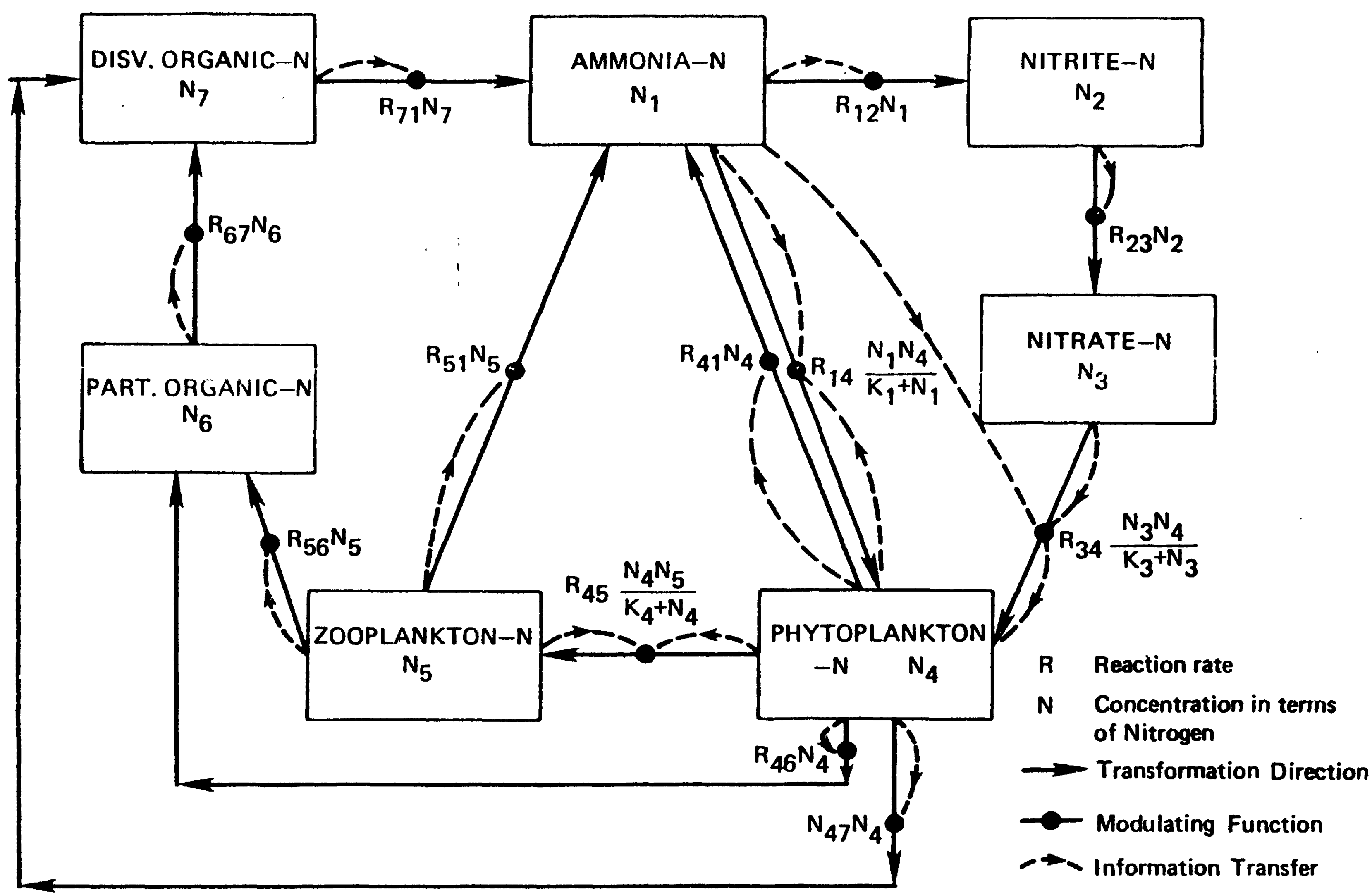

Figure 7. Nitrogen cycle modeled by the MIT model (adapted from Harleman and others, 1977)

organic nitrogen is excreted by living zooplankton and phytoplankton, and cells die and become a source of organic nitrogen. The model simulates the conversion of phytoplankton to particulate and dissolved organic nitrogen and zooplankton to particulate organic nitrogen. Particulate organic nitrogen conversion to dissolved organic nitrogen is simulated as is the hydrolysis of dissolved organic nitrogen to ammonia to complete the nitrogen cycle. Coefficients control the rate of each conversion.

Fecal coliform bacteria are simulated using a first-order reaction with inputs from tributaries and waste sources (injections). Salinity is simulated as a conservative substance.

Program utility

The input data are extensive and complex, but coding forms, detailed instructions, and examples simplify coding. Extensive internal checks are made to determine if program dimensions are sufficient. Results are organized efficiently in tables.

The program can first be run without a complete execution to check the input data. This is a useful option to debug input data without increasing computing cost. The model also has the option to execute the hydraulic solution without going through water-quality calculations.

Allowing or forcing a user to enlarge the program limits adds flexibility that is offset by the need to apply the preprocessing program. When the model is used on computer systems that do not accept the preprocessing program or do not have an equivalent utility program, 4,445 cards or lines must be sorted or edited to find and modify COMMON and DIMENSION statements.

The program does not plot the results, but an option is available to write results to a file. This file can then be plotted using a post-processing program suggested by Harleman and others (1977).

The user is required to determine the time step and element length which may be a disadvantage to the occasional user. When working with irregular cross sections, the user must have a good understanding of the stability of numerical solution schemes. The MIT model does not check for convergence and the documentation hints that some art is involved in choosing time steps. Even the steady-state solution is susceptible to numerical oscilliations of discharge and depth in the longitudinal direction as noted from this work.

In the time frame of this study, the MIT model could not be implemented because of these numerical instabilities. The source program was compiled on an IBM 3033 computer and checked with example steady-flow river data given in the documentation. Those results were the same as the results listed in Harlemann and others (1977).

The model was then applied to data collected on the Chattahoochee River in Georgia (to be described later). The program rejected the field data because limits in DIMENSION and COMMON statements were too small. The redimensioning program recommended by the documentation was applied; the program was compiled once more; and then the compiled version was rechecked with the example data.

After the input Chattahoochee data were corrected, the model ran without declaring DIMENSION and COMMON statements out of bounds. The water-quality solution was turned off to check the hydraulic solution. The hydraulic solution converged but was unstable longitudinally for a steady river flow. Continuity was not preserved.

This instability persisted despite some corrective measures. Element lengths or mesh spacing and time steps were varied over a wide range with limited effect. Next, all inflows and withdrawals, except the headwater, were removed, and the study segment was reduced to a short simple reach. Finally, the channel geometry was greatly simplified. Despite these measures, the longitudinal instability in the hydraulics solution remained without significant change.

It was unclear why this instability persisted. Possibilities include input data error, time-step and mesh-spacing error, or program error. While it was likely that an error was made in specifying the time-step or other data, it was possible that the steady-flow river options have not been fully tested. This writer has not been able to locate published accounts in which the MIT model has been applied to nontidal steady upland streams.

The MIT model is written in Fortran IV. The program has 4,445 statements and 47 subroutines. The model has been applied using the

MIT computing system, the Tennessee Valley Authority CDC system, and the U.S. Department of Energy IBM 370 system in Washington, D.C. The source code and documentation can be requested from the following address:

Corvallis Environmental Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency
Corvallis, OR 97330

Models Comparison

Modeling capabilities

Table 2 summarizes the previous sections on model capabilities and offers a comparison of these capabilities. The comparison is divided into three sections that include: (1) hydraulics computation schemes; (2) water-quality computation schemes; and (3) program utility. The first section compares hydraulic regimes, discretization schemes, and hydrodynamic solution techniques. The second section compares the water-quality solution schemes, the constituents that are modeled, and the sinks and sources for each constituent. The third section compares the utility of each model by outlining the usefulness of input data and results and by describing the general utility of each program.

Although each model was designed to simulate different ranges of conditions with formulations of differing complexity, the models have in common the capability to simulate water quality under conditions of steady flow and constant inflow of water-quality loads, making it possible to compare modeling results. The difference in model complexity depends on the capabilities of each model. The models can be ranked in order of least complex to most complex as Streeter-Phelps, QUAL II, MIT, and WQRRS. The Streeter-Phelps model is designed to simulate steady-state water quality that occurs during critical low-flow periods that last from several days to one or two months. Stream temperature and hydraulics are not simulated and must be specified. The QUAL II model can simulate conditions of steady flow and water quality for critical low-flow periods plus simulate diurnal variation of water quality due to changes in meteorological conditions. The MIT model was designed for dynamic and

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steady-state simulation of flow and water quality in nitrogen-limited waters for time periods on the order of days to months. The WQRRS model simulates dynamic and steady-state flow and water quality for periods up to a year. The water-quality simulation of the WQRRS model covers the widest range of interacting constituents of the four models evaluated. Carbon, nitrogen, and DO cycles, in addition to the food chain, are simulated by the WQRRS model. Unfortunately, the data available for this study were limited to steady flow and water quality. Thus, only part of the range of the dynamic model capabilities were utilized.

Hydraulics modeling

The Streeter-Phelps model does not predict hydraulic conditions. Instead, time of travel or reach-averaged velocity must be specified. In addition, the discretization scheme allows the model to accurately represent the streams.

The QUAL II model can simulate steady low flow in streams by one of two options. Velocity and depth are computed as simple functions of discharge or a trapezoidal cross section is assumed and velocity and depth are derived from a trial-and-error solution of the Manning equation. The discretization scheme allows simulation of multiple branched streams but is limited by the requirement that all reaches must consist of an integer number of equal length elements.

The WQRRS model simulates steady or unsteady discharge with six different options. In addition to simulating vertically-stratified lakes, the model simulates stream networks that may include reversing direction of flow. The discretization scheme can accurately depict stream geometry but is limited to 41 channel cross sections and 10 inflows, withdrawals, and nonpoint source zones for water-quality simulation.

The MIT model was formulated to simulate steady and unsteady flows in rivers and estuaries using a form of the continuity and momentum equations solved by a finite element technique. The MIT model formulation offers the flexibility of allowing the user to determine discretization limits but forces the user to make coding changes for DIMENSION and COMMON statements. Variable element lengths allow an accurate representation of stream geometry.

Stream ecology and water-quality modeling

All four models have the capability to simulate DO, BOD, ammonia, nitrite, nitrate, and coliform bacteria. In addition to these common components, each model has a wide range of other capabilities, some of which are unique among these four models.

Each model simulates water-quality constituents using a mass balance in each element. However, the models treat longitudinal dispersion differently and include different sinks and sources. The Streeter-Phelps model neglects dispersion. The QUAL II model under-predicts dispersion using Elder's equation for straight infinitely wide channels. The WQRRS model includes dispersion, but no details are given of how it is computed. The MIT model formulation potentially offers the greatest flexibility in computing dispersion. An estuary dispersion parameter and Taylor's dispersion coefficient can be specified.

The QUAL II, WQRRS, and MIT models were formulated to simulate temperature, whereas the Streeter-Phelps model was not. The QUAL II and MIT model formulations do not include the moderating effect of a heat flux to the bed. In addition, the MIT model was not designed to simulate solar and atmospheric radiation. The QUAL II and WQRRS models neglect tree shading, but the MIT model was formulated to include tree shading effects in that net solar radiation is required data. When solar radiation is estimated or measured, the estimate or measurement should include effects of tree shading. Overall, the WQRRS temperature submodel offers the greatest flexibility and accuracy. The QUAL II model is simplest to apply to steady-state temperature modeling.

The DO formulation for the Streeter-Phelps, QUAL II, and WQRRS models are similar. Each considers reaeration, BOD decay, nitrification, photosynthesis, respiration, and benthic demand. The MIT model was limited to a DO formulation that considered reaeration, BOD decay, and nitrification. Despite the fact that the nitrogen content of phyto-plankton and zooplankton were included in the formulation for nitrogen, photosynthesis and respiration effects on the DO balance were not mentioned in Equation 3.29 of the MIT model documentation (Harleman and others, 1977, p. 50). Equation 3.29 did include a constant source or sink term that could be used to specify mean photosynthesis, respiration, or benthic demand. However, the documentation does not give details on how to specify that term in the input data, indicating that the computer code may not make allowances for these additional sources or sinks.

The Streeter-Phelps, QUAL II, and WQRRS models are flexible enough to predict or allow specification of K_2. The WQRRS model allows specification of reaeration at a point such as rapids, dams, and waterfalls, and reaeration of tributary inflows between measurement points and the actual entrance into the main stem. Point reaeration can be simulated with a short reach using the Streeter-Phelps model or an inflow with high DO using the Streeter-Phelps model or the QUAL II model.

The MIT model formulation for K_2 was limited to Equation 4 (or see Table 2 under reaeration for the MIT model). Specification of K_2 is not an option, but limited control is available to determine K_2 by specifying the coefficient C in Equation 4.

BOD decay is treated as a first-order process in all four models. The MIT model was formulated to neglect benthic interactions. The Streeter-Phelps and QUAL II models treat sources and sinks of BOD as a first-order process. No allowance is made to simulate the scour or release of BOD from benthos at a constant rate. The WQRRS model simulates dissolved BOD, detritus, and organic sediment. Dissolved BOD and detritus plus organic sediment decay at different first-order rates.

The Streeter-Phelps and QUAL II models simulate benthic DO demand or sediment oxygen demand with a constant rate. The WQRRS model couples organic sediment decay and benthic plant photosynthesis and respiration to the DO balance. The MIT model was designed so that benthic interactions with DO were neglected.

For the Streeter-Phelps model, net daily primary productivity and chlorophyll a concentrations can be specified. Neither are simulated and chlorophyll a only affects orthophosphate concentrations. The QUAL II model simulates phytoplankton and primary productivity. The QUAL II phytoplankton submodel links phytoplankton growth to levels of light, nitrate, and orthophosphate. The WQRRS model simulates benthic algae, phytoplankton, zooplankton, aquatic insects, fish, and benthic animals. These WQRRS model components are linked to DO, nitrogen, carbon, and phosphorus balances. The MIT model was formulated to simulate phytoplankton and zooplankton effects on the nitrogen balance.

There are also a number of other capabilities. Except for the MIT model, all the models were formulated to simulate orthophosphate. The Streeter-Phelps and QUAL II models simulate three arbitrary conservative substances. The Streeter-Phelps model simulates anoxic conditions. The QUAL II model simulates one arbitrary non-conservative substance and computes flow augmentation and BOD reduction needed for point sources to meet specified levels of DO in the stream being simulated. The QUAL II model also makes diurnal predictions of water quality given a record of meteorological data. The WQRRS model simulates organic and inorganic sediments; unit toxicity; and pH, alkalinity, and inorganic carbon. The MIT model was formulated to simulate salinity.

Program utility

Data coding requirements and the usefulness of the four models are related to the manner in which the models are applied. The Streeter-Phelps and QUAL II models are well adapted for steady-state modeling. The WQRRS model is best adapted for dynamic modeling of stream water-quality. The MIT model was designed to simulate water quality in nitrogen-limited estuaries. The added dynamic modeling capabilities of the MIT and WQRRS models make it much more difficult to code data and calibrate the models to simulate steady-state water quality.

In terms of program utility, the QUAL II model is very good. The documentation offers the necessary detail about the theory and formulation of the model in one section (Roesner, Giguere, and Evenson, 1977A) and describes the use of the model and gives coding sheets and instructions in a second section (Roesner, Giguere, and Evenson, 1977B). Perhaps the QUAL II documentation is one of the best examples of how documentation for models should be written.

The data coding formats are well organized and the QUAL II model makes numerous checks of the data. The printed results are also efficiently organized and options are available to suppress unwanted output. However, without an option to plot results and measurements, calibration can be tedious. Nevertheless, other versions of the QUAL II model make allowances to plot the results.

The Streeter-Phelps model documentation lacks some detail on model formulation and coding instructions. However, two good example applications compensate for some lack of detail. Care must be exercised in coding and checking the data because data coding formats are inefficient and the model does not check the coded data. Model output is lengthy but the results are efficiently summarized in plots of results and measurements that greatly simplify model calibration.

The WQRRS model documentation is also vague in some areas. A model listing and coding sheets are not given. The example simulation seems to be based on hypothetical or idealized data. However, the program listing is available upon request, a common data coding format and detailed instructions simplify coding, and separate reports describing actual field applications are available to compensate for these minor deficiencies. The model checks the coded data and adjusts the time step to converge to a stable condition.

The MIT model documentation also lacks some detail, especially in how to implement the model. The choice of the time step and mesh spacing requires some experience in dynamic flow routing simulation. Coding sheets compensate for the lack of detailed instructions for data coding.

Based on the model documentation, the MIT model seems best adapted to modeling water quality in estuaries in which phytoplankton is nitrogen limited, benthic interactions are insignificant, and reaeration is not very important. The WQRRS model seems best adapted to dynamic stream water-quality modeling. Steady-state modeling with these dynamic models is tedious and cannot be justified unless the simulation of benthic algae and other aquatic plants and animals, pH, inorganic carbon, suspended sediment, or organic sediment is necessary. The QUAL II model simulates diurnal variations of water quality for steady discharge. The model also has the added flexibility of modeling one arbitrary nonconservative and three conservative substances. Both the QUAL II and Streeter-Phelps models are well adapted to steady-state conditions. The Streeter-Phelps model simulates nitrogenous BOD and anoxic conditions.

PART IV: DESCRIPTION OF FIELD DATA

Data Base

Selection criteria

An important aspect of this model comparison study involved applying each model to a common data base collected under actual field conditions. That data base was formed from three data sets derived from USGS studies using the following criteria. First, the study had to be an intensive synoptic data-collection effort that collected high quality data. Second, the three data sets had to cover a wide range of physical, chemical, and biological conditions. Third, the widest possible geographic distribution was desired. Fourth, the data sets had to provide the information necessary to apply each of the four models, including channel geometry for the dynamic models. Fifth, data was chosen so that as-many-as-possible independent determinations of model coefficients were available. Finally, calibration and verification required at least two independent data-collection surveys in each river study. The calibration data were necessary to derive criteria for the choice of model coefficients. The independent verification data were used to test the predictive capability of models after calibration.

Unfortunately, the data base was limited to steady-state conditions. Dynamic water-quality data, adequate for model evaluation, could not be located. Therefore, the dynamic models considered in this evaluation were used to simulate the steady conditions described by the data.

Selected data

Based on these criteria, data sets collected on the Chattahoochee River in Georgia, Willamette River in Oregon, and Arkansas River in Colorado were selected. The Chattahoochee and Willamette River studies were river-quality assessments in which the USGS studied a wide range of water-quality problems, developing and verifying new methods of sampling, analysis, and evaluation as needed. The Arkansas River study was a cooperative study undertaken by USGS personnel.

These studies of the Willamette River between Salem and Portland, Oregon; the Chattahoochee River between Atlanta and Whitesburg, Georgia; and the Arkansas River between Pueblo and Nepesta, Colorado, covered a wide range of conditions. During these critical flow studies, the Willamette River had high discharge, great depth, low velocity, and mild bed slope. The Arkansas River had high velocity, steep bed slope, low discharge, and shallow depth. The Chattahoochee River had moderate depth and bed slope.

Each study has a unique set of measurements useful for testing model options. The Willamette River had significant benthic oxygen demand in the lower reaches. The Chattahoochee River received loadings of organic nitrogen and ammonia that was oxidized to nitrate that builds up in the stream. In the Arkansas River, reaeration coefficients were measured and 5-day BOD was approximately equal to ultimate BOD.

Decay rates of BOD were low for the Willamette and Chattahoochee rivers and high for the Arkansas River. The Willamette River stayed at a steady state for 1- to 3-month periods. The normal hydropower generation schedule for the Chattahoochee River was modified to maintain 2 to 4 days of steady flow for the four study periods. At low flow (2.8 cubic meters per second or 100 cubic feet per second), over half the flow in the Arkansas River was due to a municipal waste source, making mild fluctuations from steady state in the waste source more important to in-stream water quality.

Chattahoochee River-Quality Data

Chattahoochee River studies

The Chattahoochee River in Georgia has been the subject of an intensive river-quality assessment undertaken by the USGS. This three-year study was one of several demonstration projects designed to assess and provide information concerning the water resources of the Nation's rivers. The specific purpose of this study was to assess point and nonpoint source pollution effects on river DO levels and phytoplankton populations in West Point Lake (Stamer and others, 1979), an impoundment downstream of the study segment.

Stamer and others (1979) studied the DO content in the Chattahoochee River between Atlanta and Whitesburg, Georgia. Data were collected July 11-12, 1976; August 28-31, 1976; September 5-8, 1976; and May 30 - June 2, 1977 to identify and estimate the effects of point and nonpoint sources. A steady-state Velz water-quality model was calibrated, verified using independent data sets, and then used to predict future water quality for a number of resource management alternatives. The predictions included increased waste loads expected from the growth of the Atlanta metropolitan area.

Other studies focused on transient flows in the Chattahoochee River. Jobson and Keefer (1979) modeled transient flows and temperatures below Buford Dam, a peaking power facility. Dye concentrations, temperature, and discharge were accurately simulated, but other water-quality constituents were not measured or simulated.

Faye, Jobson, and Land (1979) studied the thermal and flow regimes of the Chattahoochee River between Atlanta and Whitesburg, Georgia. Dynamic simulation of temperature and flow led the authors to conclude that waste heat from coal-fired power plants near Atlanta balanced cold water discharged upstream at Buford Dam. Resulting mean annual temperatures of the combined effects were within 0.5 degrees Centigrade (1 degree Fahrenheit) of natural temperatures. The unsteady operation of Buford Dam led to larger temperature variations than those expected under natural conditions.

Miller and Jennings (1979) and Miller (1981) simulated the steady-state water quality of the Chattahoochee River using the Streeter-Phelps model. Their studies focused on the nitrogen and DO balance of the river. McConnell (1979) studied the quality of urban runoff into the Chattahoochee River.

The complete water-quality data set collected during the Chattahoochee river-quality assessment can be retrieved from the WATSTORE data management system of the USGS (U.S. Geological Survey, 1977). WATSTORE entries are transferred to the EPA data-management system STORET on a weekly basis. Edwards (1980) describes the overall water data-management network that includes WATSTORE and STORET. Data retrieval by any interested party is possible for a nominal charge at 41 locations nationwide.

Basin description

The upper Chattahoochee River lies within the Atlanta Plateau of the southern Piedmont physiographic province (Faye, Jobson, and Land, 1979). The topography is characterized by low hills separated by narrow valleys. Small mountains, not exceeding 610 meters (2,000 feet) in elevation are found along the northern divide in the Blue Ridge Mountains. Mean basin elevation is about 305 meters (1,000 feet). The basin above West Point Dam has 5,158 square kilometers (1,990 square miles) of area (Figure 8).

The climate of the area is influenced by the Gulf of Mexico and Blue Ridge Mountains. Rainfall averages 127 centimeters (50 inches) per year. The average temperature is 16 degrees Centigrade (61 degrees Fahrenheit). Air temperatures are highest from June to August but rarely exceed 38 degrees Centigrade (100 degrees Fahrenheit).

Stream description

The Chattahoochee River flows southwest in the study reach between Atlanta and Whitesburg. The channel drains metropolitan Atlanta between the Atlanta gage and the gage near Fairburn. This reach receives tributary inflows from urban areas, waste treatment plant discharges, and power-plant waste heat discharges. Water is withdrawn at the Atlanta Waterworks and at the power-plant complex consisting of plants Atkinson and McDonough. Between Fairburn and Whitesburg, forests and farmlands are drained by tributaries. Table 3 and Figure 8 give each tributary and the location at which it enters the river.

The forty-one channel cross sections measured between RK 487.48 and 418.10 (Atlanta to Whitesburg, RM 302.97 to 259.85) are approximately trapezodial in shape with high steep banks and sand beds. However, rock beds and shoals do occur. These can be found near the Atlanta gage (RK* 487.48 or RM* 302.97), below the mouth of Nickajack Creek (RK 474.36 or RM 295.13), and between Capps Ferry Bridge (RK 436.34 or RM 271.19)

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* Abbreviation for river miles or river kilometers upstream of the mouth of the Chattahoochee River at its intersection with the Flint River.

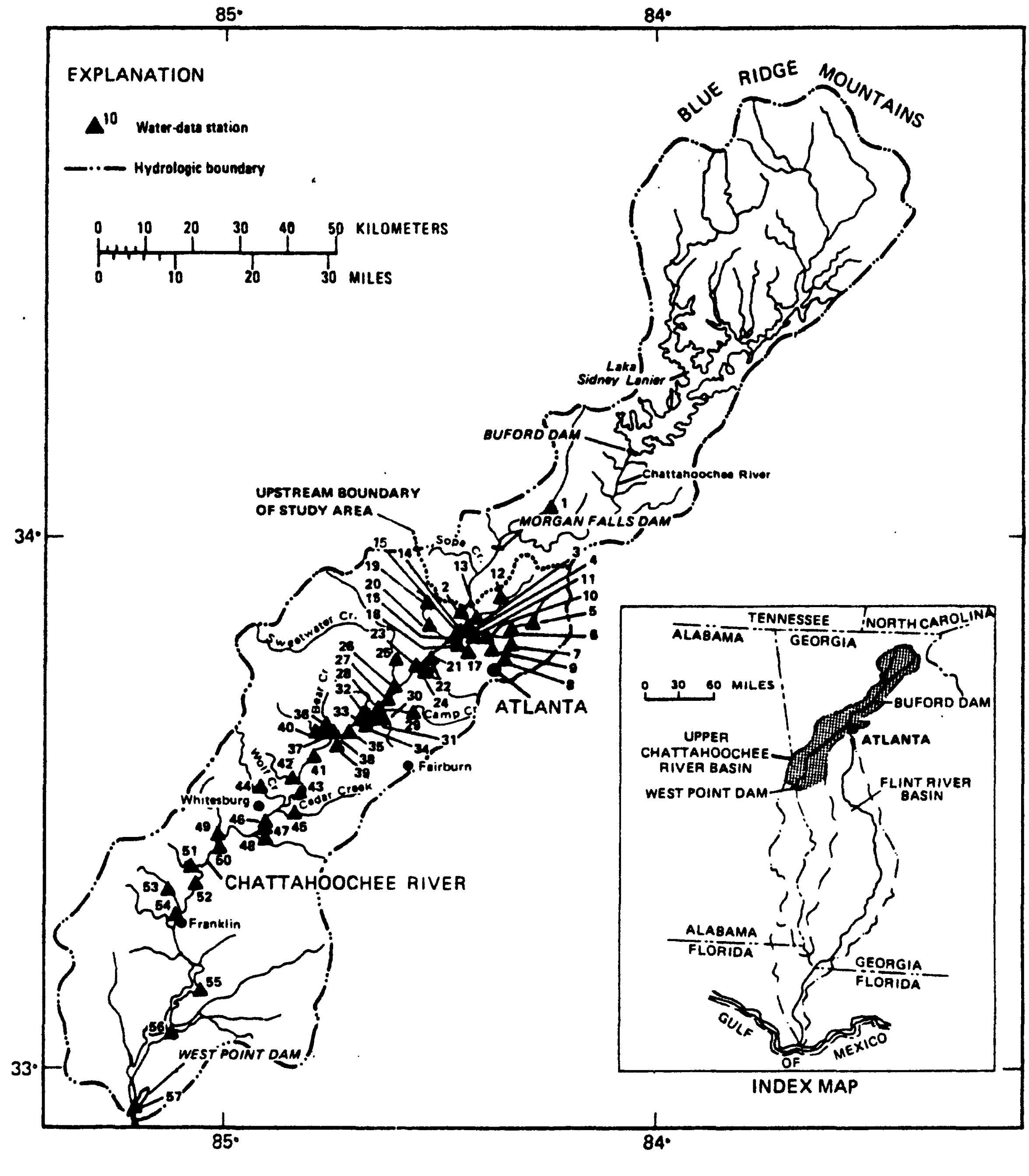


Figure 8. Map showing the upper Chattahoochee River Basin; the study reach between Atlanta and Whitesburg, Georgia; and water-quality sampling sites (adapted from Stamer and others, 1979). See Table 3 for identification of sampling sites.

Table 3

Identification of Sampling Sites Shown in the Chattahoochee Basin Map (Figure 8)*

Map Reference No.	Station Name	River Mile (1 mi = 1.6 km)
1	Big Creek near Alpharetta	317.37
2	Chattahoochee River at Atlanta	302.97
3	Chattahoochee River (Atlanta Intake) at Atlanta	300.62
4	Cobb Chattahoochee WTF near Atlanta	300.56
5	North Fork Peachtree Creek Tributary (Meadowcliff Drive) near Chamblee	------
6	North Fork Peachtree Creek at Buford Highway near Atlanta	------
7	South Fork Peachtree Creek at Atlanta	------
8	Clear Creek at Piedmont Park at Atlanta	------
9	Tanyard Branch at 26th Street extension at Atlanta	------
10	Peachtree Creek at Atlanta	300.52
11	Woodall Creek at DeFoors Ferry Road at Atlanta	------
12	Nancy Creek Tributary near Chamblee	------
13	Nancy Creek at Randall Mill Road at Atlanta	------
14	R. M. Clayton WTF at Atlanta	300.24
15	Plants Atkinson-McDonough at Atlanta	299.11
16	Chattahoochee River at SR 280 at Atlanta	298.77
17	Hollywood Road WTF at Atlanta	------
18	Proctor Creek at SR 280 at Atlanta	297.50
19	Nickajack Creek (USAF Plant No. 6 outfall) near Smyrna	------
20	Nickajack Creek at Cooper Lake Road near Mableton	295.13
21	Chattahoochee River at SR 139 near Mableton	294.65
22	South Cobb Chattahoochee WTF near Mableton	294.28
23	Utoy Creek WTF near Atlanta	291.60
24	Utoy Creek at SR 70 near Atlanta	291.57
25	Sweetwater Creek near Austell	288.58
26	Sweetwater Creek WTF near Austell	288.57
27	Chattahoochee River (SR 166) near Ben Hill	286.07
28	Camp Creek WTF near Atlanta	283.78
29	Camp Creek at Enon Road near Atlanta	283.54
30	Deep Creek at SR 70 near Tell	283.27
31	Chattahoochee River (SR 92) near Fairburn	281.88
32	Anneewakee Creek at SR 166 near Douglasville	281.47
33	Anneewakee Creek WTF near Douglasville	281.46
34	Three-river interceptor	281.45
35	Pea Creek at SR 70 near Palmetto	277.40
36	Bear Creek at SR 166 near Douglasville	275.95
37	Bear Creek (SR 166) WTF near Douglasville	275.94
38	Chattahoochee River (above Bear Creek) near Rico	275.81
39	Bear Creek at SR 70 near Rico	274.49
40	Dog River at SR 166 near Fairplay	273.46
41	Chattahoochee River (Capps Ferry Bridge) near Rico	271.19
42	Wolf Creek at SR 5 near Banning	267.34
43	Chattahoochee River at Hutcheson's Ferry near Rico	265.66
44	Snake Creek near Whitesburg	261.72
45	Cedar Creek at SR 70 near Roscoe	261.25
46	Chattahoochee River (U.S. Alt. 27) near Whitesburg	259.85
47	Plant Yates	259.70
48	Wahoo Creek at Arnco Mills	256.55
49	Whooping Creek near Lowell	250.87
50	Plant Wansley	249.20
51	Chattahoochee River at Bush Head Shoals near Franklin	246.93
52	Pink Creek near Centralhatchee	244.89
53	Centralhatchee Creek at U.S. 27 near Franklin	236.51
54	Chattahoochee River at U.S. 27 at Franklin	235.46
55	Chattahoochee River at SR 219 near LaGrange	221.26
56	Chattahoochee River at SR 109 near Abbottsford	210.67
57	West Point Lake at dam pool near West Point	202.36

* Adapted from Stamer and others (1979).

and Whitesburg (RK 418.10 or RM 259.85) (Figure 9). Thirty-six sections were obtained from the COE. Five others were measured during the water-quality assessment (Faye, Jobson, and Land, 1979).

Channel geometry data from the COE flood study were supplemented with data collected during a steady low-flow period. USGS personnel floated down the reach in a boat and measured widths and depths at about 366-meter (1,200 feet) intervals. Depth (Stamer and others, 1979) was measured near cross sections previously measured by the COE. Water-surface elevations were measured at bridges where known elevation markers were located. River discharge was measured at several points. In addition, tributaries, withdrawals, and treatment plant discharges were measured or estimated.

These data were used for two purposes. First, Stamer and others (1979, p. 38) used reach-averaged depth and velocity, along with discharge, to calculate reach volumes and travel times. Second, reach volumes and travel times were used in a Velz rational model to calculate reaeration. These unpublished data are presented in Tables 4 and 5 for completeness.

The stream had a moderate slope of 0.0003. Figure 9 shows the channel thalweg and the water-surface profile at low flow. Weirs at RK 487.48 (RM 300.62) and RK 481.25 (RM 299.1) created pumping pools for the Atlanta water-supply facility and the Atkinson and McDonough power plants.

Stream hydrology

Streamflows were affected by basin rainfall and regulation by Buford, Morgan Falls, and West Point dams. Streamflow at Atlanta was dominated by regulation upstream. Flood peaks increased in the downstream direction as basin area increased and reservoir regulation effects were moderated. The cyclic nature of discharge was due to the weekly (7-day) schedule of power production taking place on weekdays and a minimum flow being maintained on weekends.

Faye, Jobson, and Land (1979) analyzed long-term affects of regulation on streamflow and temperature and found that peak flows were smaller in magnitude and duration and minimum flows were higher when regulated flows were compared to previous unregulated flows. Stamer and

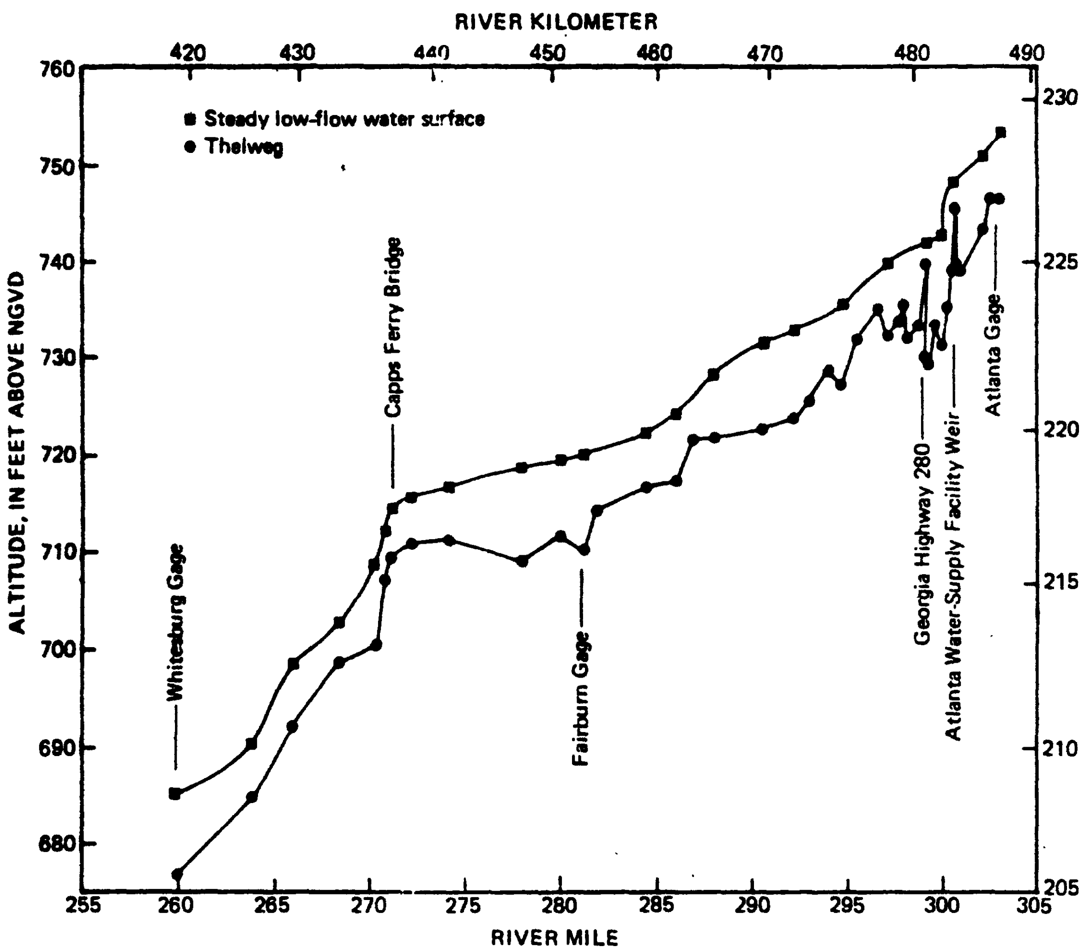


Figure 9. Thalweg and steady low-flow profile of the Chattahoochee River from Atlanta to Whitesburg, Georgia (adapted from Faye, Jobson, and Land, 1979). NGVD is the National Geodetic Vertical Datum of 1929.

Table 4

Reach-Averaged Depths, Widths, Travel Times, and Steady Low Flows for the Chattahoochee River, Georgia*

Tributary or Location	Beginning** River Mile	Cumulative Travel Time hrs.	Measured Discharge of Trib. or River ft^3/s	Depth ft	Width ft
Atlanta Gage	302.97	0	1248	4.25	234
Cobb Co. STP	300.56	3.45	16	4.02	236
Peachtree Ck.	300.52	3.52	37	4.32	236
Clayton STP	300.24	3.86	128	4.26	237
Power Plants	299.11	5.50	†	5.14	238
SR-280 Bridge	298.77	5.94	1288	4.24	238
I-285 Bridge	297.75	7.08	1429	5.12	239
Proctor Ck.	297.50	7.20	6	4.22	239
Nickajack Ck.	295.13	9.56	13	5.12	241
SR-139 Bridge	294.65	9.96	1448	4.25	242
South Cobb STP	294.28	10.30	13	5.09	242
Utoy STP	291.60	12.98	19	4.29	244
Utoy Ck.	291.57	13.01	10	5.18	244
CR at Buzzard Is.	290.57	13.65	1490	4.34	244
Sweetwater Ck.	288.58	15.73	130	5.14	246
SR-166 Bridge	286.07	18.33	1620	5.09	248
Camp Ck. STP	283.78	20.78	7	5.06	250
Camp Ck.	283.54	21.06	10	5.10	250
Deep Ck.	283.27	21.35	10	5.11	250
Fairburn (SR-92)	281.88	22.75	1646	5.13	251
Anneewakee Ck.	281.47	23.19	17	5.08	252
Pea Ck.	277.40	27.53	4	5.16	255
Upper Bear Ck.	275.95	29.00	11	5.02	256
CR near Bear Ck.	275.81	29.20	1678	5.19	257
Lower Bear Ck.	274.49	30.66	6	5.00	257
Dog River	273.46	31.80	42	5.20	258
Capps Fy. Bridge	271.19	34.03	1726	5.20	260
Wolf Ck.	267.34	38.12	13	5.10	263
CR at Hutcheson's Fy.	265.66	39.40	1739	5.19	264
Snake Ck.	261.72	42.56	53	5.01	268
Cedar Ck.	261.25	42.96	14	5.39	268
CR at Whitesburg††	259.85	44.28	1799	5.03	269

* Interpreted from office notes furnished by USGS, Doraville, Ga. District Office.

** Faye, Jobson, and Land (1979), Stamer and others (1979), and field notes, disagree on exact river locations; river mile designations of Stamer and others are used.

† Discharge not measured.

†† End of the study reach.

Note: 1 mile = 1.61 kilometers, 1 ft^3/s = 0.028 m^3/s, 1 foot = 0.3048 meters, SR is an abbreviation for State Route, CR is Chattahoochee River, Fy. is Ferry, STP is Sewage Treatment Plant, and Ck. is Creek.

Table 5

Water-Surface Elevations and Discharge Measurements on the Chattahoochee River, Georgia

Location	River Mile	Water-Surface Elevation ft above 1929 NGVD	River Discharge* ft^3/s
Atlanta gage	302.97	753.9	1140
Atlanta water treatment plant weir	300.62	748.0	995**
Power plant weir	299.11	742.4	1277†
State Route 280 Bridge	298.77	740.9	1277
State Route 139 Bridge	294.65	734.4	1288††
State Route 166 Bridge	286.07	724.4	1349‡
Fairburn‡‡	281.88	720.6	1770
		720.97	2020
Capps Ferry Bridge	271.19	715.00	2223
Whitesburg	259.85	685.59	2350

* Some tributary discharges were not available.

** Water plant withdrawal = 140 ft^3/s.

† South Cobb STP + Peachtree Ck. ≅ 142 ft^3/s, Clayton STP ≅ 140 ft^3/s.

†† Proctor Ck. ≅ 10 ft^3/s, Nickajack Ck. ≅ 30 ft^3/s.

‡ South Cobb STP + Utoy STP + Utoy Ck. ≅ 30 ft^3/s, Sweetwater Ck. ≅ 297 ft^3/s.

‡‡ Survey began at Atlanta gage and proceeded downstream to Capps Ferry Bridge where the survey stopped for nightfall and began anew the next morning. Discharge and water-surface elevation increased overnight.

Note: 1 mile = 1.61 kilometers, 1 ft^3/s = 0.028 m^3/s, and 1 foot = 0.3048 meters.

Data in this table were adapted from a written communication by Harvey Jobson, Hydrologist, USGS, Bay St. Louis, Miss.

others (1979) concluded that late autumn flows were normally the lowest on an annual basis. Winter stream temperatures above the Atkinson and McDonough power plants were increased while summer stream temperatures were decreased. The power plant increased river temperatures by an annual average of 2 degrees Centigrade (4 degrees Fahrenheit). On an annual basis, heated water discharges balanced cold water discharged by upstream dams.

Water-quality description

The quality of the Chattahoochee River below Atlanta was influenced by two factors related to man's activities. Point sources and distributed sources add organic material and nutrients causing degradation of stream water quality. Upstream hydropower releases affect water quality by dilution and flushing out the stream on a weekly basis. Typically, point sources exert considerable influence during weekend periods of minimum flow. When weekday peaking-power operations begin, dilution and flushing cause water-quality improvements illustrated in Figure 10.

During summer minimum flows, DO is depressed to 4 to 5 milligrams per liter below Atlanta from near saturation above Atlanta. The recovery of DO levels begins between Franklin and Whitesburg, depending on travel times and loadings. Nutrients, BOD, and coliform bacteria increase from low levels upstream of Atlanta to high levels downstream.

Stamer and others (1979, p. 37) indicated photosynthesis was not significant in the study reach. This was supported by measurements of phytoplankton (cells per milliliter), periphyton biomass, and chlorophyll a and b collected at three sites over the three-year study period. In addition, the weekly flushing scours the channel as indicated by upstream erosion problems (Jobson and Keefer, 1979). Not only will channel scour remove attached aquatic plants, but the associated turbidity and short travel times will also restrict phytoplankton growth in the nutrient-enriched waters below Atlanta. This is confirmed later with modeling results showing that nitrate builds up from nitrification in the stream and nitrate and orthophosphate are not removed from the stream. In addition, the DO balance does not indicate a significant benthic oxygen demand.

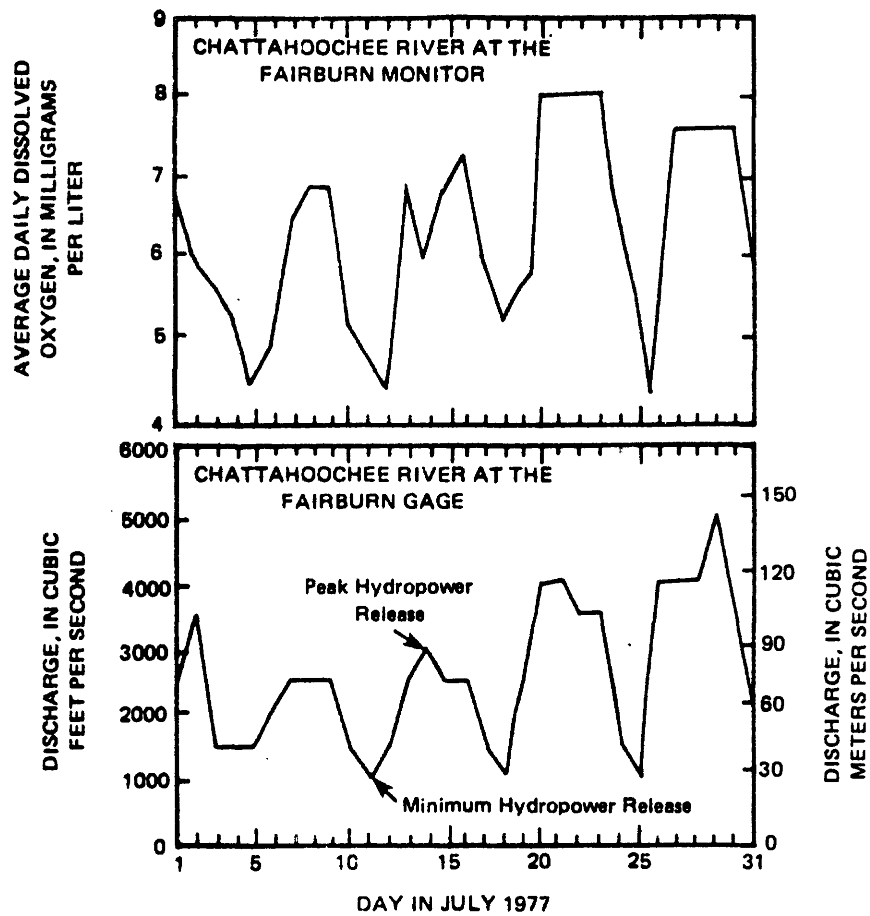


Figure 10. Dissolved-oxygen concentrations and mean daily discharge during July 1977

Data collection

Four intensive synoptic data-collection studies were undertaken in the summers of 1976 and 1977 when problems associated with low flow were compounded by high water temperatures. These steady-state studies occurred on the following dates:

1. July 11-12, 1976.
2. August 28-31, 1976.
3. September 5-8, 1976.
4. May 30-June 2, 1977.

These intensive studies were part of an overall data-collection program that extended from October 1975 to September 1977. Data were collected at a fewer number of points over longer time increments during the overall study period than in the intensive studies.

A tabular summary of measured water-quality constituents is given in the "Data Comparison" section that follows. Temperature, specific conductance, pH, DO, and streamflow were measured in situ. Water samples were width and depth integrated. Limited sampling across the stream at Fairburn showed little or no lateral variation. Samples were chilled and filtered in the field as needed. Analysis of samples and field measurements were made by standard methods documented by the USGS (Stamer and others, 1979, p. 17-18).

Reaeration coefficients for the Chattahoochee River were measured previously by Tsivoglou and Wallace (1972) during several studies that covered the middle segment of the reach considered in the USGS study. Tsivoglou and Wallace (1972) found reaeration coefficients did not change significantly as discharge varied during low flow, making it possible in this specific study to use the same reach-averaged coefficients for all four data sets. Reaeration coefficients for the other reaches were predicted by an equation developed by Tsivoglou and Wallace from these Chattahoochee River data. The coefficients used for this study are listed in Table 6.

The May-June 1977 data set best documents waste loads into the river. Multiple samples were taken at all seven wastewater treatment plant effluents and all tributaries flowing into the river. Mean constituent concentrations were determined from grab samples collected

Table 6

Reaeration Coefficients for the Chattahoochee River, Georgia, Atlanta to Whitesburg*

Reach (RM)	Reaeration Coefficient (1/day base e 25°C)
302.97 - 300.50	1.75
300.50 - 299.00	0.54
299.00 - 294.00	1.21
298.77 - 292.80	1.46
292.80 - 288.54	1.03
288.54 - 286.18	1.03
286.18 - 281.91	0.84
281.91 - 280.21	0.47
281.50 - 277.50	0.29
277.50 - 276.00	0.57
276.00 - 271.25	0.98
271.25 - 267.25	2.66
267.25 - 262.25	2.77
262.25 - 259.75	1.17

* Coefficients for the middle reaches were derived from tracer measurements by Tsivoglou and Wallace (1972, p. 149); coefficients for the upstream and downstream reaches follow from calculations using the equation developed by Tsivoglou and Wallace (1972, p. 248) from Chattahoochee River data.

by the USGS. Tributary discharges were measured, but treatment plant discharges were computed from monthly plant-operator reports submitted to the Georgia Environmental Protection Division. A comparison of a few waste loads reported by treatment plant operators and loads computed by the USGS was favorable. Despite this, loads from the R. M. Clayton Plant were still underestimated according to modeling results presented later. The May-June 1977 data set did not include as many water-quality constituents as the August 1976 data set but discharge, DO, BOD, nitrogen species, and temperature were better documented with multiple samples of waste inflows and tributary inflows.

The August 1976 data set had the most extensive range of constituents measured. In addition to data describing DO, BOD, nitrogen species, and temperature, this data set contained information describing coliform bacteria, arsenic, chromium, copper, lead, zinc, suspended solids, chemical oxygen demand, and pH. Nevertheless, these data were limited to single samples of most inflows and infrequent river sampling.

The other two data sets were less complete. Natural tributaries were not sampled during the September 5-8, 1976 period and most waste treatment plant effluents were only sampled for 5-day BOD. Measurements of DO and water temperature were made for a couple of waste outfalls. Single grab samples were taken from those waste effluents sampled. No data were collected to describe nitrogen species, but information was available from the previous week, collected at the time of the August 30-31, 1976 data collection survey.

The July 1976 data included single samples from all significant tributaries but none from wastewater treatment plant effluents. Laboratory analysis included determination of organic nitrigen, ammonia, nitrite, and nitrate. Missing data for waste treatment plants were estimated from data collected during the August 1976 study.

Willamette River-Quality Data

Willamette River studies

The river-quality assessment of the Willamette River Basin, Oregon, was the first intensive river-quality assessment undertaken by the USGS and served as a prototype study for other river-quality studies (Rickert and Hines, 1975). Specific problems studied were (Rickert, Hines, and McKenzie, 1976):

1. Effects of waste discharge on DO resulting from population and industrial growth.
2. Potential for nuisance algal growths.
3. Possibility of trace-metal accumulations in bottom sediments.
4. Potentially harmful effects on river quality and land use due to accelerated erosion resulting from population and industrial growth.

The Willamette study was interesting for historical reasons. Severe DO depletion had occurred in the lower reaches of the river. On occasion, DO levels dropped to zero and hampered upstream migration of salmon. Recreation was curtailed and esthetic values diminished as well as other water uses being affected. DO-related problems were compounded by high fecal coliform bacteria concentrations, floating and benthic sludge, sulfurous odors, and sewage fungus (Hines and others, 1977).

These problems have been overcome by basinwide secondary treatment of point sources, chemical-recovery processes implemented by paper and pulp mills, routing the combined sewer overflows from Portland out of the basin to the new Columbia River treatment plant, and flow augmentation from headwater reservoirs. DO levels are now acceptable and water-contact recreation has returned. As of 1977 the Willamette River was the largest river basin in the Nation in which all point sources received secondary treatment.

Recent studies indicate changes in the physical, chemical, and biological characteristics of the river have occurred. Dredging and channelization have changed water travel times, reaeration, and benthic deposits. The releases from headwater reservoirs have increased the annual low flow and have controlled algae growths. Implementation of basinwide secondary treatment has changed in-stream deoxygenation rates of organic material.

Data surveys were conducted during the summers of 1973 and 1974 to define the water quality of the Willamette River. The results were documented in the USGS Circular Series 715-A through 715-M. Most of the Willamette River data used in this study were taken from these reports. A limited amount of information was obtained from the investigators involved in more recent studies and from engineers involved in flood insurance studies of the lower Willamette River.

Several problems were encountered in gathering these data. Raw data or basic measurements were not recorded in a data report or stored on a computer data-storage system. Therefore, much of the necessary data had to be interpolated from charts and graphs. The first study (July-August 1973) lacked nitrogen data. Finally, channel geometry data were gathered from Willamette River flood insurance studies where data collection was tailored to high flow conditions rather than low flow conditions.

Basin description

The Willamette River Basin is located in northwest Oregon, as shown in Figure 11, and has an area of 29,800 square kilometers (11,500 square miles). The basin is roughly rectangular in shape and is bound on the east by the Cascade Mountains, on the west by the Coastal Range, on the south by the Calapooya Mountains, and on the north by the Columbia River. The State's three largest cities, Portland, Salem, and Eugene, are located within the basin, representing 70 percent of the population of Oregon.

Land elevations vary from less than 3.1 meters (above 1929 NGVD) (10 feet) at the mouth of the Willamette River below Portland, to 140 meters (450 feet) near Eugene on the valley floor (see Figures 11 and 12), and to more than 3,050 meters (10,000 feet) in the Cascade Range. The Coast Range varies in elevation from 300 to 600 meters (1,000-2,000 feet) with some peaks exceeding 1,200 meters (4,000 feet).

The Willamette Basin has a modified marine climate controlled by surrounding mountain ranges (Figure 11) and the Pacific Ocean. The climate is characterized by wet, cloudy winters and clear, dry summers. Daily average temperatures range from 1.7 degrees Centigrade (35 degrees Fahrenheit) to 28 degrees Centigrade (83 degrees Fahrenheit) on the valley floor and from -6.7 degrees Centigrade (20 degrees Fahrenheit) to 24 degrees Centigrade (75 degrees Fahrenheit) on the crest of the Cascade Range.

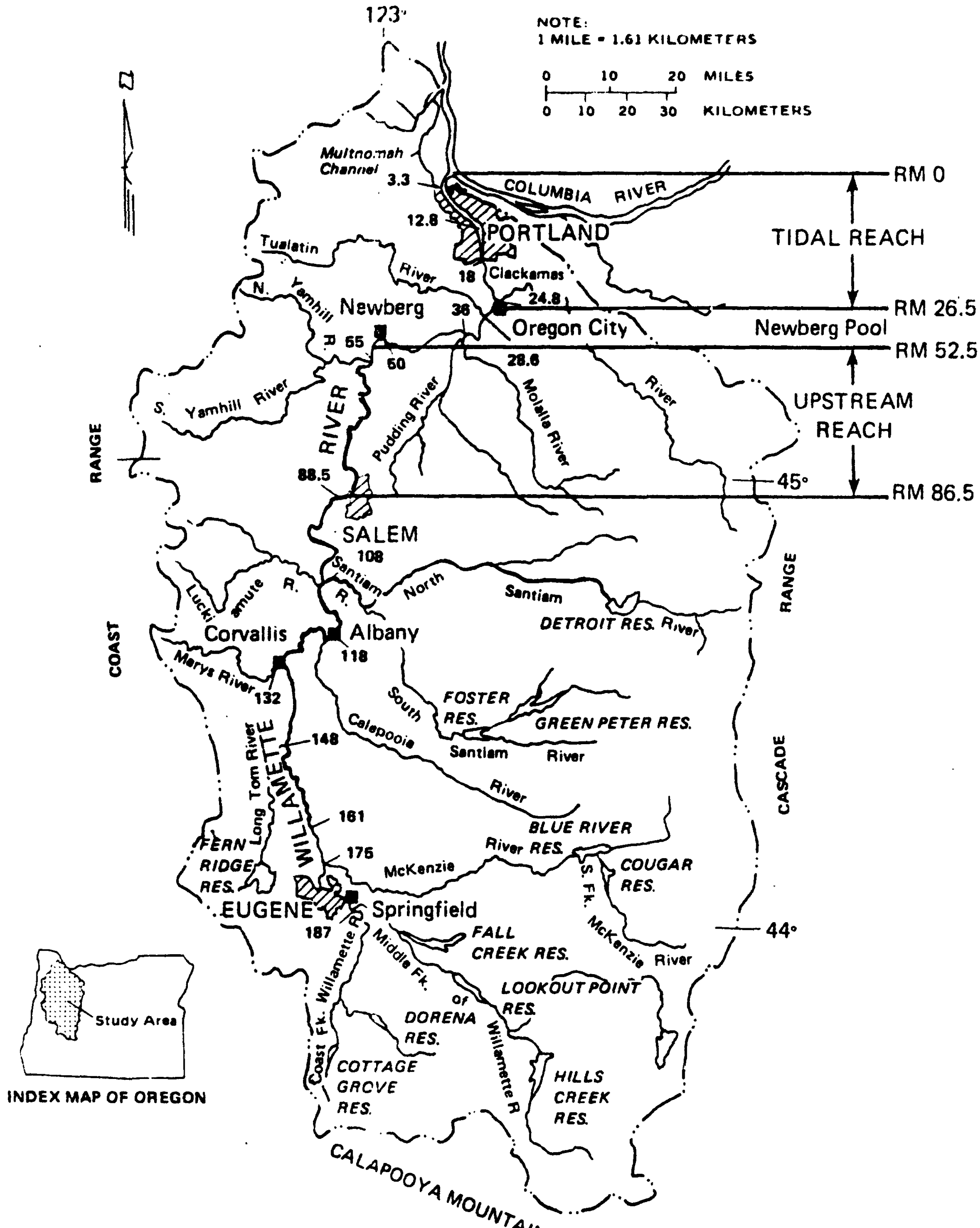


Figure 11. Willamette River Basin, Oregon, showing main stem river miles, significant tributary streams, headwater reservoirs, and three distinctly different reaches in the study area (adapted from Hines and others, 1977)

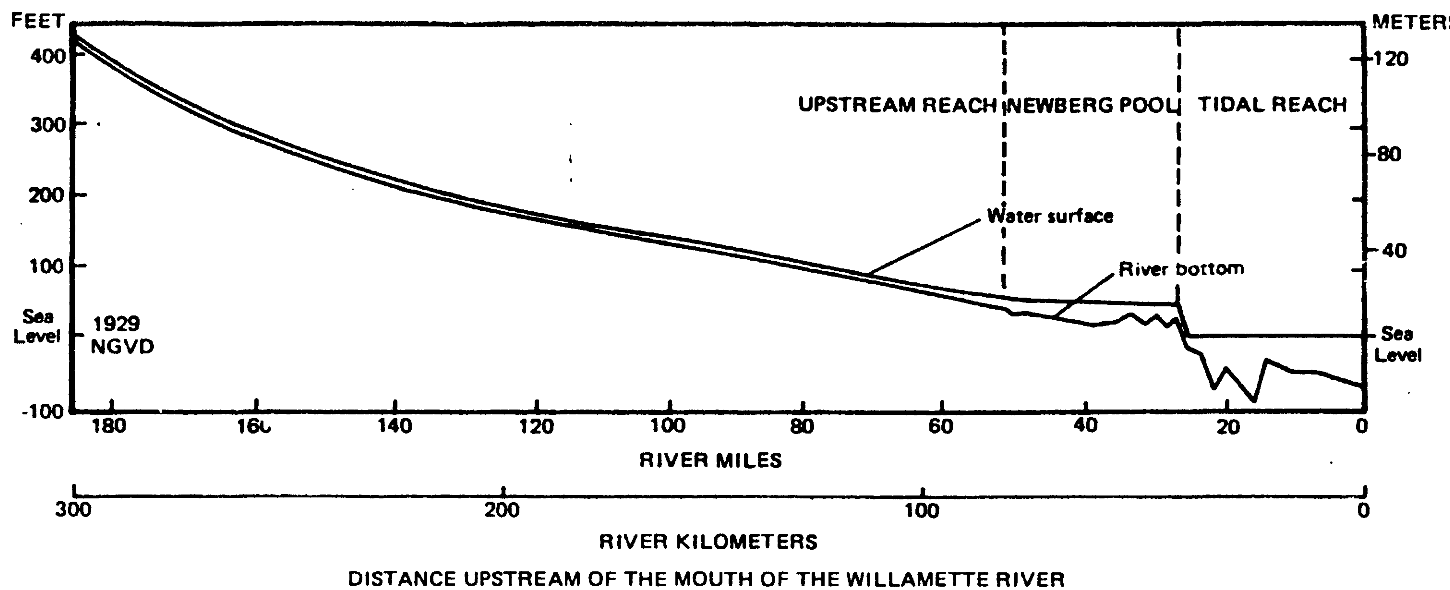


Figure 12. Water surface and bottom elevations of the Willamette River, Oregon, during the mid-July 1973 low-flow period (adapted from Rickert and others, 1977)

Only five percent of the annual precipitation falls in June to August. Extended drought periods may occur in late summer and early fall, in which rainfall may not occur over 30 to 60 day periods. Hines and others (1977) noted that this seasonal dry period has great impact on the summer and fall quantity and quality of streamflow in the Willamette River.

Mean annual precipitation for the Willamette Basin is 1600 millimeters (63 inches), but large areal variations occur because of elevation and topography. Heavy snowpack and high storage and yield of water by volcanic rock in the Cascade Range results in higher than expected summer baseflows in the Willamette River and Cascade tributaries compared to basins with similar rainfall patterns.

Channel characteristics

The main stem of the Willamette River begins at the confluence of the Coast and Middle Forks near Eugene, moves northward 301 kilometers (187 miles) through Corvallis, Albany, Salem, Newberg, Oregon City, and Portland, and flows into the Columbia River 159 kilometers (99 miles) from its mouth at the Pacific Ocean. The main stem can be segmented into three distinct reaches based on physical, chemical, and biological characteristics. Table 7 defines some of the differing characteristics.

The "Upstream Reach" is 217 kilometers (135 miles) in length and extends from Eugene to upstream of Newberg, RK 301 (RM 187) to RK 84 (RM 52)*. This shallow reach (Table 7) has a steep bed slope and large average flow velocities that are 10 to 20 times higher than those in downstream reaches. The bed consists primarily of cobbles and gravel that are covered with biological growth during the summer. Although segments of the Upstream Reach have been channelized, numerous meanders, islands, and side channels still exist. Gravel bars are visible at low flows and wide shallow sections occur. Large velocities and steep slopes indicate this is an eroding reach. High flows transport significant quantities

* River kilometers or miles upstream of the mouth of the Willamette River at its intersection with the Columbia River.

Table 7

Selected Physical Characteristics of the Willamette River, Oregon

Representative of Low-Flow Conditions (170 m^3/s at Salem)

Reach	Length mi	Approximate Bed Slope ft/mi	Bed Material	Representative Midchannel Water Depth ft	Average Velocity ft/s	Approximate Travel Time in Reach days
Tidal Reach	26.5	<0.1	Intermixed clay, sand, and gravel	40	0.16*	10.0
Newberg Pool	25.5	0.12	Intermixed clay, sand, and gravel with some cobbles	25	0.40*	3.9
Upstream Reach	135	2.8	Mostly cobbles and gravel	7	2.9**	2.8

* Calculated by volume displacement method using channel cross-sectional data.
** Calculated from dye study conducted by USGS.
Note: 1 mile = 1.61 kilometers, 1 foot = 0.304 meters.
Information adapted from Rickert and others (1977).

of cobbles and gravel as bedload (Rickert and others, 1977). Cross sections and channel locations are unstable over a yearly period, but such changes were estimated to have little impact on the time of travel through the reach (McKenzie and others, 1979).

The middle reach, referred to as the "Newberg Pool," extends from upstream of Newberg downstream to Willamette Falls at Oregon City, RK 84 to RK 43 (RM 52 - RM 26.5). Willamette Falls is formed by a 15-meter (50-feet) high basaltic sill. The river in the Newberg Pool is slow moving and deep. The bottom profile (Figure 12), low velocities, and the presence of fine bottom sediments indicate the Newberg Pool is a depositional reach.

Comparison of travel-time data given in Table 7 with previous studies referred to by Rickert, Hines, and McKenzie (1976) indicate dredging and gravel removal have increased low-flow (198 cubic meters per second or 7000 cubic feet per second) travel time 30 percent in the Newberg Pool.

Most of the summer, low flow at Willamette Falls is diverted through power generation turbines or over a fish ladder. These river-management activities lead to mild fluctuations in water elevation and velocity throughout the Newberg Pool (Rickert, Hines, and McKenzie, 1976). These mild variations do not seem to significantly effect one-dimensional, steady-state modeling of the reach (McKenzie and others, 1979).

The final reach, known as the "Tidal Reach," extends from Willamette Falls at RK 43 (RM 26.5), through Portland Harbor, to the mouth on the Columbia River. The Tidal Reach is also deep and slow moving. Tides on the Pacific Ocean affect velocity and water-surface elevation of the Willamette River near the mouth. A 12-meter (40-feet) channel is maintained by dredging from RK 0 (RM 0) to RK 22 (RM 14) in the Portland Harbor. The primary reach of sediment deposition for the Willamette River extends from RK 5 upstream to RK 16 (RM 3 - RM 10).

During the summer low-flow period, net downstream movement is slow and tidal effects cause flow reversals and large changes in velocity. Tidal effects are more pronounced in the lower 16 kilometers (10 miles) of the Willamette River. Depending on tide and river stages, Willamette River water may move downstream into the Columbia River, or Columbia River water may move upstream in the Willamette channel. The Columbia River water usually moves upstream as a density underflow as far upstream as St. John's Bridge at RK 10 (RM 6) (Rickert, Hines, and McKenzie, 1976). Downstream movement of water below RK 7 (RM 4) is also complicated by movement through the Multnomah Channel (bifurcation).

Because of the density underflow, pronounced tidal effects, and Multnomah Channel, river quality cannot be described with one-dimensional steady-state approximations from about RK 9 (RM 6) to the mouth. Upstream of RK 9 (RM 6), one-dimensional steady-state approximations seem valid for mean daily predictions. Tidal effects are moderated; travel times are large (10 days); and biological decay is slow (deoxygenation rate K_1 = 0.07 per day base e).

Hydrology

Most of the annual streamflow occurs from November to March in response to persistent winter rainstorms and spring snowmelt. Snowmelt in the High Cascades at elevations above 1,500 meters (5,000 feet) tends to prolong the higher streamflows until June or early July. Periods of low flow extend from July to September. In September, flows are increased with flow augmentation from headwater reservoirs to assist fish migration.

Flow augmentation from headwater reservoirs has a significant impact by increasing summer base flows, in addition to shortening summer low-flow periods. The 30-day low flow at Salem has increased from 104 cubic meters per second (3,670 cubic feet per second), measured prior to 1953 when the construction of headwater reservoirs began, to 170 cubic meters per second (6,010 cubic feet per second), measured between 1953 and 1970 (Rickert, Hines, and McKenzie, 1976).

Water temperatures of the Willamette River and major tributaries reach a maximum during July and August at the onset of low flow. During this critical period of low flow and high water temperature, temperatures show a tendency to increase in the downstream direction. Reservoir releases do not influence summer water temperatures below RK 192 (RM 120).

Water-quality description

The critical water-quality period of July and August corresponds to the summer period of low flow and high water temperature. Waste loads receive less dilution, and biological reactions place larger demands on the waste assimilative capacity of the river due to increases in reaction rates with temperature.

Based on water-quality studies undertaken in 1973 and 1974 (Table 8), a number of factors that influence water quality can be discerned. Reservoir releases controlled critical summer water quality by providing low-flow augmentation over the summer. The low-flow period was usually ended in early September by reservoir releases of water that aided fish migration. Temperature effects of reservoir releases were limited to the upstream reaches of the river. Seasonal increases of phosphorus in the Willamette River were related to spring and fall overturn of reservoirs and did not effect critical low-flow water quality (Rickert and others, 1977).

Waste loading was another significant factor that affected water quality. During the low-flow periods studied, 55 percent of the total carbonaceous BOD load was contributed by municipal and industrial discharges, whereas 45 percent was derived from nonpoint sources. Carbonaceous BOD from point sources was affected by basinwide implementation of secondary biological treatment.

Sixty-one percent of the point loads of carbonaceous BOD was due to industrial sources, and the remaining 39 percent was due to municipal sources. Point loads of carbonaceous BOD were distributed over the length of the main stem. Industrial loads were almost exclusively due to wood-product industries. The municipal loads included seasonal canning waste and other small industry waste (Hines and others, 1977).

Unlike carbonaceous BOD, nitrogeneous BOD was contributed, mainly by point sources, with 91 percent introduced by municipal and industrial discharges and 9 percent by nonpoint sources. The discharge of Boise Cascade Corporation paper mill at RK 136 (RM 85) was the overwhelming nitrogen source. Ninety percent of the nitrogenous BOD load was in the form of ammonia.

Table 8

Summary of Studies of the Willamette River Conducted during 1973 and 1974

Description	Date	Sampling Sites
Reconnaissance (review of historical data, preliminary sampling, methods testing, formulation of preliminary hypotheses)	Jan.-July 1973	Numerous sites throughout 187-mi main stem, major tributaries, and wastewater outfalls.
DO-BOD study, RM 26.5-0	July 24-26, 1973	RM's 28.6, 25.5, 21.1, 16.8, 12.8, 7.0, 6.0, 3.5, 1.5; all major tributaries just above main-stem confluence; all major wastewater outfalls.
DO-BOD study, RM 187-86.5	August 6-12, 1973	RM's 185, 161, 134, 120, 96, 86.5; McKenzie River, RM 7.1; Santiam River, RM 6; all major wastewater outfalls.
DO-BOD study, 86.5-26.5	August 15-18, 1973	RM's 86.5, 72, 50.0, 46.0, 39.0, 34.0, 28.6; plus all major tributaries just above main-stem confluence; all major wastewater outfalls.
Nonpoint-source study of BOD and nutrient loading.	June-Aug. 1974	Coast Fork Willamette River RM's 6.4 and 29.5; Middle Fork Willamette River RM 8; McKenzie River, RM's 7.1 and 14.9; South Santiam RM's 7.6 and 23.3; Clackamas River RM 0.5.
DO-BOD study, RM 86.5-0	August 6-7, 1974	RM's 86.5, 72.0, 50.0, 39.0, 28.6, 21.0, 12.8, 10, 7.0, 6.0; all major tributaries just above main-stem confluence; all major wastewater outfalls.
Nitrification study, RM 120-0.	August 12-14, 1974	RM's 120, 114, 86.5, 72.0, 60.0, 55.0, 50.0, 39.0, 28.6, 12.8, 7.0; all tributaries just above main-stem confluence; all major wastewater outfalls.

Note: 1 mile = 1.61 kilometers.
Information in the table was adapted from Hines and others (1977).

Processes affecting DO were well defined. Flow augmentation, reaeration, and mixing with Columbia River water added DO. Nitrification, deoxygenation, and benthic deposits exerted oxygen demands (Hines and others, 1977). Rickert and others (1980) found oxygen demands for low-flow conditions compared as follows:

	Percent
Deoxygenation of point source loads	28
Deoxygenation of nonpoint source loads	22
Nitrification of point source loads	32
Nitrification of nonpoint source loads	2
Benthic oxygen demand	16
Total	100

The results show that nitrification of point sources is the largest oxygen demand and that benthic demand is significant.

Because of the difference in the three reaches, deoxygenation and nitrification occurred at different rates. Nitrification occurred rapidly in the Upstream Reach but was insignificant in the Newberg Pool and Tidal Reach. Deoxygenation was higher in the Upstream Reach than the Newberg Pool and Tidal Reach. Higher reaction rates occurred because the Upstream Reach was a shallow surface-active reach. The gravel and cobbles that lined the bottom were covered with biological growth. In the Newberg Pool and Tidal Reach, there was no attached growth on the river bottom. The oxidizing bacteria were suspended or attached to suspended particles. In addition, deoxygenation rates were affected by differences in river depths and surface-area-to-volume ratios.

In addition to the significant benthic oxygen demand, significant amounts of carbonaceous BOD were resuspended or added in the Portland Harbor. This addition of carbonaceous BOD in areas of low average velocity may have resulted from resuspension by propwash from passing ships, reversing tidal currents, navigation channel dredging, ship discharges, or from the sewer overflows (Hines and others, 1977).

Deoxygenation rates for BOD samples of bottom materials were within the range of rates determined for river water samples. Other studies by the USGS Oregon District Office* showed the benthic oxygen

* Oral communication, June 1980, Stuart McKenzie and Frank Rinella, Hydrologists, USGS, Portland, Oregon.

demand to be 1.2 grams of DO per square meter per day (0.11 grams of DO per square foot per day) between RK 8 and 22 (RM 5 and 14) (3,000,000 square meters or 3.2 x 10^{-7} square feet of bottom area).

Nuisance algal growths did not occur in the Willamette River nor did phytoplankton productivity have a significant effect on mean DO levels between RK 0 and 188 (RM 0 and 86.5). Over the reach, photosynthesis balanced respiration. Vertical differences in DO occurred where photosynthesis increases DO in the upper zone (euphotic zone) and respiration decreases DO in the lower zone.

Data collection

Water-quality data were collected for the Willamette River to describe the DO balance during steady-state low flows between RK 138 and 0 (RM 86 and 0). The kinds of data collected are summarized in the following "Data Comparison" section. A number of observations were made to confirm that the Willamette River could be described by a one-dimensional steady-state model. Regions of two-dimensional flow in the downstream section of the river near the mouth were excluded. A number of different samples were taken to determine that lateral and vertical differences were unimportant or composite samples were taken to estimate the mean concentration.

Two-hundred and sixty segments were defined in the study reach from RK 8 to 138 (RM 5 to 86.5) on the basis of river morphology, location of major waste discharges and tributaries, logistical considerations, and stream-gage locations. Sampling-site locations were chosen on the basis of point source outfalls, tributary inflows, travel times, availability of boat launches, and bridge sites. Reconnaissance studies determined that sample sites were well mixed over the cross section.

Channel geometry data were obtained from fathometer soundings and from maps furnished by the COE Portland office. Travel time for each segment was calculated from the volume of the segment and discharge at that location. Supplementary staff gages and rating curves were set up to define discharge throughout the study reach (Hines and others, 1977). Travel time, average cross-sectional area, and average widths were tabulated from office notes of the USGS Portland District Office (Table 9).

Table 9

Channel Geometry and Cumulative Travel Time for the Willamette River, Oregon, RM 5 - 86.5

	Segment Beginning RM	Area ft^2	Width ft	Depth ft	Travel Time days
UPSTREAM REACH	86.5	2,400	455	5.27	0.0
	85.8	2,400	285	8.42	0.01519
	85.2	2,400	370	6.49	0.02821
	85.0	2,350	400	5.88	0.03250
	84.0	2,350	545	4.31	0.05367
	83.9	1,460	415	3.52	0.05538
	83.0	1,460	515	2.83	0.06722
	82.0	1,460	530	2.75	0.08037
	81.0	1,460	460	3.17	0.09352
	80.0	1,460	375	3.89	0.10667
	79.0	1,460	245	5.96	0.11982
	77.9	1,460	290	5.03	0.13428
	77.8	2,020	400	5.05	0.13584
	77.0	2,020	295	6.85	0.15031
	76.0	2,020	380	5.32	0.16839
	75.0	2,020	400	5.05	0.18648
	74.0	2,020	360	5.61	0.20457
	73.0	2,020	360	5.61	0.22265
	72.0	2,020	445	4.54	0.24074
	71.8	1,980	385	5.14	0.24432
	71.0	1,980	390	5.08	0.25850
	70.0	1,980	270	7.33	0.27623
	69.0	1,980	260	7.62	0.29396
	68.0	1,983	300	6.60	0.31169
	67.0	1,980	370	5.35	0.32942
	66.0	1,980	325	6.09	0.34714
	65.0	1,980	375	5.28	0.36487
	64.8	2,510	335	7.49	0.36889
	64.0	2,510	295	8.51	0.38687
	63.0	2,510	380	6.61	0.40935
	62.0	2,510	360	6.97	0.43182
	61.0	2,510	350	7.17	0.45429
	60.2	2,510	260	9.65	0.47227
	60.0	1,250	285	4.39	0.47564
	59.0	1,250	365	3.42	0.48683
	58.0	1,250	210	5.95	0.49802
	57.0	1,250	210	5.95	0.50921
	56.0	1,250	210	5.95	0.52041
	55.2	1,250	210	5.95	0.52936
NEWBERG POOL	55.2	1,250	210	5.95	0.0
	55.0	3,326	367	9.06	0.00410
	54.8	1,904	337	5.65	0.00873
	54.6	2,326	510	4.55	0.01248
	54.45	4,649	650	7.15	0.01711
	54.2	5,626	470	11.97	0.02848
	54.0	11,295	570	19.82	0.04347
	53.8	4,819	618	7.80	0.05774
	53.6	4,016	680	5.91	0.06556
	53.4	3,488	640	5.45	0.07221
	53.2	3,303	610	5.41	0.07822
	53.0	6,000	467	12.85	0.08646
	52.8	5,202	505	10.30	0.09638
	52.6	14,076	600	23.46	0.11346
	52.4	8,660	438	19.77	0.13359
	52.2	6,460	720	8.97	0.14698
	52.05	5,870	790	7.43	0.15517
	51.8	8,380	710	11.80	0.17095
	51.6	7,500	710	10.56	0.18501
	51.4	8,510	770	11.05	0.19919
	51.0	10,190	810	12.58	0.23231
	50.8	15,065	950	15.86	0.25468
	50.65	8,456	680	12.44	0.27030
	50.5	7,885	690	11.43	0.28116
	50.4	7,314	700	10.45	0.28789
	50.2	13,808	650	21.24	0.30659
	50.0	9,949	470	21.17	0.32763
	49.8	11,648	520	22.40	0.34675
	49.5	12,993	520	24.99	0.36851
	49.4	10,830	520	20.83	0.38955
	49.2	10,348	480	21.56	0.40825
	49.0	13,763	600	22.94	0.42955
	48.8	14,528	580	25.05	0.45454
	48.6	10,887	440	24.74	0.47698
	48.4	10,065	520	19.36	0.49548
	48.2	9,718	570	17.05	0.51296
	48.0	10,690	680	15.72	0.53098
	47.75	9,827	638	15.40	0.55363
	47.6	10,087	620	16.27	0.56682
	47.4	10,815	680	15.90	0.58528
	47.2	13,412	580	23.12	0.60668
	47.0	12,471	550	22.67	0.62953
	46.8	11,706	489	23.94	0.65089
	46.55	13,729	570	24.09	0.67897
	46.4	15,387	600	25.64	0.69825
	46.2	13,887	650	21.36	0.72411
	46.0	19,173	650	29.50	0.75330
	45.85	19,436	600	32.39	0.77888
	45.6	15,552	657	23.67	0.81750
	45.4	12,287	465	26.42	0.84209
	45.2	16,330	725	23.21	0.86780
	45.0	13,940	780	17.87	0.89498
	44.75	13,217	520	25.42	0.92496
	44.55	15,818	570	27.75	0.95060
	44.4	16,191	580	27.92	0.97180
	44.2	17,594	657	26.78	1.00164
	44.05	17,400	580	30.	1.02482
	43.8	17,575	570	30.83	1.06343
	43.6	17,594	580	30.33	1.09449
	43.4	16,168	650	24.86	1.12430

Note: 1 mile = 1.61 kilometers; 1 foot = 0.3048 meters.

(Continued)

Table 9 (Continued)

	Segment Beginning RM	Area ft^2	Width ft	Depth ft	Travel Time days		Segment Beginning RM	Area ft^2	Width ft	Depth ft	Travel Time days
Newberg	43.2	17,448	580	30.08	1.15398		33.3	7,611	780	9.76	2.53640
Pool (contd.)	43.0	13,257	566	23.42	1.18109		33.05	17,822	1010	17.65	2.56388
	42.8	20,525	580	35.39	1.21093		33.0	16,286	945	17.23	2.57124
	42.6	15,818	680	23.26	1.24303		32.8	14,750	680	16.76	2.59807
	42.4	14,072	580	24.26	1.26942		32.6	12,541	750	16.72	2.62165
	42.2	16,445	550	29.90	1.29637		32.45	27,520	827	33.40	2.64768
	42.0	13,029	635	20.52	1.32240		32.4	26,072	500	52.14	2.65928
	41.8	17,031	616	27.65	1.34895		32.25	17,334	730	23.75	2.68741
	41.6	15,926	530	30.05	1.37806		32.05	6,146	870	7.06	2.70771
	41.40	13,161	565	23.29	1.40375		31.8	11,160	920	12.13	2.72640
	41.2	15,290	690	26.51	1.43152		31.6	21,508	910	23.64	2.75463
	41.0	15,456	680	22.73	1.46132		31.42	19,990	620	32.24	2.78691
	40.8	13,233	650	20.36	1.48666		31.2	33,211	780	42.58	2.83748
	40.6	14,329	620	23.11	1.51100		31.0	19,155	780	24.56	2.88274
	40.4	17,471	600	29.12	1.53909		30.56	10,767	1100	9.79	2.95539
	40.2	14,540	520	23.45	1.56736		30.2	8,309	420	20.97	2.98584
	40.0	13,615	640	21.27	1.59222		30.0	17,506	310	56.47	3.00859
	39.8	16,576	565	29.34	1.61889		29.8	11,995	570	21.04	3.03408
	39.6	15,126	600	25.21	1.64688		29.4	25,868	1090	23.73	3.10750
	39.4	13,852	617	22.45	1.67248		29.2	25,971	1170	22.2	3.15230
	39.25	13,091	636	20.58	1.69032		29.0	21,773	1300	16.75	3.19356
	39.0	13,486	605	22.29	1.71966		28.90	17,570	1200	14.64	3.21056
	38.75	11,796	550	21.45	1.74757		28.65	20,510	1090	18.82	3.25170
	38.6	15,029	600	25.03	1.76531		28.45	33,091	790	41.89	3.29807
	38.4	18,675	650	28.73	1.79509		28.35	31,175	740	42.13	3.32564
	38.2	17,226	550	31.32	1.82679		28.15	29,501	780	37.82	3.37779
	38.0	17,115	570	30.03	1.85712		28.02	30,507	910	33.52	3.41131
	37.8	17,942	550	32.62	1.88808		28.0	31,639	960	32.96	3.41665
	37.6	17,598	650	27.07	1.91947		27.86	28,994	960	30.20	3.45303
	37.4	17,506	700	25.01	1.95047		27.8	30,683	990	30.99	3.46838
	37.2	15,877	690	23.01	1.97995		27.6	30,314	1010	30.01	3.52065
	37.0	14,775	650	22.73	2.00702		27.4	30,527	910	33.55	3.57264
	36.8	13,755	~~650~~	21.16	2.03222		27.2	26,888	730	36.83	3.62170
	36.6	16,430	612	26.85	2.05887		27.0	30,674	1090	28.14	3.67085
	36.4	16,383	650	25.20	2.08785		26.99	2,000	200	10.00	3.67228
	36.2	16,061	730	22.	2.11650		26.53	2,000	200	10.00	3.68014
	36.0	17,648	520	33.94	2.14627						
	35.75	14,765	600	24.61	2.18205	**TIDAL**	26.53	2,000	200	10.00	0.0
	35.6	11,067	570	19.42	2.19880	**REACH**	26.52	11,990	400	29.97	0.00060
	35.45	16,381	700	23.40	2.21659		26.37	9,360	265	35.32	0.01428
	25.2	13,870	620	22.37	2.24927		26.2	21,580	395	54.63	0.03675
	35.0	16,527	520	31.78	2.27554		26.0	11,350	440	25.80	0.06489
	34.8	19,883	630	31.56	2.30701		25.74	13,520	870	15.54	0.09252
	34.6	16,600	750	22.13	2.33854		25.56	12,500	585	21.37	0.11253
	34.4	13,566	780	17.39	2.36461		25.37	4,200	360	11.67	0.12609
	34.26	22,012	580	37.95	2.38614		25.21	4,870	420	11.60	0.13229
	34.2	17,315	700	24.74	2.39633		25.0	4,400	405	10.85	0.14060
	34.0	32,391	860	37.65	2.43929		24.82	5,750	380	15.13	0.14841
	33.8	16,489	550	29.98	2.48154		24.58	7,200	660	10.91	0.15965
	33.6	14,227	600	23.71	2.50809		24.34	19,240	785	24.51	0.18261

(Continued)

Table 9 (Concluded)

	Segment Beginning RM	Area ft^2	Width ft	Depth ft	Travel Time days	Segment Beginning RM	Area ft^2	Width ft	Depth ft	Travel Time days
Tidal Reach (contd.)	24.14	11,585	725	15.98	0.20491	14.52	24,730	735	33.65	2.57947
	24.0	16,370	640	25.58	0.21907	14.35	30,000	655	45.80	2.61307
	23.34	21,930	660	33.23	0.24123	14.18	26,400	720	36.67	2.64770
	23.69	12,520	660	18.97	0.25993	14.0	41,950	1560	26.89	2.69213
	23.53	12,238	610	20.06	0.27426	13.82	48,000	1320	36.36	2.75060
	23.28	12,640	530	23.85	0.29676	13.57	39,780	1290	30.84	2.82985
	23.0	21,480	924	23.25	0.33131	13.41	38,500	1270	30.31	2.87509
	22.78	7,350	475	15.47	0.35426	13.22	42,600	1480	28.78	2.93073
	22.61	10,520	290	36.28	0.36525	13.0	44,950	1180	38.09	3.00029
	22.45	17,590	450	39.09	0.38152	12.84	37,600	1005	37.41	3.04799
	22.27	17,570	460	38.20	0.40441	12.57	35,800	825	43.39	3.11956
	21.87	25,770	425	60.64	0.46712	12.46	36,930	820	45.04	3.14846
	21.68	13,900	685	20.29	0.49439	12.23	46,360	1060	43.74	3.21764
	21.46	13,800	365	37.81	0.51643	12.0	45,500	820	55.49	3.29394
	21.23	40,620	424	95.80	0.56171	11.77	36,740	745	49.32	3.36255
	20.86	21,150	485	43.61	0.64438	11.60	40,560	1000	40.56	3.40971
	20.68	26,820	660	40.64	0.67562	11.40	54,400	1265	43.00	3.47830
	20.52	26,650	510	52.25	0.70657	11.20	48,900	1185	41.27	3.55291
	20.23	26,760	565	47.36	0.76259	11.00	52,270	1110	47.09	3.62598
	19.9	27,990	665	42.09	0.82791	10.73	57,180	1145	49.94	3.73270
	19.77	32,230	900	35.81	0.85620	10.6	53,150	1160	45.82	3.78450
	19.57	40,400	670	60.30	0.90869	10.4	51,160	1120	45.68	3.85984
	19.4	35,260	870	40.53	0.95517	10.2	52,700	1210	43.55	3.93486
	19.14	45,360	640	70.86	1.03090	10.0	75,150	1710	43.95	4.02720
	19.0	20,230	515	39.28	1.06408	9.8	80,990	2430	33.33	4.13998
	18.86	39,900	905	44.09	1.09450	9.6	59,570	1640	36.32	4.24150
	18.68	53,850	1110	48.51	1.15548	9.4	57,800	1630	35.46	4.32628
	18.50	45,050	948	47.52	1.21980	9.2	62,830	1670	37.62	4.41341
	18.40	45,050	948	47.52	1.25236	9.0	80,060	1710	46.82	4.51661
	18.39	45,050	948	47.52	1.25561	8.8	91,250	2025	45.06	4.64035
	18.36	36,250	785	46.18	1.26442	8.6	95,860	2360	40.62	4.77549
	18.17	29,280	845	34.65	1.30938	8.4	92,720	2250	41.21	4.91170
	18.10	28,400	810	35.06	1.32397	8.23	85,240	1760	48.43	5.02096
	17.92	32,400	690	46.96	1.36349	7.9	124,170	2385	52.06	5.27053
	17.73	34,730	750	46.31	1.40955	7.8	133,300	2835	47.02	5.36351
	17.54	31,080	690	45.04	1.45471	7.6	105,300	2555	41.21	5.53585
	17.35	40,180	760	52.87	1.50360	7.4	91,340	2060	44.34	5.67788
	17.16	49,160	955	51.48	1.56491	7.2	84,130	2000	42.06	5.80462
	17.1	63,660	1090	58.40	1.58935	7.0	72,350	1740	41.58	5.91764
	16.68	42,400	1231	34.44	1.68128	6.86	56,670	1355	41.82	5.98287
	16.68	25,250	1000	25.25	1.72525	6.6	62,200	1414	43.99	6.09449
	16.43	52,730	960	54.93	1.79566	6.4	56,340	1366	41.24	6.18011
	16.16	32,020	720	44.47	1.87830	6.2	55,900	1340	41.72	6.26118
	16.0	31,060	1260	24.65	1.91474	6.0	63,940	1500	42.63	6.34773
	15.82	37,300	1505	24.78	1.95918	5.8	52,840	1210	43.67	6.43208
	15.57	25,290	1770	26.82	2.22122	5.6	54,500	1320	41.29	6.50961
	15.39	34,360	1190	28.87	2.40797	5.4	56,710	1290	43.96	6.58994
	15.22	33,340	1216	27.42	2.44954	5.2	61,560	1340	45.94	6.67536
	15.1	31,280	1075	29.10	2.47754	5.0	68,260	1500	45.51	6.76913
	14.78	20,320	800	25.40	2.53717					

Because dynamic flow models require information on cross-sectional properties at a number of discrete points rather than segment-averaged geometry data, data describing individual cross sections were collected. These data were derived from flood insurance studies by the COE Portland office; James M. Montgomery Engineers, Inc.; and CH2M HILL. The cross sections were measured over the reach at intervals that varied from 0.02 kilometers to 4.8 kilometers (0.01 - 3 miles). Cross sections were defined by 20 to 50 points, but many of these are on the floodplains, and it was unclear how low-flow channel sections were defined.

Samples of BOD were taken and DO and water temperature were measured from dawn to dusk over the two- to six-day study periods listed in Table 8. At each site, 12 to 20 BOD samples were collected and 100 to 350 discrete DO and water temperature measurements were made.

Every two hours during the day, vertical profiles of DO and temperature were measured at the water surface, 1, 2, 3, 5, 6, 9, 12, and 16 meters below the surface (3, 6, 10, 15, 20, 30, 40, and 50 feet). In addition, DO and temperature were measured at 0.6 of the channel depth. Vertical profiles were measured at three locations across the river to confirm vertical and lateral homogeneity.

Samples of BOD were collected at four-hour intervals, near mid-channel at 0.6 of the channel depth. Reconnaissance studies indicated little variation of BOD over the cross section. Water samples were collected with a four-liter Scott-modified Van Dorn bottle.

Samples of BOD were collected one to two times daily at tributaries and waste outfalls. Grab samples were collected on tributaries just above the confluence with the main stem of the river. Municipal effluent samples were composited over 24-hour periods. Grab samples were taken from pulp and paper mill effluents since diurnal variations were very small. Municipal wastewater samples were collected by the Oregon Department of Environmental Quality with the cooperation of each treatment plant staff. Some industrial effluent samples were collected by the technical service organization for wood product industries, NCASI. Preliminary measurement and sampling of the inflows and the river was begun 2 to 7 days before each study began.

Nitrogen samples were collected August 12 to 14, 1974 during a rapid downstream boat trip. Three samples were taken at sites listed in Table 8. Major waste effluents and tributaries were sampled during the boat trip as the boat passed each inflow.

Arkansas River Waste Assimilative Capacity Data

Arkansas River study

The Arkansas River in Pueblo County, Colorado, was studied by the USGS under a cooperative agreement with the Pueblo Area Council of Governments. Water-quality data, including data describing reaeration coefficients, were collected April 1 to 2, 1976; October 13 to 15, 1976; and September 19 to 20, 1979. These data were used to calibrate and verify the USGS Streeter-Phelps model.

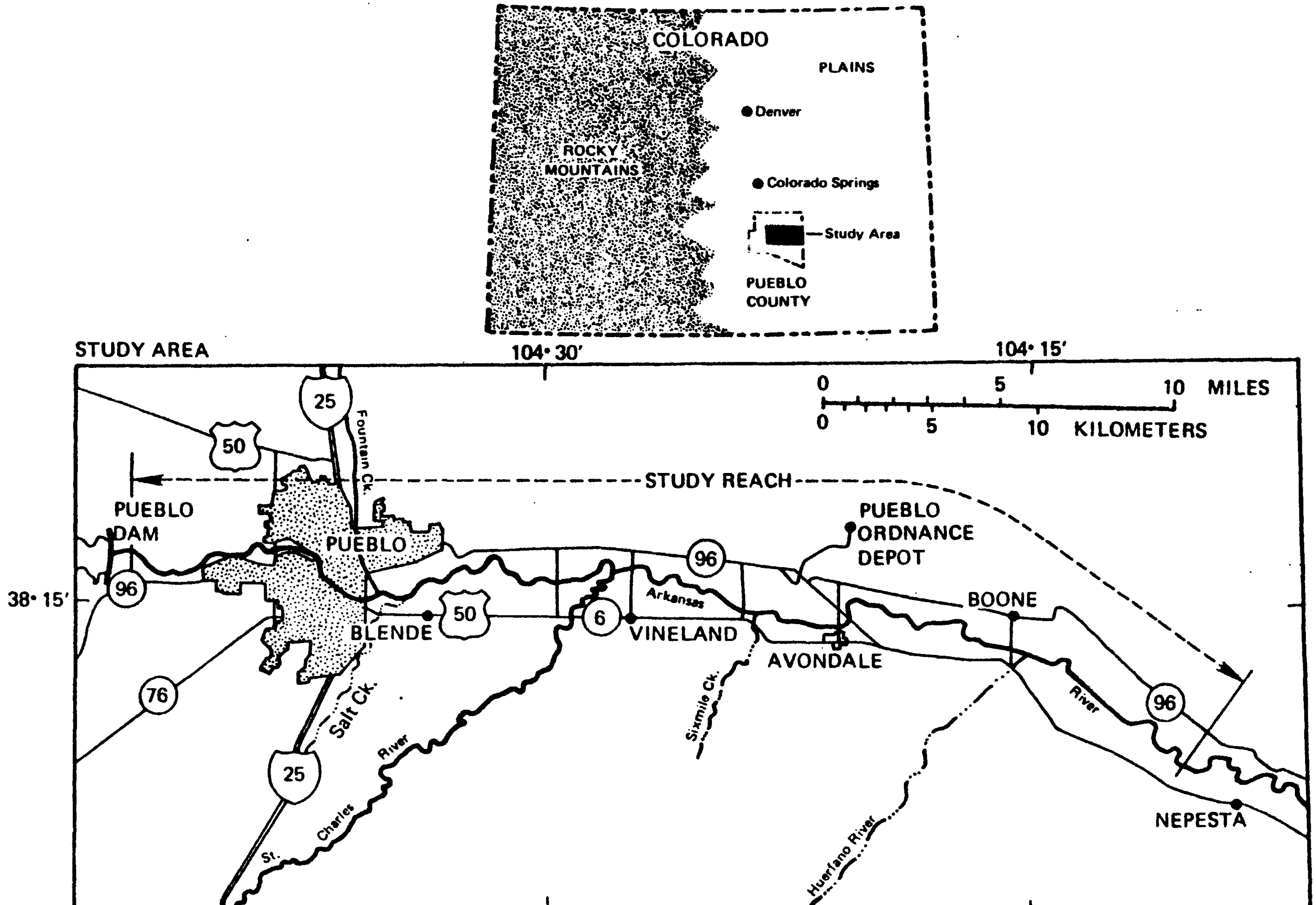
The data are contained in Goddard (1980); Cain, Baldridge, and Edelmann (1980); and Cain and Edelmann (1980). In addition, most of the data can be accessed through the USGS water data-management system WATSTORE or the EPA system STORET.

Basin description

Pueblo County, Colorado, is located on the plains of the eastern part of the state about 32 kilometers (20 miles) east of the Front Range of the southern Rocky Mountains (Figure 13). The Arkansas River originates near the Continental Divide in the Rocky Mountains and flows several hundred miles through the mountains before entering the plains west of Pueblo County. Snowmelt in the Rocky Mountains is a significant source of flow. On the plains, irrigation farming is an important basin activity affecting quantity and quality of the river flow.

Stream description

The stream channel had a slope of 0.0015 over the 67.6-kilometer (42-mile) study reach (Figure 14). The channel was braided in places with numerous islands and sand bars. The bed consisted of cobbles and gravel in the upstream reaches and sand in the lower reaches. The depth of flow



Base from U.S. Geological Survey
Pueblo, 1954 1:250,000

Figure 13. Arkansas River and significant tributaries between Pueblo Dam and Nepesta, Colorado (adapted from Goddard, 1980)

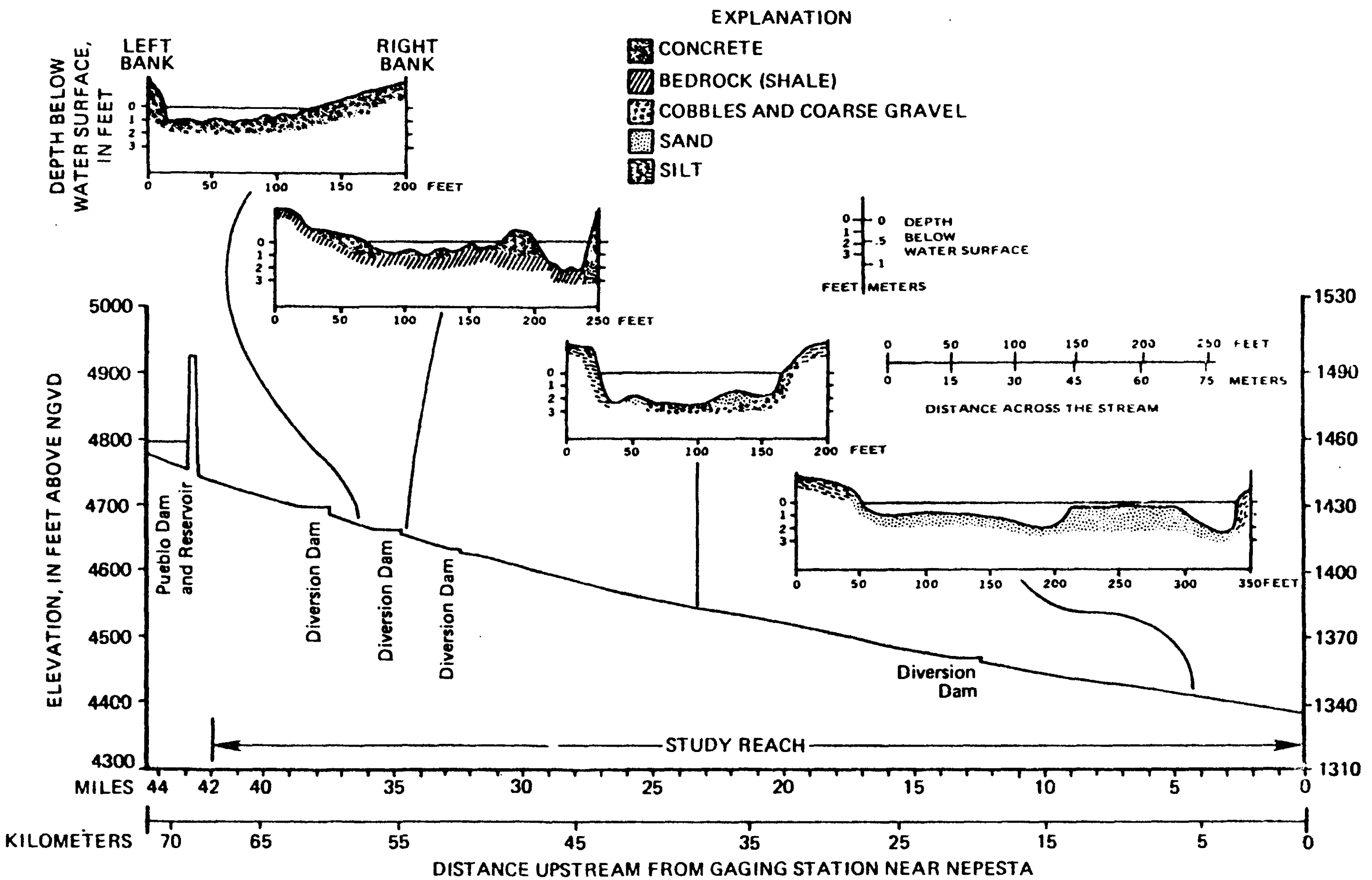


Figure 14. Channel slope, selected cross sections, and bed material for the Arkansas River study reach in Colorado (adapted from Goddard, 1980)

varied from 0.1 to 0.4 meters (0.4 to 1.3 feet).

At low flow, the upstream reach consisted of a series of pools separated by riffles. Longer pools were formed by four diversion dams located in the study reach. These low dams or weirs allowed water to be withdrawn for irrigation and municipal water supply.

Hydrology

Five factors affected the low-flow discharge in the study reach. First, Pueblo Reservoir, just upstream of the study area, stored flood flows and released flow during low-flow periods in accordance with water rights. Second, discharge rates were affected by interbasin transfer of water. Some water in Pueblo Reservoir was transferred to the Platte River Basin. Another upstream reservoir received water from the Colorado River Basin. Third, irrigation water was diverted and returned to the study segment by an arrangement determined by water rights. Fourth, snowmelt of May and June in the Rocky Mountains divided low-flow periods into two separate events. Finally, waste inflows from the Pueblo Wastewater Treatment Plant and the CF&I Steel Company, which were originally derived from groundwater and upstream diversions, increased flow in the stream by about 100 percent (Figure 15).

Water was diverted from the upstream portion of the reach into Bessemer Ditch and into the Pueblo water-treatment plant (Figure 16). Water diverted from an upstream reservoir in the mountains was returned to the river as cooling water from a coal-fired power plant. Downstream, water was diverted into the Colorado Canal, Rocky Ford Highline Canal, and Oxford Farmers Ditch. The Bessemer Ditch discharged into the Huerfano River that in turn flowed into the Arkansas River near Boone.

Two critical low-flow periods occurred in the upstream reaches of the Arkansas River. One occurred from March to early May before the occurrence of snowmelt in the Rocky Mountains. Later, after the snowmelt ended, a second low-flow period occurred from mid-August to mid-October. Both periods are critical in terms of water quality, but higher temperatures in late summer compounded problems during the second low-flow period.

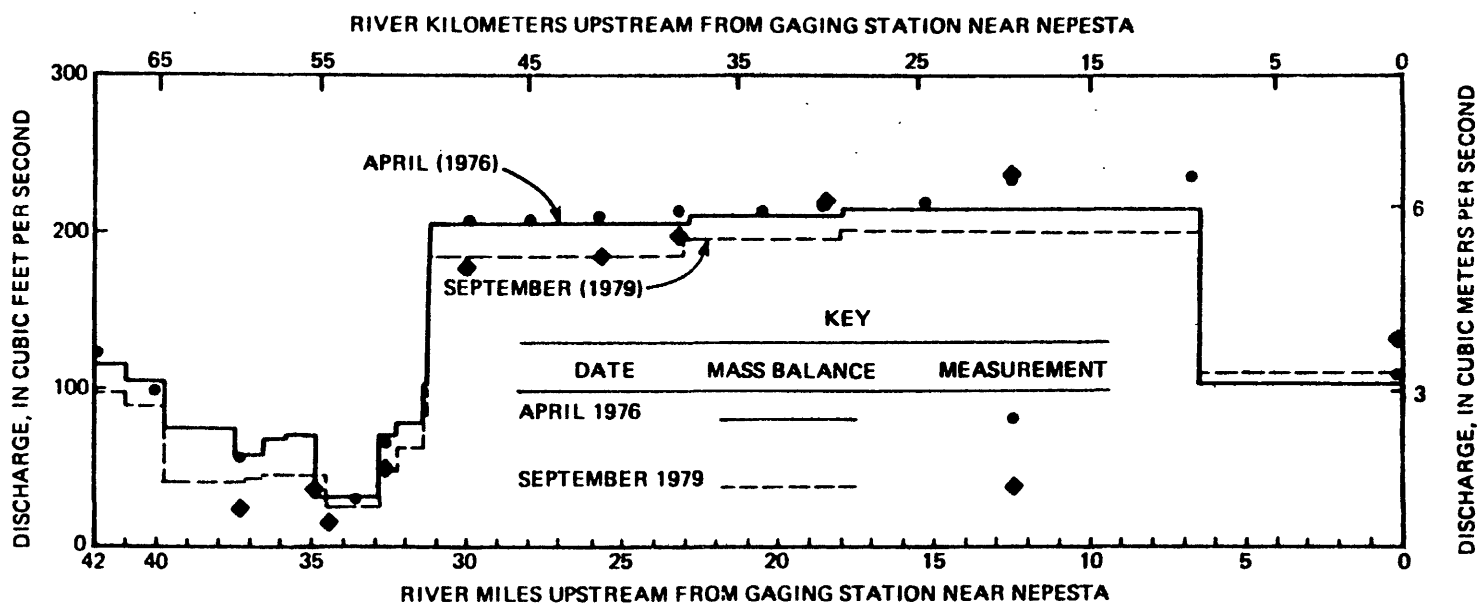


Figure 15. Discharge profiles and discharge measurements for the Arkansas River in Colorado, April 1976 and September 1979 (1 ft^3/s = 0.028 m^3/s and 1 mile = 1.6 km)

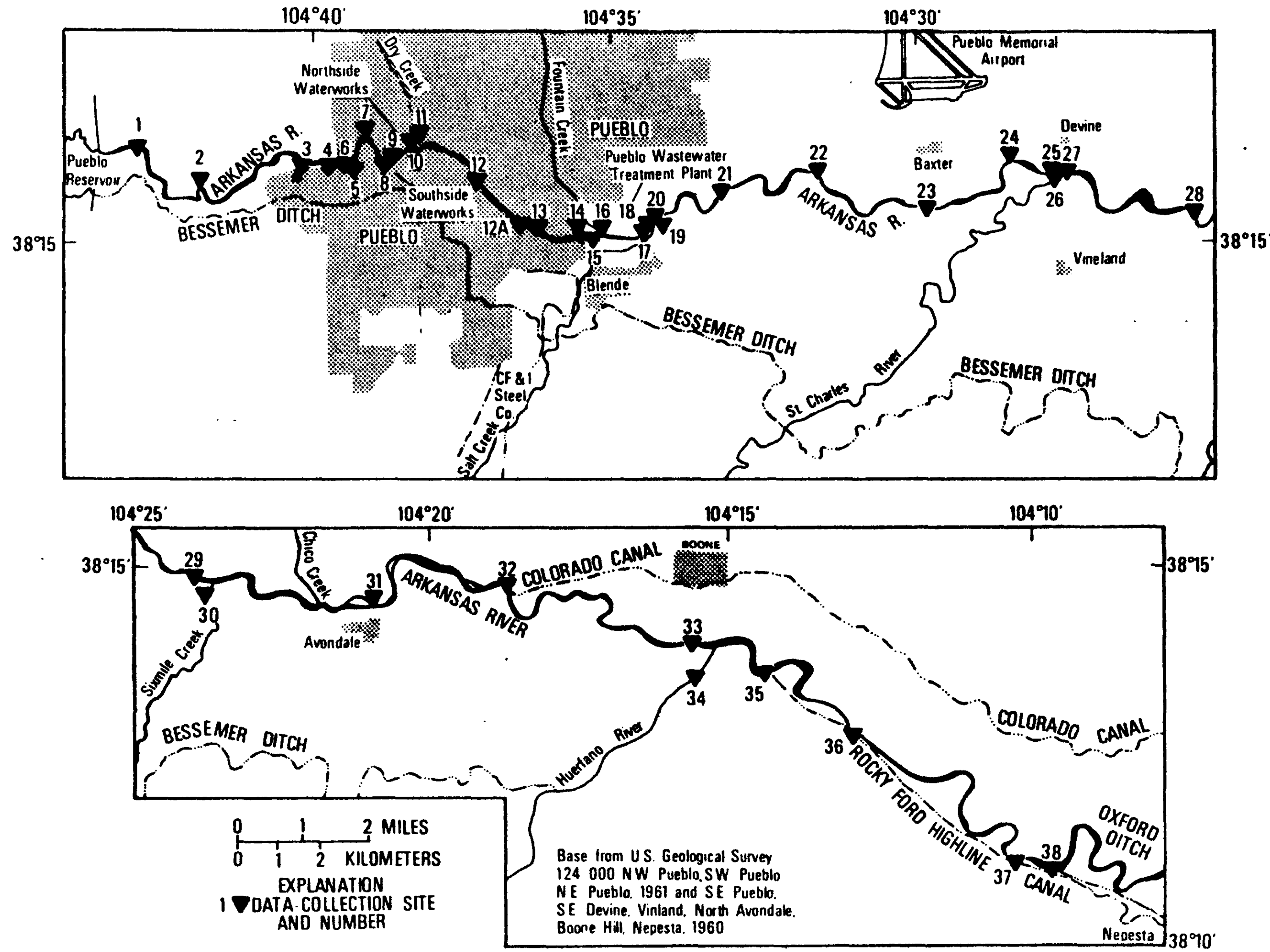


Figure 16. Location of sample sites on the Arkansas River between Pueblo and Nepesta, Colorado; and sample sites on tributaries and point sources (adapted from Goddard, 1980)

Water-quality description

The water quality of the Arkansas River in Pueblo County, Colorado, was severely affected by discharges of BOD and ammonia from the Pueblo sewage treatment plant (STP) and the CF&I Steel Corporation plant. Water temperatures were increased upstream of these point sources by the effluent of cooling water from a Southern Colorado Power plant. Nonpoint sources entering through several drains, creeks, and rivers were not of great importance during low-flow periods.

River DO levels dropped from saturation upstream of Pueblo to a minimum value 8 to 16 kilometers (5 to 10 miles) downstream of the urban area. This minimum value violates state water-quality standards of 5 milligrams per liter for DO during the spring and fall low-flow periods.

The DO balance was affected by reaeration and the oxidation of carbonaceous and nitrogenous materials. The significance of benthic oxygen demand and photosynthesis was unknown.

Data collection

For the 67-kilometer (42-mile) study reach, water-quality data were collected at 23 sites on the Arkansas River, 7 outfalls of drainage networks, 5 tributaries, and 4 wastewater treatment plant outfalls. These sampling sites are shown in Figure 16 and listed in Table 10.

Specific conductance, DO, pH, and temperature were measured at the time each water sample was collected. Samples were analyzed for 5-day BOD with nitrification inhibited, total Kjeldahl nitrogen, total ammonia nitrogen, total nitrite nitrogen, and total nitrite-plus-nitrate nitrogen. This information is summarized in the next section.

Eleven to twelve samples were collected over a 24-hour period at treatment plants having diurnal variations in discharge. Constant inflows were sampled 4 to 5 times over the 2-day study periods. The river was sampled about 4 times at each site (Goddard, 1980, and Douglas Cain*).

Nitrogen samples were chilled to 4 degrees Centigrade (39 degrees Fahrenheit) in transit to the laboratory and were analyzed within 24 to

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* Written communication, January 1980, Douglas Cain, USGS, Pueblo, Colorado.

Table 10

Data-Collection Sites on the Arkansas River

Site Number*	Site Identifier**	Site Type†	River Mile††	Description
1	381617 1044306 00 (07099400)	M	42	Arkansas River upstream of Pueblo
2	381544 1044144 00	M	40.1	Arkansas River near Goodnight
3	381604 1044005 00	D	38	Goodnight drain at mouth
4	381604 1043942 00	D	37.5	Pueblo Blvd. storm drain at mouth
5	381603 1043922 00	D	37.3	City Park Drain Number One at mouth
6	381602 1043926 00 (07099500)	M	37.2	Arkansas River near Pueblo
7	381623 1043905 00	D	36.7	Northside Waterworks sluice at mouth
8	381608 1043838 00	D	36.2	City Park Drain Number Two at mouth
9	381624 1043835 00	M	36.2	Arkansas River near Southside Waterworks
10	381621 1043820 00	D	35.9	Northside Waterworks drain at mouth
11	381628 1043817 00	T	35.7	Dry Creek at mouth
12	381607 1043725 00	M	34.9	Arkansas River at 4th Street Bridge
12A	381515 1043631 00	T	33.6	I-25 Tributary at mouth
13	381607 1043725 00	M	33.5	Arkansas River at Santa Fe Avenue
14‡	381508 1043544 00	W	32.8	Southern Colorado Power outfall
15	381510 1043509 00	M	32.5	Arkansas River near Colorado Highway 227
16	381515 1043519 00	T	32.3	Fountain Creek at mouth
17	381520 1043420 00	M	31.4	Arkansas River upstream of Pueblo STP outfall
18	381522 1043421 00	W	31.3	Pueblo STP outfall
19‡‡	381522 1043418 00	T,W	31.2	Salt Creek at mouth
20	381523 1043416 00	M	31.2	Arkansas River at Salt Creek

* Site number refers to number on Figure 16.
** Latitude (first six digits), longitude (next seven digits), and sequence code (last two digits); USGS station number given in parenthesis for established gaging stations.
† M = main channel of Arkansas River; D = drainage ditch or pipe; T = natural tributary; W = wastewater discharge.
†† River miles upstream from the gaging station, Arkansas River near Nepesta. 1 mi = 1.61 km.
‡ Discharge of cooling water from Southern Colorado Power's electrical generating facility, at Pueblo. Flows originate at Runyon Lake upstream from site.
‡‡ Wastewater from CF&I Steel Corp. is discharged to Salt Creek.
Note: Information in this table was adapted from Goddard (1980).

(Continued)

Table 10 (Concluded)

Site Number	Site Identifier	Site Type	River Mile	Description
21	381547 1043308 00	M	29.8	Arkansas River at 23rd Lane
22	381601 1043130 00	M	27.9	Arkansas River at 28th Lane
23	381530 1042946 00	M	25.8	Arkansas River at Colorado Highway 233
24	381609 1042826 00	W	24.2	Meadowbrook STP outfall
25	381600 1042726 00	M	23.3	Arkansas River upstream of St. Charles River
26	381556 1042733 00	T	23.2	St. Charles River at mouth
27	381613 1042726 00	M	23.1	Arkansas River at Colorado Highway 231
28	381532 1042521 00	M	20.5	Arkansas River at 40th Lane
29	381453 1042355 00 (07109500)	M	18.5	Arkansas River near Avondale
30	381440 1042342 00	T	18.1	Sixmile Creek at mouth
31	381432 1042055 00	M	15.3	Arkansas River at Avondale
32	381443 1041842 00	M	12.4	Arkansas River at Colorado Canal headgate
33	381401 1041537 00	M	7.8	Arkansas River at Boone
34	381332 1041539 00	T	7.4	Huerfano River near mouth
35	381336 1041424 00	M	6.7	Arkansas River at Rocky Ford Highland Canal headgate
36	381247 1041259 00	M	4.0	Arkansas River downstream of Rocky Ford Highland Canal headgate
37	381103 1041022 00	M	0	Arkansas River near Nepesta
38	381054 1040941 00	M	-0.7	Arkansas River at Oxford Farmers Canal headgate

48 hours. Nitrogen samples were analyzed using standard USGS procedures (Skougstad and others, 1979) at the USGS Central Laboratory in Denver, Colorado.

Samples of BOD were also chilled upon collection. Five-day tests were run with an inhibitor added to prevent nitrification.

Discharge measurements for the two data-collection surveys were made on all inflows and at selected points in the stream. Multiple discharge measurements were made for inflows that varied by more than 25 percent.

Travel time and channel geometry were measured from the Pueblo STP outfall to the end of the reach (sites 20 to 37, Table 10) on September 17 to 21, 1979. Travel-time measurements were made using Rhodamine WT fluorescent dye and a fluorometer. Distance from the bank and depth were measured at multiple distances across the stream for 72 sites (Cain, Baldridge, and Edelmann, 1980) in order to estimate mean depth and area. Measured stream depths were referred to the water surface at each cross section and were not referred to a common datum.

During October 1976, Goddard (1980) measured reaeration coefficients between sites 9 to 17, sites 21 to 23, sites 23 to 27, and sites 36 to 38 using the modified hydrocarbon gas-tracer technique of Rathbun, Shultz, and Stephens (1975). The October 1976 reaeration data, collected when the discharge at the head of reach was 11.3 cubic meters per second (400 cubic feet per second), were generalized for other flow conditions by determining which reaeration equation best fit measurements for this reach of the Arkansas River. Comparing 19 previously published reaeration equations to the measured data, Goddard determined that the Padden and Glovna equation (Cain, Baldridge, and Edelmann, 1980)

$$K_2 = 6.86\ (\bar{u}/H^{1.5})^{0.703} \tag{6}$$

best described reaeration in the Arkansas River at this location. In Equation 6, $\bar{u}$ is the reach-averaged velocity in feet per second; H is the reach-averaged depth in feet; and K_2 is the reaeration coefficient at 20 degrees Centigrade (68 degrees Fahrenheit) for a natural logarithm in units of per day.

Data Comparison

Contrast and comparison of data

The data collected in water-quality studies of the Chattahoochee River in Georgia, Willamette River in Oregon, and Arkansas River of Colorado define a wide range of physical, chemical, and biological conditions. The data base describes the DO balance of each stream. In addition, some information is available to describe nitrogen, phosphorus, coliform bacteria, and heavy metals.

Table 11 compares and contrasts the three sets of data. The Willamette River is a large sluggish stream while the Arkansas River in Colorado is a small fast-moving stream. The Chattahoochee River falls between the two.

There are roughly orders of magnitude differences in the three streams as illustrated by reaeration coefficients: Chattahoochee River, K_2 = 0.3 to 11; Willamette River, K_2 = 0.05 to 0.4; and Arkansas River, K_2 = 6 to 15. The hydrology and geology of each basin along with man's activities lead to this wide range of differences. Snowmelt is important in the Willamette and upper Arkansas basins but not important in the Chattahoochee basin. The Arkansas River has a steep bed slope and shallow depths, whereas the Willamette River is deep and has a mild bed slope. The Chattahoochee River is flushed clean of benthic material on a weekly basis by upstream peaking-power production, but the Willamette still has appreciable benthic demand from excessive discharge of organic materials into the river in the recent past.

Data collected

A wide range of data was collected in each of the three studies. The data fall into three classes that include hydraulic measurements, field measurements of water quality, and laboratory analysis of water quality.

Table 12 compares the hydraulic data available for each of the three separate studies. Discharge measurements were excellent for the

Table 11

Contrast of Physical, Chemical, and Biological Characteristics from the Chattahoochee, Willamette, and Arkansas River Studies

Study	Discharge ft^3/s	Slope	Length mi	Width ft	Depth ft	Travel Time days	Reaeration Coefficient day^{-1}	Deoxygenation Rate day^{-1}	DO Balance Components
Chattta-hoochee River	1100-1800	0.0003	43	234-269	4-6	1.8	0.3-11.	0.16	Reaeration Deoxygenation Nitrification
Willa-mette River	6000-8000	0.0005-0.0	83	370-1300	2-60	16.7	0.05-0.4	0.07-0.14	Reaeration Deoxygenation Nitrification Benthic demand
Arkan-sas River	25-200	0.0015	42	65-190	0.4-1.3	1.7-2.0	6-15	1.5	Reaeration Deoxygenation Nitrification

Note: 1 ft^3/s = 0.028 m^3/s.
1 mile = 1.61 kilometers.
1 foot = 0.3048 meter.

Table 12

Hydraulic Data Collected in Studies of the Chattahoochee, Willamette, and Arkansas Rivers

Study	Discharge	Cross Sections	Width	Depth	Travel Time	Reaeration
Chattahoochee River	Measured by USGS	Measured by COE and USGS	Measured by USGS	Measured by USGS	Computed from width, depth, length, and discharge	From Tsivoglou and Wallace (1972)
July 1976	(STP* lacking)					
August 1976	(From STP operator)					
September 1976	(From STP operator; Trib Q** not measured)					
May-June 1977	(From STP operator)					
Willamette River						
1973 and 1974	Measured by USGS (point sources from operators)	Measured by COE, James Montgomery Engineers, and CH2M HILL	Measured by USGS	Measured by USGS	Computed from dye studies and measurements of width, depth, length, and discharge	Computed by Velz Rational Method
Arkansas River						
1976 and 1979	Measured by USGS	Measured by USGS but not referred to a common datum	Measured by USGS	Measured by USGS	Computed from dye studies and measurements of width, depth, length, and discharge	USGS gas-tracer measurements

* STP - Sewage treatment plant.
** Trib. Q - Tributary discharge.

Arkansas and Willamette studies, but were less satisfactory for point sources in the Chattahoochee study. For the Chattahoochee data, monthly average discharges, derived from STP records, were used and other tributaries were gaged infrequently. For the Arkansas River, all inflows were gaged and unsteady inflows (when discharge varied more than 25 percent over a 24-hour study period) were measured several times and averaged. Wastewater treatment plant records were used in the Willamette study, but the greater dilution effect of that river over-shadowed possible errors in inflow discharge measurements.

Cross-section data were more accurate in the Chattahoochee data than in the other two studies. Actual channel coordinates at 41 cross sections were measured in the field and were related to mean sea level. Width and depth were also measured during a boat trip down the river. Flood studies also provided cross-section data related to mean sea level for the Willamette River. In addition, estimates of reach-averaged width and depth were made by the USGS from field measurements. However, channel coordinates collected for high-flow conditions may lack detail necessary for adequate representation of the channel during low-flow conditions. The cross-section data collected during the Arkansas River study lacked completeness because those data were not referred to a common datum. Nevertheless, because of the steep river channel slope, a flow-routing model makes reasonable estimates of travel time and depth based on the measured channel cross-sectional properties and local channel slopes taken from topographic maps.

Travel times measured or estimated in each study seem to be reliable. Dye studies were used in the Arkansas River study and on the upstream segment of the Willamette River. Accurate channel volume measurements on the Chattahoochee River and lower Willamette River compensate for the lack of dye studies.

Three different techniques were used to determine reaeration coefficients. Reaeration coefficients for the Arkansas River were measured with a hydrocarbon-gas tracer. Reaeration coefficients were estimated using the Velz rational method for the Willamette River in which reaeration coefficients were low (0.05 to 0.4 per day). The in-stream balance of DO and BOD tended to confirm the reaeration estimates. In the Chattahoochee study, direct measurements were made using radioactive-gas tracers. These measurements compared well with reaeration estimates using the Velz rational method (Velz, 1970).

Table 13 lists water-quality measurements and sampling techniques used in each study. For some of the Chattahoochee study periods, DO and water temperature measurements were made infrequently (once for each inflow), and data were missing for some STPs and tributaries. Sampling techniques were well adapted to the size of each river except samples were taken infrequently for some Chattahoochee River tributaries. Grab samples were appropriate for the Arkansas River except at three sites on the river below inflows at which the inflow was not laterally mixed in the river. Cross-sectional integrated sampling was necessary for the Willamette and Chattahoochee rivers because of their greater widths.

Table 14 lists the laboratory analyses performed in USGS laboratories. When deviations occurred from standard practice, new procedures were fully tested beforehand. The Willamette study samples were analyzed in the Portland Office, but quality-control samples were sent to the USGS Central Laboratory in Salt Lake City, Utah.

Information collected for the Chattahoochee River study described the most constituents, but some of the data, such as coliform bacteria and metals, were collected infrequently. The Willamette study concentrated on the DO balance in the stream. Separate studies of metals and nutrients were undertaken.

Table 13

Field Water-Quality Measurements and Sampling Methods in Chattahoochee, Willamette, and Arkansas River Studies

	Water-Quality Measurements				Sampling Method		
Study	DO	Temp.	Sp. Cond.	pH	Tributaries	Point Sources	River
Chattahoochee River:							
July 1976	✓	✓	✓	✓	Grab	Not taken	X-section integrated
August 1976	(Some STP data lacking)	(Some STP data lacking)	✓	✓	Grab	Grab	X-section integrated
September 1976	(Some STP data lacking)	(Some STP data lacking)			Not taken	Grab for BOD_5	X-section integrated
May-June 1977	✓	✓	✓		Grab	Grab	X-section integrated
Willamette River:							
July-August 1973	✓	✓			Grab	Composite (non USGS)	X-section integrated
August 1974	✓	✓			Grab	Composite (non USGS)	X-section integrated
Arkansas River:							
April 1976	✓	✓	✓	✓	Grab	Grab	Grab
September 1979	✓	✓	✓	✓	Grab	Grab	Grab

Table 14

Laboratory Analyses of Samples for Studies of the Chattahoochee, Willamette, and Arkansas Rivers

Study	BOD	COD	TOC	Organic Nitrogen	Ammonia	Nitrite	Nitrate	Soluble Phosphorus	Ortho-phosphates	Suspended Sediment	Coliforms	Heavy Metals
Chattahoochee River:												
July 1976	5-day			total & diss.	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.		fecal & strep.*	
August 1976	5-day	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.	Size distribution & concentration	fecal & strep.*	As, Cr, Cu, Pb, Zn*
September 1976	5-day											
May-June 1977	5-day ultimate	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.	total & diss.			
Willamette River:												
July-August 1973	ultimate											Separate study
August 1974	ultimate & NBOD			total	total	total	total					Separate study
Arkansas River:												
April 1976	5-day with N-inhibitor			total	total	total	total		total		fecal & total	
September 1979	5-day with N-inhibitor			total	total	total	total					

* Single grab samples at points with incomplete coverage of all inflows.

PART V: MODEL APPLICATIONS WITH CHATTAHOOCHEE RIVER DATA

Model Preparation

Application

The data collected from the Chattahoochee River were reduced to fit the data requirements of each of the four models outlined in Part III. The August 1976 and May-June 1977 data sets were used for calibration, and the July 1976 and September 1976 data sets were used for verification. The Streeter-Phelps and QUAL II models were applied to all four data sets. The WQRRS model was applied to the August 1976 data. The MIT model was not used to simulate any of the data because of program difficulties. In addition, comparable results were available from the Velz rational method (Stamer and others, 1979) for parts of the August 1976 and May-June 1977 data.

The Streeter-Phelps, QUAL II, and WQRRS models were used to simulate DO, BOD, organic nitrogen (Streeter-Phelps model only), organic detritus (WQRRS model only), ammonia, nitrite, and nitrate. Orthophosphate, fecal coliform bacteria, chromium, zinc, and lead were simulated with the Streeter-Phelps and QUAL II models. Ultimate BOD was predicted using the Streeter-Phelps and QUAL II models, whereas 5-day BOD was predicted using the WQRRS model. Stream temperature was predicted using the QUAL II and WQRRS models. Dissolved lead, chromium, and zinc were simulated as conservative substances. Predictions from Stamer and others (1979) based on the Velz method included BOD, ammonia, nitrate, and DO for the May-June 1977 data and DO for the August 1976 data.

Stream discretization and hydraulics

Reaches and computational elements were standardized when possible. In the case of the Chattahoochee River, 24 reaches were defined for the Streeter-Phelps and QUAL II models. These were based on the headwaters, 21 inflows, 1 withdrawal, and 1 point where hydraulic characteristics changed significantly. Some reaches varied in length between the two models by as much as 0.3 kilometers (0.2 miles) because reaches in the

QUAL II model had to be an integer multiple of the element length.

The stream discretization for the WQRRS model was involved. The limitation of 10 inflows dictated that the study reach be broken into two separate applications and three insignificant creeks were not included. The withdrawal and discharge of Atkinson and McDonough power plants were dropped, and the heat content of that effluent was shifted upstream to the headwaters and the R. M. Clayton Sewage Treatment Plant, which affected plots of temperature versus distance for RK 487.78 to 481.55 (RM 302.97 to 299.1). Two control structures dictated that three reaches be defined in the upstream segment. One reach was defined in the downstream segment. Computational elements varied in length from 0.84 to 1.22 kilometers (0.52 to 0.76 miles) compared to the 0.40-kilometer (0.25 mile) elements used in the Streeter-Phelps and QUAL II models.

Travel time was specified for the Streeter-Phelps model in simulating the August 1976 data, and travel time was calculated from discharge and reach volume for the July 1976, September 1976, and May-June 1977 data. The QUAL II model was applied utilizing the option that approximated the channel cross-section shapes with a trapezoid and routed the flow using the Manning equation. The Manning roughness coefficients were adjusted until the simulated August 1976 travel times matched measured travel times. The steady backwater routing option of the WQRRS model was also used to simulate the travel time, average depth, and average velocity also from data describing cross sections and discharge. Roughness coefficients were adjusted until the simulated travel time agreed with measured travel times.

Water-quality coefficients

The same reaction rates and coefficients were used for each model except for wind-speed coefficients. This exception was relatively unimportant since temperature predictions were insensitive to wind-speed coefficients in this case. For the other coefficients, deoxygenation and reaeration coefficients were deduced from measurements; nitrification coefficients were deduced by model calibration; finally, there was evidence indicating that benthic demand and photosynthesis were not important.

Wind-speed coefficients, necessary for temperature simulation, were among the coefficients that were determined independently. Jobson and Keefer (1979) measured wind speed, short- and long-wave radiation, dry- and wet-bulb air temperature, and vapor pressure on July 12-19 and August 1-8, 1976 at the R. M. Clayton Sewage Treatment Plant. Using these data, they determined that the wind-speed function for this part of the river was 70 percent of 3.01 + 1.13 • (wind speed). The coefficient 3.01 has units of millimeters per day per kilopascal. The coefficient 1.13 has units of millimeters per day per kilopascal per meter per second.

The reaction rate for BOD was chosen as 0.16 per day at 20 degrees Centigrade (68 degrees Fahrenheit) from previous studies by Stamer and others (1979) using the Velz method and Miller and Jennings (1979) using the Streeter-Phelps model. Stamer and others (1979) developed the BOD rates from extensive analysis of BOD samples.

Fecal coliform bacteria data were limited to single samples of each inflow (a few inflows were not sampled) and single samples at seven points in the river, all measured during the August 1976 study. The die-off rate was estimated as 0.08 per day from an EPA compilation of published die-off rates (Zison and others, 1978) because the in-stream data were not accurate enough to estimate the die-off rate. The rate was estimated so that modeling results could be compared.

Reaeration coefficients were specified as input data from Table 6 for the Streeter-Phelps, QUAL II, and WQRRS models and were not changed for the four different surveys (July 1976, August 1976, September 1976, and May-June 1977) except to correct for temperature for the Streeter-Phelps model data. Miller and Jennings (1979) noted little or no change in the reaeration coefficient with discharge in the range of flows found during this study (in general, K_2 varies with changes in discharge).

Orthophosphate and nitrate uptake rates were set to zero since biomass growth and nutrient cycling were not indicated. Diurnal changes in DO and pH were small at all sampling sites except at the Atlanta gage, which was at the head of reach. On August 30, 1976 at the Atlanta gage, DO varied from 8.4 to 10.1 milligrams per liter and pH varied from 7.2

to 8.1. Attached plants were observed at this point and downstream of the Whitesburg gage at the end of the study reach. In addition, each diurnal DO record at all sampling sites in the study segment were analyzed using the Odum technique (Stephens and Jennings, 1976) indicating that the net productivity of DO was insignificant. As an example, measurements at the Fairburn gage in the middle of the reach on August 30, 1976 showed diel variations of 26 to 27.2 degrees Centigrade (78.8 to 80.6 degrees Fahrenheit), 4.1 to 5.2 milligrams per liter of DO, and no change in pH from 6.9, despite low buffer capacity. Alkalinity varied from 12 to 22 milligrams per liter as calcium carbonate and phytoplankton varied from 210 to 2500 cells per milliliter over the period January 1976 to June 1977. Finally, later modeling results confirm that orthophosphate was not removed by biota and nitrate builds up in the stream without removal.

The rate of decay for organic nitrogen and detritus was estimated from a plot of concentration versus travel time and modified slightly in the calibration phase. Decay rates of ammonia and nitrite were estimated and modified during calibration. Those decay rates were 0.2 per day for organic nitrogen, 0.3 per day for ammonia, 2.6 per day for nitrite, and 0.0 per day for nitrate at 20 degrees Centigrade (68 degrees Fahrenheit).

Benthic oxygen demand was assumed to be negligible. Except for attached plants in a short segment at the head of the reach and bedrock outcroppings, the bottom material consisted of sand. Deposits of organic materials downstream of wastewater treatment plants were not detected. Peak hydropower releases could be expected to scour the channel clean on a weekly basis. This is consistent with the bank erosion problems (Jobson and Keefer, 1979) that occurred just upstream of this study reach.

Missing and inconsistent data

Several difficulties were encountered in preparing the data that describe the water quality of the river and tributaries entering the river. These involved measurement of organic loads at the STPs, estimation of the quality of the power-plant effluent, the conversion of 5-day BOD to ultimate BOD, and the conversion of organic nitrogen to organic detritus.

Some organic loads from the wastewater treatment plants and tributaries were underestimated for all four data collection periods. Single grab samples collected during the August and September 1976 surveys were not representative of average loads (July 1976 loadings were estimated from August 1976 loadings). Since discharge was estimated from daily treatment plant records, the problem was compounded. Multiple grab samples collected during the May-June 1977 survey were representative of the average waste treatment loads except the R. M. Clayton plant loading was underestimated.

Estimates of wastewater treatment plant and tributary loads were revised if needed, using in-stream measurements of discharge and water quality. The procedure involved a mass balance in the stream using the next upstream and the next downstream sites bracketing the location where the questionable load entered the river. This procedure was valid because the in-stream measuring sites were originally chosen such that the stream was laterally mixed. For further assurance, samples were composited from four depth-integrated aliquots taken across the stream.*

The withdrawal and discharge of the power-plant cooling water at RK 481.6 (RM 299.1) were treated in a similar fashion. Effluent discharge was estimated, and then the effluent temperature was calculated from upstream and downstream measurements of water temperature. It was assumed that other water-quality parameters did not change as the water was withdrawn and returned and that the water withdrawn was equal to the amount returned to the stream.

Five-day BOD was reported for the July, August, and September 1976 studies, while 5-day and ultimate BOD was reported for the May-June 1977 study. Since the QUAL II and Streeter-Phelps models work on the basis of ultimate BOD, a conversion factor was needed (the QUAL II model has a 5-day BOD option, but it converts 5-day BOD to ultimate BOD using a fixed deoxygenation rate of 0.23 per day). Optimum deoxygenation rates should not vary whether ultimate or 5-day BOD is used, but if 5-day BOD is used, less DO is consumed through BOD decay.

The BOD data collected in May-June 1977 show ultimate BOD is 2.5

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* Personal communication, June 1981, Robert Faye, USGS, Atlanta, Georgia.

times larger than 5-day BOD. The ratio varied from 2.1 to 3.5 without a noticeable difference between BOD samples collected in the river or from sewage treatment plants and tributaries. The 5-day BOD data collected during July, August, and September 1976 were multiplied by 2.5 to convert that data to ultimate BOD.

Because the WQRRS model simulated organic detritus rather than separate components that include organic nitrogen, organic nitrogen data were used to estimate organic detritus concentrations. Organic detritus was assumed to contain 8 percent organic nitrogen because that factor was used as a default conversion factor in the WQRRS model.

Model Results

Calibration

Model results for the Streeter-Phelps and QUAL II models were obtained through calibration using the August 1976 and May-June 1977 data. The WQRRS model was calibrated with the August 1976 data. The Streeter-Phelps model was calibrated to simulate BOD; organic nitrogen; ammonia; nitrite; nitrate; DO; orthophosphate; coliform bacteria; and dissolved chromium, lead, and zinc, in that order. Travel-time and stream-temperature measurements were specified as input data. The QUAL II model was calibrated to simulate velocity; depth; temperature; BOD; ammonia; nitrite; nitrate; DO; orthophosphate; coliform bacteria; and dissolved chromium, lead, and zinc, in that order. The WQRRS model was calibrated to simulate velocity, depth, temperature, BOD, detritus, ammonia, nitrate, and DO, in that order.

Deoxygenation, reaeration, nitrate uptake, orthophosphate uptake, and coliform bacteria die-off coefficients were estimated independently and were not changed during calibration. Nitrification coefficients for organic nitrogen or detritus decay, ammonia decay, and nitrite decay were chosen using the calibration data. Equivalent coefficients were used in all three models.

For the August 1976 data, the QUAL II model predicted a travel time from Atlanta to Whitesburg (RK 487.8-418.4, RM 302.97-259.85) of

45.3 hours. The WQRRS model predicted a travel time of 44.9 hours. The measured travel time specified in the Streeter-Phelps model was 44.3 hours. Figures 17 and 18 compare depth and velocity simulated by the QUAL II and WQRRS models to reach-average depth and velocity specified for the Streeter-Phelps model.

Next, the QUAL II and WQRRS models were calibrated to simulate temperature. Jobson and Keefer (1979) and Faye, Jobson, and Land (1979) determined that the wind-speed function for the Chattahoochee River should be 70 percent of the wind-speed function derived by an energy balance in the San Diego Aqueduct (Jobson and Keefer, 1979, p. 6). However, because the WQRRS and QUAL II models simulated short- and long-wave radiation and did not explicitly compensate for tree shading, some differences result that must be taken into account with the wind-speed function. In addition, both models had an atmospheric-turbidity factor that was estimated.

In calibrating the QUAL II model, it was discovered that a program error existed in the steady-state temperature submodel (see NCASI, 1980, for a detailed explanation). Using Jobson's wind-speed coefficients $(a + bW)$ of $a = 2.44 \cdot 10^{-9}$ meter per second per millibar (0.001 feet per hour per inch of mercury) and $b = 9.16 \cdot 10^{-10}$ per millibar (0.00016 feet per hour/inch of mercury/miles per hour) and estimating the dust-attenuation coefficient as 0.04, the QUAL II model underpredicts temperature by as much as 4.5 degrees Centigrade (8.1 degrees Fahrenheit) compared to the August 1976 data.

After correcting the program error, the optimum wind-speed function was determined such that the root mean square (RMS) error was minimized. For the August 1976 data, the optimum wind-speed function was 0.55 of the function determined for the San Diego Aqueduct. The optimum for the May-June 1977 data was 0.80. The sum of the RMS error for both data sets indicated 0.65 was the optimum factor, compared to 0.70 determined by Jobson and Keefer (1979) for Chattahoochee River upstream of Atlanta, and Faye, Jobson, and Land (1979) for this reach downstream of Atlanta. Because the RMS difference between the factor 0.65 and 0.70 was 1.33 versus 1.36 and the determination of 0.70 by

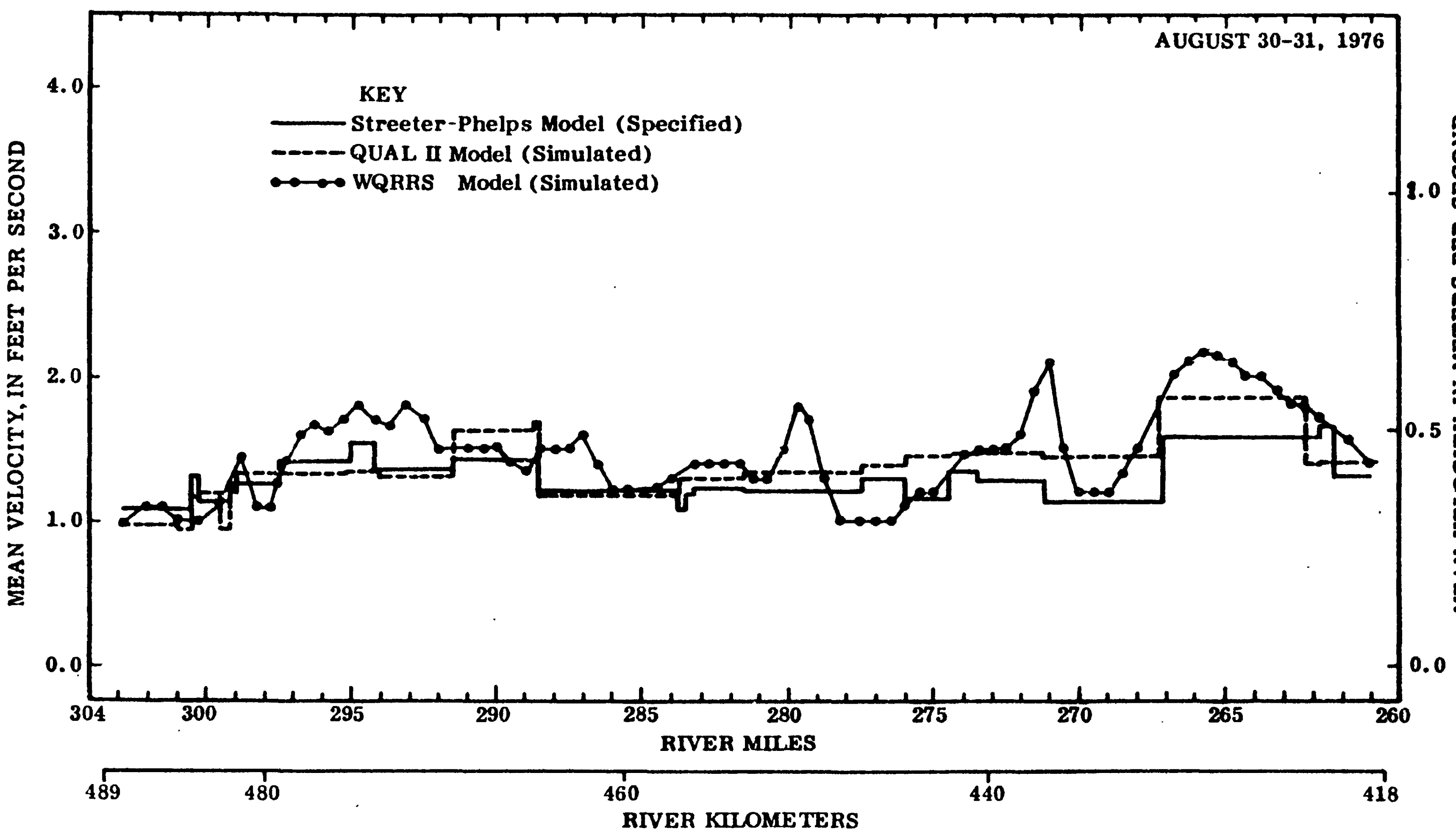
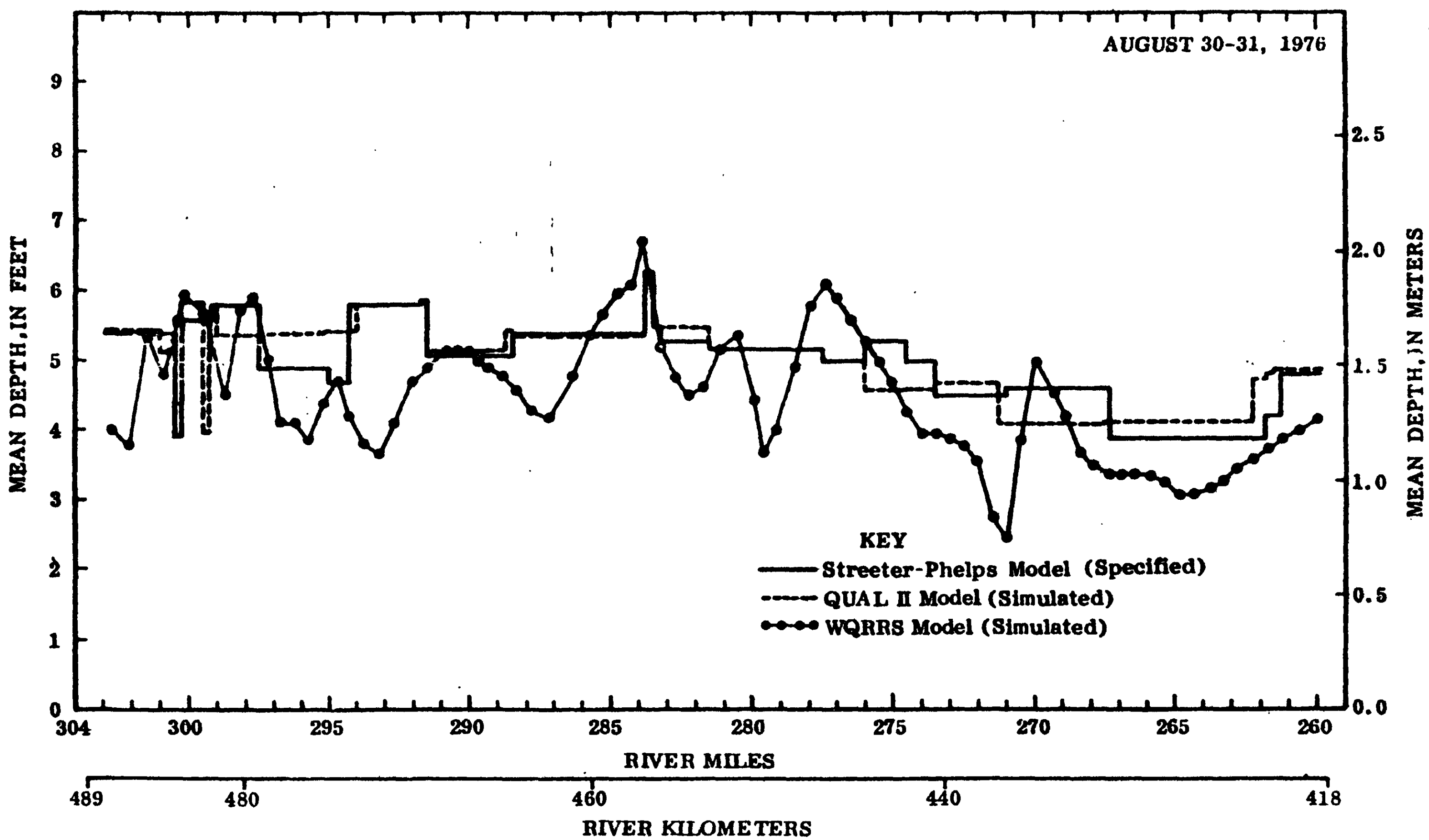


Figure 17. Comparison of mean velocities specified for the Streeter-Phelps model to mean velocities simulated by the QUAL II and WQRRS models for the Chattahoochee River



DISTANCE UPSTREAM OF THE MOUTH OF THE CHATTAHOOCHEE RIVER

Figure 18. Comparison of mean depths specified for the Streeter-Phelps model to mean depths simulated by the QUAL II and WQRRS models for the Chattahoochee River

the investigators cited in the preceding sentence was based on longer periods of time and two different reaches of the river, the factor 0.7 [a = $2.44 \cdot 10^{-9}$ meters per second per millibar (0.001 feet per hour per inch of mercury) and b = $9.16 \cdot 10^{-10}$ per millibar (0.00016 feet per hour/inch of mercury/miles per hour)] was adopted along with a dust attenuation factor of 0.04 in calibrating the QUAL II model. These results are shown in Figure 19.

The WQRRS model underpredicted temperature using Jobson's coefficients, but only by 2 degrees Centigrade (4 degrees Fahrenheit) at most for the August 1976 data. The WQRRS model simulation shown in Figure 19 is based on default coefficients of a = 0.0, b = $1.5 \cdot 10^{-10}$ per millibar (0.000026 feet per hour/inch of mercury/miles per hour) and an atmospheric-turbidity factor = 2.0. Results based on these coefficients are illustrated in Figure 19 and were used as final calibration values because these results showed that the WQRRS model could make predictions under these conditions to within 2 degrees Centigrade (4 degrees Fahrenheit) of measurements, without prior calibration, using coefficients recommended in the model documentation (Smith, 1978). This difference in wind-speed coefficients was the only case where different model coefficients were used.

The high temperature predictions made by the WQRRS model between RK 487.8 to RK 481.6 (RM 302.97-299.1) were due to the upstream shift of the heat load entering at RK 481.6 (RM 299.1). This did not effect the validity of the results downstream of RK 481.6 (RM 299.1).

In the third step, BOD predictions based on a K_1 of 0.16 per day were checked. Figure 20 indicates that 0.16 per day was appropriate. Comparison with data in Figure 20 showed that the May-June 1977 BOD predictions from the Streeter-Phelps, QUAL II, and Velz models were equivalent despite the differences in solving the mass balance equations. The slight difference in predictions of BOD between the Streeter-Phelps and QUAL II models indicated that numerical dispersion in the QUAL II model was insignificant for these steady-state simulations.

Five-day BOD predictions from the WQRRS model shown in Figure 20 were not equivalent to ultimate BOD predictions from the Streeter-Phelps

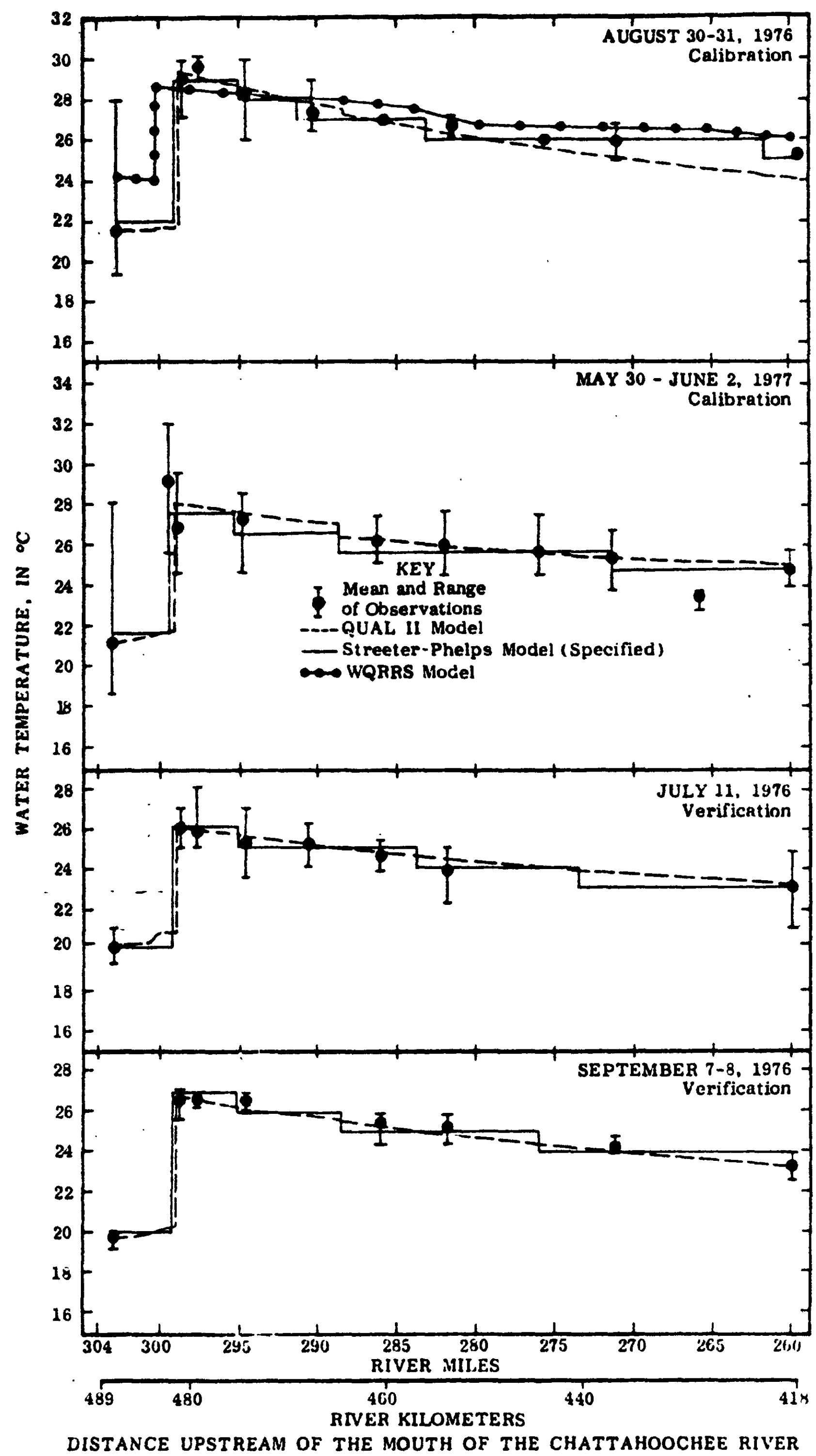


Figure 19. Observed and predicted water temperatures in the Chattahoochee River

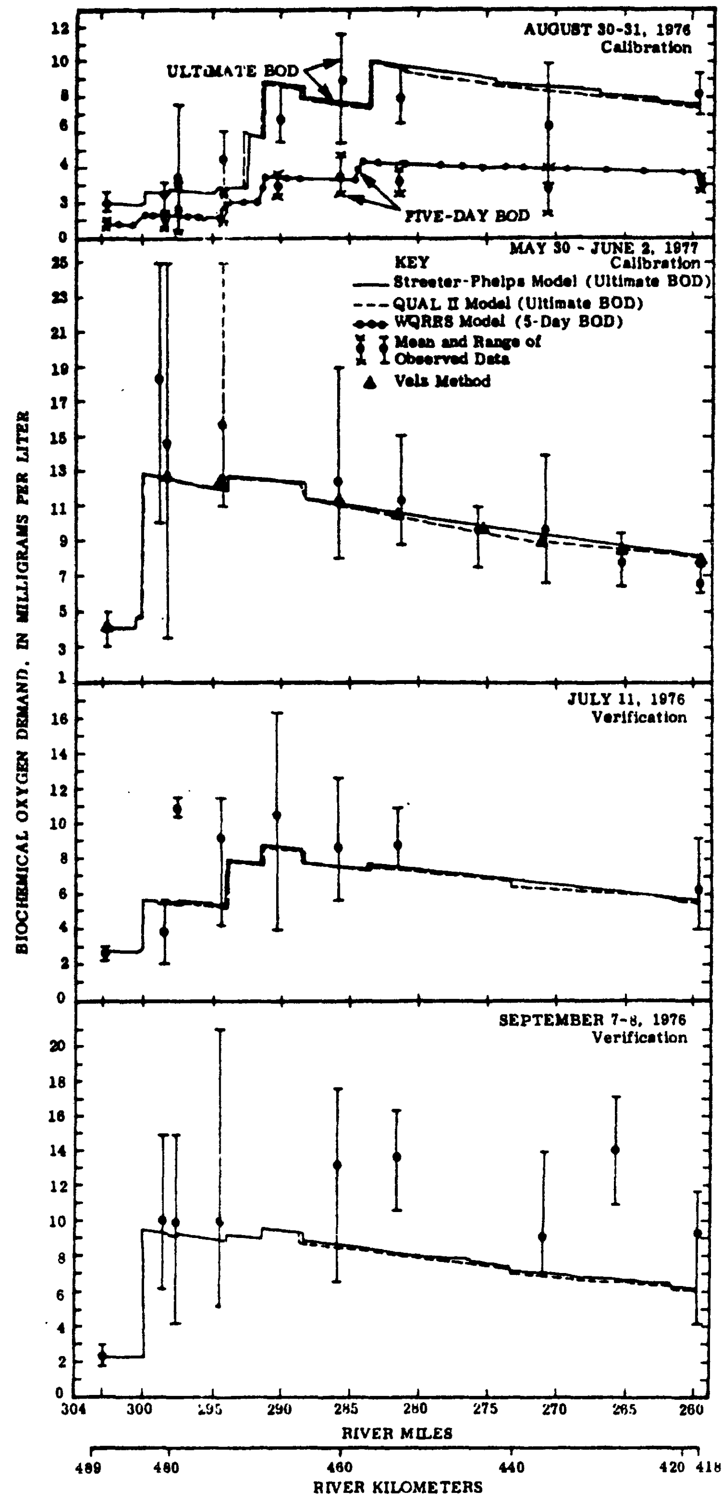


Figure 20. Observed and predicted biochemical oxygen demand, Chattahoochee River

and QUAL II models. The ratio of ultimate to 5-day BOD varied from 2.5 at the head of the reach to 2.1 at the downstream end of the reach rather than remaining constant at 2.5 over the entire reach. This occurred despite the fact the same deoxygenation rate and BOD loads were specified for the WQRRS, Streeter-Phelps, and QUAL II models. The WQRRS model appeared to use the same temperature correction for K_1, and temperature predictions were about the same (Figure 19). The manner in which dispersion was included in the numerical solution of the mass balance equations for the WQRRS model was unclear.

The fourth step of the calibration involved determination of coefficients for the nitrification process. At this point, the Streeter-Phelps, QUAL II, and WQRRS models diverge in formulation. The Streeter-Phelps model simulates organic nitrogen, ammonia, nitrite, and nitrate. The QUAL II model simulates ammonia, nitrite, nitrate, and chlorophyll a. The WQRRS model predicts organic detritus (8 percent organic nitrogen), ammonia, nitrite (not printed), nitrate, and several different forms of biota.

Figure 21 illustrates the calibration of the Streeter-Phelps and WQRRS models to predict total organic nitrogen or detritus using the August 1976 data. In order to predict detritus with the WQRRS model, organic sediment had to be simulated. Since no data existed, the initial amount of organic sediment was specified as zero and the settling velocity of detritus was specified as zero. However, a recent update corrects this problem so that detritus can be modeled without modeling organic sediment.

Figure 21 indicates first-order decay with a decay rate of 0.2 per day used in the Streeter-Phelps model was adequate for simulating organic nitrogen in the Chattahoochee River. The tendency for total organic nitrogen to decrease with distance downstream of the waste treatment plants may also confirm the initial hypothesis that phytoplankton growth was not significant.

Like BOD, a difference in organic nitrogen decay in the WQRRS model was indicated. Figure 21 shows the detritus prediction of the WQRRS model for August 1976 data using a decay rate of 0.2 per day and

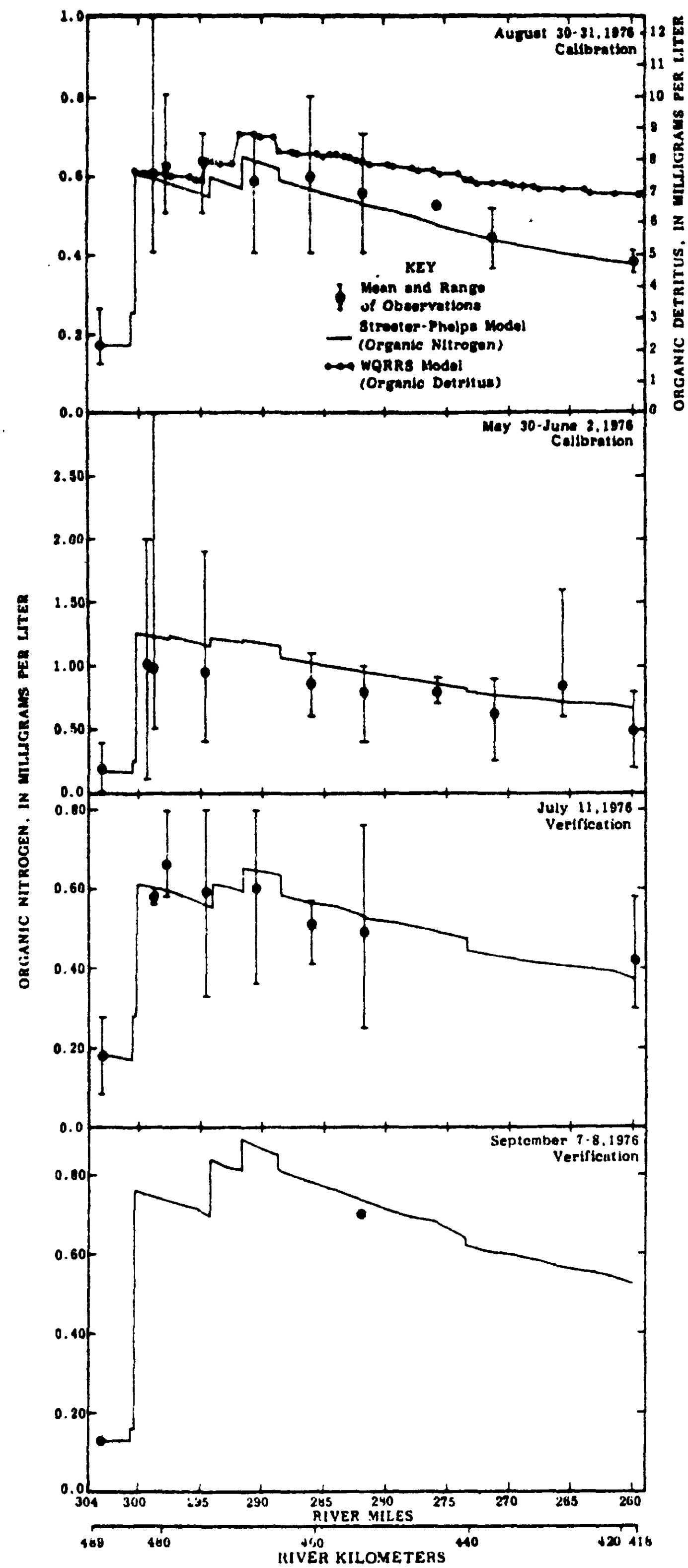


Figure 21. Predictions of organic nitrogen (Streeter-Phelps model) and organic detritus (WQRRS model) compared to measurements for the Chattahoochee River

the same waste loads. Compared with the Streeter-Phelps model simulation, detritus was removed at a slower rate than organic nitrogen.

Next, the ammonia decay rate was estimated as 0.3 per day for the Streeter-Phelps model using the agreements of ammonia, nitrite, nitrate, and DO predictions with measurements made August 1976 and May-June 1977 as criteria. Following that, the nitrite decay rate was adjusted slightly to 2.6 per day. The nitrate removal rate remained zero.

Following this, the ammonia decay rate of 0.3 per day and nitrite decay rate of 2.6 per day were specified for the QUAL II and WQRRS models. Biomass was not modeled. The QUAL II model simulation confirmed the Streeter-Phelps model calibration. Figures 22 and 23 compared model predictions for ammonia and nitrate to measurements. Nitrite was not plotted since the QUAL II and WQRRS models did not print those results in the model summaries of results. However, the Streeter-Phelps model simulations of nitrite were accurate. In general, the predictions of nitrite were higher than measurements, but the differences were minor.

The May-June 1977 plot in Figure 22 illustrates good agreement between the Velz model and the QUAL II model in predicting ammonia which was to be expected since neither model simulated organic nitrogen decay. The August 1976 and May-June 1977 applications of the Streeter-Phelps and QUAL II models demonstrated that organic nitrogen decay had a small effect on ammonia predictions. At most, the Streeter-Phelps model predicted 0.2 milligrams per liter more ammonia nitrogen than the QUAL II model at RK 418.4 (RM 259.85) for May-June 1977. The August 1976 application indicated that smaller amounts of ammonia were removed in the WQRRS model simulation when compared to the Streeter-Phelps model simulation using equivalent ammonia decay rates and loadings. This followed the trend noted from the simulation of BOD and detritus.

Figure 23, illustrating nitrate measurements and predictions, indicated a difference in the Velz model and the QUAL II model. However, since details for the Velz model were not available, the cause of this difference was unknown. In addition, the expanded scale in Figure 23 better illustrates the effect of modeling organic nitrogen. The Streeter-Phelps model predicted 0.1 to 0.2 milligrams per liter of

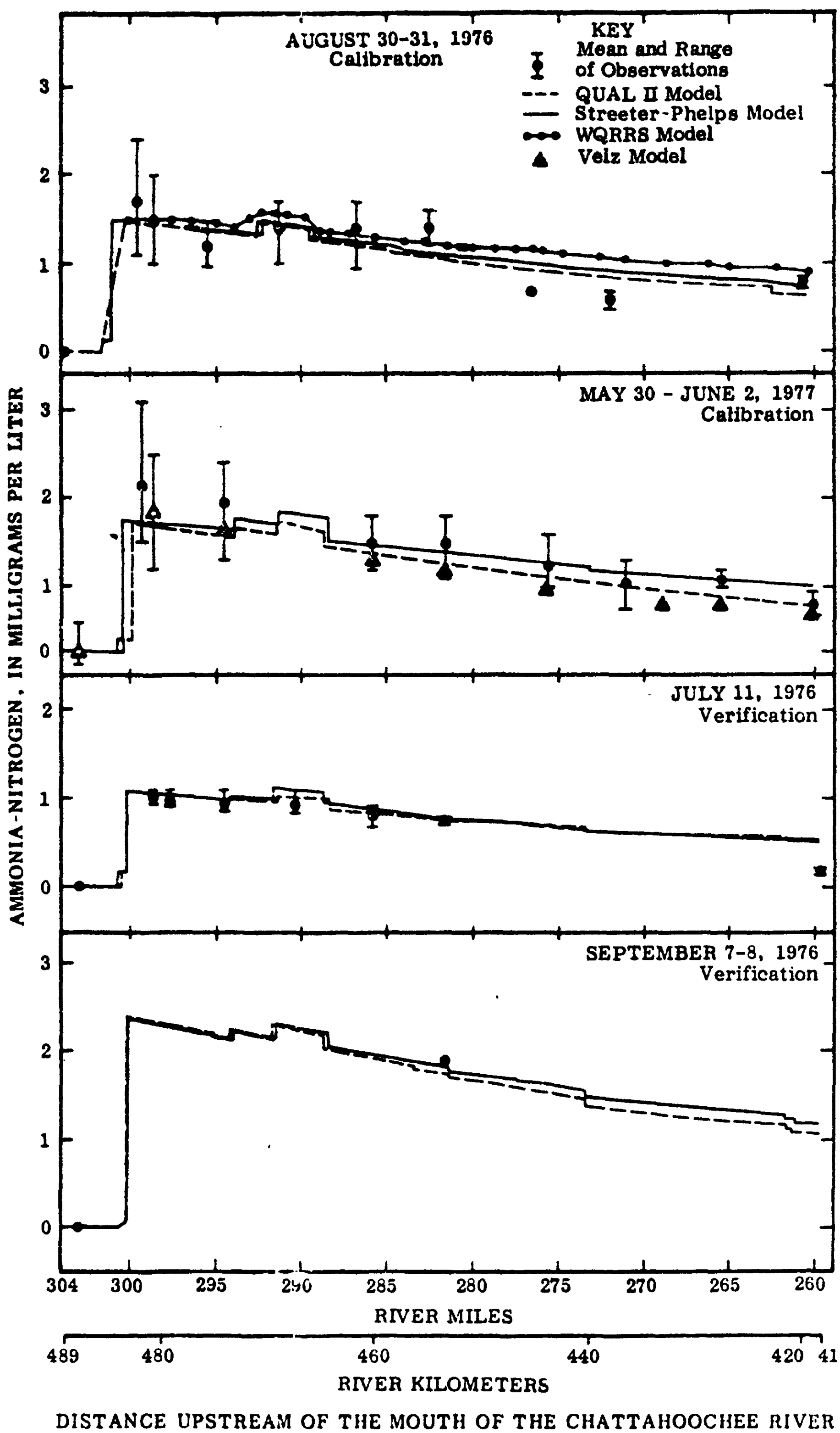


Figure 22. Observed and predicted ammonia-nitrogen, Chattahoochee River

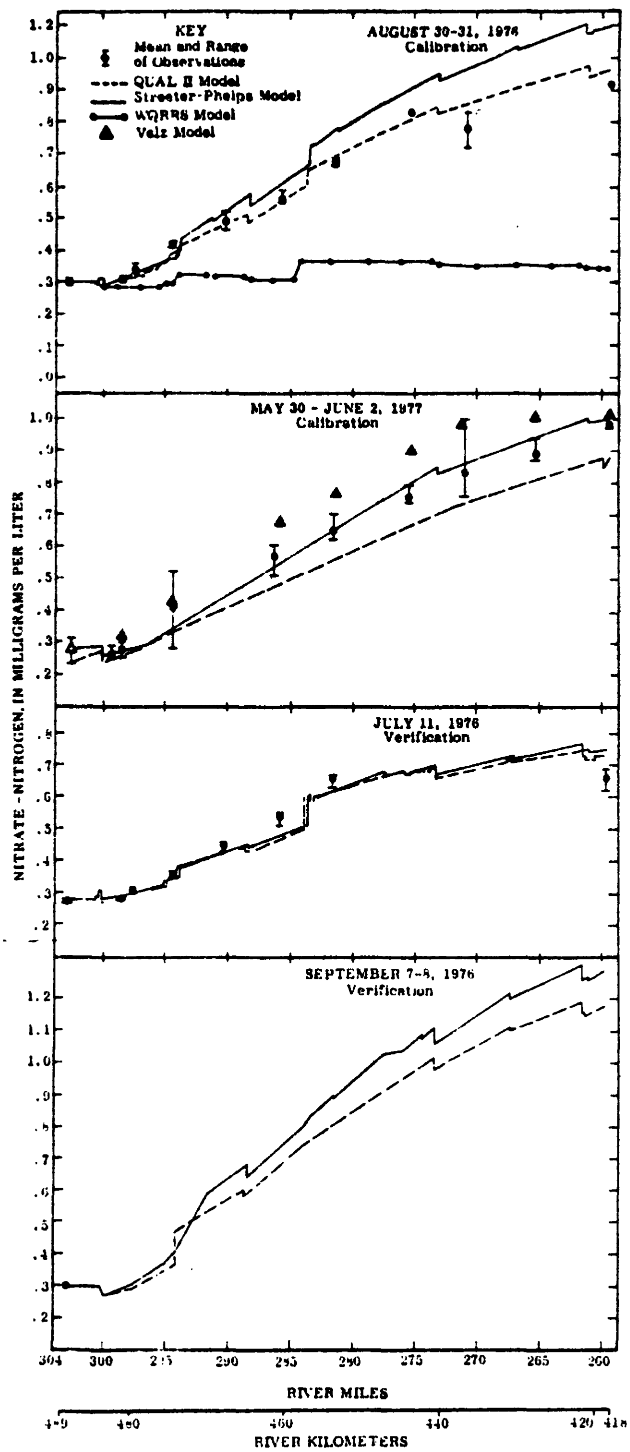


Figure 23. Observed and predicted nitrate-nitrogen, Chattahoochee River

nitrate nitrogen more than the QUAL II model because of organic nitrogen simulation.

The WQRRS model predictions for nitrate, for the August 1976 application, indicated that the nitrate balance was uncoupled from ammonia decay. Simulated nitrate concentrations changed at inflows but did not respond to ammonia decay.

In summary, the Streeter-Phelps, QUAL II, and Velz models gave about the same results for ammonia and nitrate. Simulation of organic nitrogen decay had a minor effect on the results. The WQRRS model predicted slower ammonia decay and nitrate predictions were incorrect.

In the fifth step, DO predictions were checked. Measured reaeration coefficients were specified for the Streeter-Phelps, QUAL II, and WQRRS models. Reaeration coefficients were computed by Stamer and others (1979) using the Velz technique in the Velz model. Benthic demand and photosynthesis were assumed to be insignificant.

Figure 24 illustrates DO predictions and measurements used for calibration. The Streeter-Phelps, QUAL II, and WQRRS models tended to overpredict DO for August 1976 in the downstream reaches compared to the few data collected from that segment. Differences between the single measurement at RK 418.36 (RM 259.85) and predictions from the Streeter-Phelps, QUAL II, and WQRRS models were 1.5, 2.0, and 2.5 milligrams per liter, respectively. The more reliable May-June 1977 data were in better agreement with predictions from the Streeter-Phelps and QUAL II models. Both models also slightly overpredict DO for the May-June 1977 data. The greatest difference between mean observations and predictions was 0.7 and 1.0 milligrams per liter for the Streeter-Phelps and QUAL II models, respectively.

The Velz model simulation was in close agreement with both sets of calibration data. However, the Velz iterative technique was used to calculate reaeration forcing a better fit to the data than was achievable using reaeration measurements by Tsivoglou and Wallace (1972).

Different model predictions of DO were attributed to several factors. First, the simulation of organic nitrogen resulted in small differences in predictions of ammonia and nitrate for the Streeter-Phelps

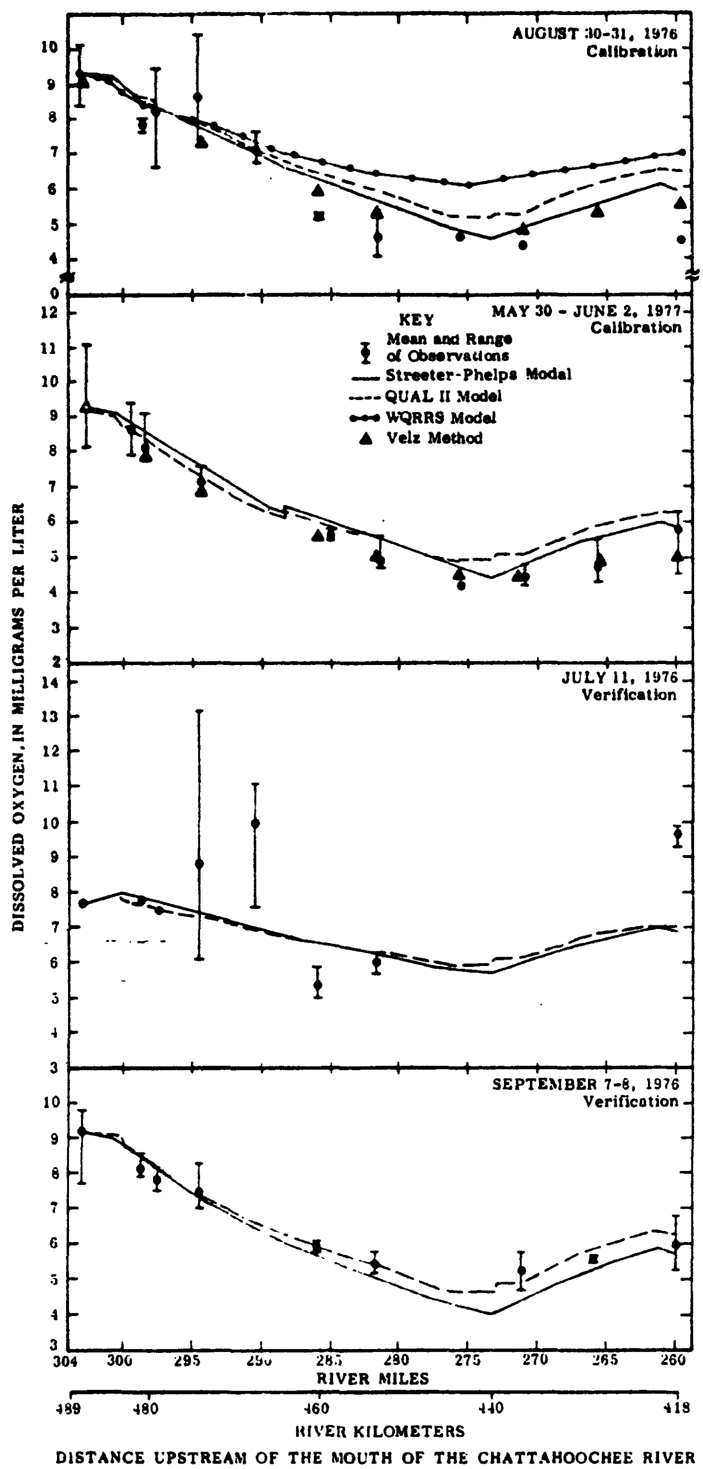


Figure 24. Observed and predicted dissolved oxygen, Chattahoochee River

and QUAL II models. However, these small differences in nitrate translate into larger oxygen demands by a factor of 4.57. Second, detritus, BOD, and ammonia decayed at slower rates in the WQRRS model simulation.

Overall, the calibrations for the Streeter-Phelps, QUAL II, and WQRRS models were reasonable. Temperature predictions from the QUAL II and WQRRS models were accurate. Measurements of BOD showed considerable scatter but predictions adequately described mean concentrations. Nitrogen predictions were also adequate and confirm that photosynthesis was not important. DO predictions were reasonable, but showed a tendency to overestimate.

Following DO calibration, orthophosphate was simulated with a zero uptake rate by biomass to confirm that photosynthesis was not significant. The data describing orthophosphate were limited to the August 1976 and May-June 1977 studies. Figure 25 confirms that orthophosphate-phosphorus can be simulated as a conservative substance for this segment of the Chattahoochee River using the Streeter-Phelps or QUAL II models.

Finally, limited data describing fecal coliform bacteria and dissolved chromium, lead, and zinc from the August 1976 study were used to evaluate options to predict coliform bacteria and three conservative substances in the Streeter-Phelps and QUAL II models. Figure 26 shows that the Streeter-Phelps and QUAL II models gave equivalent predictions for fecal coliform bacteria with a die-off rate of 0.08 per day estimated from Zison and others (1978). The data in Figure 26, resulting from single grab samples, were not suitable to determine the validity of the first-order die-off formulations for predicting fecal coliform bacteria.

Dissolved metal measurements in Figure 27 were also based on single grab samples. In modeling the data, both the Streeter-Phelps and QUAL II models gave the same predictions as was to be expected since the simulations were based on the conservative substances options in the models. The predictions indicate that chromium seems to behave as a conservative substance for this segment of the Chattahoochee River, whereas lead and zinc does not. In comparing the two models, the QUAL II model has the greater flexibility in that units of the conservative

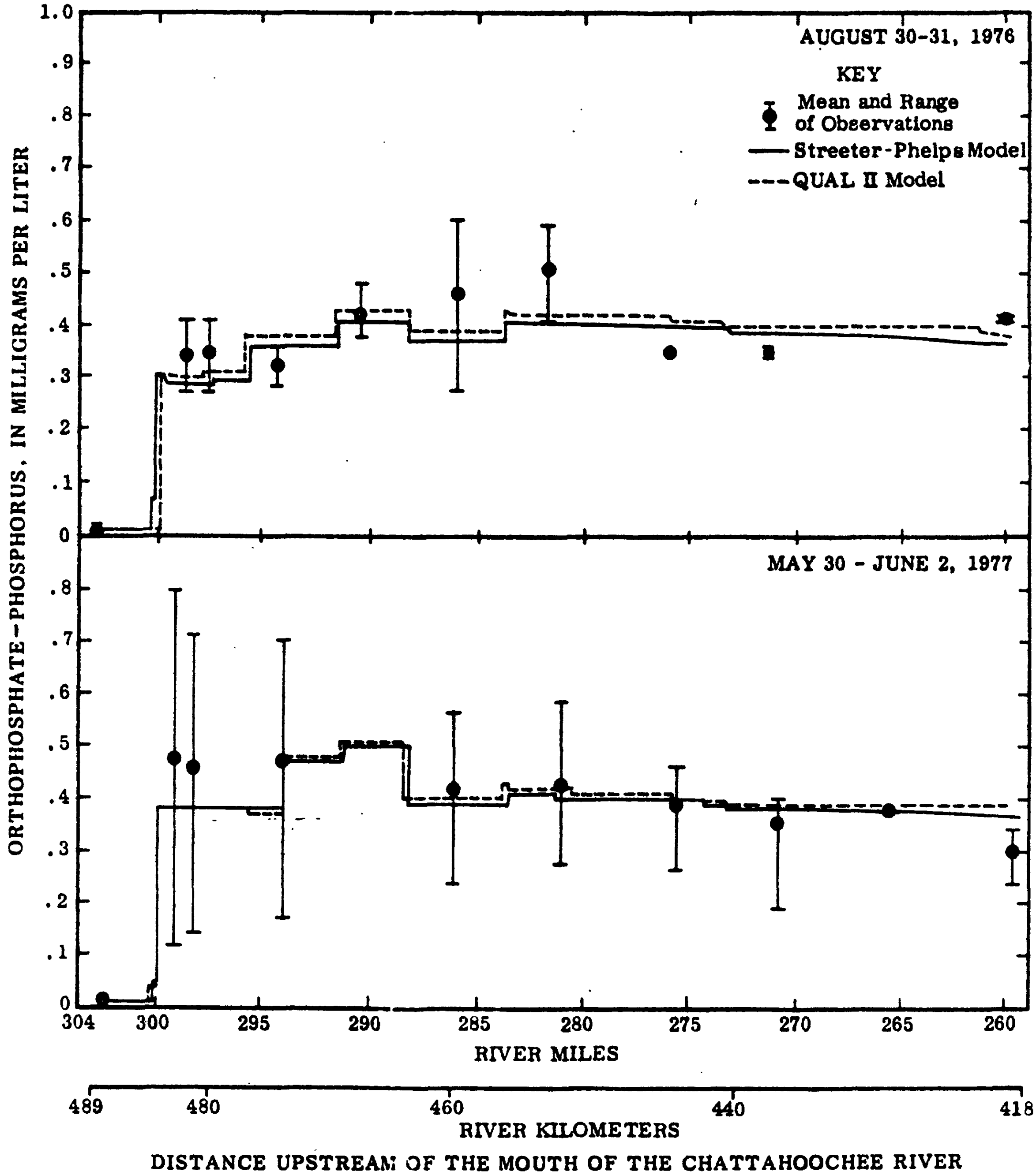


Figure 25. Observed and predicted orthophosphate-phosphorus, Chattahoochee River

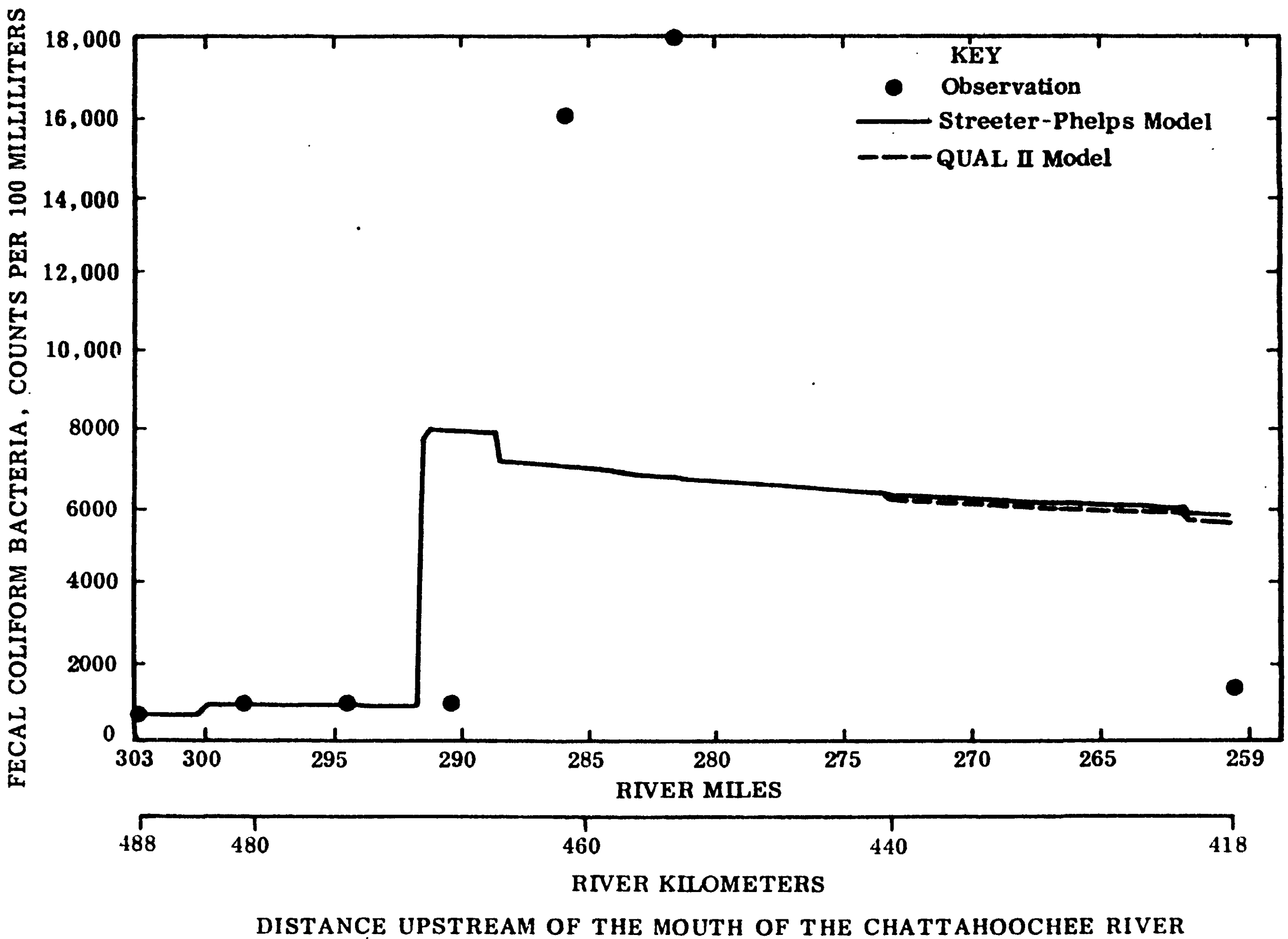


Figure 26. Observed and predicted fecal coliform bacteria, August 30-31, 1976, Chattahoochee River

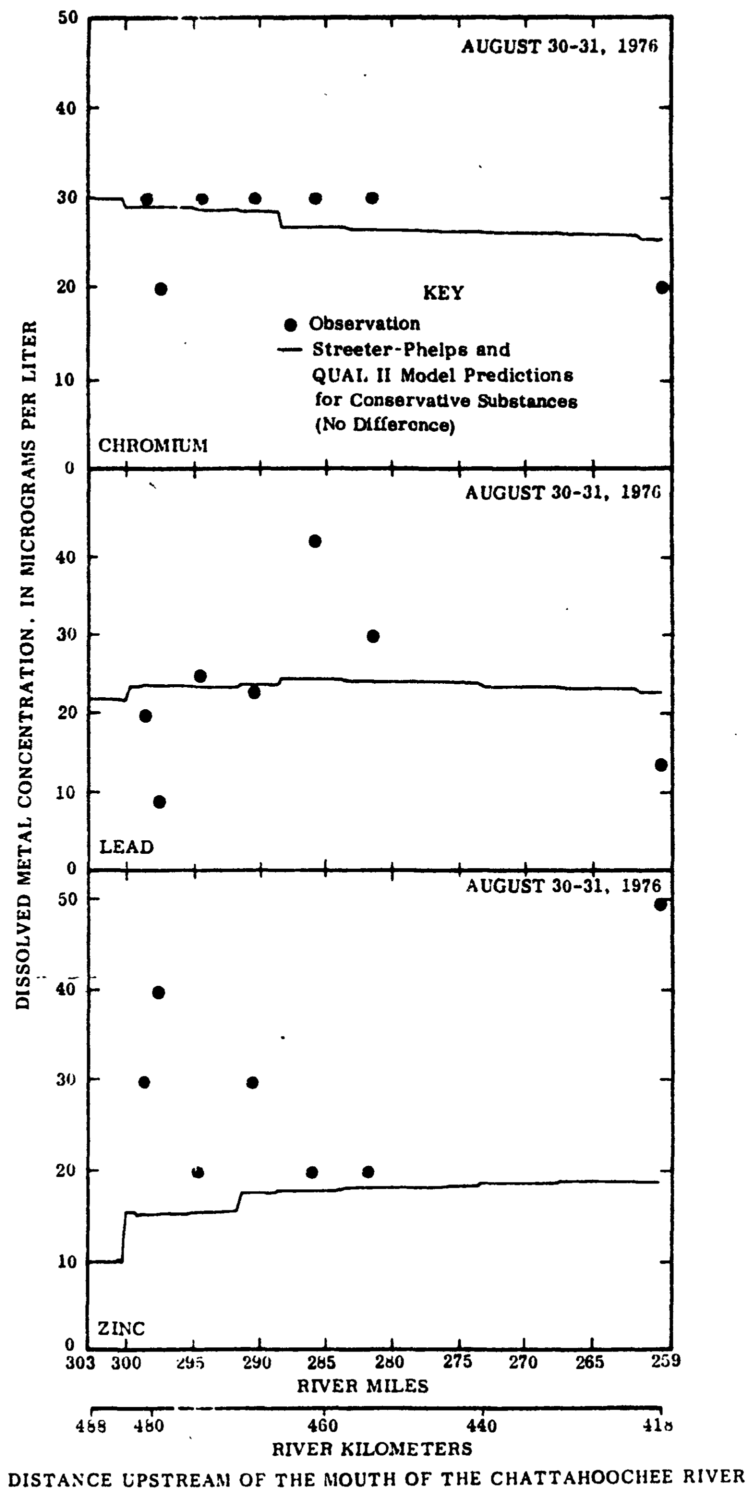


Figure 27. Observations and predictions of dissolved metals (assuming the metals behave as conservative substances), Chattahoochee River

substance can be specified. The Streeter-Phelps model is limited to the concentration units of milligrams per liter.

<u>Verification</u>

Following calibration, the temperature predictions of the QUAL II model, the organic nitrogen predictions of the Streeter-Phelps model, and the BOD, ammonia, nitrate, and DO predictions of the Streeter-Phelps and QUAL II models were verified using the coefficients determined by calibration. These predictions were compared to the July and September 1976 data in Figures 19 through 25.

Figure 19 shows that the temperature predictions from the QUAL II model were accurate to 0.6 degrees Centigrade (1.1 degrees Fahrenheit) compared to the verification data. The greatest difference between predictions and the mean of measurements at a point in the calibration data was 1.7 degrees Centigrade (3.1 degrees Fahrenheit). The large difference between the May-June 1977 observation at RK 481.55 (RM 299.1) and the prediction from the QUAL II model was due to discretization error.

Figure 20 shows the verification of the BOD formulations for the July and September 1976 data. Despite considerable scatter in the data, results from the Streeter-Phelps and QUAL II models seem to be adequate. The greatest difference between predictions and mean observations was 7.4 milligrams per liter or 53 percent.

Figures 21 to 23 confirm nitrification predictions. The greatest difference between organic nitrogen predictions from the Streeter-Phelps model and mean observations from July 1976 was 0.07 milligrams per liter. The greatest difference between ammonia predictions and mean observations from July 1976 was 0.15 milligrams per liter for the Streeter-Phelps model and 0.25 milligrams per liter for the QUAL II model. The greatest difference between nitrate predictions and mean observations from July 1976 was 0.09 milligrams per liter for the Streeter-Phelps model and 0.07 milligrams per liter for the QUAL II model.

Figures 21 to 23 also include predictions of organic nitrogen, ammonia, and nitrate for September 1976 despite the fact that few data were collected to compare with predictions. The plots show that the range of predictions are not greatly different from the range for which the models were calibrated and illustrate the difference due to organic nitrogen.

Figure 24 confirmed the predictive capability for DO. There was a good fit to September 1976 measurements but a less-than-satisfactory fit to the July 1976 data. The greatest difference between mean observation and prediction was 3.0 milligrams per liter or 30 percent for the July 1976 data. The July 11, 1976 DO data were collected in a period of less than a day and measurements at RK's 474.39, 467.82, and 418.36 (RM's 294.65, 290.57, and 259.85) exceed DO saturation values where there was no indication that supersaturated conditions existed. Because this was the first study, problems may have occurred in calibrating DO meters or the flow may have not been steady throughout the reach for this short period.

Comparison

These model applications using the Chattahoochee River data indicates that the Streeter-Phelps, QUAL II, and Velz models give about the same results despite slightly different formulations. The Streeter-Phelps and Velz models were limited to steady-state conditions and did not simulate travel time and temperature. Slight differences in BOD and coliform bacteria predictions between the Streeter-Phelps and QUAL II models indicated that numerical dispersion in the QUAL II model was small. The QUAL II model did not simulate organic nitrogen, leading to small differences in nitrogen predictions and greater differences in DO predictions compared to the Streeter-Phelps model. The data were not precise enough to determine the significance of this difference.

Temperature predictions from the QUAL II and WQRRS models were quite accurate. These results indicate that the QUAL II model needs calibration to achieve this accuracy. The WQRRS model seems capable of making accurate temperature predictions without calibration based on the guidance given in the documentation and accurate inflow data.

The WQRRS model uses first-order decay formulations to describe detritus, ammonia, and BOD like the Streeter-Phelps and QUAL II models. However, despite using the same loading and decay coefficients in all three models, the WQRRS model predicts less detritus, ammonia, and BOD removal. This difference can probably be attributed to higher dispersion computed in the WQRRS model.

In addition, 5-day BOD utilized by the WQRRS model had an internal conversion to ultimate BOD using the factor 1.46. For these data that factor should have been 2.5. This led to an overprediction of DO since the oxygen demand due to BOD was underpredicted by a factor of 0.58.

Nitrate predictions from the WQRRS model proved to be invalid. Nitrite decay was not coupled to the nitrate formulation. This problem has since been corrected in the HEC library version of the model and did not affect ammonia and nitrite simulation.

Despite the flexibility of the WQRRS model discretization scheme, the model proved difficult to apply to the Chattahoochee River. The crucial limitation involved the limit of 10 inflows, withdrawals, and nonpoint sources.

PART VI: MODEL APPLICATIONS WITH WILLAMETTE RIVER DATA

Model Preparation

Application

The Willamette River data contained in McKenzie and others (1979) were transformed to fit the requirements of the Streeter-Phelps, QUAL II, and WQRRS models. The calibration and verification procedure was similar to that used to simulate water quality in the Chattahoochee River. The August 1974 data were used for the calibration of the three models. The Streeter-Phelps and QUAL II models were verified with the July-August 1973 data.

The Streeter-Phelps model was used to simulate BOD, nitrogenous BOD, and DO. The QUAL II and WQRRS models were used to simulate BOD, ammonia nitrogen, nitrite nitrogen (not printed), nitrate nitrogen, and DO. Temperature was simulated with the WQRRS model. Results from the Velz method used by McKenzie and others (1979) to describe BOD and DO were available for comparison.

Stream discretization and hydraulics

Unlike the Chattahoochee River, distinct changes in physical, chemical, and biological conditions occur in the Willamette River. Therefore, reaches were based on these changes and tributaries entering the river. River conditions changed between the Upstream Reach and Newberg Pool and between the Newberg Pool and the Tidal Reach. A benthic DO demand occurred downstream of RK 23.3 (RM 14.5) in the Tidal Reach. Tributaries included four papermill effluents, ten municipal effluents, the headwater inflow at the beginning of the reach, four tributary rivers, and two tributary creeks.

For the Streeter-Phelps model discretization scheme, the Willamette River was divided into 23 reaches and the computational element length was chosen as 3.2 kilometers (2 miles). The upstream ends of the 23 reaches were chosen to coincide with the headwaters, the 20 tributaries, the beginning of the Newberg Pool reach, the beginning of the Tidal Reach at Willamette Falls, and a break point in the Tidal Reach where benthic

DO demand begins. Nineteen reaches were defined using the 20 tributaries. Johnson Creek and the Milwaukie municipal STP effluent enter at the same point on opposite sides of the river and were combined into a single inflow.

For the QUAL II model, the stream was divided into 5 reaches which included the Upstream Reach, Newberg Pool, and 3 reaches in the Tidal Reach. In the Tidal Reach, one segment included a short reach, one element in length, downstream of Willamette Falls in which the reaeration coefficient was increased in an abortive attempt to mimic the reaeration of 0.35 milligrams per liter of DO due to the falls. The remainder of the Tidal Reach was divided at RK 23.3 (RM 14.5) so that benthic demand could be specified in the reach RK 23.3 to 7.2 (RM 14.5 to 4.5).

The computational element length for the QUAL II model was chosen as 3.2 kilometers (2 miles). This choice matched the Streeter-Phelps model element length and was the largest integer number that would fit the model limitations of 20 elements per reach and 100 elements per study segment.

The Willamette River proved to be the most difficult stream to discretize with the WORRS model. The model limitations of 41 cross sections and 10 inflows, along with the fact that Willamette Falls is a natural control, required that the study segment be modeled by five separate applications of the model. These five reaches were RK 139.3 to 105.6 (RM 86.5 to 65.58), RK 105.6 to 76.70 (RM 65.58 to 47.64), RK 76.70 to 42.54 (RM 47.64 to 26.42), RK 42.54 to 22.41 (RM 26.42 to 13.92), and RK 22.41 to 5.64 (RM 13.92 to 3.5). For these reaches, the computational element lengths were, in the above order, 1.2, 1.5, 2.8, 0.98, and 0.84 kilometers (0.75, 0.90, 1.77, 0.52, and 0.63 miles).

Travel times taken from Table 9 were specified for the Streeter-Phelps model, and it was assumed that travel times were not significantly different for the low flows of July-August 1973 and August 1974 for which the discharges at the head of the study reach were 168 and 189 cubic meters per second (6000 and 6760 cubic feet per second), respectively. The QUAL II model was calibrated to simulate the same travel times using $\overline{u} = aQ^{0.05}$, where the small exponent 0.05 was chosen so that the velocity variation between the July-August 1973 application and the August 1974 application would be minor. The coefficient a was calculated for each reach from the reach length, travel time, and discharge.

The WQRRS model was calibrated to simulate the measured travel times using the steady-state backwater option and the cross-sectional geometry measured during flood studies. The channel roughness coefficients derived from the flood studies were reduced in the Upstream Reach to reproduce measured travel times. Travel times in the Newberg Pool and Tidal Reach were controlled by river stage at Willamette Falls and at the end of the Tidal Reach, respectively.

Water-quality coefficients

Water temperatures were specified as input data in the Streeter-Phelps and QUAL II models from measurements made during the water-quality surveys. Temperature was simulated with the WQRRS model using default wind-speed coefficients and estimated meteorological conditions. Despite indications by Smith (1978), temperature could not be specified as initial data and held constant in the WQRRS model. In addition, the option to simulate temperature by the equilibrium temperature method was also not functioning. These errors have been corrected in the latest update.

Reaeration coefficients, calculated by the Velz iterative technique (Hines and others, 1977, p. I29. Note that values in Figure 16 of the first printing should be reduced by a factor of 1/2.303 to be expressed as base 10 per day.) were specified as input data for the Streeter-Phelps, QUAL II, and WQRRS models. For the Streeter-Phelps model, the reaeration coefficient was increased in the short segment just below Willamette Falls to introduce 0.35 milligrams per liter of DO. For the QUAL II model, DO in the Clackamas River, just upstream of the Willamette Falls, was increased to introduce an extra 0.35 milligrams per liter of DO. The Velz simulation added 6078 kilograms per day (13,400 pounds per day) of DO at Willamette Falls. Reaeration at Willamette Falls was not simulated with the WQRRS model.

Deoxygenation rates were taken from McKenzie and others (1979), who in turn derived these rates from BOD bottle decay rates and verified the rates by modeling BOD in the river. Those rates were 0.14 per day for the Upstream Reach and 0.07 for the Newberg Pool and Tidal Reach at 20 degrees Centigrade (68 degrees Fahrenheit). Because the modeling results of McKenzie and others (1979) indicated that a benthos source of BOD existed in the Portland Harbor, a source rate was estimated by trial and error during calibration. The benthic oxygen demand associated with bottom sediments in the Portland Harbor (lower end of the Tidal Reach) was estimated to be 1.2 grams of DO per square meter per day (0.11 grams of DO per square foot per day) or 23,000 milligrams per meter per day (75,300 milligrams per foot per day).

Nitrogenous BOD decay rates of 1.6 per day for the Upstream Reach and zero per day for the two downstream reaches were also taken from the modeling results of McKenzie and others (1979). In the Upstream Reach, the ammonia decay rate was assumed to equal the nitrogenous BOD decay rate of 1.6 per day. The nitrite decay rate was estimated to be 4.5 per day. The nitrate uptake rate was assumed to be zero since photosynthesis was determined to be insignificant (Hines and others, 1977, p. I26). These rates were assumed to be zero in Newberg Pool and Tidal Reach.

Missing and inconsistent data

Several difficulties were encountered in applying these models to the Willamette River data. First, nitrogen data were reported as nitrogenous BOD. The Streeter-Phelps model has a nitrogenous BOD option to simulate nitrogenous BOD decay as a first-order process, but the QUAL II and WORRS models simulate ammonia, nitrite, and nitrate. Therefore, nitrogenous BOD was converted to ammonia using the factor 4.57 milligrams nitrogenous BOD per milligram of ammonia and assuming that nitrogenous BOD was 100 percent ammonia (McKenzie and others, 1979).

Second, nitrogen data were not collected during the July-August 1973 study so that nitrogenous BOD, ammonia, nitrite, and nitrate predictions could not be directly verified. The DO predictions for July-August 1973 served as indirect confirmation criteria for nitrogenous BOD, ammonia, and nitrate predictions.

Third, the July-August 1973 BOD tests did not include a nitrification inhibitor. These data were adjusted by McKenzie and others (1979) using the August 1974 BOD tests, which were run with a nitrification inhibitor.

Model Results

Calibration

The Streeter-Phelps, QUAL II, and WQRRS models were calibrated to predict carbonaceous BOD; nitrogenous BOD or ammonia, nitrite, and nitrate; and DO for the August 1974 data. Temperature was simulated using the WQRRS model.

The measured travel time specified in the Streeter-Phelps model was 263.3 hours. The QUAL II model simulated a travel time of 270.7 hours. The WQRRS model simulated a travel time of 263.1 hours.

Figure 28 shows that the WQRRS model temperature predictions were equivalent to the measurements specified in the Streeter-Phelps and QUAL II models. Default wind-speed coefficients and estimated meteorological data were used for this simulation, indicating the WQRRS model is capable of making future stream-temperature predictions for rivers of this type when calibration and verification are impossible. Heat loads and the effects of upstream reservoirs were insignificant in this reach.

Figure 29 shows the calibration for BOD predictions using the August 1974 data. The Streeter-Phelps, QUAL II, WQRRS, and Velz models gave the same predictions for the Upstream Reach, where the temperature was 20 degrees Centigrade (68 degrees Fahrenheit). In the Newberg Pool and the Tidal Reach, where water temperature was as high as 23 degrees Centigrade (73 degrees Fahrenheit), the WQRRS model simulated lower BOD removal compared to the Streeter-Phelps, QUAL II, and Velz models.

The Streeter-Phelps and QUAL II models were applied such that the addition of BOD to the water in the Tidal Reach was simulated. This addition of BOD was simulated with the Streeter-Phelps model as the difference of two first-order reactions (0.07 per day and 0.01 per day at 20 degrees Centigrade or 68 degrees Fahrenheit). The addition of BOD was simulated with the QUAL II model by specifying a negative BOD

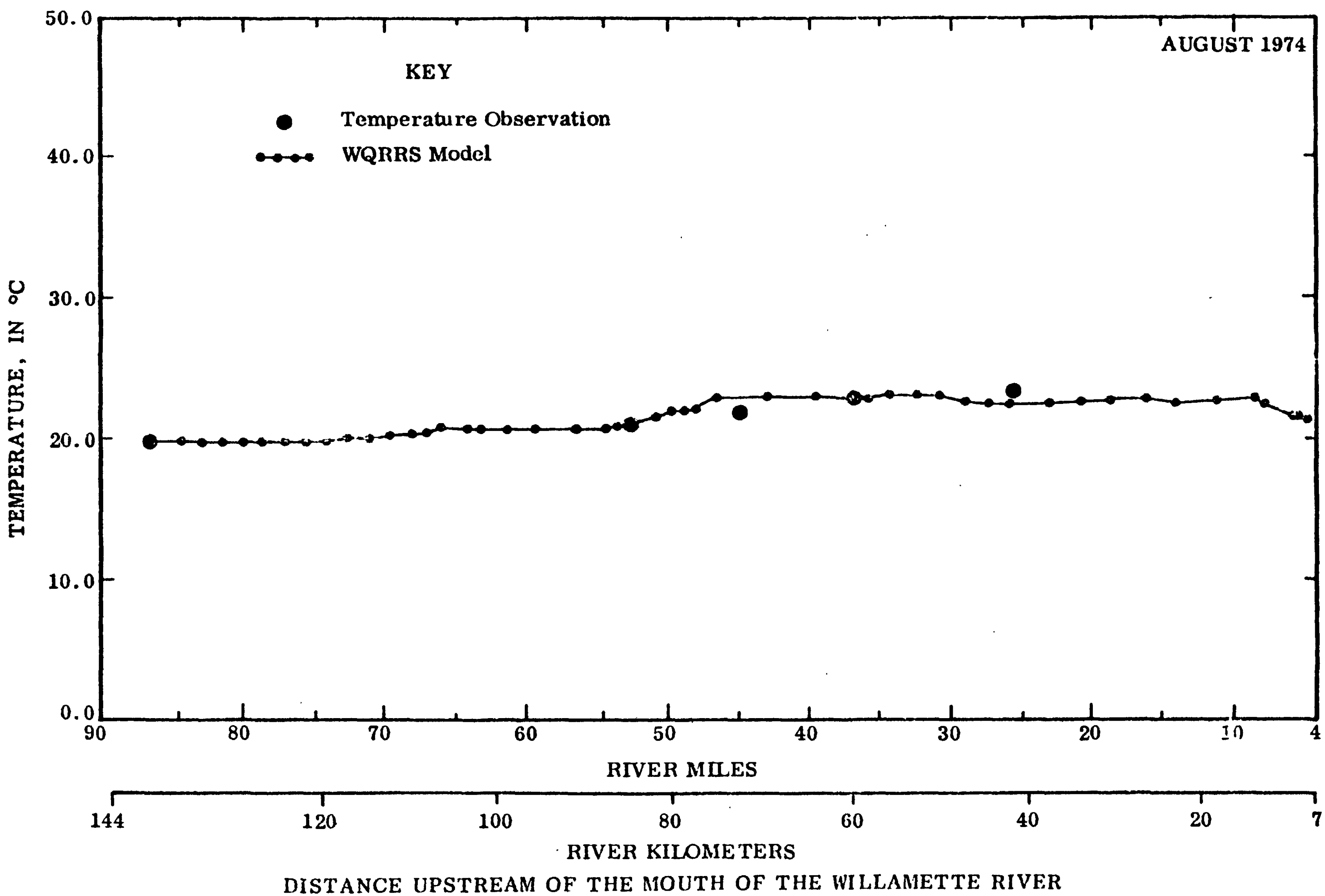


Figure 28. Observed water temperatures compared to WQRRS model predictions, Willamette River

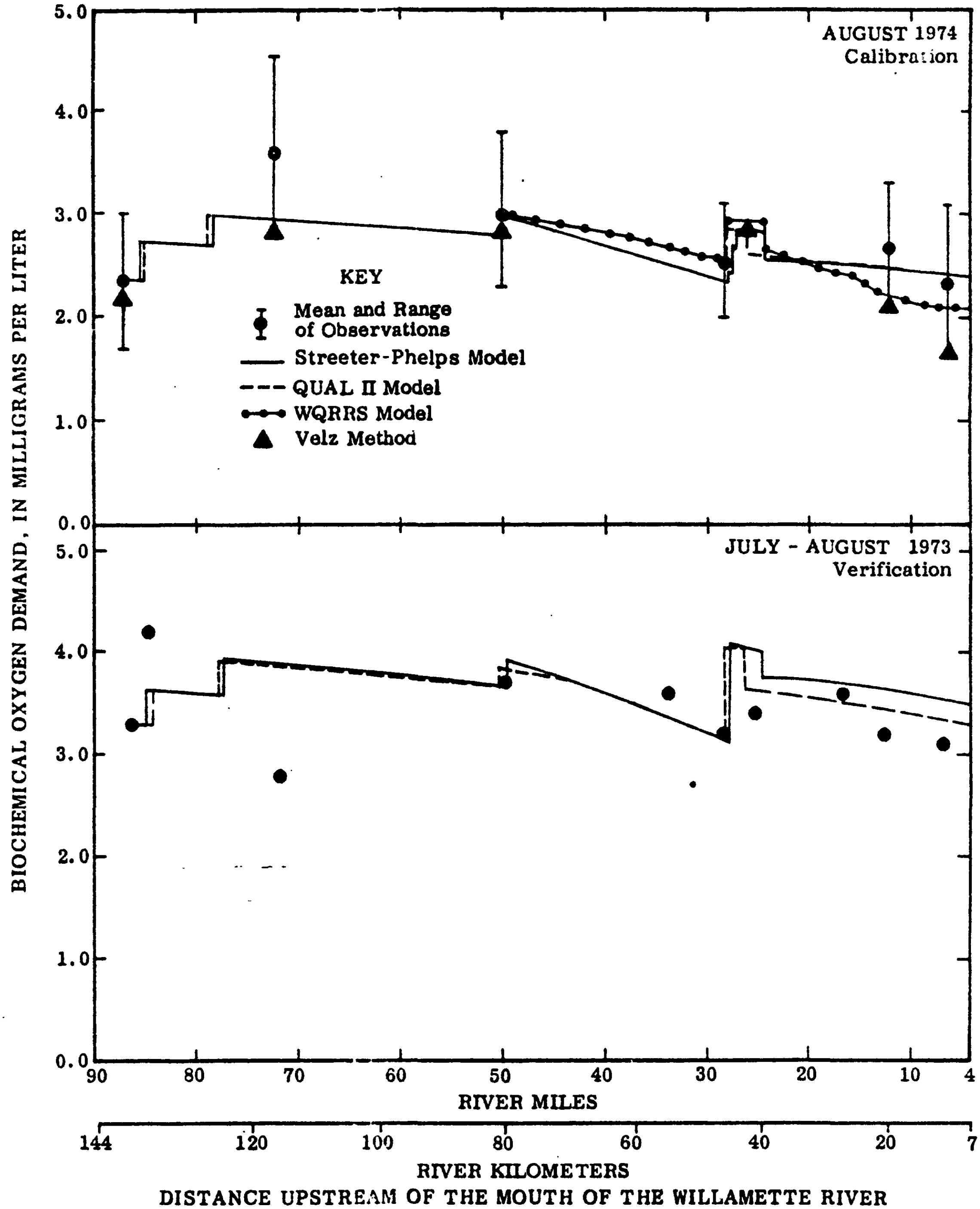


Figure 29. Observed and predicted biochemical oxygen demand, Willamette River

sedimentation rate. However, since the QUAL II model does not apply a temperature correction to the BOD settling rate, the rate of -0.06 per day at 20 degrees Centigrade (68 degrees Fahrenheit) indicated from the Streeter-Phelps model simulation was adjusted to -0.07 per day at 23 degrees Centigrade (73 degrees Fahrenheit) for the QUAL II model simulation. This gave equivalent results for BOD over the entire study reach for the Streeter-Phelps and QUAL II models.

The smaller BOD predictions of the WQRRS and Velz models in the Tidal Reach result because a benthos source was not simulated. A benthos source of BOD is not explicitly included in the WQRRS model. McKenzie and others (1979) neglected this source in modeling the Willamette River with the Velz rational method.

The addition of BOD was simulated as a first-order process because neither the Streeter-Phelps or QUAL II models allow a constant benthos source of BOD. The QUAL II model had a coefficient that was labeled "benthos source rate for BOD." However, that coefficient was actually the benthic or sediment oxygen demand rate. It does not affect BOD predictions.

In summary, the Velz simulation of McKenzie and others (1979) indicated a need for a distributed benthos source of BOD. This distributed benthic source of BOD was simulated with the Streeter-Phelps and QUAL II models as a first-order process by specifying a negative BOD sedimentation rate despite the fact that distributed benthic sources of BOD are usually assumed to release BOD at a constant rate (zero-order process). These data lacked the detail and precision necessary to determine the importance of this deviation from standard practice. In addition, caution should be used when simulating BOD sedimentation or release. Temperature affects on particle settling or BOD release would have to be considered separately when these affects are important.

Finally, when the WQRRS model simulation for the Upstream Reach at 20 degrees Centigrade (68 degrees Fahrenheit) was compared to the other simulations, it showed no difference in BOD decay. This indicated the apparent differences in decay between the WQRRS model and the Streeter-Phelps, QUAL II, and Velz models, in the Newberg Pool and Tidal Reach of

Willamette River and the Chattahoochee River were also due to temperature corrections for decay rates. The WQRRS model allows specification of the temperature correction coefficient θ (defined in: $K_T = K_{20}\ \theta^{(T-20)}$) and prints the factor $\theta^{(T-20)}$ for various temperatures T. However, these results indicate a different technique [possibly a recent update not mentioned in Smith (1978)] may be used to correct decay rates for temperature.

Figure 30 shows results of the nitrogenous BOD calibration for the Streeter-Phelps model using the August 1974 data. The few data available in the Upstream Reach (RK 139-84.5, RM 86.5-52.5) indicated that a decay rate of 1.6 per day was appropriate. Hines and others (1977, pp. I25-I26) measured concentrations of nitrosomonas and nitrobactor bacteria in river water and on rock slimes and used this information in concluding that nitrification was insignificant in the Newberg Pool and Tidal Reach.

Figure 31 illustrates the ammonia calibration for the QUAL II and WQRRS models using the August 1974 data. Like the nitrogenous BOD data, few calibration data were available describing ammonia. Between RK 139 to RK 84.5 (RM 76 to 52.5), the WQRRS model appeared to predict slower ammonia decay compared to the QUAL II model prediction for equivalent loads of ammonia and specified ammonia decay rates. Because of the way the river was discretized, the WQRRS model application includes the simulation of ammonia decay between RK 84.5 to RK 76.7 (RM 52.5 to 47.64) that was not included in the QUAL II model application. For a correct comparison, 0.08 milligrams per liter of ammonia should be added to the WQRRS model predictions between RK 76.7 to RK 6.4 (RM 76.7 to 4) to compensate for this difference.

Neither the WQRRS or QUAL II models report nitrite in a final summary. However, the reported nitrate predictions from the QUAL II and WQRRS models are illustrated in Figure 32 for the calibration using the August 1974 data. The QUAL II model predictions were fair in matching the few nitrate calibration data. The calibration indicates that some nitrate loads were not measured. However, the in-stream measurements were not sufficient for estimating nitrate loads. Again, as in the Chattahoochee River comparison, the WQRRS model did not adequately predict nitrate.

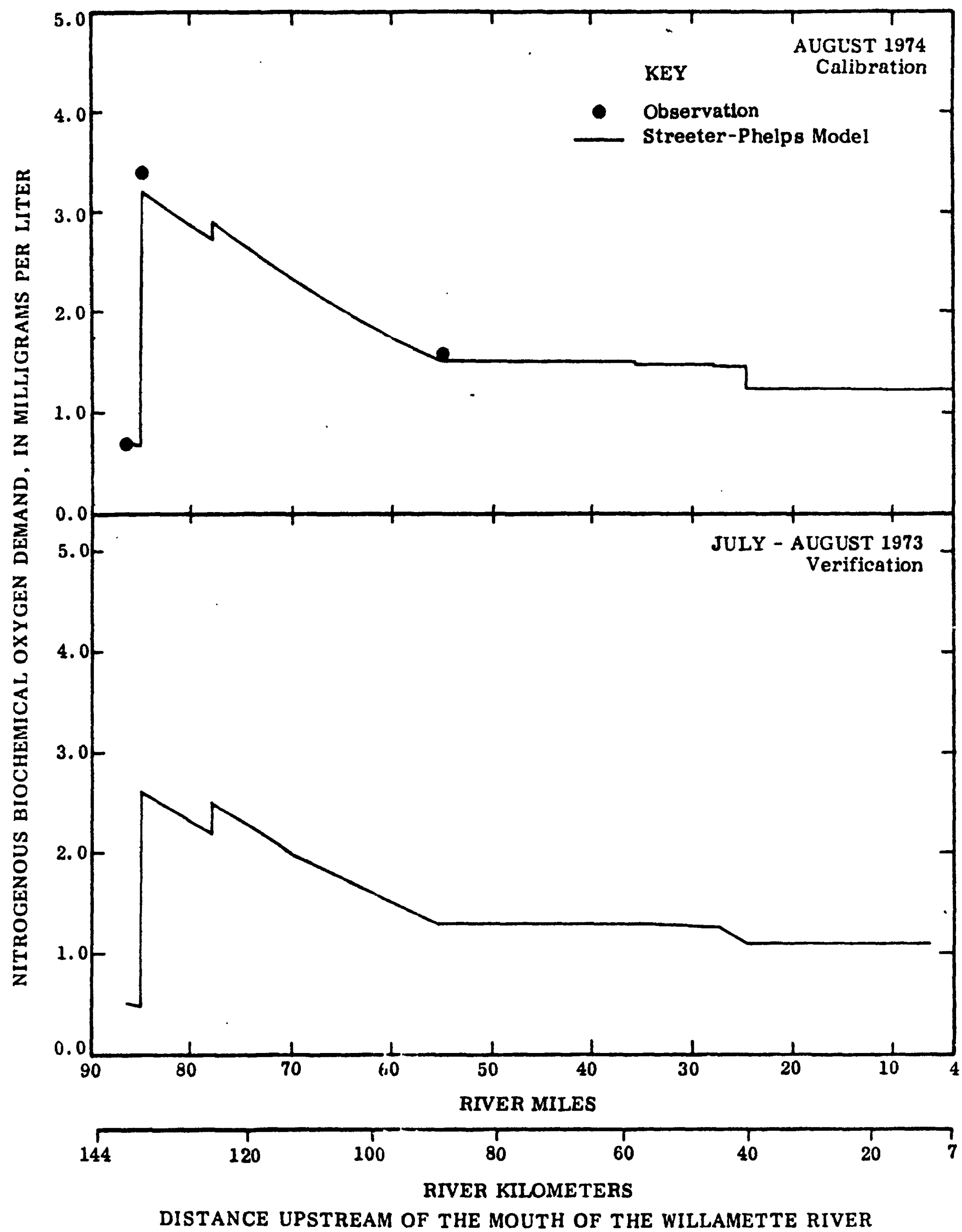


Figure 30. Observed and predicted nitrogenous biochemical oxygen demand, Willamette River

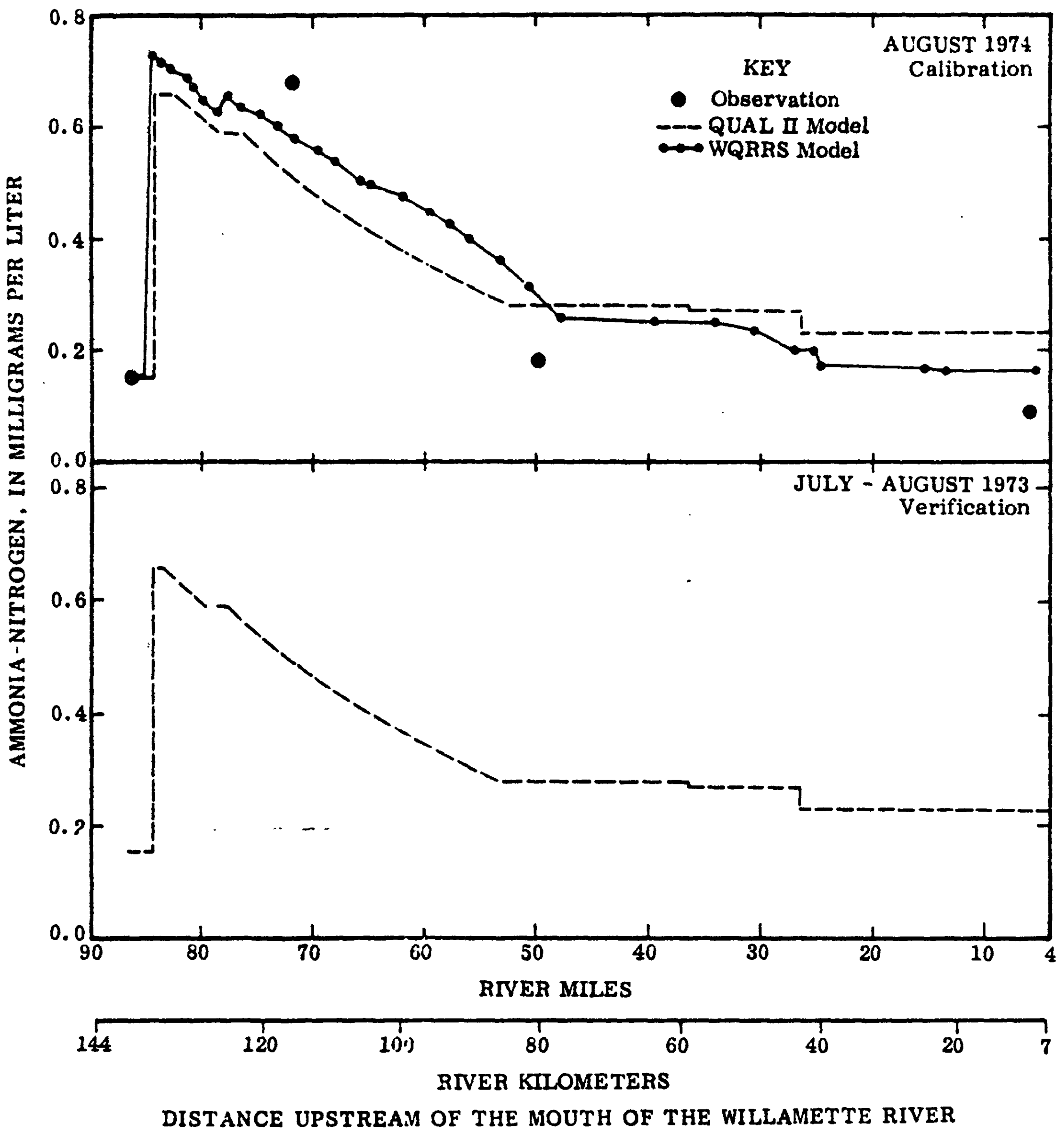


Figure 31. Observed and predicted ammonia-nitrogen, Willamette River

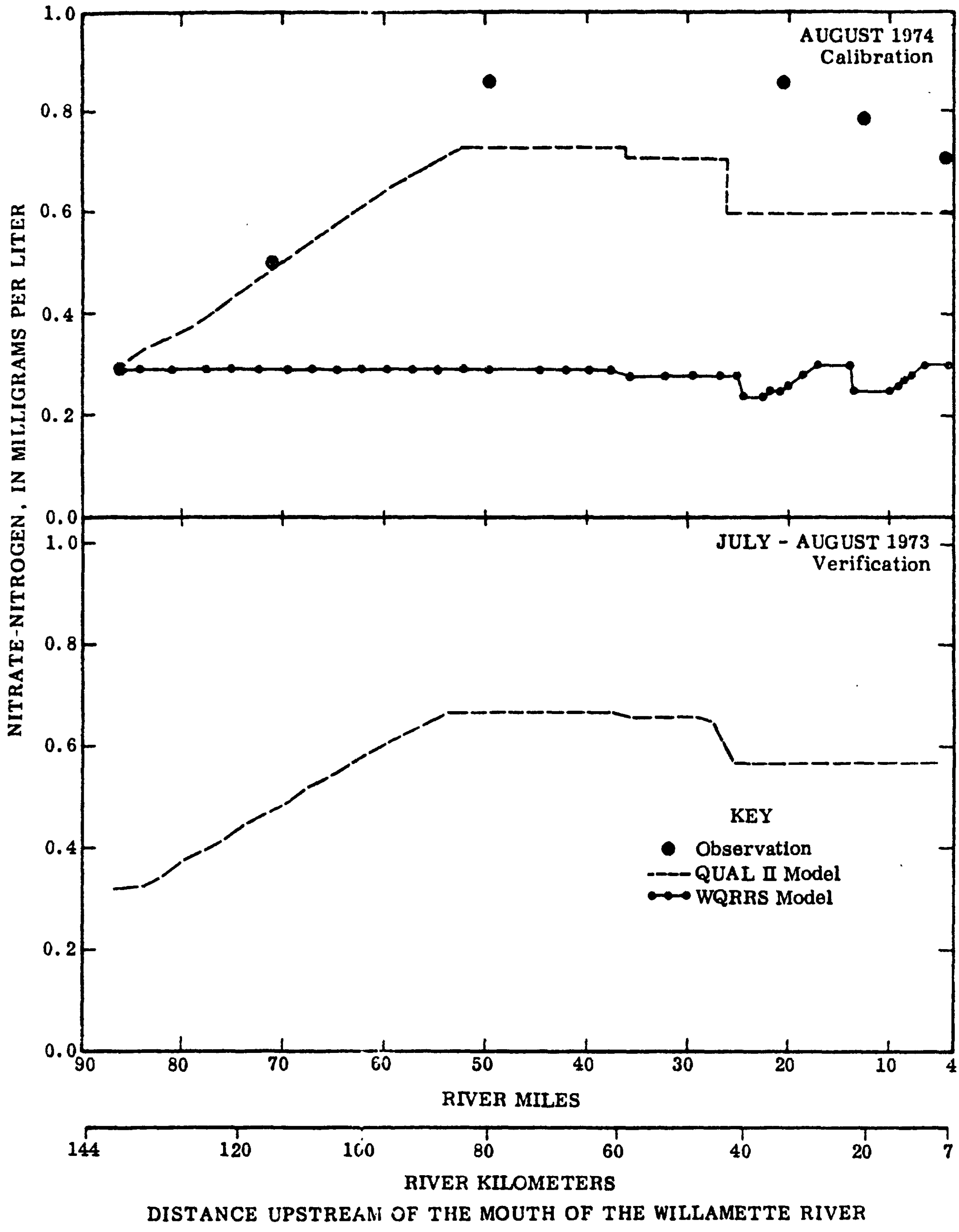


Figure 32. Observed and predicted nitrate-nitrogen, Willamette River

In the final calibration step, DO predictions were compared with measurements from August 1974 as illustrated in Figure 33. The Streeter-Phelps, QUAL II, WQRRS, and Velz models gave approximately the same predictions for the Upstream Reach where the water temperature was near 20 degrees Centigrade (68 degrees Fahrenheit). The Streeter-Phelps, QUAL II, and Velz models gave approximately the same predictions over the entire reach.

The less adequate agreement between the WQRRS model predictions and measurements of DO in the Newberg Pool and Tidal Reach result because benthic oxygen demand was not explicitly included in the model, the stream was not properly discretized, and reaeration at Willamette Falls was neglected. In addition, BOD and ammonia seemed to decay at a slower rate compared to the other models for equivalent waste loads, stream temperature, and specified decay rates. Since questions remain concerning the DO sinks of BOD and ammonia decay, improvement of stream discretization, the implicit simulation of a benthic demand as organic sediment decay, and the simulation of reaeration at Willamette Falls were deferred.

Verification

Verification data collected in July and August 1973 were limited to mean values of BOD and maximum, minimum, and average DO. The Streeter-Phelps and QUAL II model simulations for the July-August 1973 verification data were plotted with calibration simulations in Figures 29 to 33. Nitrogenous BOD, ammonia, and nitrate predictions were plotted despite the lack of data for comparison to show that the range of predictions were within the range of application established by calibration.

The BOD data used for verification indicated that the Streeter-Phelps and QUAL II models tended to overpredict mean BOD in the Tidal Reach by approximately 0.5 milligrams per liter. The DO data used for verification indicated that the QUAL II model can predict DO to $\pm$ 0.6 milligrams per liter and the Streeter-Phelps model to $\pm$ 0.5 milligrams per liter of DO. While confirmation was not possible, these results for DO indicated that the nitrogenous BOD, ammonia, and nitrate predictions were adequate.

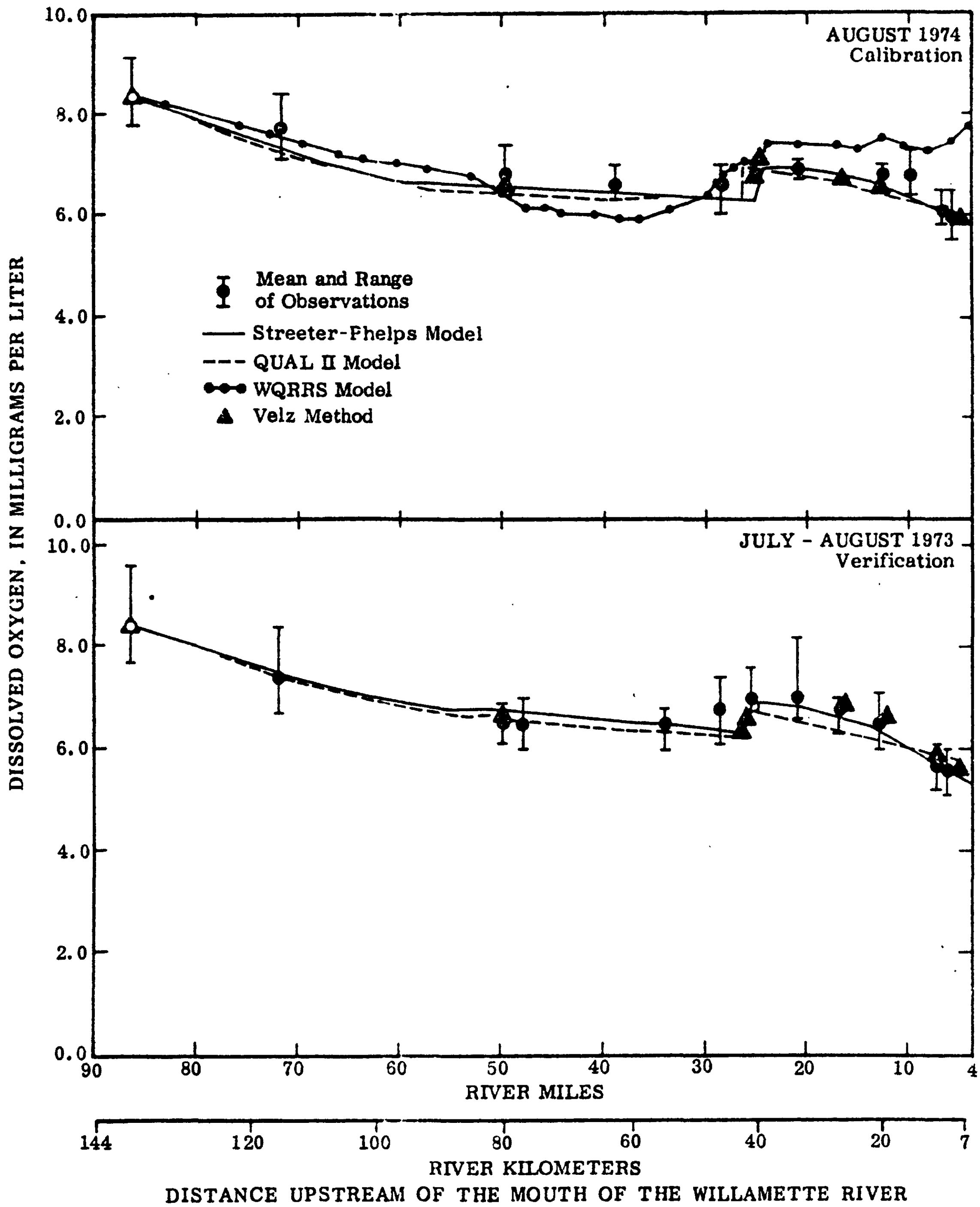


Figure 33. Observed and predicted dissolved oxygen, Willamette River

Comparison

This large river with 20 tributaries proved difficult to discretize using the WQRRS model. The crucial limitations were 10 inflows, 47 points at which cross-section coordinates were specified, and water-quality coefficients could not be modified for different reaches. The capability to store and access results on magnetic tape made simulation of five separate segments easier but data coding and calibration in five model applications prove time consuming and tedious.

The QUAL II model proved to have the most flexible discretization scheme. Discretization errors proved to be minor and the stream was modeled with five reaches in a single application of the model. The Streeter-Phelps model simulated the stream with a single application but required greater effort to code data for 23 reaches.

Minor errors were noted with the direct temperature specification option and equilibrium temperature options of the WQRRS model. However, the heat balance option seems to be quite accurate.

In summary, neither the Streeter-Phelps, QUAL II, or WQRRS models will simulate a constant benthic source of BOD. The Streeter-Phelps and QUAL II models do not simulate reaeration due to a lock and dam, fish ladder, or waterfall. Finally, these results indicate that an apparent difference in decay of BOD and ammonia between the WQRRS model and the other models may be due to temperature corrections of the decay rates in addition to the differences due to dispersion calculations.

PART VII: MODEL APPLICATIONS WITH ARKANSAS RIVER DATA

Model Preparation

Application

The Arkansas River data were transformed to fit requirements of the Streeter-Phelps, QUAL II, and WQRRS models. The September 1979 data were used for model calibration and the April 1976 data served to verify the results from the steady-state models. Organic nitrogen, ammonia, nitrite, nitrate, DO, and BOD were modeled. Temperature was simulated using the WQRRS model. The QUAL II model did not include organic nitrogen. In addition, the dynamic simulation option of the QUAL II model was tested by predicting diurnal variations of temperature, DO, BOD, and nutrients for the September 1979 data.

Cain, Baldridge, and Edelmann (1980) had converted the original data to fit the format of the Streeter-Phelps model. The data were further transformed to fit the formats of the QUAL II and WQRRS models. To avoid some bias in favor of Streeter-Phelps model, the September 1979 data were used to check the calibration and the April 1976 data were used for verification. This was the reverse of the procedure used by Cain, Baldridge, and Edelmann (1980).

Stream discretization and hydraulics

The Arkansas River data proved to be a strict test for the discretization schemes of the three models. The 68-kilometer (42-mile) reach has numerous inflows, some of which entered the river near another inflow, making it difficult to discretize the stream so that the individual effects of each inflow were retained. In addition, cross-section properties were measured at 61 sites and 21 inflows exceeded the limitations of the WQRRS model, which made three separate applications necessary.

In applying the Streeter-Phelps model, Cain, Baldridge, and Edelmann (1980) defined 27 reaches for the September 1979 calibration data and 25 for the April 1976 verification data. The September 1979 data described 4 withdrawals and 15 tributary inflows. The April 1976 data described 5 withdrawals and 15 tributaries. Seven extra reaches were defined for the September 1979 application and 4 for the April 1976 application in a test of management alternatives by Cain, Baldridge, and Edelmann (1980).

Reaches for the QUAL II model application were defined, first, to meet the limitations of the model and, second, to correspond to reaches used by the Streeter-Phelps model so that results would be comparable. Based on these two criteria, 25 reaches were defined for the April 1976 and September 1979 data. This included combining four reaches defined for the Streeter-Phelps model into two reaches for the QUAL II model and subdividing two other reaches into four. Inflows from two drains were combined into a single inflow near RK 60 (RM 37), and the Salt Creek inflow and the Pueblo STP effluent near RK 52 (RM 32) were combined. Two long reaches at the end of the study segment were subdivided into four reaches to fit the QUAL II model requirements of 20 elements per reach. As a result, some reach endpoints and inflows were shifted by as much as 0.3 kilometers (0.2 miles) to conform to the QUAL II model discretization limitation of a constant element length.

The discretization scheme for the WQRRS model adequately described the stream without serious discretization error but required greater data coding effort to achieve this. Whereas, the 68-kilometer (42-mile) reach could be modeled with one application of the Streeter-Phelps and QUAL II models, three applications were required to simulate the river with the WQRRS model. This was necessary because tributaries and withdrawals exceeded the limit of ten and cross-sectional properties were measured at more than 41 sites. Based on this, a reach from RK 67.6 to 55.9 (RM 42 to 34.7) was defined for the 10 inflows and withdrawals farthest upstream in the study segment. The downstream segment was divided at RK 19.3 (RM 12) so that 41 points, having measured cross-sectional properties, were contained in the downstream reach.

Overall, the QUAL II model required greater data-coding effort to match stream geometry than did the Streeter-Phelps model, which was relatively easy to apply. The WQRRS model required the greatest effort to simulate stream geometry.

Computational element lengths chosen for the Streeter-Phelps, QUAL II, and WQRRS models were not well matched for this application. The smallest length acceptable for the QUAL II model was approximately 0.40 kilometer (0.25 mile). Cain, Baldridge, and Edelmann (1980) used an interval of 0.2 kilometer (0.1 mile) for the Streeter-Phelps model. To fit stream geometry, the WQRRS model needed interval lengths of 1.2, 1.1, and 1.1 kilometers (0.73, 0.70, 0.69 miles) for reaches RK 67.6 to 55.9, 55.9 to 19.3, and 19.3 to 0 (RM 42.0 to 34.7, 34.7 to 12.0, and 12.0 to 0) measured upstream of the Nepesta, Colorado, stream gage.

Travel time in the stream was matched as close as possible in all three model simulations for comparison of results. For the Streeter-Phelps model, travel times (Cain, Baldridge, and Edelmann, 1980) were specified as input data from dye measurements. For the QUAL II model, the coefficients a and b of $\overline{u} = aQ^b$ were calculated for each reach from the April 1976 and September 1979 data. This gave two pairs of velocity $\overline{u}$ and discharge Q (one for September 1979 and one for April 1976) to calculate the two coefficients in each reach.

It was more difficult to model travel time with the WQRRS model because cross-section data were incomplete. The steady-state backwater routing option is normally calibrated with water-surface profiles, which were missing in this case. Therefore, the Manning coefficient at each cross section was varied until predicted travel time matched measured travel time.

Water-quality coefficients

Temperature was simulated with the WQRRS model by estimating meteorological conditions. Default wind-speed coefficients were used.

Reaeration coefficients were specified directly for all three models. These coefficients were taken from Cain, Baldridge, and Edelmann (1980), who derived their reach by reach values from direct measurements using hydrocarbon gas tracers.

Decay rates for BOD, organic nitrogen, ammonia, nitrite, and nitrate were estimated from Cain, Baldridge, and Edelmann (1980). They assumed that photosynthesis and nutrient cycling did not affect average daily concentrations of DO. These rates were deduced by a trial-and-error calibration of the Streeter-Phelps model.

Missing and inconsistent data

Several types of data were either missing or inconsistent. First, 5-day BOD (with nitrification inhibited) was measured rather than ultimate BOD. Ultimate BOD is required by the Streeter-Phelps and QUAL II models for proper simulation of DO (the QUAL II model has a 5-day BOD option, but it incorrectly converts BOD decay to oxygen demand unless the BOD decay rate is 0.23 per day). Five-day BOD is required by the WQRRS model. The conversion factor relating 5-day BOD to ultimate BOD was not available because ultimate BOD was not measured. However, preliminary modeling results indicated the in-stream deoxygenation rate was 1.5 per day (base e, 20 degrees Centigrade or 68 degrees Fahrenheit). Therefore, at such a high decay rate, 5-day BOD was a very good approximation of ultimate BOD ($BOD_5 = BOD_{ULT} \{1 - e^{-(1.5)5}\} = BOD_{ULT} \{0.999\}$). This also set up a good comparison between BOD predictions made by the WQRRS model and the other models.

Second, BOD, organic nitrogen, ammonia, nitrite, and nitrate were sampled as total constituents in the water column. Particulate matter, which may include bacteria and phytoplankton, tends to interfere with sample preservation of ammonia, nitrite, and nitrate. Since phytoplankton may be present, the analysis for total organic nitrogen will include nitrogen bound in active biomass that is not immediately available for decay to ammonia. In addition, the BOD samples will include the effects of phytoplankton respiration and detritus decay. The effect of this sampling technique could not be accurately determined. Although Kjeldahl nitrogen (organic nitrogen plus ammonia) was smaller than ammonia in a few cases, sample processing times of 24 to 48 hours should have minimized sample preservation problems. Effects of phytoplankton on organic nitrogen and BOD depend on phytoplankton concentrations.

Third, cross-sectional measurements were referred to the September 17-21, 1979 water surface rather than a common vertical datum such as mean sea level. Therefore, the elevation of the river channel was estimated from topographic maps and it was assumed that the September 17-21, 1979 water-surface elevation was at the cross-sectional average depth above the estimated channel bottom. This gave adequate estimates of the vertical relationship between cross sections because the study reach has a steep slope that prevented significant backwater effects.

Finally, grab samples taken on the Arkansas River at RK 59.9 and 53.9 (RM 37.2 and 33.5) were not representative of the cross-sectional average concentration. A mass balance using specific conductivity as a measure of dissolved solids showed that these samples were taken from plumes originating from upstream drains and that the inflows were not well mixed at these cross sections. These questionable data were labeled as such in the following results section.

Model Results

Calibration

Calibration results from the three models were obtained in the following way. The Streeter-Phelps, QUAL II, and WORRS models, in that order, were calibrated using the September 1979 data. Travel time was specified as input data for the Streeter-Phelps model, and was simulated using the QUAL II and WQRRS models. Temperature was specified for the Streeter-Phelps and QUAL II models and simulated using the WQRRS model. Following that, the models were calibrated to predict BOD. Then, organic nitrogen was simulated using the Streeter-Phelps model. Finally, all three models were calibrated to predict ammonia, nitrite, nitrate, and DO.

In applying the Streeter-Phelps model, travel times of 48 and 43 hours were specified for September 1979 and April 1976, respectively. The QUAL II model simulated travel times of 46 and 41 hours for September 1979 and April 1976, respectively. The WQRRS model simulated a travel time of 47 hours for the September 1979 data.

Mean temperatures, derived from the observations shown in Figure 34, were specified for the Streeter-Phelps and QUAL II models. Temperature was simulated with the WQRRS model as shown in Figure 34 because the option to specify temperature as input data was not functioning at

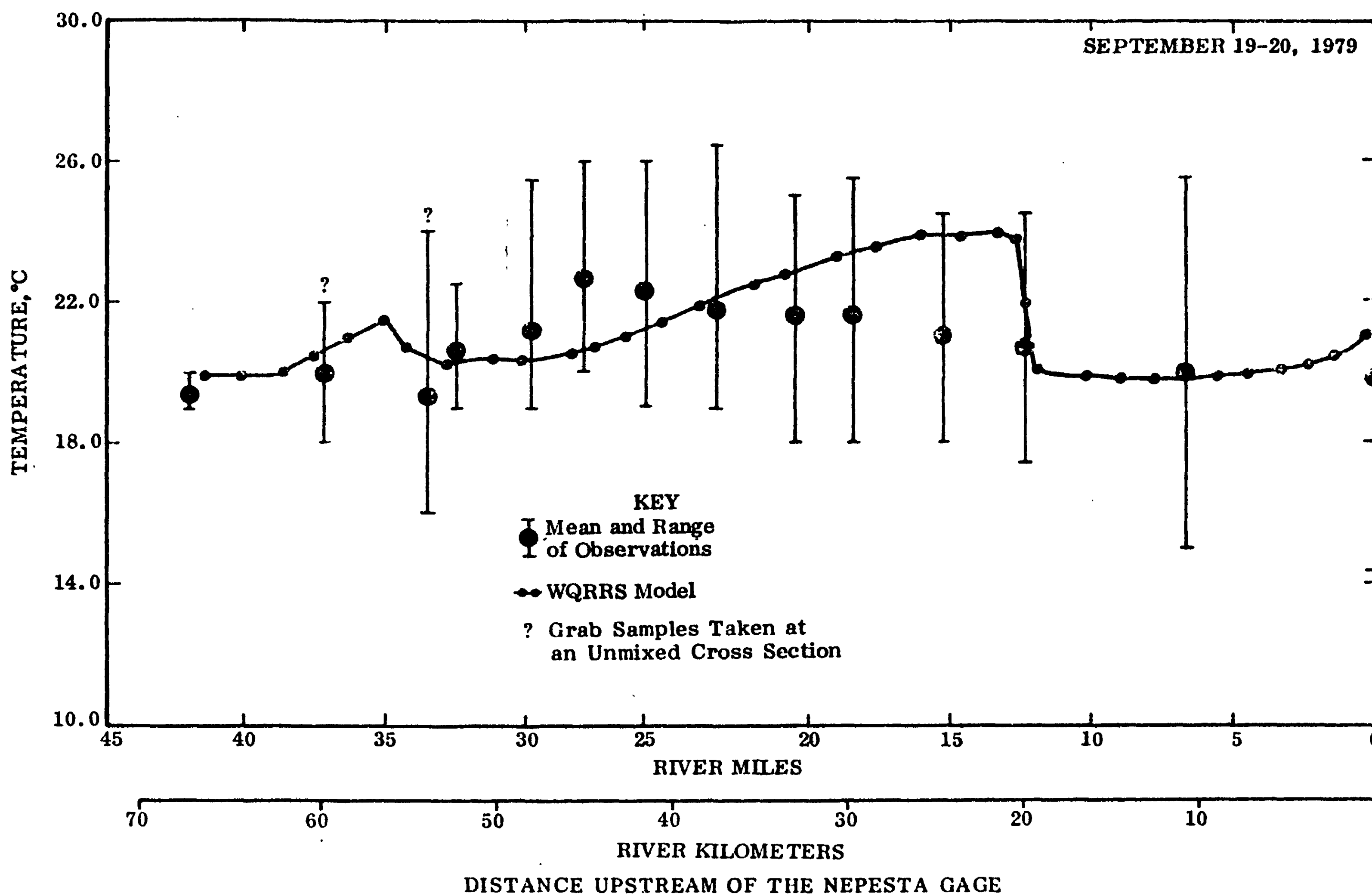


Figure 34. Water temperatures predicted by the WQRRS model compared to observed temperatures for the Arkansas River

the time of this study. Based on default wind-speed coefficients and estimated meteorological data, the WQRRS model was capable of predicting mean stream temperatures to within 3 degrees Centigrade (5 degrees Fahrenheit) of the mean of observations.

Figure 35 compared BOD predictions from the Streeter-Phelps, QUAL II, and WQRRS models to the September 1979 calibration data. These results were based on a deoxygenation rate of 1.5 per day. The results from the Streeter-Phelps and QUAL II models were essentially the same with the exception of one short segment between the Pueblo STP effluent (RK 50.4 or RM 31.3) and Salt Creek (RK 50.2 or RM 31.2). Here the Streeter-Phelps model predicted a peak concentration of 12.9 milligrams per liter, whereas the QUAL II model predicted a peak concentration of 6.4 milligrams per liter, because the QUAL II model treated the sewage effluent and Salt Creek as a single inflow diluting the effects of the Pueblo STP effluent before it reached the river.

The WQRRS model predicted a peak BOD concentration of 12.3 milligrams per liter and shifted that peak upstream of the wastewater treatment plant outfall. In addition, BOD decay occurred at a slower rate compared to the other two models.

The deoxygenation rate of 1.5 per day, chosen by Cain, Baldridge, and Edelmann (1980), was high but not unreasonable. Zison and others (1978, pp. 171, 176, and 179-180) showed that shallow, steep mountain streams typically have deoxygenation rates of 0.1 to 3.4 per day, base e at 20 degrees Centigrade (68 degrees Fahrenheit). In addition, Velz (1970, p. 183) indicated that stream deoxygenation rates vary from 0.46 to 2.3 per day (base e) or more.

Figure 36 shows the calibration for organic nitrogen using the Streeter-Phelps model with a decay rate of 0.2 per day at 20 degrees Centigrade (68 degrees Fahrenheit) chosen by Cain, Baldridge, and Edelmann (1980). Discounting questionable measurements at RK 59.9 and 53.9 (RM 37.2 and 33.5), the Streeter-Phelps model tended to overpredict organic nitrogen; and from RK 52.3 to 25.0 (RM 32.5 to 15.5), the model predicted a slight decrease in organic nitrogen concentration, whereas the data indicated an increase of 0.4 milligrams per liter.

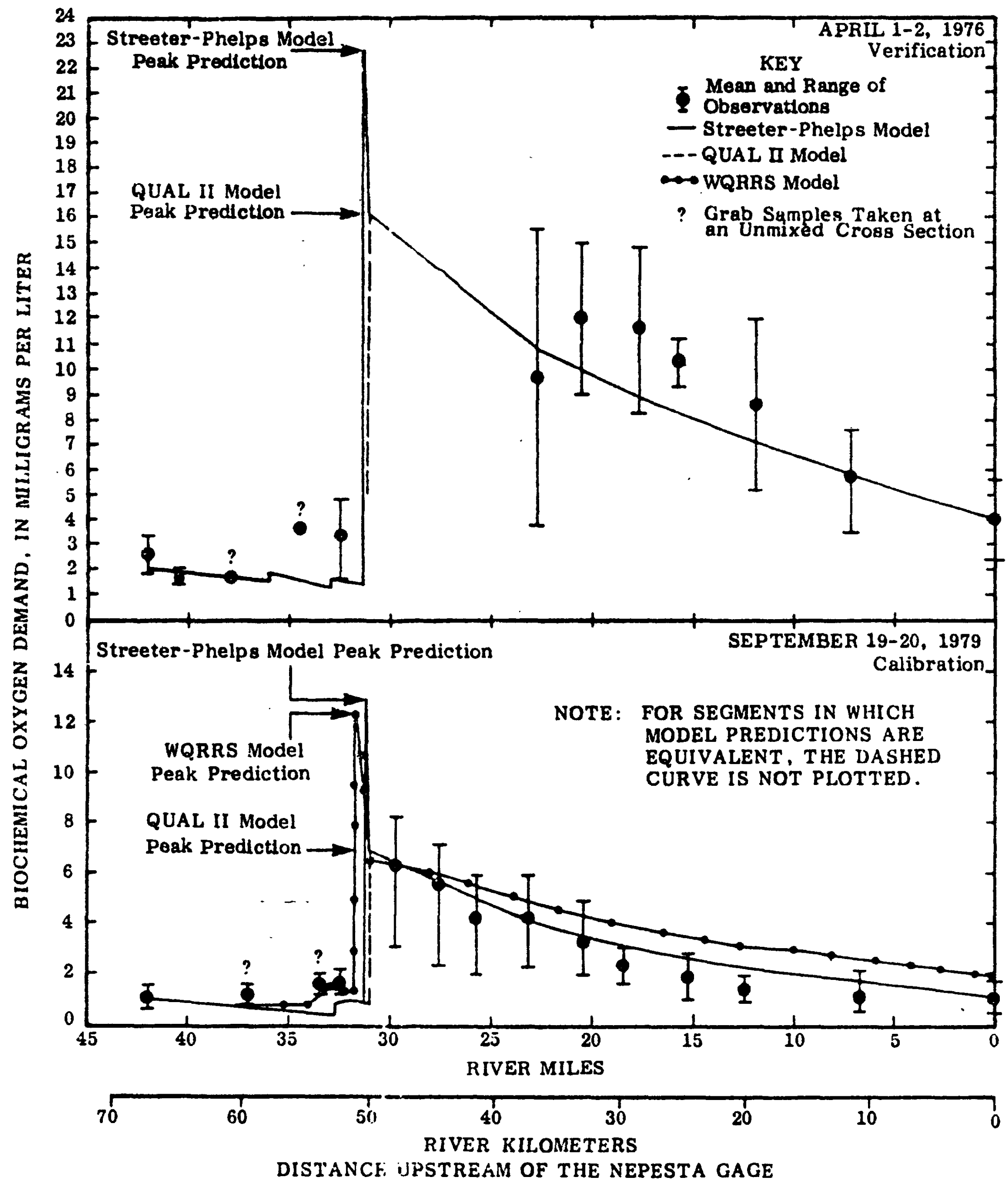


Figure 35. Observed and predicted biochemical oxygen demand, Arkansas River

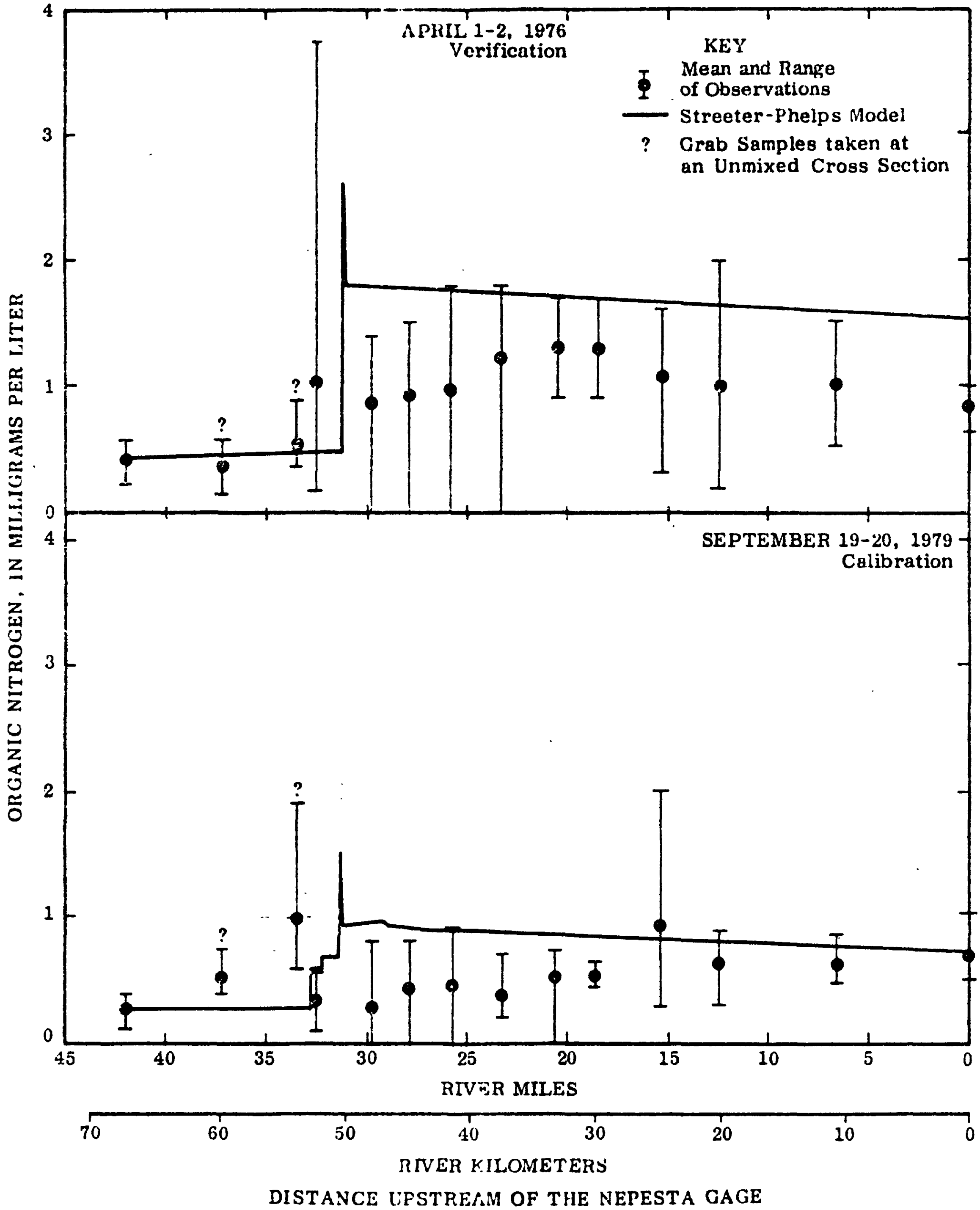


Figure 36. Observed and predicted organic nitrogen, Arkansas River

This disagreement is most likely due to two problems with sample analysis. Cain, Baldridge, and Edelmann (1980) indicated that some Kjeldahl nitrogen (organic nitrogen plus ammonia) determinations were open to question. Ammonia nitrogen exceeded Kjeldahl nitrogen in several samples. More likely, the trend of increasing total organic nitrogen from RK 52.3 to 25.0 (RM 32.5 to 15.5) may have been due to an increase in phytoplankton.

Based on these results, the capability of the Streeter-Phelps model to predict total organic nitrogen in this river segment was questionable. Furthermore, the calibration was insufficient.

Ammonia measurements and predictions for the September 1979 calibration data were compared in Figure 37 for an ammonia decay rate of 2.5 per day at 20 degrees Centigrade (68 degrees Fahrenheit). The results of the calibration showed that predictions from the Streeter-Phelps and QUAL II models were equivalent despite the transformation of organic nitrogen to ammonia simulated in the Streeter-Phelps model. Like the BOD simulation, the same discretization differences occurred between the Pueblo STP and Salt Creek. Decay also occurred at a slower rate in the WQRRS model simulation. Unlike the organic nitrogen simulation, there was good agreement between the measured calibration data for ammonia and all three model predictions. The Streeter-Phelps and QUAL II model simulations were in excellent agreement with the data. The WQRRS model simulation could be improved to better match the data if the decay rate was increased.

The calibration for ammonia indicated that the decomposition of organic nitrogen simulated by the Streeter-Phelps model, but not the QUAL II model, did not have a large effect on ammonia predictions. In this case, the decay rate for organic nitrogen was much smaller than the ammonia decay rate [K(Org.N) = 0.2 per day and $K(NH_3)$ = 2.5 per day at 20 degrees Centigrade or 68 degrees Fahrenheit]. The decay of ammonia was so rapid that the slower decay of organic nitrogen had little effect. This indicated that the failure to calibrate the Streeter-Phelps model to predict organic nitrogen was of lessened importance.

Figure 38 illustrates results from the Streeter-Phelps and QUAL II

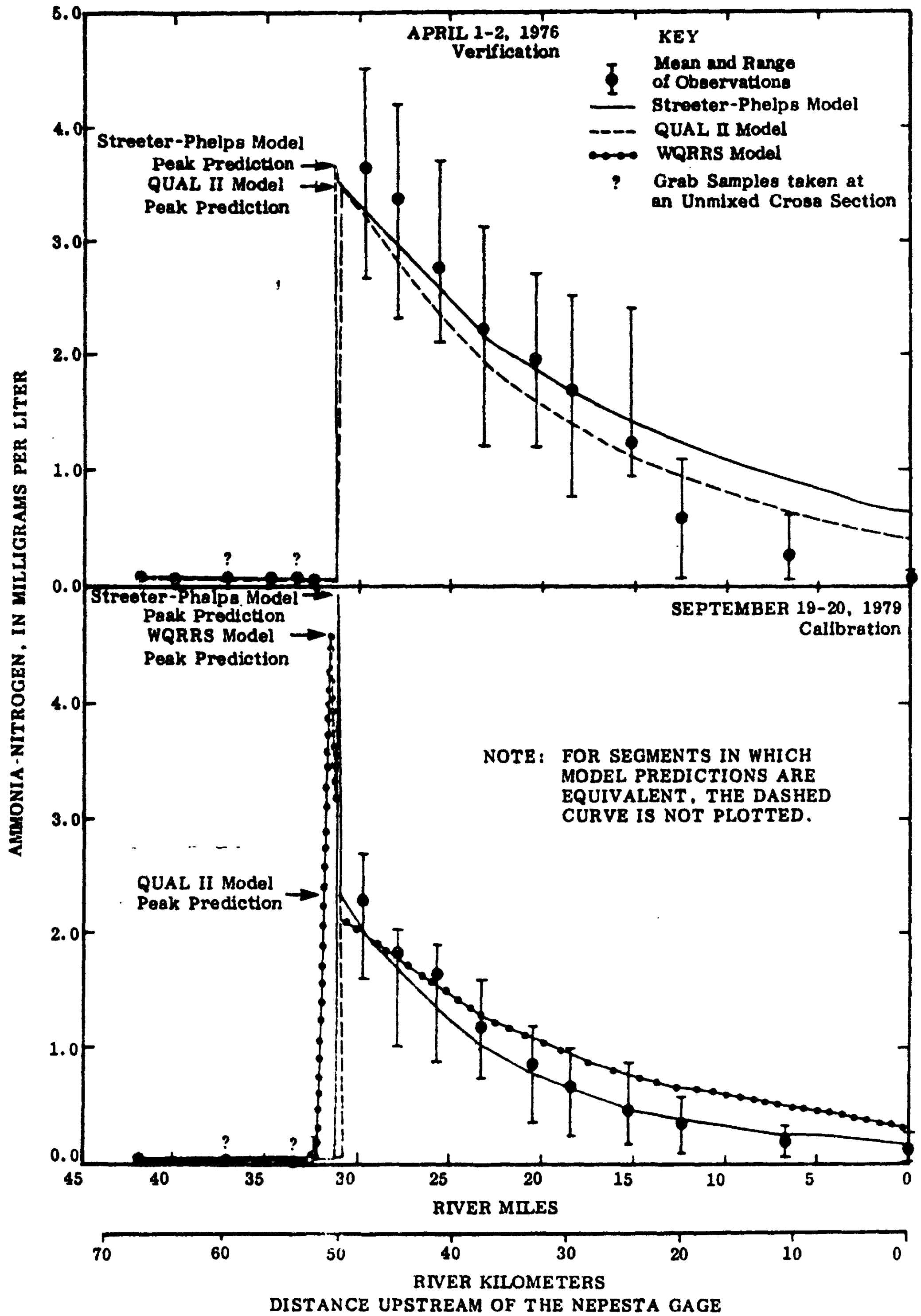


Figure 37. Observed and predicted ammonia-nitrogen, Arkansas River

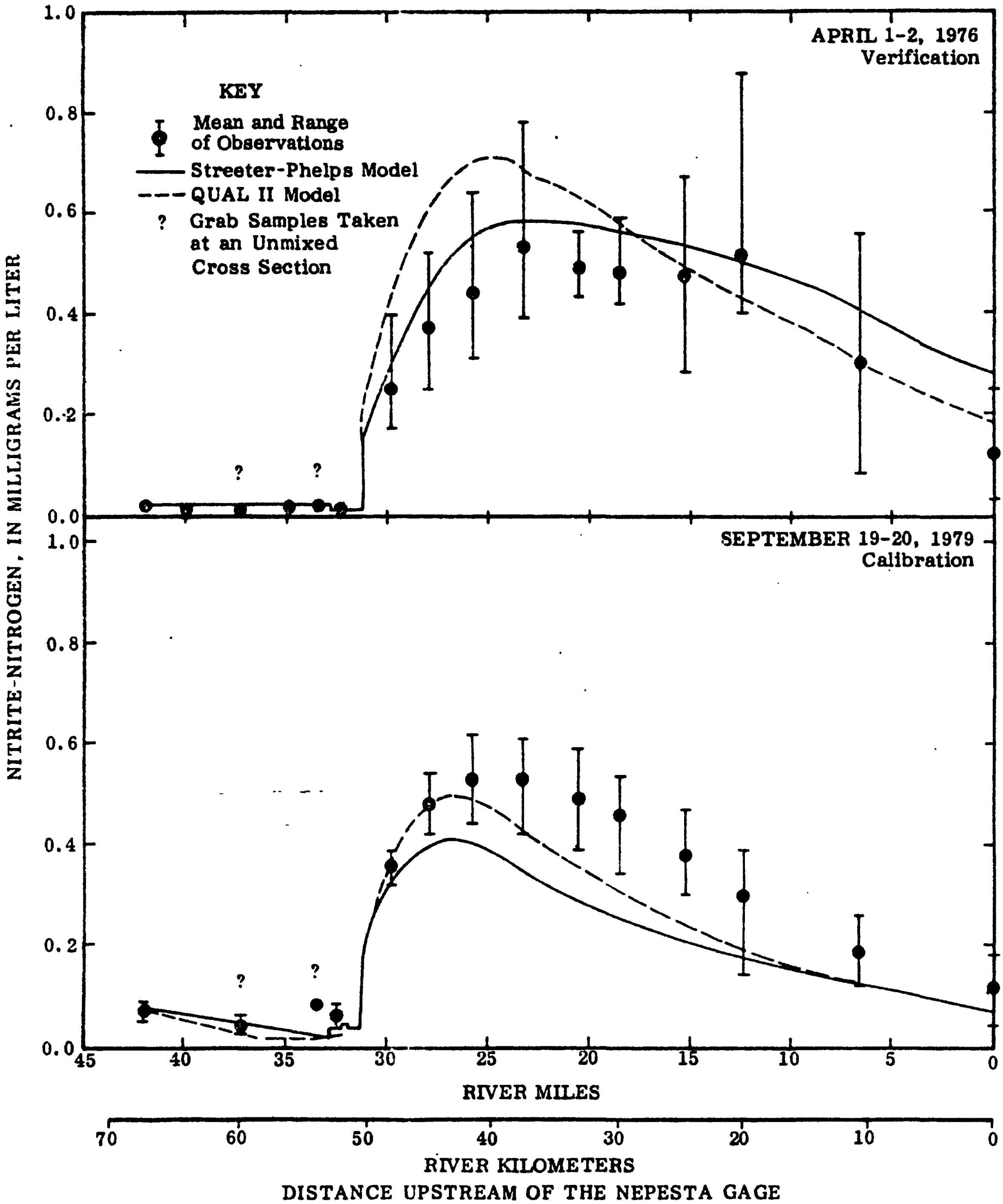


Figure 38. Observed and predicted nitrite-nitrogen, Arkansas River

models for the nitrite calibration using a decay rate of 7.5 per day at 20 degrees Centigrade (68 degrees Fahrenheit). The removal of ammonia by aquatic plants or desorption of ammonia gas was simulated with the Streeter-Phelps model by specifying a forward reaction rate of 2.0 per day at 20 degrees Centigrade (68 degrees Fahrenheit) compared to an ammonia decay rate of 2.5 per day. This difference was responsible for the lower nitrite predictions from the Streeter-Phelps model compared to the QUAL II model predictions for the September 1979 data.

Both models tend to underpredict nitrite. However, the greatest difference between the mean nitrite observation and prediction was 0.2 milligrams per liter for the Streeter-Phelps model and 0.15 milligrams per liter for the QUAL II model.

Figure 39 shows the calibration of the Streeter-Phelps, QUAL II, and WQRRS models to predict nitrate using the September 1979 data. Since the chlorophyll a content of phytoplankton was not simulated with the QUAL II model, predicted nitrate built up in the stream. The WQRRS model simulation, again, showed that the algorithm that transforms nitrite to nitrate was not functioning. The Streeter-Phelps model simulation matched the calibration data. However, this simulation was based on the removal of ammonia and nitrate by first-order processes having decay rates of 0.5 per day for ammonia removal and 1.7 to 0.4 per day at 20 degrees Centigrade (68 degrees Fahrenheit) for nitrate removal. Over the entire reach, the removal of 1.8 milligrams per liter of nitrogen was simulated with Streeter-Phelps model that was not simulated using the QUAL II and WQRRS models. In addition, this Streeter-Phelps simulation was based on the assumption that plant photosynthesis did not effect mean levels of DO.

Figure 40 illustrates the results of model calibration to predict DO for the September 1979 data. All three models gave approximately the same predictions and all three slightly overpredicted DO in the downstream two-thirds of the study reach.

The three models gave equivalent DO predictions despite several differences. The Streeter-Phelps model calibration included ammonia removal by aquatic plants. This simulated process removed a part of the

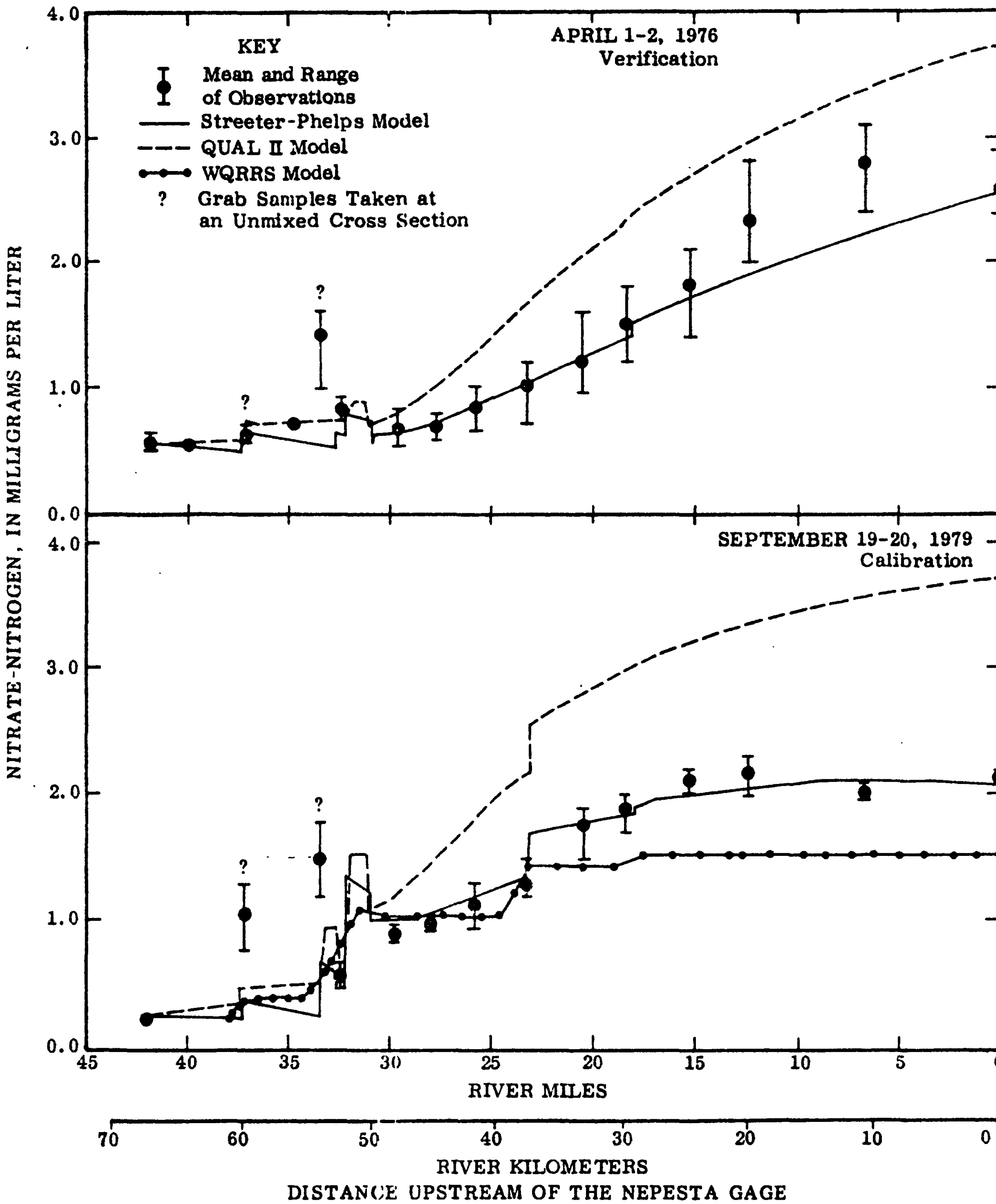


Figure 39. Observed and predicted nitrate-nitrogen, Arkansas River

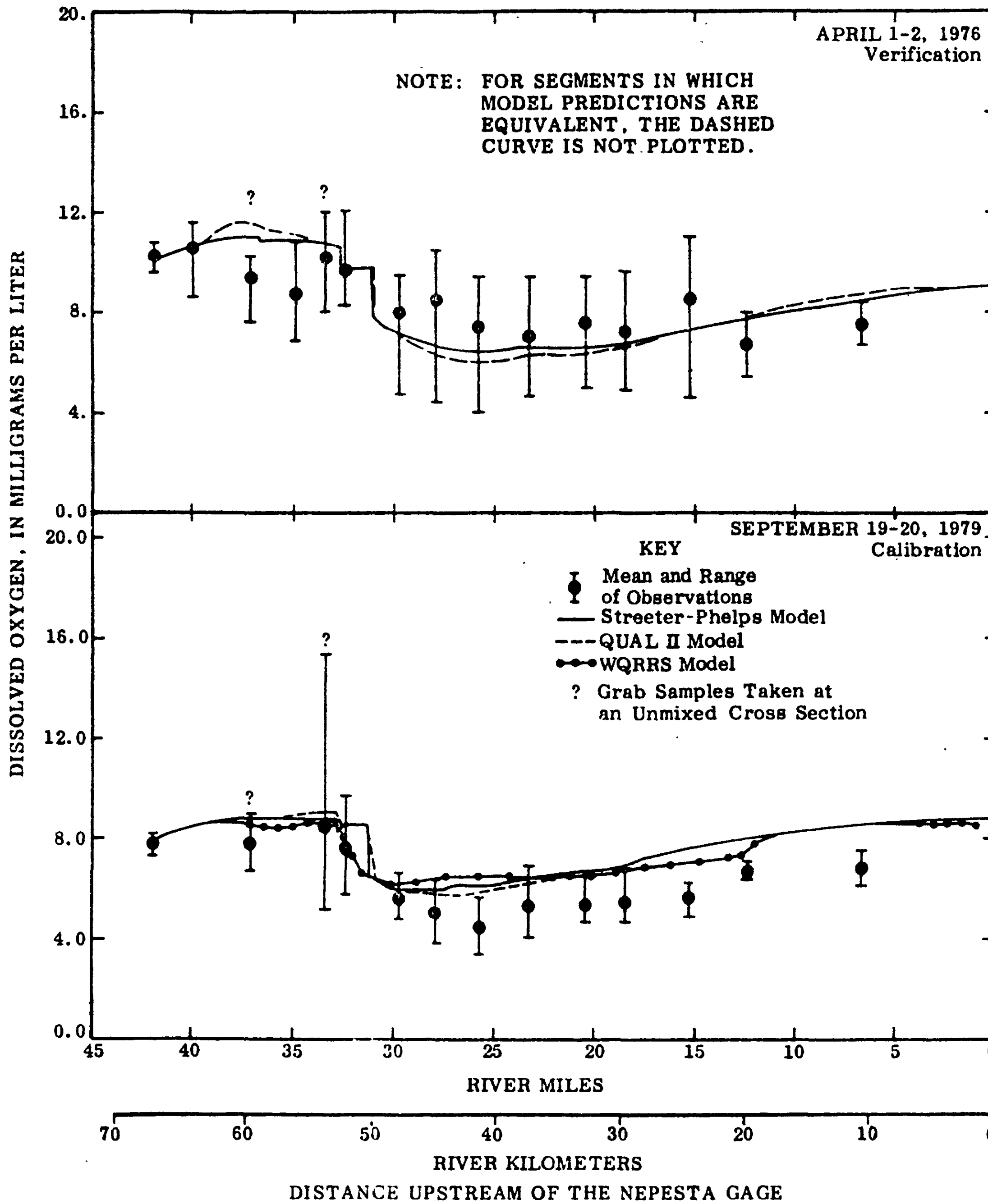


Figure 40. Observed and predicted dissolved oxygen, Arkansas River

nitrogenous oxygen demand that was included in the QUAL II and WQRRS models. In addition, the QUAL II model applied a smaller temperature correction to the reaeration coefficient so that simulated reaeration occurred at a slower rate than reaeration simulated with the Streeter-Phelps and WQRRS models. In part, this difference was compensated for because the QUAL II model did not correct computed saturation values of DO for the difference in atmospheric pressure at sea level and local atmospheric pressure. That correction should have been 633 millimeters of mercury/760 millimeters of mercury (25 inches of mercury/29.92 inches of mercury) or 0.84 of the saturation value computed for mean sea level (or NGVD of 1929). This difference was noticeable near RK 60 (RM 37) at which river DO approached saturation. The higher computed DO saturation caused the QUAL II model to predict DO higher by 0.6 milligrams per liter for the April 1976 data at RK 60 (RM 37). Finally, the WQRRS model simulated lesser amounts of BOD and ammonia decay, compared to the other two models. Either these differences were minor or they were compensating so that the DO predictions of all three models were approximately the same.

In general, this calibration was reasonable for all three models. The WQRRS model gave reasonable temperature predictions. The WQRRS model tended to overpredict BOD and ammonia, but the Streeter-Phelps and QUAL II models gave excellent predictions of BOD and ammonia. Greater decay rates would have improved the WQRRS model simulation for BOD and ammonia. The Streeter-Phelps model did not correctly predict organic nitrogen. However, the ammonia calibration indicated organic nitrogen simulation was unimportant. The Streeter-Phelps and QUAL II models underpredicted nitrite by a small amount. Nitrite results from the WQRRS model were not available. The Streeter-Phelps simulation of nitrate agreed closely with measurements but was based on the assumption that photosynthesis did not affect mean DO. Since algae were not modeled, the QUAL II model was not calibrated to predict nitrate. A program error prevented the WQRRS model from adequately simulating nitrate. The DO calibration was reasonable, with all three models tending to overpredict DO.

Because DO and nitrate were overpredicted using the QUAL II model, several attempts were made to improve the calibration. First, reaeration coefficients were examined. Because reaeration coefficients were calculated from the Padden and Gloyna equation that best fit gas and dye tracer measurements made in October 1976 at higher flows, these coefficients were not modified. The Padden and Gloyna equation gave higher K_2 values as discharge decreased.

Next, the possibility of benthal oxygen demand was considered but discounted as a significant DO sink. Field crews measured the channel shape at 67 points between RK 52.3 and 0 (RM 32.5 to 0) and qualitatively described channel materials, vegetation, and animal life. Between RK 52.3 and 24.2 (RM 32.5 to 15), bottom materials consisted of about 90 percent sand and gravel and 10 percent silt. Sludge banks or deposits of organic material were not noted. Aquatic vegetation was described as "light moss" in this segment. Vegetation was described as "heavy moss" at a few points, and vegetation was not observed at a few other points. Between RK 24.2 to 0 (RM 15 to 0), attached vegetation was rare and the channel bed material ranged from 10 to 50 percent silt.

Finally, the simulation of algae, using the dynamic simulation option of QUAL II, was attempted for the September 1979 data. This attempt provided additional information about this river but did not significantly improve the calibration.

While the QUAL II model was formulated to simulate floating rather than attached algae, it seemed possible that the model was flexible enough to predict the affect on nitrate and DO from attached plants. However, in this case the data describing nitrate and DO are inconsistent with the QUAL II model formulation (see Figures 39 and 40). Except for primary productivity (diurnal changes in DO ranged from 0.4 to 3.9 milligrams per liter), sources and sinks of nitrate and DO seem to be adequately described. The inconsistency arises because the DO balance indicated that net respiration exceeds net photosynthesis, whereas the nitrogen balance indicated that net photosynthesis exceeded net respiration. Nitrate predictions indicated that significant biomass growth occurred in which nitrate was removed from the water. Predictions of DO indicated that a decrease in biomass was necessary to further decrease DO so that predictions agreed with measurements. Lacking data describing chlorophyll a or biomass, it was not possible to resolve this problem.

Several model simulations did show that temperature simulation was necessary to model diel DO variation. Furthermore, nutrient cycling where net respiration exceeded net photosynthesis, described most of the diel variation of DO in the upstream segment between RK 68 and 52 (RM 42 to 32.5).

Verification

Following calibration, the predictions of the Streeter-Phelps and QUAL II models were verified using the April 1976 data (Figures 35 to 40). Data describing coefficients determined by calibration were unchanged for verification tests. In general, there was good agreement between observations and predictions with the exception of organic nitrogen and nitrate. In April 1976, BOD, organic nitrogen, ammonia, nitrite, nitrate, and DO occurred at greater concentrations and showed greater variation from a mean value compared to the September 1979 calibration data.

Figure 35 shows that the Streeter-Phelps and QUAL II models gave equivalent BOD predictions for the April 1976 data except between Pueblo STP and Salt Creek. Despite the occurrance of greater BOD concentrations and greater variation of BOD in April 1976 compared to September 1979, predictions agreed with measurements. The greatest difference between a mean observation and prediction was 2.5 milligrams per liter or 22 percent compared with 0.7 milligrams per liter or 54 percent for the September 1979 calibration data.

Figure 36 confirms the lack of agreement between observed organic nitrogen and predictions from the Streeter-Phelps model using the April 1976 verification data. As in the calibration, organic nitrogen was overpredicted and the predicted trend of decreasing organic nitrogen between RK 48 to 24 (RM 30 to 15) did not agree with the measured trend of increasing organic nitrogen.

Figure 37 verifies agreement between predicted and observed ammonia except between RK 24 to 0 (RM 15 to 0), where ammonia was overpredicted. Because the water in this segment (RK 24 to 0 or RM 15 to 0) entered the study reach prior to the time when sampling began, ammonia samples taken on April 1 to 2, 1976 from the Pueblo STP and Salt Creek may not have been representative of water quality in this downstream segment. Unlike the September 1979 predictions, these predictions reflected the difference between the Streeter-Phelps and QUAL II models in correcting ammonia decay rates for temperature. The empirical temperature corrections were:

Model	K_T
Streeter-Phelps	$K_{20} \cdot (1.09)^{T-20}$
QUAL II model	$K_{20} \cdot (1.047)^{T-20}$

In addition, some difference was due to modeling organic nitrogen with the Streeter-Phelps model.

The greatest difference between April 1976 observations and the Streeter-Phelps model predictions was 0.62 milligrams per liter occurring at RK 20.1 (RM 12.5). For the QUAL II model predictions, the greatest difference was 0.59 milligrams per liter at RK 44.9 (RM 27.9). The greatest difference between September 1979 observations and predictions from the Streeter-Phelps and QUAL II models was 0.3 milligrams per liter. Based on these data, it was not possible to determine how to best correct ammonia decay rates for temperature changes.

Figure 38 indicates reasonable agreement between the April 1976 observations of nitrite and predictions of the Streeter-Phelps and QUAL II models. Predictions were within 0.23 milligrams per liter of observations. Model predictions from the Streeter-Phelps and QUAL II models differed because the same temperature corrections applied to ammonia decay rates were applied to nitrite decay rates. In addition, the Streeter-Phelps model simulation included ammonia removal by aquatic plants.

Figure 39 shows verification results for nitrate predictions compared to April 1976 observations. The Streeter-Phelps model simulation for mean nitrate, based on the premise that mean daily photosynthetic oxygen production was balanced by respiration, was accurate to within 0.66 milligrams per liter. The QUAL II model was not calibrated to predict nitrate.

Figure 40 shows the agreement between the predictions from the Streeter-Phelps and QUAL II models and DO observed in April 1976. Both models gave about the same results despite several differences. These included different formulations for DO saturation and temperature corrections for ammonia and nitrite decay. The greatest difference between predictions and observations was 2.2 milligrams per liter compared with 2.0 milligrams per liter for the September 1979 data.

Comparison

In comparison, this application to the Arkansas River data indicated that several model differences existed. First, the Streeter-Phelps model had a superior discretization scheme. The QUAL II model required more coding effort because equal length computational elements were required. Despite the extra effort, the QUAL II model did not produce a proper simulation of the water quality between the Pueblo STP effluent and Salt Creek. The WQRRS model required the greatest coding effort to simulate stream geometry. Three separate applications were required and the Pueblo STP effluent was overdiluted and shifted upstream. These differences did not seriously detract from the flexibility of the QUAL II model, whereas discretization limits of 10 inflows and 41 cross sections did detract from the flexibility of the WQRRS model.

Other differences were noted in modeling BOD, ammonia, nitrate, and temperature. The WQRRS model simulated slower BOD and ammonia decay for the same coefficients, failed to simulate nitrate, and did not allow specification of stream temperature as input data. These same problems occurred in simulating water quality in the Chattahoochee and Willamette rivers.

The cooler stream temperatures recorded in April 1976 helped define several differences between the Streeter-Phelps and QUAL II models. These two models applied different temperature corrections to reaeration coefficients and nitrogen decay rates. In addition, the QUAL II model does not correct DO-saturation calculations for local atmospheric pressure.

Finally, the Streeter-Phelps model did not properly predict organic nitrogen in this river. This was related to the failure to simulate biomass and the measurement of total instead of dissolved organic nitrogen. The QUAL II model does include chlorophyll *a*, but the model could not be calibrated so the biomass component removed DO and nitrate. The WQRRS model had a wider range of capabilities in simulating biomass, but these applications were deemed beyond the scope of this study.

PART VIII: SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

Summary

Four models were examined to determine the validity and usefulness of the models for modeling stream water quality downstream of reservoirs. These models included the U.S. Geological Survey One-Dimensional Steady-State Stream Water-Quality Model (the modified Streeter-Phelps model), QUAL II Stream Quality Model (Southeast Michigan Council of Governments version), U.S. Army Corps of Engineers Water Quality for River-Reservoir Systems (WQRRS) Model, and the MIT Transient Water Quality Network Model. Modeling capabilities listed in model documentations were examined and summarized in Table 2 for reference. The models were evaluated and compared using a comprehensive data base compiled from previous studies of the Chattahoochee, Willamette, and Arkansas rivers.

The data base included information from USGS studies of three rivers having widely varied characteristics. The Chattahoochee River is a moderate size eastern stream with a moderate channel slope. Nitrification is important and aquatic plants and benthic interactions are unimportant. The Willamette River is a large sluggish West Coast stream having three distinct reaches with different water-quality characteristics. Decay and reaeration rates are low. Nitrification occurs in the upstream reach and benthic demand occurs in the downstream reach of the Willamette River. The Arkansas River in Colorado is a small cool stream with steep channel slopes and large decay and reaeration rates. Waste inflows make up a majority of the Arkansas River flow.

The data base, which should be useful in establishing the credibility of other models, was limited to steady flow conditions. The water-quality data from the Chattahoochee and Willamette rivers best describes steady conditions. Frequent measurements at several sites on the Arkansas River makes it possible to model diurnal changes using these water-quality data. A review of USGS files and a brief literature review failed to reveal the existence of a comprehensive synoptic data collection study for dynamic flow conditions. For that reason the evaluation of dynamic models was limited to their steady-state capabilities.

A number of differences existed between models because each model was designed for different conditions. Except for the MIT model, the models performed as expected based on model documentation. The QUAL II model showed the greatest flexibility in simulating steady flow and water quality. However, the Streeter-Phelps model was inexpensive and easy to apply and calibrate. Because of complex coding and discretization requirements, the WQRRS model should be limited to applications involving dynamic conditions.

The Streeter-Phelps, QUAL II, and WQRRS models proved to be of comparable accuracy and equally valid under steady-state stream conditions. Despite the fact that the MIT model has been used successfully in estuary water-quality studies, it was not possible to confirm the validity or usefulness of the model using the steady-state data compiled for this study. While the Velz rational method was not originally included in this model evaluation, use of the Chattahoochee and Willamette river data made it possible to use previous work to compare the Velz technique to the other models. Compared to the Streeter-Phelps, QUAL II, and WQRRS models, the Velz rational technique was equally valid and of comparable accuracy. Examination of the utility and full capabilities of the Velz technique was beyond the scope of the work.

The minor differences among the Streeter-Phelps, QUAL II, and WQRRS models included organic nitrogen modeling, nitrate uptake, nitrogenous BOD, temperature corrections for nitrification and reaeration coefficients, distributed sources of BOD, benthic oxygen demand, 5-day BOD modeling, and calculation of DO saturation. These differences were considered minor for two reasons. The models are flexible enough to compensate for the differences, or the difference in prediction was smaller than the variation in the data due to measurement.

Conclusions

The examination of the Streeter-Phelps model, QUAL II model, WQRRS model, and MIT model emphasized several differences. As outlined in Table 2, the Streeter-Phelps model was formulated for steady flow and waste loads and does not simulate temperature and stream hydraulics. The QUAL II model also simulates steady water quality but has the added capability to model temperature, stream hydraulics, and diurnal changes in water quality for steady discharge. The MIT model was formulated to simulate unsteady flow and water quality for nitrogen-limited estuaries. The usefulness of the MIT model could not be confirmed with the steady-state river data selected for this study. The WQRRS model was the most general model considered. Its capabilities include dynamic modeling of flow, water quality, and stream biota.

Minor program errors were noted with the Streeter-Phelps, QUAL II, and WQRRS models. However, most of these problems have been cleared up in recent updates. Major difficulties and minor programming errors were encountered in applying the MIT model. Overall, the Streeter-Phelps, QUAL II, and WQRRS models performed according to expectations derived from the user's guide for each model. The MIT model did not.

Recent updates to the Streeter-Phelps model has corrected several problems encountered in this study. These included updating the DO saturation formulation to compute DO saturation as a function of salinity in addition to temperature and barometric pressure. Several different reaeration options have been added, and the fecal coliform die-off option was updated to allow temperature corrections to the die-off rate. Finally, an error in the DO mass balance at tributary inflows has been corrected.

The NCASI (1980) recently reviewed the SEMCOG version of the QUAL II model and corrected errors in the steady-state temperature submodel; one of the reaeration coefficient formulations; and the data specification for the algae submodel. Therefore, the most reliable and up-to-date version of the QUAL II model is the SEMCOG version with the NCASI updates.

The WQRRS model was recently updated to correct several problems noted in this study. Errors in the options to directly specify temperature or use the equilibrium temperature method were corrected. The reaeration coefficient can now be directly specified. Ammonia and nitrite decay were coupled to the nitrate formulation. Finally, organic detritus can be simulated without simulating organic sediment.

A comparison of the Streeter-Phelps and QUAL II models indicated that the Elder equation for longitudinal dispersion may in fact under-predict dispersion for natural channels. Furthermore, these results indicated that numerical dispersion in the QUAL II model was insignificant. However, comparing the WQRRS model to the two steady-state models indicated that either dispersion calculations or temperature corrections for decay rates led to the simulation of less removal of BOD, detritus, and ammonia for the same specified decay rates and loadings.

The Streeter-Phelps model simulates organic nitrogen but not algae. The QUAL II model lumps organic nitrogen with algae. The WQRRS model includes organic nitrogen with detritus. The comparison indicates that the QUAL II and WQRRS models have sufficient flexibility such that organic nitrogen simulation is not crucial.

Both the Streeter-Phelps and QUAL II models were limited to modeling benthos sources of BOD as a first-order process rather than as a constant source. In addition, different temperature corrections were applied to nitrogen decay rates, reaeration coefficients, and BOD settling rates. Furthermore, DO saturation was not corrected for salinity and barometric pressure in the QUAL II model.

The major difference between the Streeter-Phelps, QUAL II, and WQRRS models involved the stream discretization scheme. Because of the discretization scheme, coding data for the QUAL II model was easiest except where extra effort was needed to match the discretization scheme of other models. This was offset by greater discretization errors resulting from the requirement of equal-length elements. The Streeter-Phelps model better matched stream geometry and had less discretization error. Discretization limits of 10 inflows and 41 cross sections severely limited the flexibility of the WQRRS model.

The WQRRS model has two other significant limitations for a general unsteady water-quality model. First, the scour of solid material has been neglected. As Kreutzberger and others (1980) noted, scour of benthic materials can significantly affect water quality during unsteady flows. Second, the WQRRS model simulates 5-day BOD and converts that to ultimate BOD using a constant factor of 1.43. The constant factor should be specified as input data or computed from the in-stream decay rate. For the data compiled in this study, the factor varied from 1 to 2.5.

Like the WQRRS model, the MIT model capabilities noted from the documentation did not include benthic interactions. In addition, the documented model was not formulated to simulate orthophosphate and the reaeration coefficient option was severely limited.

In general, the QUAL II model was best suited to simulate water quality for steady flow. However, the Streeter-Phelps model is inexpensive and easy to apply. Options for nitrogenous BOD and anaerobic conditions are available in the Streeter-Phelps model. In addition, calibration of the Streeter-Phelps model is simplified by printed graphs of results. Because of cost and data coding effort required, the WQRRS model should be limited to unsteady flow and water-quality simulation or conditions where complex plant and animal communities contribute to water-quality problems.

The MIT model should be selected for modeling studies only after a serious consideration of alternatives and objectives. Unlike the other three models it was not possible to apply the MIT to steady-state river quality data using the computer code furnished by the EPA and using the documentation as a guide. This difficulty in applying the model is not unlike the difficulties experienced by other users.* While the model has proven useful in other studies, the results of this study indicates that the assistance of an experienced user of the MIT model may be needed to successfully apply the model.

Recommendations for Further Study

The data base compiled for this study was adequate for testing the steady-state capabilities of the Streeter-Phelps, QUAL II, and WQRRS models. However, field studies are needed to gather dynamic water-quality data so that a similar comparison for dynamic models will be possible. The most likely models include the WQRRS model and the

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* Personal communications: June 1980, Thomas Barnwell, Civil Engineer, EPA, Athens, Ga.; June 1980, Frank Tatom, Consultant, Engineering Analysis, Inc., Huntsville, Ala.; and July 1981, Frank Parker, Professor, Vanderbilt University, Nashville, Tenn.

Lagrangian model developed by Jobson (1980). In addition, further study is needed to collect data on aquatic plants and animals along with data on the water quality in order to test model formulations for biota.

Each of the four models considered in this study could benefit from further development. All four models, with the exception of the updated Streeter-Phelps model, could benefit from an improved DO-saturation formulation. The USGS recently surveyed the literature* and determined that the formulations from Weiss (1970) and Standard Methods (Franson, 1980, p. 86) best related DO saturation to temperature, salinity, barometric pressure, and water vapor pressure.

The Streeter-Phelps model could be improved by:

1. Adding a temperature and stream hydraulics subroutine.
2. Internal checks of the input data.
3. Better organization of output data.
4. Dividing the code into modules having a specific purpose.
5. Adding a reaeration formulation for dams or rapids.
6. Updating the documentation to provide more detail.

The QUAL II model could be improved by:

1. Adding formulations for dissolved organic nitrogen and periphyton and reaeration at dams and rapids.
2. Revising the formulation for dispersion.
3. Revising the formulation for 5-day BOD so a variable conversion ratio BOD_{ULT}/BOD_5 can be specified.
4. Adding the option to directly specify travel time.
5. Revising the ammonia formulation to allow the escape of ammonia gas to the atmosphere.
6. Adding a plotting subroutine to assist in calibration.

The WQRRS model could be improved by:

1. Simulating ultimate BOD rather than 5-day BOD times the factor 1.46.
2. Allowing reaction coefficients to vary by reach.
3. Adding formulations describing benthos sources or simulating scour.

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* Written communications, May 8, 1981, Jack Pickering, USGS Quality of Water Branch, Reston, Va.

4. Increasing discretization limits to allow 40-50 inflows, withdrawals, and nonpoint sources; and 70-80 cross sections.

5. Revising the documentation to provide more detail and up-to-date examples.

The MIT model could be updated to provide a more detailed documentation. In addition, the computer code may require a careful review. Some attention to the utility and ease-of-use might improve the credibility of the MIT model as a general dynamic water-quality model.

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APPENDIX A: NOTATION

The following list defines abbreviations, acronyms, and symbols used in the report.

a	Coefficient used in the wind-speed function or to relate velocity to discharge
A	Total cross-sectional area
b	Coefficient used in the wind-speed function or to relate velocity to discharge
B	Total top width
BOD	Biochemical oxygen demand
BOD_{ULT}	Ultimate carbonaceous BOD
C	Coefficient relating a function of velocity and depth to K_2
$CaCO_3$	Calcium carbonate
COD	Chemical oxygen demand
COE	U.S. Army Corps of Engineers
CO_2	Carbon dioxide
d	Depth
D_L	Longitudinal dispersion coefficient
DO	Dissolved oxygen
EL	Environmental laboratory
EPA	U.S. Environmental Protection Agency
EWQOS	Environmental and Water Quality Operation Studies
GCHC	Gulf Coast Hydroscience Center
H	Depth
HEC	Hydrologic Engineering Center
HEC-2	Second in the series of the HEC models
K_T	Decay rate for temperature T
K_1	BOD decay rate
K_2	Reaeration rate
MIT	Massachusetts Institute of Technology
n	Manning's roughness coefficient
NCASI	National Council of the Paper Industry for Air and Stream Improvement

NGVD	National Geodetic Vertical Datum of 1929
NH_3	Ammonia
NO_2	Nitrite
NO_3	Nitrate
PO_4	Orthophosphate
Q	Discharge
QUAL II	Title of a steady-state stream water-quality model
RK	River kilometer
RM	River mile
RMS	Root mean square
SEMCOG	Southeast Michigan Council of Governments
SR	State route
STORET	EPA water-quality data management system
STORM	Urban Storm Water Runoff model
STP	Sewage treatment plant
T	Temperature
TOC	Total organic carbon
u	Reach averaged velocity
USGS	U.S. Geological Survey
V	Velocity
W	Windspeed
WATSTORE	USGS hydrologic data management system
WES	Waterways Experiment Station
WQRRS	Water Quality for River Reservoir Systems
WTF	Waste treatment facility
α	Coefficient relating depth to discharge
β	Exponent coefficient relating depth to discharge
θ	Temperature correction coefficient