Modeling Water Quality in the Tualatin River, Oregon 1991–1997

Prepared in cooperation with the
Unified Sewerage Agency of Washington County, Oregon

U.S. DEPARTMENT OF THE INTERIOR
U.S. GEOLOGICAL SURVEY

Water-Resources Investigations Report 01–4041

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Cover Photographs:

Top: Tualatin River, Oregon, at Stafford Road, river mile 5.5.
(Photograph by Tirian Mink, Portland, Oregon.)

Middle: Treatment basin at Unified Sewerage Agency's Rock Creek Treatment Facility, Hillsboro, Oregon.
(Photograph by Stewart A. Rounds.)

Bottom: U.S. Geological Survey scuba divers checking a sediment oxygen demand chamber in the Tualatin River, Oregon.
(Photograph by Charles Collins.)

U.S. Department of the Interior
U.S. Geological Survey

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By STEWART A. ROUNDS AND TAMARA M. WOOD

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Portland, Oregon: 2001

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CONTENTS

FIGURES

TABLES

Modeling Water Quality in the Tualatin River, Oregon, 1991–1997

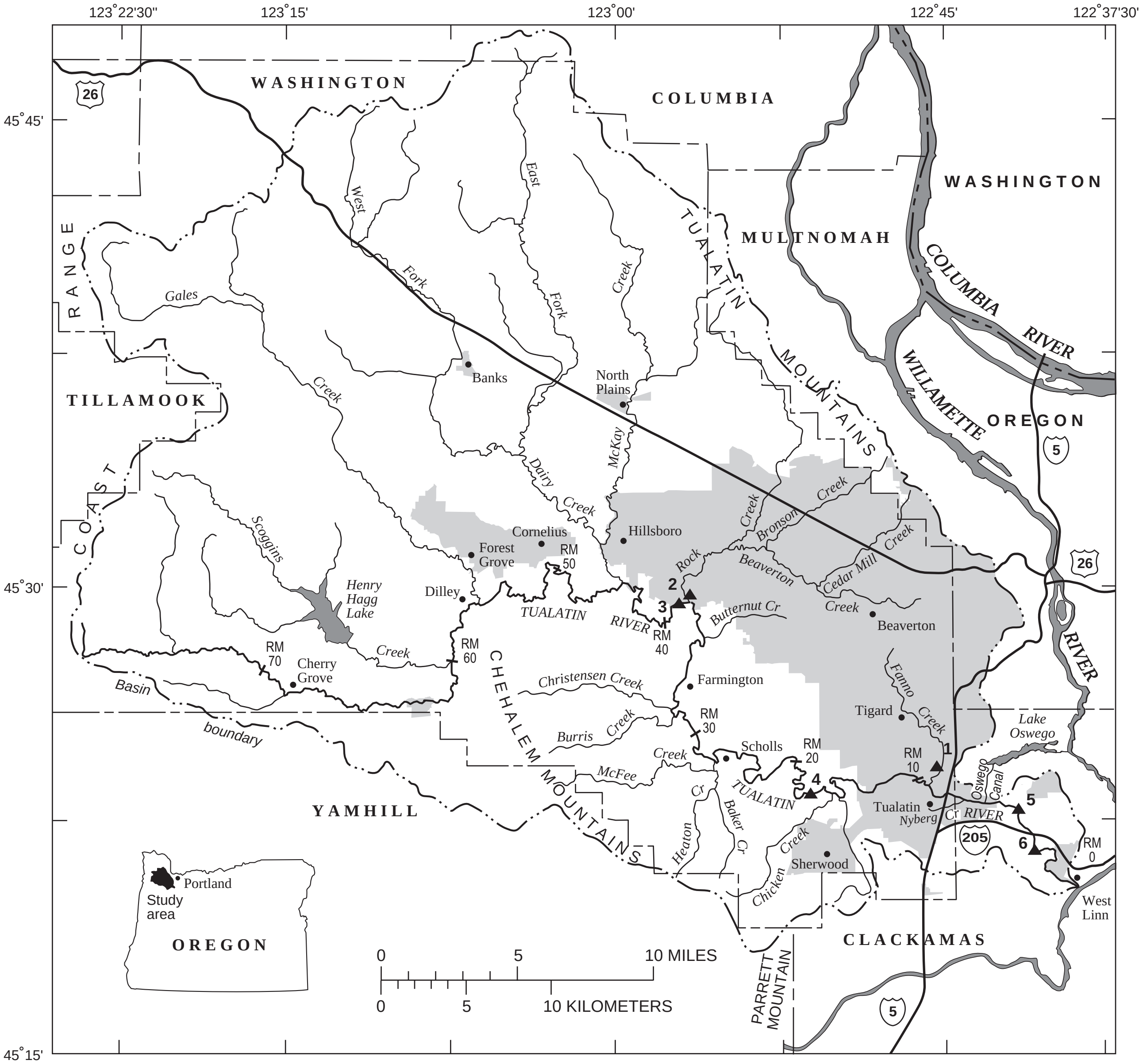
By Stewart A. Rounds *and* Tamara M. Wood

ABSTRACT

The calibration of a model of flow, temperature, and water quality in the Tualatin River, Oregon, originally calibrated for the summers of 1991 through 1993, was extended to the summers of 1991 through 1997. The model is now calibrated for a total period of 42 months during the May through October periods of 7 hydrologically distinct years. Based on a modified version of the U.S. Army Corps of Engineers model CE-QUAL-W2, this model provides a good fit to the measured data for streamflow, water temperature, and water quality constituents such as chloride, ammonia, nitrate, total phosphorus, orthophosphate, phytoplankton, and dissolved oxygen. In particular, the model simulates ammonia concentrations and the effects of instream ammonia nitrification very well, which is critical to ongoing efforts to revise ammonia regulations for the Tualatin River. In addition, the model simulates the timing, duration, and relative size of algal blooms with sufficient accuracy to provide important insights for regulators and managers of this river. Efforts to limit the size of algal blooms through phosphorus control measures are apparent in the model simulations, which show this limitation on algal growth. Such measures are largely responsible for avoiding violations of the State of Oregon maximum pH standard of 8.5 in recent years, but they have not yet reduced algal biomass levels below the State of Oregon nuisance phytoplankton growth guideline of 15 μg/L chlorophyll-*a*. Most of the dynamics of the instream dissolved oxygen concentrations are captured by the model. About half of the error in the simulated dissolved oxygen concentrations is directly attributable to error in the size of the simulated phytoplankton population. To achieve greater accuracy in simulating dissolved oxygen, therefore, it will be necessary to increase accuracy in the simulation of Tualatin River phytoplankton. Future efforts may include the introduction of multiple algal groups in the model. This model of the Tualatin River continues to be used as a quantitative tool to aid in the management of this important resource.

INTRODUCTION

The Tualatin River Basin is located on the west side of the Portland metropolitan area in northwest Oregon (fig. 1). From its headwaters in the forested Coast Range mountains, the river flows east into a fertile agricultural valley. The river meanders through the valley bottom, skirting to the south and west of most of the urban areas, home to more than 350,000 people. Finally, the river flows through the south-western edge of the Portland metropolitan area before discharging to the Willamette River at West Linn. The population of the Tualatin Basin depends on the river as a source of municipal, industrial, and irrigation water; habitat for fish and other wildlife; and a place to recreate.



Base modified from U.S. Geological Survey
1:100,000, topographic quadrangles, 1978-84

EXPLANATION

2 ▲ **Reference location**

Map number	Site identification number	Site name and river mile location
1	452359122454500	Durham Wastewater Treatment Plant (RM 9.3)
2	452938122565500	Rock Creek Wastewater Treatment Plant (RM 38.1)
3	14206440	Tualatin River at Rood Bridge Road (RM 38.4)
4	14206740	Tualatin River at Elsner Road (RM 16.2)
5	14207050	Tualatin River at Stafford Road (RM 5.5)
6	14207200	Tualatin River at Oswego diversion dam (RM 3.4)

Designated urban growth area—From Metro, 1998

RM 10 **River mile**

Figure 1. Tualatin River Basin, Oregon.

The Tualatin River is not a large river by many criteria, although its size varies with the time of year. Streamflow in western Oregon reflects the seasonal variation in precipitation. Most of the precipitation in the basin falls as rain during the November through April "wet" season. In the center of the basin at Hillsboro, the mean annual precipitation total for the 1991–1997 period was 40.08 inches, of which an average of 30.72 inches fell between November and April. The May through October "dry" season generates much less rain; the months of July and August are particularly dry, producing an average of only 0.62 and 0.50 inches of rain during 1991–1997 (Oregon Climate Service, no date). Flows in the Tualatin River near its mouth at West Linn typically decrease from more than 2,000 ft^3/s (cubic feet per second) at the beginning of May to only 100 or 200 ft^3/s during the low-flow period from July through October. During the low-flow summer period, the river's flow is augmented from stored water in Henry Hagg Lake (fig. 1).

Just as the size of the river varies seasonally, the water quality problems also are seasonal phenomena. Bacteria levels are a concern all year, but most water quality concerns for the Tualatin River are manifested during the warm and relatively dry summer. The warm summers in western Oregon often cause the river's water temperature to exceed Oregon State requirements for the passage of fish such as salmon and steelhead (17.8˚C). Long travel times, when combined with ample nutrients (phosphorus and nitrogen) and sunny summer weather, produce blooms of phytoplankton in the reservoir-like reach of the river from river mile (RM) 3.4 to 30. Such blooms can impair the river's aesthetic qualities, produce violations of the Oregon State maximum pH standard (8.5), and contribute to problems with the river's dissolved oxygen concentrations (< 6.5 mg/L). Indeed, such problems were prevalent during the 1980s. The river also is most susceptible to the effects of treated municipal wastewater during the low-flow period. In the mid-1980s, instream nitrification of ammonia loads from municipal wastewater typically caused or contributed to violations of the Oregon State minimum dissolved oxygen standard.

In 1984 and again in 1986, the Oregon Department of Environmental Quality recognized these water quality problems and listed the Tualatin River as an impaired waterbody. In 1988, in accordance with the Total Maximum Daily Load (TMDL) provisions of the Clean Water Act, limits were placed on the amount of ammonia and phosphorus allowed in the river and its largest tributaries (Oregon Department of Environmental Quality, 1997). The TMDL for ammonia was designed to protect the river's aquatic health by preventing excessive consumption of dissolved oxygen through nitrification. The TMDL for total phosphorus was designed to protect the river's aesthetic qualities by limiting the size of phytoplankton blooms; such limits also would protect the river's aquatic health by preventing the high pH conditions typically caused by large blooms.

At the time these regulations were created, the largest sources of ammonia and phosphorus to the river during the summer were two large wastewater treatment plants (WWTPs) operated by the Unified Sewerage Agency (USA) of Washington County. USA operates four WWTPs in the basin, but the two smaller plants do not discharge their effluent to the river during the summer. USA is the agency charged with managing most of the municipal wastewater from the urban areas of the basin. In response to both a rapidly growing urban population as well as the new TMDL regulations, USA implemented a plan to upgrade the two large WWTPs. Advanced biological nutrient removal for ammonia and phosphorus, as well as two-stage alum treatment for phosphorus removal, were added to these WWTPs in the early 1990s. These state-of-the-art facilities now are in compliance with their load limits under the ammonia and total phosphorus TMDLs.

When the Tualatin River TMDLs were created, comparatively little was known about the sources and transport of phosphorus and nitrogen, the dynamics of algal growth, and the consumption and production of dissolved oxygen in the river. In order to learn more about these processes and use that information to better manage the river, USA entered into a scientific partnership with the U.S. Geological Survey (USGS) in 1990. The objectives of this partnership were:

(1) to identify the major sources of nutrients to the main-stem Tualatin River,

(2) to assess the transport and fate of those nutrients in the main stem,

(3) to quantify processes that affect dissolved oxygen concentrations in the main stem, and

(4) to construct and use a mechanistically based, process-oriented model of nutrients and dissolved oxygen for the main stem.

These objectives were accomplished and documented in several reports. Nutrient sources and transport in the Tualatin River are discussed in a report by Kelly and others (1999). Several reports on processes affecting dissolved oxygen and the calibration and performance of the USGS Tualatin River model also are available (Rounds and others, 1999; Rounds and Wood, 1998; Wood and Rounds, 1998, Rounds and Doyle, 1997).

The USGS Tualatin River model was developed to better understand and quantify the processes controlling nutrient transport, algal dynamics, and dissolved oxygen, then use that understanding to test the efficacy of potential management strategies for the river. Based on a modified version of the U.S. Army Corps of Engineers model CE-QUAL-W2 (Cole and Buchak, 1995), the Tualatin River model was originally calibrated using data from May–October of 1991, 1992, and 1993 and was documented by Rounds and others (1999). Since 1993, however, at least five important factors affecting the river have changed. First, 1991–1993 was a period of transition for the operation of the two large USA WWTPs. During those summers, the capacities of the WWTPs were being increased and their treatment capabilities were being tested. As a result, there were periods when phosphorus and ammonia removal by the WWTPs was greater or less than what one would expect during normal operations. While this variability created a wide range of river conditions for testing the model, it does not reflect the more stable operating conditions of the WWTPs since 1993. The 1994–1997 period is more typical for WWTP operations (fig. 2). Second, loads of phosphorus from some of the Tualatin River tributaries have decreased since 1993 as a result of specific efforts by the nonpoint-source Designated Management Agencies to reduce total phosphorus concentrations in stormwater. Third, population growth has continued since 1993, causing WWTP capacities to be increased again. Fourth, closer attention to the level of flow augmentation during the 1994–1997 period resulted in better overall management of river flow for purposes of water quality. Finally, climate data show that since 1995 the Pacific Northwest has entered into a period of higher than normal precipitation. Although the 1991–1993 period showed a wide variation in hydrologic conditions (1992 was a severe drought year), the 1995–1997 period was generally wetter than the 1991–1994 period, causing base flows to the river to be greater and making more water available for flow augmentation. Indeed, flows in the Tualatin River at RM 1.8 (West

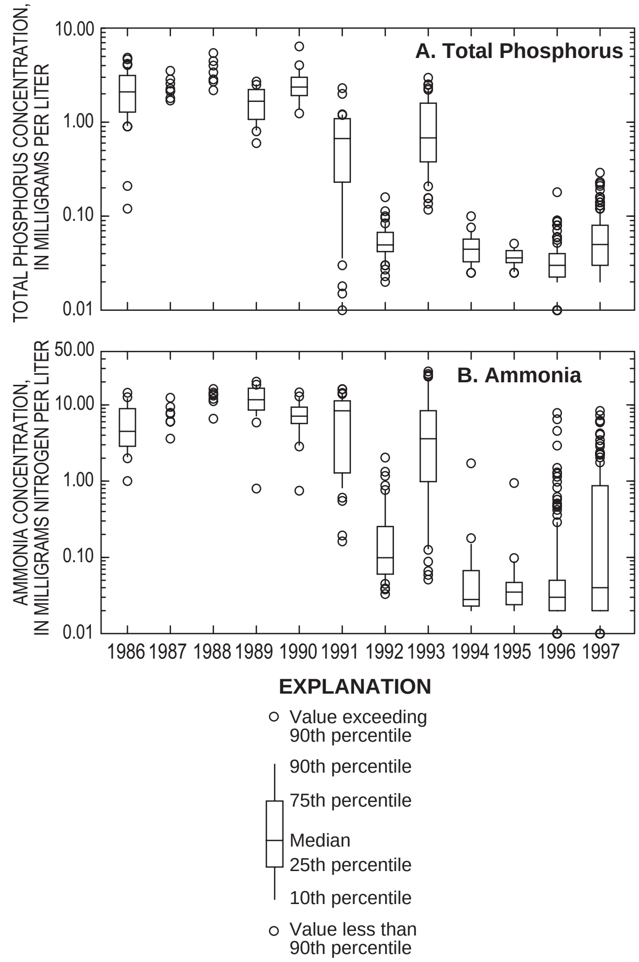


Figure 2. Concentrations of (A) total phosphorus and (B) ammonia nitrogen in the effluent of the Durham wastewater treatment plant during the May through October periods of 1986–1997.

Linn) for the summers of 1996 and 1997 were significantly higher than the post-Hagg Lake historical average (1976–1997) and the measured flows from 1991–1995 (fig. 3). These climate cycles generally last 20 to 25 years; therefore, the wetter-than-normal period could continue for another 20 or more years (Taylor and Hannan, 1999).

PURPOSE AND SCOPE

In order to retain the value of the USGS Tualatin River model as a management tool and build upon the knowledge gained from the original modeling study, the period of model calibration was expanded from the summers of 1991–1993 to include the summers of 1991–1997. (For the purposes of this study, summer is

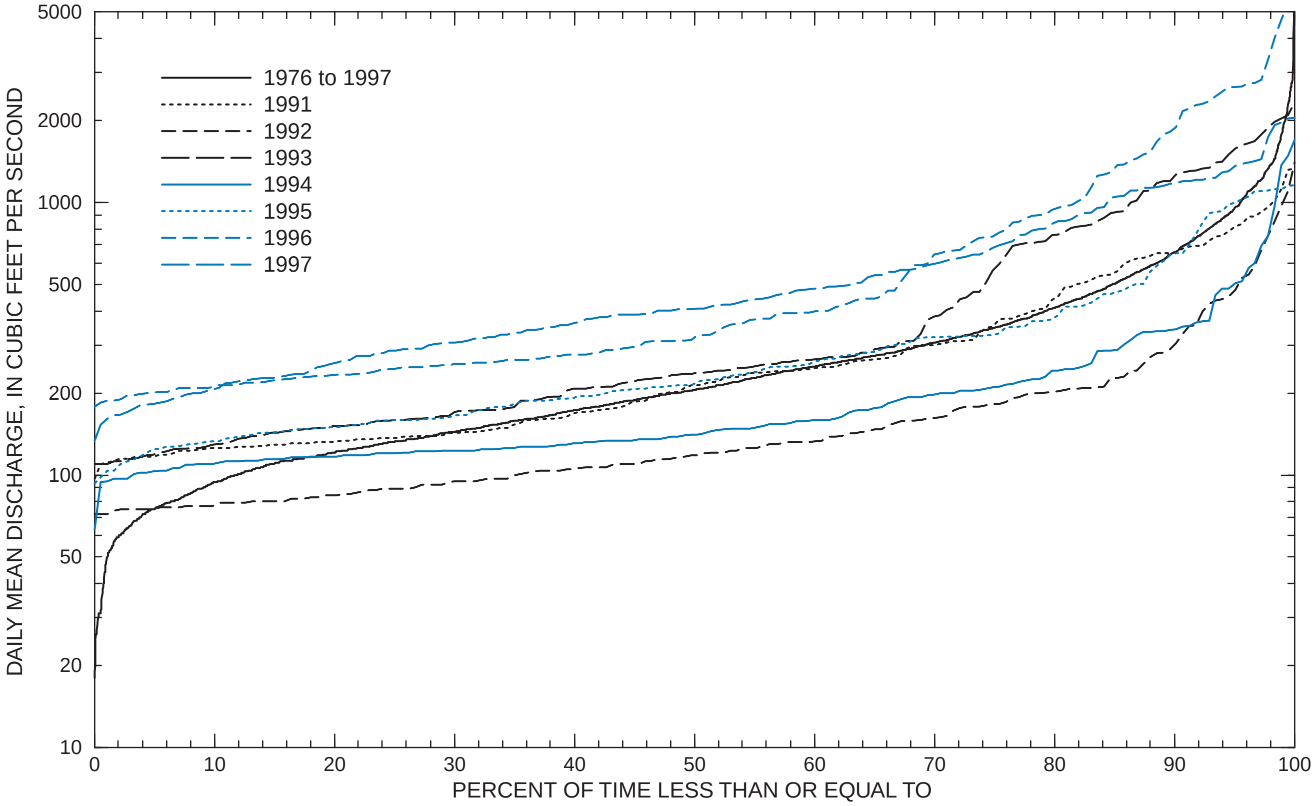


Figure 3. Flow duration curves for the Tualatin River at river mile 1.8 (West Linn), May through October of 1991–1997.

defined as May 1 through October 31.) In the process of expanding the period of calibration, the model was updated and modified slightly.

This report documents the performance of the USGS Tualatin River model for the summers of 1991–1997 as well as the minor changes made to the model since the conclusion of the original study. This report builds on the original report by Rounds and others (1999). The sources and frequency of data used in this study, as well as the values of all model parameters, are documented in this report. When those parameters are unchanged from the original study, however, no further details regarding those parameters are included in this report.

In addition to documenting the model's performance, this report includes discussions on a number of topics related to the processes that control the river's water quality—when the model results shed new light on these topics. Such topics include instream ammonia nitrification, the effect of the 1996 flood aftermath on algal growth, and other issues related to nutrient transport and algal dynamics. The report starts with a section on data sources and frequency followed by model modifications and model parameterization. Next, the ability of the model to describe measured conditions is quantified with four goodness-of-fit statistics. Finally, the report focusses on a comparison of measured and simulated conditions, from discharge and temperature to nutrients, algae, and dissolved oxygen, interspersed with discussions of water quality processes. As in the original report, all references to algae refer only to phytoplankton.

MODEL SETUP

Model Application

As in the original application by Rounds and others (1999), this application of CE-QUAL-W2 modeled the Tualatin River from RM 38.4 (Rood Bridge) to RM 3.4 (Oswego dam), the reach with most of the water-quality problems. Both the Rock Creek WWTP (RM 38.1) and the Durham WWTP (RM 9.3) were included as tributary inputs. Ten other tributaries were simulated as point sources: Rock, Butternut, Christensen, Burris, Baker, McFee, Chicken, Rock (South), Fanno, and Nyberg Creeks. Of these, only Rock and Fanno Creeks contributed significant amounts of water to the model reach. Ground water and small ungaged tributaries were handled as a nonpoint source. The model grid used 155 segments and 16 layers; most segments were about 0.25 mile long and most layers were 2 feet deep. The details of the river bathymetry and the model grid were documented previously (Rounds and others, 1999).

Data Sources and Frequency

Complex water quality models such as CE-QUAL-W2 require many types of boundary data, calibration data, and meteorological data as well as rate data such as the rates of algal growth and settling. The data used in this modeling study were collected by a variety of organizations for many purposes. Each of these data sets was quality-assured before use. The types and sources of most of the data used by the model are listed in table 1. These data are available upon request from the source agencies. During the 1991–1993 period of the original study, special efforts were made to obtain some of the more difficult-to-collect data such as algal primary productivity, light extinction coefficients, settling velocities, and zooplankton abundances. Extra water quality samples also were collected during 1991–1993 to augment USA's routine monitoring program. The extra efforts during 1991–1993 result in more of some types of data being available for that period. For some locations such as the WWTPs, however, more water quality data are available for 1996 and 1997 than for any of the years from 1991–1995. The frequency of available discharge, water temperature, and water quality data for the relevant boundary or main-stem sites is documented in table 2.

Table 2 shows several important characteristics of the data set used in this study. In particular, note the absence of light extinction, algal settling velocity, zooplankton abundance, and most importantly, primary productivity data for the 1994–1997 period. The calibration of the algal growth rate for 1994–1997, therefore, was forced to rely on the trends measured in the original 1991–1993 study. In addition, note the dearth of discharge and water-quality data from the smaller tributaries (Butternut, Christensen, Burris, Baker, McFee, Rock [South], and Nyberg Creeks) for 1994–1997. Data for the small tributaries were not critical to the study. On the other hand, abundant data are available for the important calibration sites (Scholls Bridge, Elsner Road, and Stafford Road) as well as the upstream boundary (Rood Bridge), the WWTPs (Rock Creek and Durham), and the larger tributaries (Rock, Chicken, and Fanno Creeks). These important sites have most data available on weekly to daily or better frequencies, which is more than adequate for the purposes of this investigation.

Model Modifications

The model used in this study is a modification of version 2.0 of the U.S. Army Corp of Engineers model CE-QUAL-W2 (Cole and Buchak, 1995). Most of the modifications made to this laterally averaged, 2-dimensional model were documented by Rounds and

Table 1. Sources of boundary data, calibration data, and forcing functions
[USGS, U.S. Geological Survey; USA, Unified Sewerage Agency; OWRD, Oregon Water Resources Department; OCS, Oregon Climate Service; BOR, Bureau of Reclamation]

Data Type	Source
Discharge and withdrawal rates	USGS, OWRD, USA
River elevation at Oswego Canal headgates	OWRD
Water temperature	USGS, USA, OWRD
Insolation	USGS
Precipitation	BOR, OCS
Wind speed and direction	USGS, BOR, OCS
Air temperature	USGS, BOR, OCS
Dew point temperature	USGS, BOR, OCS
Chloride, dissolved solids, total suspended solids, ammonia, nitrate, total kjeldahl nitrogen, orthophosphate, total phosphorus, chlorophyll-*a*, dissolved oxygen	USA, USGS
Primary productivity	USGS
Water-column light extinction	USGS
Algal settling velocity	USGS
Zooplankton abundance	USGS

Table 2. Frequency of data used, May-October, 1991–1997, for water quality properties and constituents in the Tualatin River

[Symbols used in table: ○, no data; ⊙, approximately monthly data; ◔, approximately twice per month data; ◒, approximately weekly data; ◕, approximately twice per week data; ●, approximately daily data or better; ⊕, data for select periods only. Abbreviations: WWTP, wastewater treatment plant]

1991 1992 1993 ○○○
1994 1995 1996 1997 ○○○○

Property or Constituent	38.4 Rood Bridge	38.1 Rock Creek WWTP [a]	38.1 Rock Creek	36.8 Meriwether	35.7 Butternut Creek	31.9 Christensen Creek	31.6 Burris Creek	28.2 Baker Creek	28.2 McFee Creek	26.9 Scholls Bridge
discharge	●●● ●●●●	●●● ●●●●	◕◕◕ ●●●●	○○○ ○○○○	⊙⊙○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	○○○ ○○○○
water temperature	◒◒◒ ●●◒●	●●● ●●●●	◕◕◕ ●●◒●	◒◒◒ ○●⊕○	◒◒○ ○○○○	◒◒⊙ ○○◔○	◒◒⊙ ○○◔○	◒◒⊙ ○○⊙○	◒◒⊙ ○○◔○	◕◕◕ ◕◕◕◒
chloride	◒◒◒ ◒◒◒◒	◕◕◕ ◒◒◕◕	◕◕◕ ◒◒◒◒	◒◒◒ ○○⊕○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◒◒◒◒
dissolved solids	◒◒◒ ◒◒◒◒	◕◕◕ ◒◒◒○	◕◕◕ ◒◒◒◒	◒◒◒ ○○○○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒
total suspended solids	◒◒◒ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◒◒◒◒	◒◒◒ ○○○○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒
ammonia	◒◒◒ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◒◒◒◒	◒◒◒ ○○⊕○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒
total kjeldahl nitrogen	◒◒◒ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◒◒◒◒	◒◒◒ ○○○○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒
nitrate plus nitrate	◒◒◒ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◒◒◒◒	◒◒◒ ○○⊕○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒
total phosphorus	◒◒◒ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◒◒◒◒	◒◒◒ ○○⊕○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒
soluble orthophosphate	◒◒◒ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◒◒◒◒	◒◒◒ ○○⊕○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒
chlorophyll-*a*	◒◒◒ ◒◒◒◒	○○○ ○○○○	◕◕◕ ◒◒◒◒	◒◒◒ ○○⊕○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒
primary productivity	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	◒◒◒ ○○○○
water-column light extinction [c]	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	◒◒○ ○○○○
algal settling velocity	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○
zooplankton abundance [d]	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	◔◒◔ ○○○○
dissolved oxygen	◒◒◒ ◒◒◒◒	●●● ●●●●	◕◕◕ ◒◒◒◒	◒◒◒ ○○⊕○	◒◒○ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◒◒⊙ ○○○○	◕◕◕ ◕◕◕◒

Property or Constituent	23.2 QA site	16.2 Elsner Road	15.2 Chicken Creek	15.2 Rock Creek (South)	11.6 Highway 99W	9.3 Fanno Creek [b]	9.3 Durham WWTP [a]	8.7 Boones Ferry Road	7.5 Nyberg Creek	5.5 Stafford Road
discharge	○○○ ○○○○	○○○ ○○○○	◒◒◒ ◒◒◒◔	◒◒⊙ ○○○○	○○○ ○○○○	◕●● ●●○●	●●● ●●●●	○○○ ○○○○	⊙⊙○ ○○○○	○○○ ○○○○
water temperature	◒◒◒ ●●○○	◕◕◕ ◕◕◕●	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○⊕○	◕◕◕ ●●◒●	●●● ●●●●	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
chloride	○◒◒ ○○○○	◕◕◕ ◒◒◒◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○⊕○	◕◕◕ ◒◒◒◒	◕◕◕ ◒◒◕◒	◕◕◕ ◒◒◒◒	◒◒○ ○○○○	◕◕◕ ◒◒◒◒
dissolved solids	○◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○○○	◕◕◕ ◒◒◒◒	◕◕◕ ◒◒◒○	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
total suspended solids	○◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○○○	◕◕◕ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
ammonia	○◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○⊕○	◕◕◕ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
total kjeldahl nitrogen	○◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○○○	◕◕◕ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
nitrate plus nitrate	○◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○⊕○	◕◕◕ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
total phosphorus	○◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○⊕○	◕◕◕ ◒◒◒◒	◕◕◕ ◒◒●●	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
soluble orthophosphate	○◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○⊕○	◕◕◕ ◒◒◒◒	◕◕◕ ◒◒◔⊙	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
chlorophyll-*a*	○◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○⊕○	◕◕◕ ◒◒◒◒	○○○ ○○○○	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒
primary productivity	○○○ ○○○○	◒◒◒ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	◒◒◒ ○○○○
water-column light extinction [c]	○○○ ○○○○	◒◒○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	◒◒○ ○○○○
algal settling velocity	○○○ ○○○○	◔◒◒ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	◔◒◒ ○○○○
zooplankton abundance [d]	○○○ ○○○○	◔◒◔ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	○○○ ○○○○	◔◒◔ ○○○○
dissolved oxygen	◒◒◒ ○○○○	◕◕◕ ◕◕◕◒	◒◒◒ ◒◒◒◒	◒◒⊙ ○○○○	◕◕◒ ○○⊕○	◕◕◕ ◒◒◒◒	●●● ●●●●	◕◕◕ ◕◕◕◒	◒◒○ ○○○○	◕◕◕ ◕◕◕◒

[a] Daily data for some constituents in WWTP effluent were available for the 1991 through 1995 period but were not used for reasons of data quality.
[b] Fanno Creek at Durham Road, except in 1996 when samples were collected at a site on Fanno Creek in Durham City Park.
[c] Light extinction measurements were collected in 1993 but were not used in this study.
[d] Few samples collected in either May or October.

others (1999). For this study, one additional modification was made to the model.

In the USGS version of CE-QUAL-W2, a fraction of the phosphorus liberated during the decomposition of sedimentary organic matter was sequestered in the sediments to simulate the strong sorption of phosphorus to mineral surfaces in the sediment under oxic conditions (as well as other less-important processes). Such sequestration was permanent in the original USGS model. In this study, that algorithm was modified to allow the sorbed phosphorus to be released to the overlying water column at a certain rate if the water at the sediment interface becomes anoxic. This new process allows the model to simulate the buildup of phosphorus concentrations in the deeper pools of the river during stratified conditions after the hypolimnion becomes anoxic. The rationale behind this algorithm is that most of the sequestered phosphorus is sorbed to iron oxide surfaces, and that phosphorus will be liberated when those iron oxides dissolve under anoxic conditions. This process affects the amount of orthophosphate available for algal growth downstream of reaches affected by anoxia at the sediment interface. The additional algorithm affects only a few sites in the most downstream part of the model reach and takes effect for only limited times during the warmest part of the summer; nevertheless, it is an important change. The effects of this new algorithm on phosphorus concentrations and algal growth are discussed later in this report.

Model Parameterization

All of the model parameter values used in this study are listed in table 3. For the most part, the values of the parameters used to calibrate the model for the 1991–1993 period were not changed. A few important changes, however, were made. First, the ammonia nitrification rate was increased from 0.023 day^{-1} to 0.11 day^{-1}, based on an analysis of data collected at several sites in the Tualatin River downstream of the Rock Creek WWTP during August of 1995 when that WWTP was releasing abnormally large ammonia loads. This rate is important only for those infrequent periods when the instream ammonia concentration is large, say > 0.2 mg N/L (milligrams nitrogen per liter). These new data on the nitrification rate were obtained after the original calibration had been completed. Second, two new parameters were introduced to describe the release of sequestered phosphorus from the sediments during anoxic conditions; these are the initial concentration of recoverable phosphorus in the sediments (in g P/m^2, grams phosphorus per square meter) and the release rate of that phosphorus under anoxic conditions (in day^{-1}, see table 3). Both of these new parameters are calibration parameters and were set by calibrating to the measured phosphorus concentrations. Finally, the light- and nutrient-saturated algal growth rate was varied as a step function as in the original study, but the range was expanded slightly from 4.5–6.0 day^{-1} in the original work to 4.0–6.5 day^{-1} in this study. This expanded range was needed to account for the expanded range of measured conditions in this larger data set. More details on the algal growth rate are given later in the report. The zooplankton mortality rate, as in the original study, was held constant during each summer but allowed to vary from year to year. The range of this mortality rate remained unchanged; the values used for each summer are listed in table 4.

The only other change made to the model was that more representative initial concentrations were used for many of the modeled constituents. Generally, this is not important, as the initial conditions for the water-column constituents are swept out of the model grid within a few days of simulated time. Nevertheless, better initial conditions helped improve the simulation of measured conditions early in May for each of the simulated summers.

MODEL RESULTS

Fit Statistics

The ability of the model to simulate measured conditions was tested with four goodness-of-fit statistics: the root mean squared error (RMSE), the coefficient of determination (r^2), the mean absolute error (MAE), and the mean of the relative absolute error (MRAE). The RMSE is defined as the square root of the mean of the squared difference between measured and simulated values. As such, the RMSE is similar to a standard deviation of the error; roughly two-thirds of the errors are expected to fall within +/- 1 RMSE. RMSE values have the units of the quantity of interest, and lower values indicate a better fit. For the statistic to be relevant, however, one must know the range of the fitted data to determine whether an RMSE

Table 3. Values of model parameters

[Symbol is the representation used by Rounds and others in the original Tualatin River modeling report (1999). Abbreviations: chl-*a*, chlorophyll-*a*; OM, organic matter; cv, calibration value; lv, literature value; mv, measured value; m, meter; mg, milligram; g, gram; L, liter; W, watt; —, no symbol]

Symbol	Model Parameter	Type	Value
	Parameters affecting phytoplankton		
K_{ag}	maximum light- and nutrient-saturated algal growth rate at 20˚C — see footnote a	cv	4.0–6.5 day^{-1}
K_{am}	maximum algal nonpredatory mortality rate	lv	0.0 day^{-1}
K_{ae}	maximum algal excretion rate	cv	0.15 day^{-1}
K_{ar}	maximum algal respiration rate	cv	0.15 day^{-1}
h_N	Michaelis-Menten half-saturation constant for nitrogen limitation to algal growth	lv	0.008 mg/L
h_P	Michaelis-Menten half-saturation constant for phosphorus limitation to algal growth	mv	0.005 mg/L
I_S	saturating light intensity for algal photosynthesis	mv	177 W/m^2
α_w	baseline light extinction coefficient	mv	1.00 m^{-1}
α_{ss}	light extinction due to inorganic suspended solids	mv	0.043 L/mg/m
α_a	light extinction due to phytoplankton	mv	0.13 L/mg/m
β	fraction of incident light absorbed at water surface	mv	0.53
ω_a	algal settling velocity at 20˚C	mv	0.5 m/day
θ_a	temperature-adjustment coefficient for algal processes	lv	1.072
$\sigma_{C:chla}$	ratio of carbon to chlorophyll-*a* in algal biomass	mv	25 mg C / mg chl-*a*
	Parameters affecting zooplankton		
K_{zg}	maximum zooplankton grazing rate	lv	1.8 day^{-1}
K_{zm}	maximum zooplankton mortality rate — see footnote b	cv	0.05–0.4 day^{-1}
K_{zr}	maximum zooplankton respiration rate	lv	0.1 day^{-1}
p_a	preference for algae as food	lv	1.0
p_{dt}	preference for detritus as food	lv	0.16
θ_z	temperature adjustment coefficient for zooplankton processes	lv	1.072
e_{zg}	efficiency of zooplankton grazing	lv	0.5
μ_z	threshold food concentration for zooplankton grazing	lv	0.02 mg/L
h_{zg}	half-saturation constant for zooplankton grazing	lv	0.2 mg/L
	Parameters affecting ammonia nitrification and sedimentary phosphorus		
K_{NH_3}	maximum ammonia nitrification rate	mv	0.11 day^{-1}
θ_{NH_3}	temperature adjustment coefficient for nitrification	lv	1.047
f_P	fraction of sediment P that is unrecoverable under oxic conditions	cv	0.9
—	initial concentration of recoverable P in sediments	cv	6.0 g P/m^2
—	sediment P release rate under anoxic conditions	cv	0.02 day^{-1}

Table 3. Values of model parameters —Continued

[Symbol is the representation used by Rounds and others in the original Tualatin River modeling report (1999). Abbreviations: chl-*a*, chlorophyll-*a*; OM, organic matter; cv, calibration value; lv, literature value; mv, measured value; m, meter; mg, milligram; g, gram; L, liter; W, watt; —, no symbol]

Symbol	Model Parameter	Type	Value
	Parameters affecting dissolved and particulate organic matter		
K_{lom}	maximum labile decay rate	lv	0.5 day^{-1}
K_{dt}	maximum detritus decay rate	mv	0.046 day^{-1}
K_s	maximum sediment decay rate	mv	0.0005 day^{-1}
θ_{lom}	temperature adjustment coefficient for labile decay	lv	1.065
θ_{dt}	temperature adjustment coefficient for detritus decay	lv	1.065
θ_s	temperature adjustment coefficient for sediment decay	lv	1.065
φ_s^0	initial concentration of sediment compartment	mv	2570 g OM/m^2
ω_{dt}	detrital settling velocity	cv	0.0 m/day
	Stoichiometric coefficients		
δ_{NH_3}	oxygen stoichiometric coefficient for nitrification	lv	4.33 mg O_2 / mg N
δ_{dt}	oxygen stoichiometric coefficient for detritus decay	lv	1.4 mg O_2 / mg OM
δ_s	oxygen stoichiometric coefficient for bottom sediment decay	lv	1.4 mg O_2 / mg OM
δ_{lom}	oxygen stoichiometric coefficient for dissolved OM decay	lv	1.4 mg O_2 / mg OM
δ_{ag}	oxygen stoichiometric coefficient for photosynthesis	lv	1.4 mg O_2 / mg biomass
δ_{ar}	oxygen stoichiometric coefficient for algal respiration	lv	1.1 mg O_2 / mg biomass
δ_{zr}	oxygen stoichiometric coefficient for zooplankton respiration	lv	1.1 mg O_2 / mg biomass
δ_C	stoichiometric coefficient for carbon in OM (dry weight)	lv	0.5 mg C / mg OM
δ_P	stoichiometric coefficient for phosphorus in OM (dry weight)	lv	0.011 mg P / mg OM
δ_N	stoichiometric coefficient for nitrogen in OM (dry weight)	lv	0.08 mg N / mg OM
	Miscellaneous parameters		
—	dissolved oxygen limit	lv	0.2 mg/L
—	longitudinal eddy viscosity	lv	1.0 m^2/sec
—	longitudinal eddy diffusivity	lv	2.5 m^2/sec
n	Manning's n (roughness coefficient)	lv	0.03
—	wind sheltering coefficient	lv	0.9

[a] The maximum algal growth rate, K_{ag}, was varied seasonally to simulate adjustments of the algal community to changes in flow and light conditions. Figure 29 illustrates the variation in K_{ag} used by the model.

[b] The maximum zooplankton mortality rate, K_{zm}, was held constant during each season but was varied from year to year. Table 4 lists the annual variation in K_{zm} used by the model.

indicates an excellent or poor fit. The coefficient of determination is defined as for linear regression methods (Miller and Miller, 1988). The coefficient of determination is the ratio of the explained variation to the total variation and therefore can be a good measure of how well the model fits the data. A value of 1.0 is a perfect fit. A low value for this coefficient is caused by either a poor fit of the model to the data or a small range in the fitted data. The latter may simply indicate that the tested constituent is not important, as is the case for ammonia during 1992. Similarly, the coefficient of determination tends to be higher when the range of

Table 4. Annual variation in the maximum zooplankton mortality rate

Year	Zooplankton mortality rate (day^{-1})
1991	0.05
1992	.4
1993	.2
1994	.2
1995	.4
1996	.2
1997	.4

the fitted data is large, as is the case for ammonia in 1995 and 1996.

The MAE is the mean of the absolute value of the difference between measured and simulated values. As such, the MAE is closely related to the RMSE, and is a measure of the general amount of prediction error expected for any one measurement. The MRAE is the mean of the absolute value of the relative errors. For some quantities, the relative error provides a good measure of the ability of the model to fit the data; for others, the information in this statistic is clouded by the range of the data or the choice of the units of measure. For example, the MRAE for water level elevation is always low because the river's elevation is near 100 feet above sea level—so, the calculation of relative error involves division by a number around 100. Similarly, large relative errors can be calculated due to low values in concentration, regardless of whether the absolute error at that point is significant or whether the model fit is good. None of these four statistics provides a perfect measure of the goodness-of-fit, but all provide some quantification of model performance that is useful.

Fit statistics were calculated for river discharge, pool elevation, water temperature, and eight water quality constituents at several sites. The statistics are shown in tables 5 (RMSE, r^2) and 6 (MAE, MRAE) and are referenced in the discussions that follow.

River Discharge and Pool Depth

To accurately simulate the water quality of the Tualatin River, it is important first to accurately simulate the flow and residence time of the river. If the simulated flow and/or travel time are in error, subsequent simulations of mass loadings and the effects of chemical and biological reactions will contain a component of that error. It is imperative, therefore, to calibrate the simulated discharge and pool elevation to their measured values. As in the original study, streamflow was fitted to the measured data by running the model without a nonpoint source of water, then comparing the simulated streamflow at the downstream boundary (RM 3.4) with measured streamflow at the nearest gaging station (RM 1.8). No significant sources or sinks of water exist between these two locations. The difference between the two time series was smoothed and added back into the model as a nonpoint source of water. Although this may seem like a forced fit, it was the best method of estimating the amount of water from ground water and small ungaged tributaries. After adding in the nonpoint source of water, the pool elevation was calibrated by adjusting the effective width of the Oswego diversion dam (RM 3.4) according to the number of flash boards installed on that structure. The results are shown in figures 4 and 5 for river discharge and pool elevation, respectively.

The modeled discharges and elevations match the measured quantities well. For the low-flow period (discharge < 300 ft^3/s), the RMSE indicates that simulated discharges at RM 3.4 were within about 13 ft^3/s of measured flows (< 5% MRAE, tables 5 and 6). Simulated pool elevations at RM 6.7 were within about 0.2 ft of the measured elevation. Obviously, the fit for the discharge had to be good due to the method used to estimate the nonpoint source. The nonpoint source typically is a small fraction of the total flow, however, and the fit still would have been acceptable without it. No pool elevation data were available at RM 6.7 after late September of 1996 due to the construction of a new headgate structure at that site. Elevation data were estimated after that date based on a correlation between elevation data at RMs 6.7 and 3.4 (r = 0.999, from data collected during 1996). Previous tests by Rounds and others (1999) demonstrated that when the discharge and pool elevation are calibrated accurately, the simulated travel times in the reservoir reach also are accurate.

Figures 3-5 show several flow characteristics that have a significant effect on water quality. The summers of 1992 and 1994 were characterized by lower flows than most of the other years. In particular, the flows during May of 1992 and 1994 were low enough, and therefore the residence time was long enough, to produce algal blooms as early as the end of May. In

Table 5. Goodness-of-fit statistics for the model results.

[Root mean square errors are in the units indicated for the property or constituent. The coefficient of determination is dimensionless. Abbreviations: ˚C, degrees Celsius; mg/L, milligrams per liter; µg/L, micrograms per liter; ft^3/s, cubic feet per second]

Property or Constituent	Units	Root Mean Square Error (RMSE)								Coefficient of Determination (r^2)							
		1991	1992	1993	1994	1995	1996	1997	All Years	1991	1992	1993	1994	1995	1996	1997	All Years
Tualatin River at Elsner Road (river mile 16.2)																	
water temperature	˚C	0.70	0.75	0.82	0.47	1.10	0.89	0.42	0.77	0.98	0.97	0.95	0.99	0.97	0.97	0.99	0.97
vertical temperature range	˚C	.29	.29	.24	.20	.30	.23	.16	.25	.47	.78	.78	.75	.72	.68	.37	.65
chloride	mg/L	1.4	1.7	1.6	1.8	1.4	.8	1.1	1.5	.79	.70	.86	.89	.91	.97	.94	.88
dissolved solids	mg/L	18	14	15	15	10	9.5	10	13	.51	.65	.67	.71	.93	.91	.87	.81
ammonia	mg/L as N	.039	.048	.069	.131	.074	.121	.031	.082	.49	.18	.69	.67	.94	.98	.60	.85
nitrate plus nitrite	mg/L as N	.29	.56	.31	.44	.32	.23	.17	.35	.72	.72	.79	.52	.80	.89	.61	.75
total phosphorus	mg/L as P	.039	.035	.025	.011	.016	.028	.014	.026	.32	.08	.39	.63	.52	.38	.61	.42
orthophosphate	mg/L as P	.015	.022	.013	.018	.014	.020	.013	.017	.39	.50	.59	.62	.67	.29	.67	.56
chlorophyll-*a*	µg/L	13	14	11	14	14	12	7.1	12	.74	.81	.84	.76	.81	.88	.94	.81
dissolved oxygen	mg/L	1.23	1.03	1.12	1.39	1.11	.97	1.09	1.14	.56	.86	.42	.57	.71	.75	.49	.67
Tualatin River at Stafford Road (river mile 5.5)																	
water temperature	˚C	.69	.80	.63	.55	.87	.85	.55	.72	.98	.97	.97	.99	.98	.98	.99	.98
vertical temperature range	˚C	.65	.52	.44	.55	.98	.62	.15	.60	.79	.89	.89	.87	.79	.57	.88	.79
chloride	mg/L	1.9	2.0	1.2	2.6	1.7	1.2	1.2	1.8	.78	.83	.96	.77	.93	.96	.96	.92
dissolved solids	mg/L	17	16	11	29	16	11	12	17	.69	.82	.91	.42	.87	.94	.91	.85
ammonia	mg/L as N	.165	.108	.299	.069	.063	.052	.042	.142	.82	.39	.91	.65	.82	.87	.67	.92
nitrate plus nitrite	mg/L as N	.62	.74	.49	.39	.42	.20	.22	.48	.70	.77	.85	.49	.79	.95	.81	.86
total phosphorus	mg/L as P	.045	.030	.050	.030	.016	.021	.016	.032	.68	.18	.71	.64	.44	.68	.62	.77
orthophosphate	mg/L as P	.030	.025	.037	.018	.015	.016	.017	.024	.47	.31	.76	.51	.74	.64	.62	.77
chlorophyll-*a*	µg/L	32	20	22	20	18	18	20	22	.53	.58	.67	.56	.74	.92	.88	.63
dissolved oxygen	mg/L	1.92	1.72	1.83	1.40	1.22	.95	.85	1.47	.59	.74	.61	.70	.68	.84	.80	.67
Tualatin River at Oswego Canal (river mile 6.7)																	
water surface elevation	feet	.07	.11	.14	.15	.12	.37	.73	.32	.98	.97	.97	.95	.94	.99	.86	.92
Tualatin River at Oswego diversion dam (river mile 3.4)																	
discharge (≤ 300)	ft^3/s	11	12	8.4	8.1	18	12	24	14	.98	.97	.99	.98	.95	.94	.90	.97
water temperature	˚C	.77	.76	.80	.68	.97	.99	.55	.80	.98	.97	.96	.99	.98	.98	.99	.97
dissolved oxygen	mg/L	2.67	3.11	3.03	2.16	1.68	1.37	1.02	2.28	.50	.56	.59	.53	.62	.76	.83	.55

Table 6. Additional goodness-of-fit statistics for the model results.

[Mean absolute errors are in the units indicated for the property or constituent. Mean relative absolute error is expressed as a percent, and is meaningful only relative to the units indicated for the property or constituent. Abbreviations: ˚C, degrees Celsius; mg/L, milligrams per liter; µg/L, micrograms per liter; ft^3/s, cubic feet per second; —, not available]

Property or Constituent	Units	Mean Absolute Error (MAE)								Mean Relative Absolute Error (MRAE, percent)							
		1991	1992	1993	1994	1995	1996	1997	All Years	1991	1992	1993	1994	1995	1996	1997	All Years
Tualatin River at Elsner Road (river mile 16.2)																	
water temperature	˚C	0.53	0.60	0.63	0.38	0.85	0.69	0.31	0.57	3.2	3.5	4.0	2.2	4.9	4.4	1.9	3.5
vertical temperature range	˚C	.13	.15	.12	.14	.20	.16	.11	.15	—	—	—	—	—	—	—	—
chloride	mg/L	1.1	1.2	1.2	1.4	1.2	.6	1.0	1.1	11	10	15	13	11	8.5	11	11
dissolved solids	mg/L	14	11	10	13	8.8	7.7	9.1	11	10	8.5	9.3	9.5	6.5	6.0	8.1	8.3
ammonia	mg/L as N	.031	.025	.046	.072	.044	.061	.019	.043	159	61	129	109	78	85	56	97
nitrate plus nitrite	mg/L as N	.24	.47	.25	.34	.24	.18	.14	.26	16	32	20	39	17	12	13	21
total phosphorus	mg/L as P	.028	.023	.019	.009	.013	.018	.012	.018	24	19	17	10	16	14	12	16
orthophosphate	mg/L as P	.012	.016	.010	.015	.011	.016	.011	.013	33	49	26	45	43	46	37	40
chlorophyll-*a*	µg/L	10	10	6.9	11	9.8	7.0	4.7	8.4	53	46	36	39	53	46	32	44
dissolved oxygen	mg/L	.94	.82	.80	1.11	.83	.65	.81	.85	11	10	9.5	14	11	9.1	9.8	11
Tualatin River at Stafford Road (river mile 5.5)																	
water temperature	˚C	.56	.65	.49	.45	.64	.71	.45	.57	3.3	3.7	2.9	2.6	3.5	4.4	2.9	3.3
vertical temperature range	˚C	.44	.37	.27	.33	.56	.38	.10	.35	—	—	—	—	—	—	—	—
chloride	mg/L	1.2	1.4	1.0	1.8	1.3	1.0	1.0	1.2	9.2	9.2	7.8	12	8.9	9.8	9.1	9.5
dissolved solids	mg/L	14	13	7.8	16	14	8.7	9.6	12	9.2	7.8	5.6	11	8.5	6.0	7.7	7.9
ammonia	mg/L as N	.111	.056	.201	.040	.040	.036	.030	.073	55	87	64	97	90	94	66	79
nitrate plus nitrite	mg/L as N	.36	.58	.38	.30	.31	.15	.19	.32	15	20	23	22	16	8.0	13	17
total phosphorus	mg/L as P	.034	.023	.034	.016	.013	.015	.012	.021	30	24	20	14	15	13	11	18
orthophosphate	mg/L as P	.022	.018	.027	.013	.011	.013	.013	.017	39	74	43	57	58	46	50	52
chlorophyll-*a*	µg/L	22	15	16	14	13	12	11	15	157	80	62	34	55	51	37	68
dissolved oxygen	mg/L	1.32	1.33	1.46	1.09	.87	.69	.51	1.04	17	15	18	13	11	8.8	5.7	13
Tualatin River at Oswego Canal (river mile 6.7)																	
water surface elevation	feet	.04	.08	.07	.07	.08	.25	.53	.16	.04	.08	.07	.06	.08	.2	.5	.2
Tualatin River at Oswego diversion dam (river mile 3.4)																	
discharge (≤ 300)	ft^3/s	8.0	8.2	6.9	5.8	12	8.0	14	9.0	4.5	6.4	4.0	3.8	6.4	3.2	6.0	4.9
water temperature	˚C	.59	.60	.64	.54	.77	.81	.43	.63	3.5	3.4	3.8	3.0	4.3	5.0	2.6	3.6
dissolved oxygen	mg/L	1.81	2.40	2.16	1.69	1.26	.92	.70	1.56	24	26	29	17	14	12	8.2	19

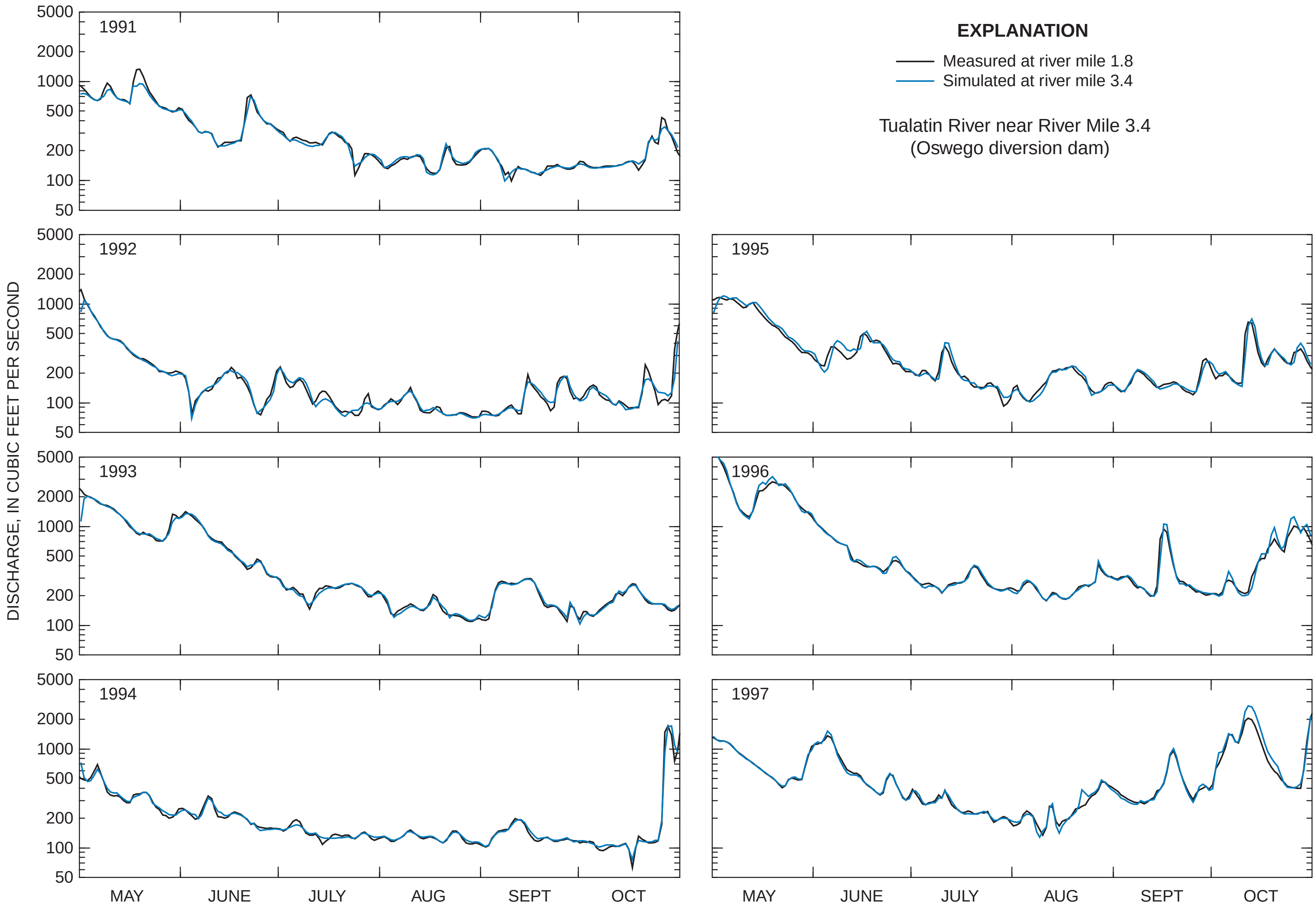


Figure 4. Calibrated Tualatin River discharge at river mile 3.4 (Oswego diversion dam) and measured discharge at river mile 1.8 (West Linn) for May-October of 1991–1997.

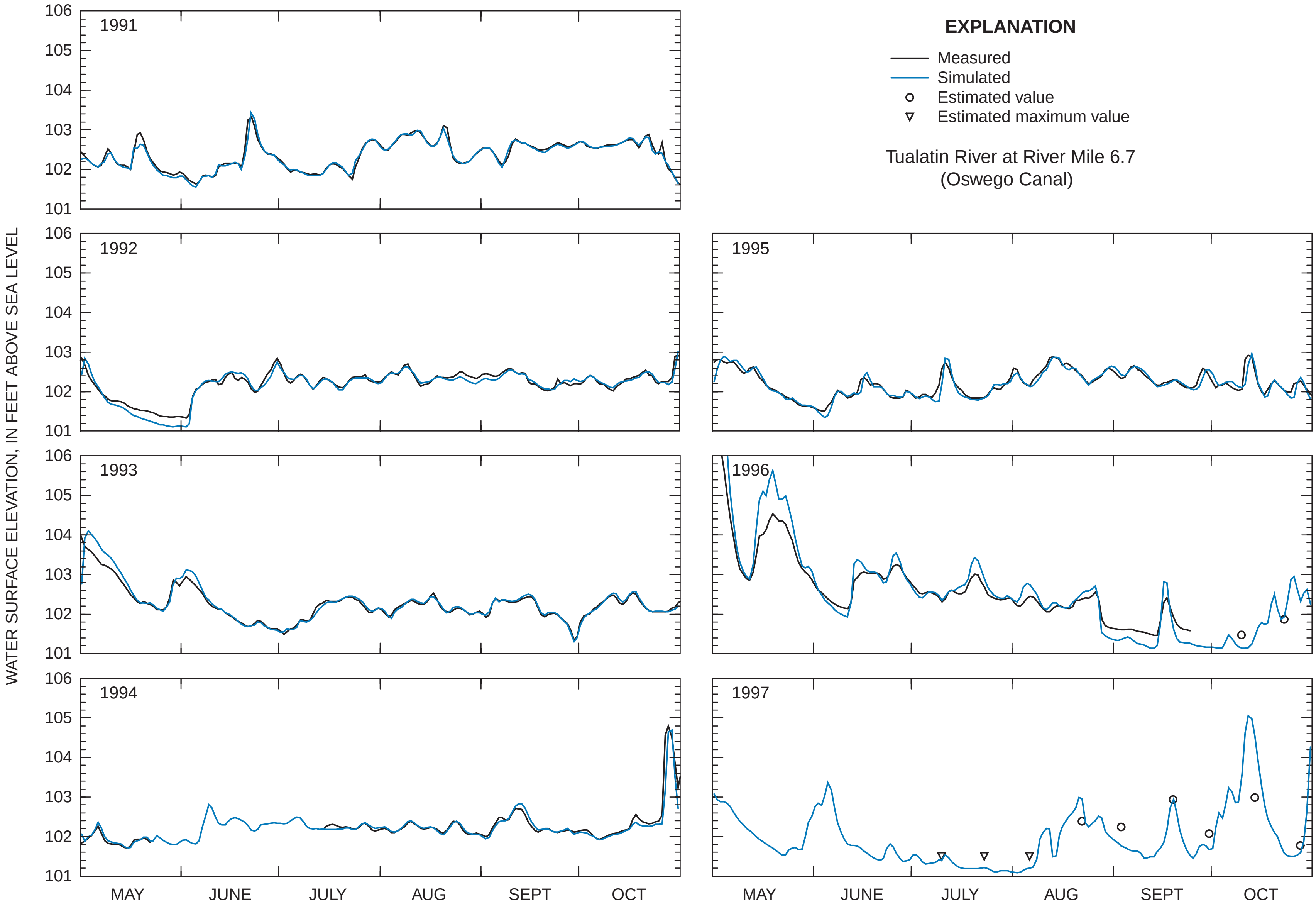


Figure 5. Calibrated and measured water-surface elevations of the Tualatin River at river mile 6.7 (Oswego Canal) for May-October of 1991–1997.

contrast, flows were high enough, and the residence time short enough, to prevent algal blooms during May and June of 1996 and 1997. Similarly high flows occurred during May of 1991 and 1993. Water quality during these high-flow periods, therefore, was not affected by algal blooms. In addition, when streamflow is less than 300 ft^3/s, discharge from the two large WWTPs (typically 25–30 ft^3/s each) comprises a significant fraction of the flow; the WWTPs can account for up to one-third of the river flow during such low-flow periods. As a result, if the WWTPs were to accidently release a large amount of ammonia, the effect on the river's quality would be important.

Water Temperature

Water temperature is an important factor in determining the solubility of oxygen as well as the rates of chemical and biological reactions. An error of 2˚C in the simulated water temperature, for example, can translate into an error of 10 to 15 percent in the rates of simulated chemical and biological reactions. Fortunately, water temperature is controlled by known physical processes and can be simulated accurately. Figures 6, 7, and 8 show the measured and simulated water temperatures at RMs 16.2 (Elsner Road), 5.5 (Stafford Road), and 3.4 (Oswego diversion dam), respectively. The measured temperatures at RMs 16.2 and 5.5 are means of all individual measurements at less than 10 feet depth, typically at 3, 6, and 9 feet; simulated temperatures are volume-weighted means from the top 10 feet of the model grid. At RM 3.4, the measured data are from a continuous monitor installed next to the fish ladder; the simulated temperatures are from the surface layer, which is the water that flows over the dam and through the fish ladder. The RMSEs for water temperature range from 0.42 to 1.10˚C (< 5% MRAE) and the r^2 values are all at least 0.95 for these sites, indicating an excellent fit to the data (tables 5 and 6). Interestingly, the two seasons with the best model fits to the water temperature data were 1994 and 1997; these summers are outside the original 1991–1993 calibration period, further demonstrating that the physics of heat transport are simulated well by the model. Although the model does have trouble simulating the water temperature at times (the simulated water temperature is too warm several times in 1995), it appears that the simulated reaction rates will not be biased significantly due to erroneous water temperatures.

Thermal Stratification

CE-QUAL-W2 simulates water quality in two dimensions: longitudinal (upstream-downstream) and vertical. The model can simulate a number of important depth-dependent processes such as light penetration (which greatly affects algal growth), as well as vertical variations in temperature and constituent concentrations. Thermal stratification, when it develops, can create or enhance vertical concentration gradients of constituents such as dissolved oxygen, phosphorus, and nitrogen. As a gross measure of the degree of thermal stratification of the river at RM 16.2 (Elsner Road), the difference between the measured temperatures at 3 and 12 feet depth was compared to the same quantity as simulated by the model (fig. 9). At RM 5.5 (Stafford Road), the temperature differences were calculated from 3 and 15 feet depths (fig. 10).

The Tualatin River at RM 16.2 does not thermally stratify for any appreciable length of time, while thermal stratification can persist at RM 5.5 for days or weeks at a time. The measured data in figures 9 and 10 illustrate this fact. Note in particular the periods of stratification at RM 5.5 during late-June through August of 1992, early August of 1993, and July of 1994. Because the turbulence associated with high flows can prevent extended periods of thermal stratification, the lack of significant stratification during 1997 is consistent with the higher flows during that summer. A comparison of measured and simulated temperature differences is complicated by the fact that the greatest differences are likely to occur in the late afternoon, especially when such differences do not persist overnight (typical at RM 16.2), and the measured data often were not obtained in the late afternoon. As a result, the simulated differences in figure 9 appear to be greater than the measured differences.

The model simulates the lack of thermal stratification at RM 16.2, and the periods of stratification at RM 5.5, very well (see tables 5 and 6 for the fit statistics), with only a few exceptions. The fit is surprisingly good (mean RMSE < 0.5˚C), given seven summers of varying hydrologic and climatic conditions and no difference in the way each summer's water temperature was simulated. This indicates that the model's simulation of light penetration and vertical mixing are very close to those occurring in the river. Simulating these processes accurately provides a good foundation for other dependent processes such as algal

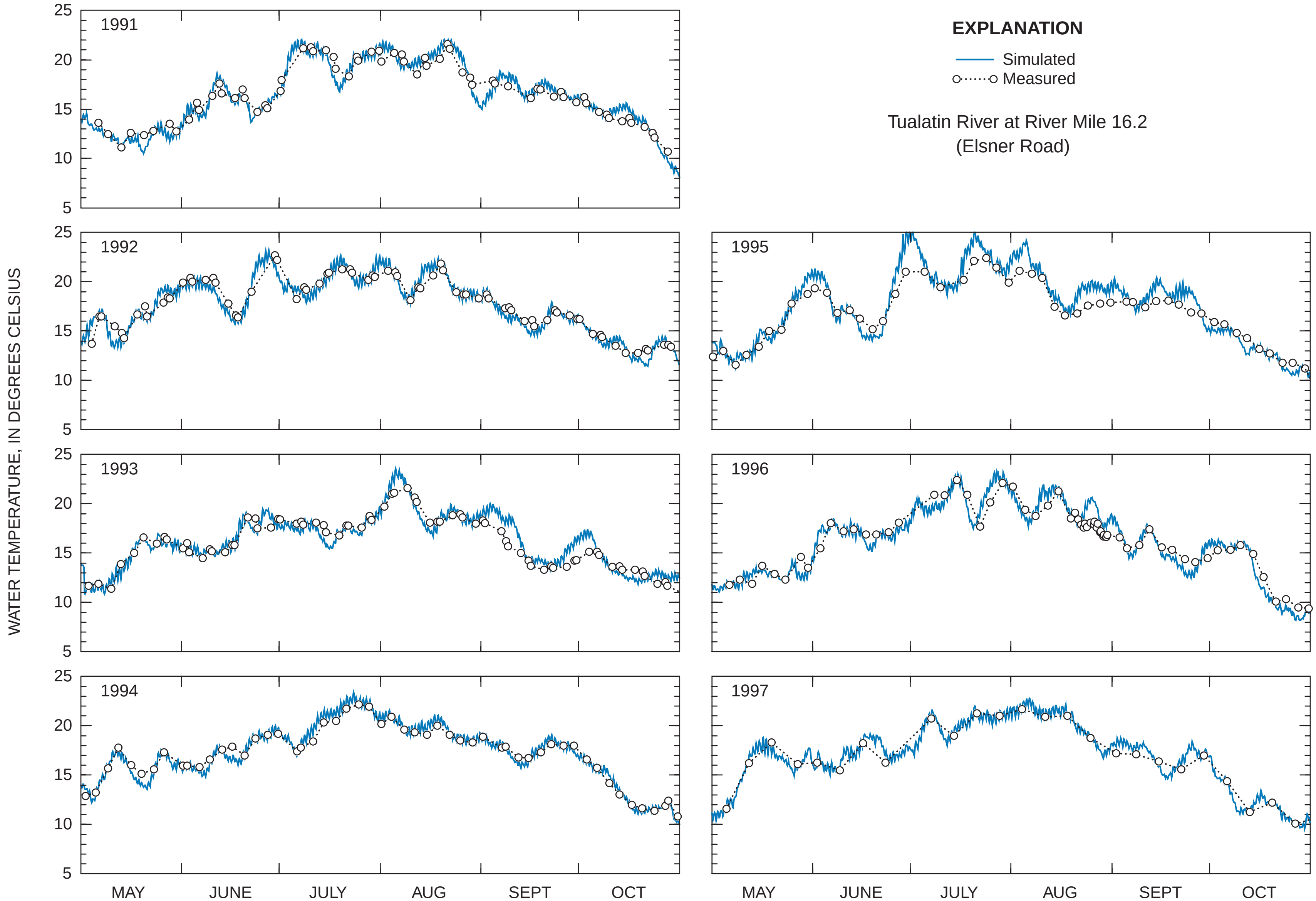


Figure 6. Simulated and measured water temperature at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured temperatures are the mean of all discrete measurements at less than 10 feet depth (typically 3, 6, and 9 feet). Simulated temperatures (every 4 hours) are volume-weighted means from the top 10 feet of the water column.)

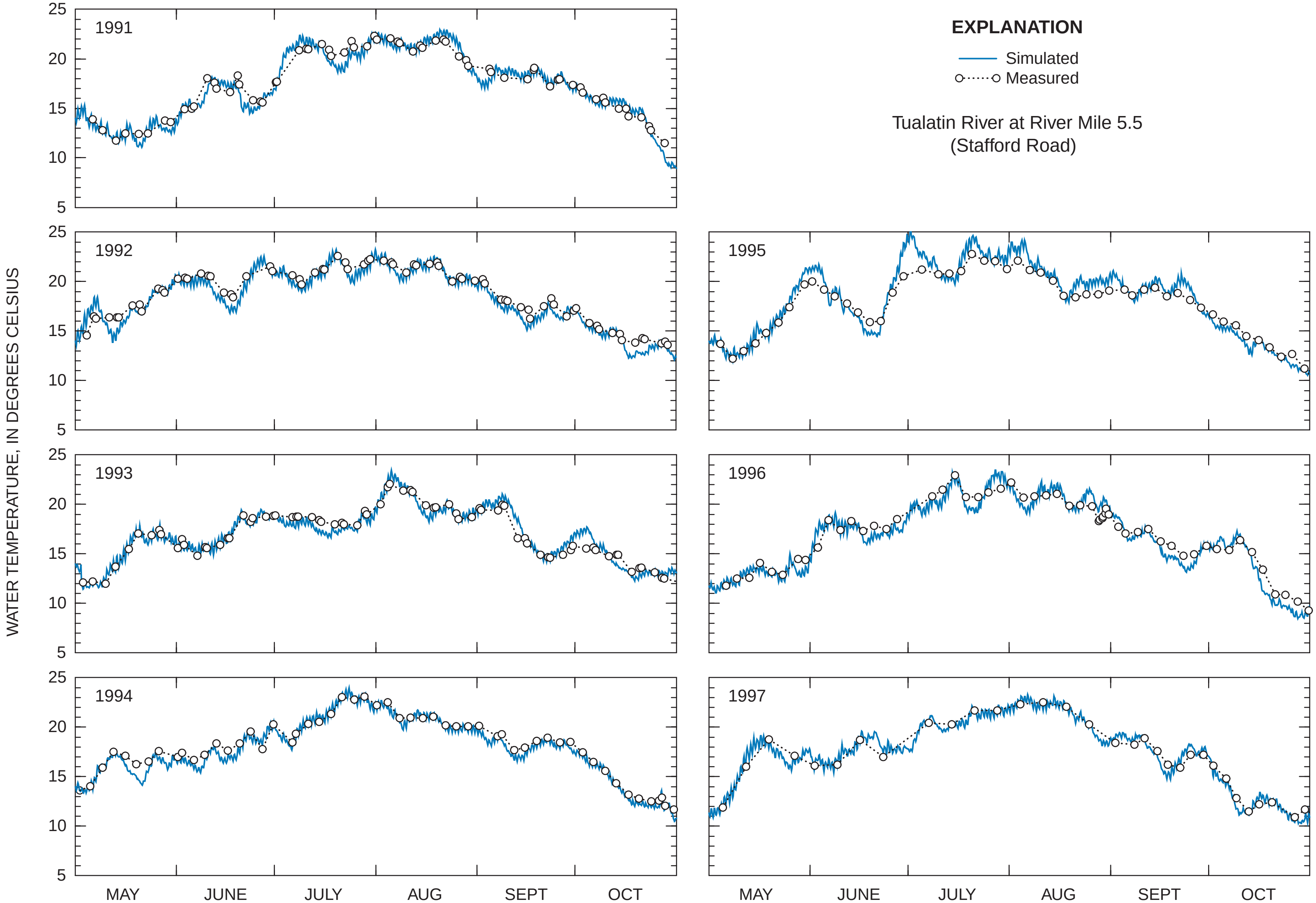


Figure 7. Simulated and measured water temperature at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured temperatures are the mean of all discrete measurements at less than 10 feet depth (typically 3, 6, and 9 feet). Simulated temperatures (every 4 hours) are volume-weighted means from the top 10 feet of the water column.)

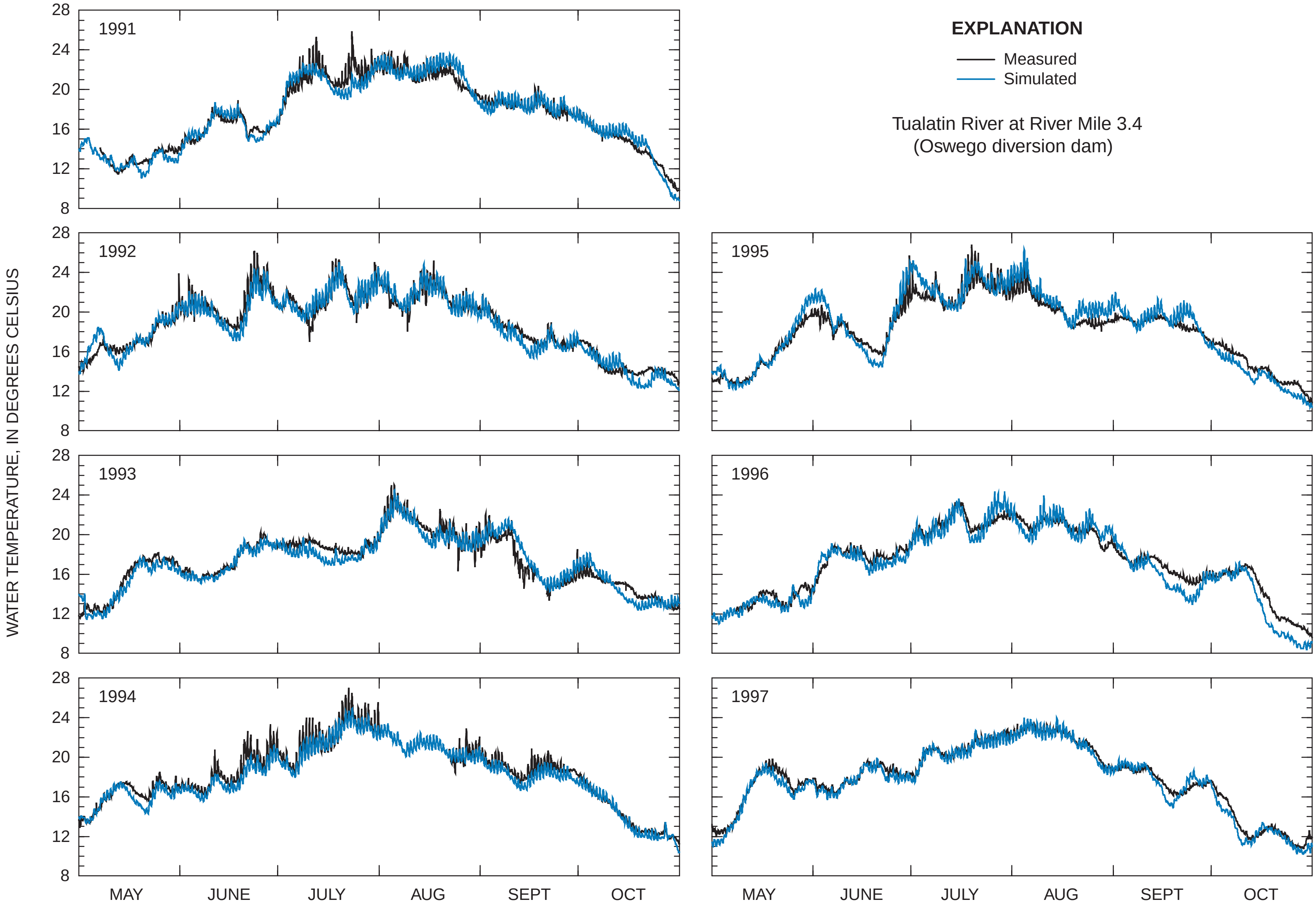


Figure 8. Simulated and measured hourly water temperature at river mile 3.4 (Oswego diversion dam) for May-October of 1991–1997.

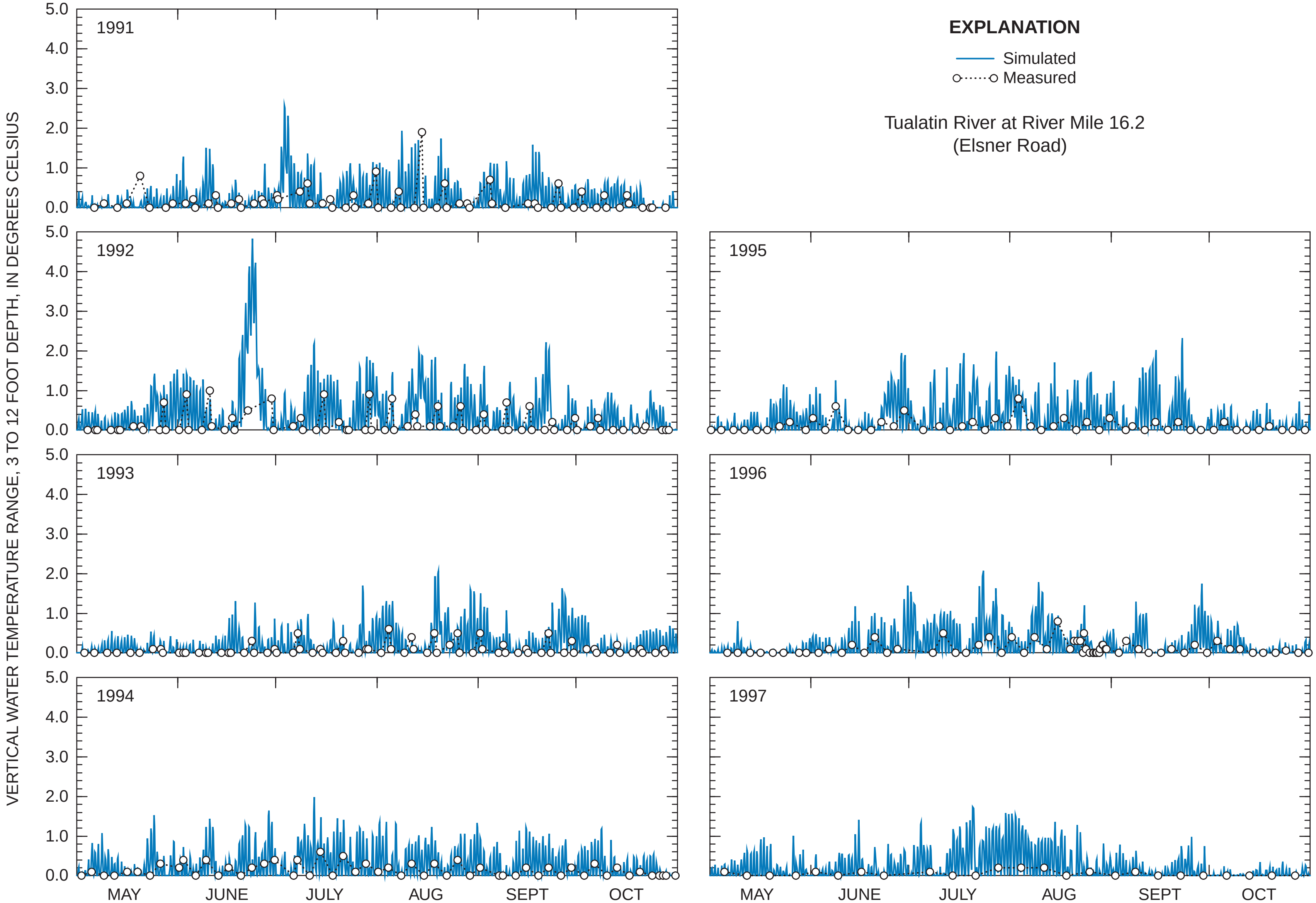


Figure 9. Simulated and measured differences between 3-foot and 12-foot water temperatures at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Simulated data are plotted at 4 hour intervals.)

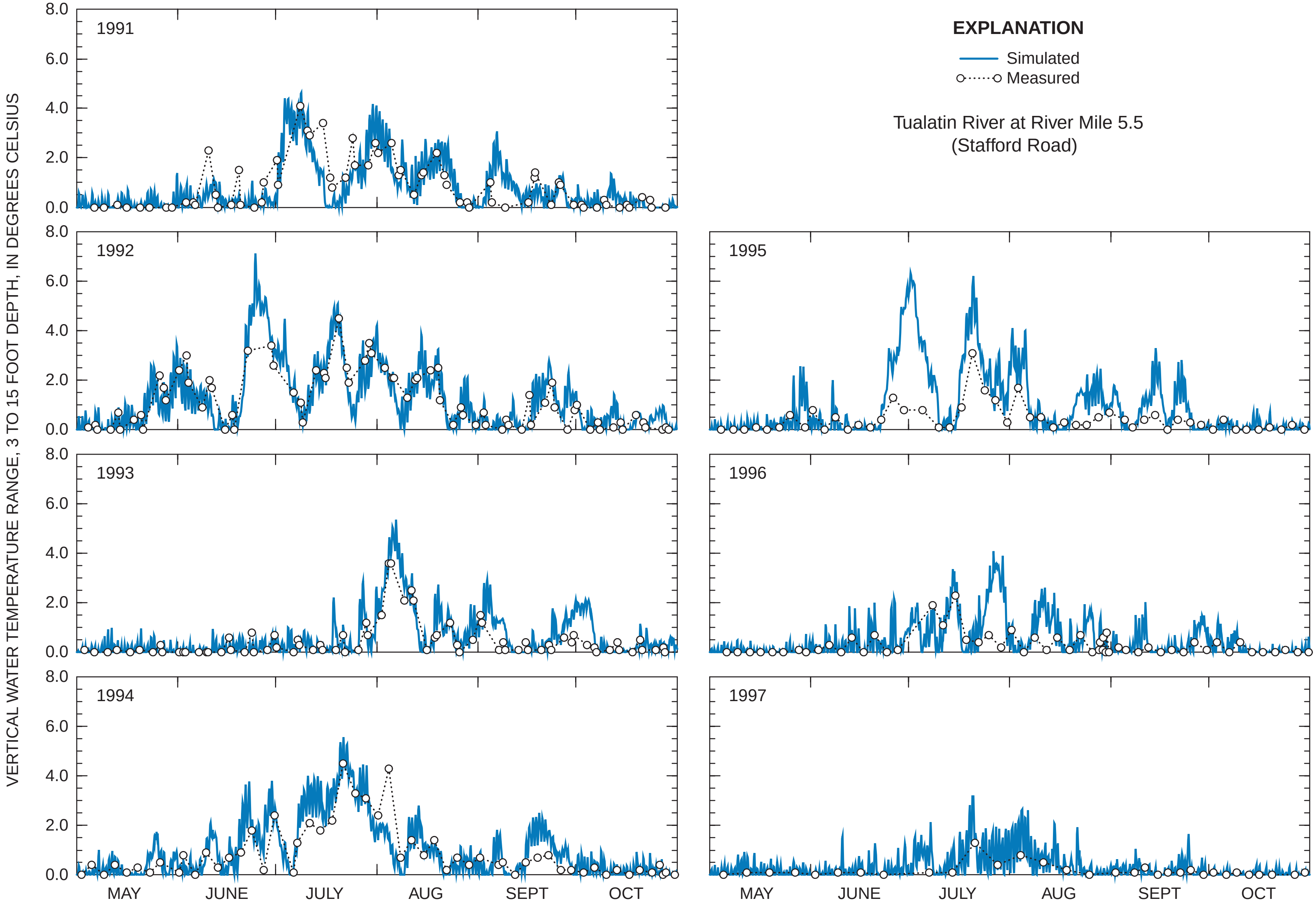


Figure 10. Simulated and measured differences between 3-foot and 15-foot water temperatures at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Simulated data are plotted at 4 hour intervals.)

growth, which is closely tied to light availability, and the development of vertical gradients of dissolved oxygen and nutrients during periods of stratification.

Conservative Tracers

The simulation of a conservative tracer, such as chloride or dissolved solids, provides a good diagnostic check for the model. In particular, the simulation of tracers is useful in determining whether a significant source or sink of water has been omitted or erroneously represented by the model. The longitudinal profile of mean chloride concentrations (a plot as a function of river mile) shows that chloride concentrations are controlled mostly by the loads of chloride in WWTP effluent (fig. 11); the Rock Creek and Durham WWTPs discharge to the river at RMs 38.1 and 9.3, respectively (fig. 1). This influence of the WWTPs on chloride concentrations was expected. The apparent "errors" associated with the measured chloride concentrations at RM 36.8 during 1991–1993 in figure 11 are actually errors in the sampling strategy that did not account for temporal variability (spatial aliasing), as discussed by Rounds and others (1999). The same is sometimes true for the measured data at RM 8.7. At RMs 16.2 (Elsner Road) and 5.5 (Stafford Road), a comparison of measured and simulated chloride concentrations shows very good agreement (figs. 12 and 13), with RMSEs near 1.5 mg/L or less and r^2 values greater than 0.70 and often greater than 0.90. These comparisons do not indicate the presence of any significant missing sources of chloride, and the simulated concentrations are within the expected analytical error of the measured data. Only in a couple of instances is any bias apparent, and that appears to be minor.

Similar conclusions may be drawn from a comparison of measured and simulated concentrations of dissolved solids (figs. 14, 15, and 16). As it was for chloride, WWTP effluent is the dominant source of dissolved solids to the river during the summer period. Again, the fit is good. No missing sources are indicated, and no significant or continuing bias is apparent. The accurate simulation of these conservative tracers shows that the advective and dispersive transport processes of the river are well represented by the model. The accurate simulation of such transport processes provides a necessary foundation for simulating the transport of constituents that are subject to chemical and biological reactions.

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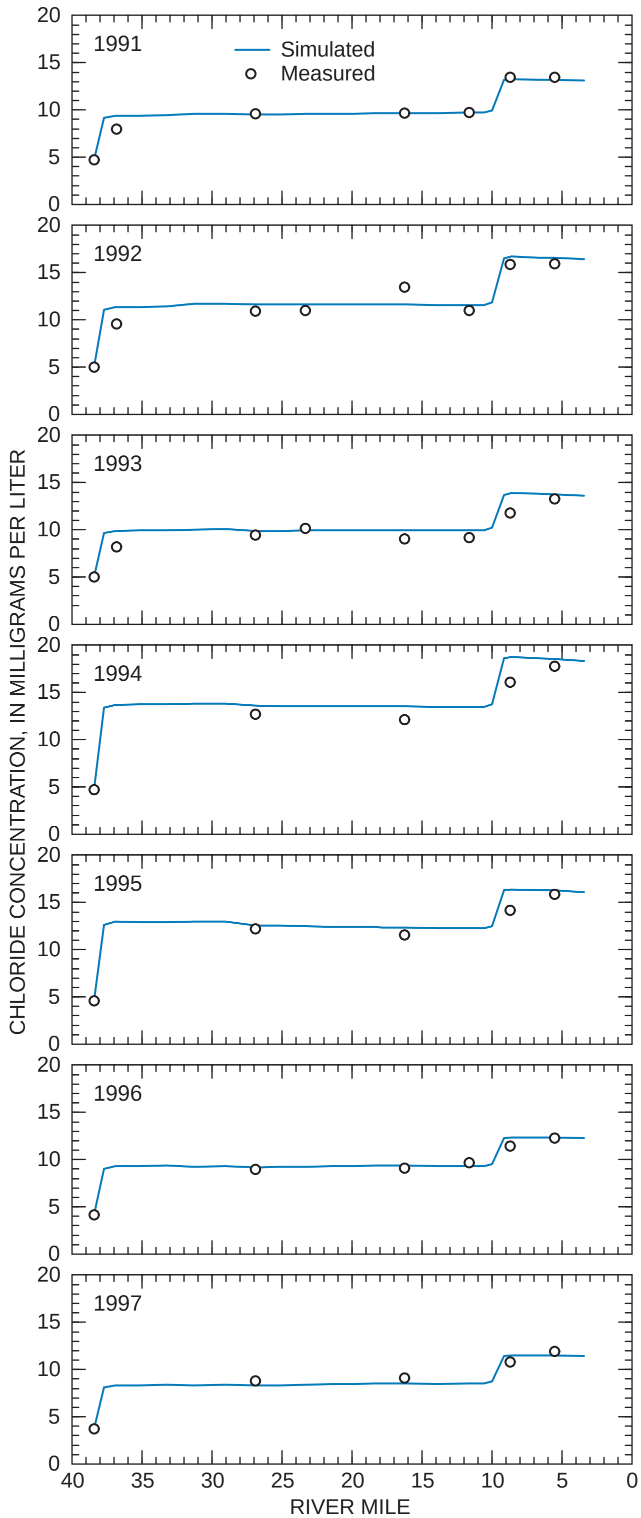

Figure 11. Simulated and measured mean chloride concentrations as a function of river mile for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations are volume-weighted means, also from the top 10 feet of the water column.)

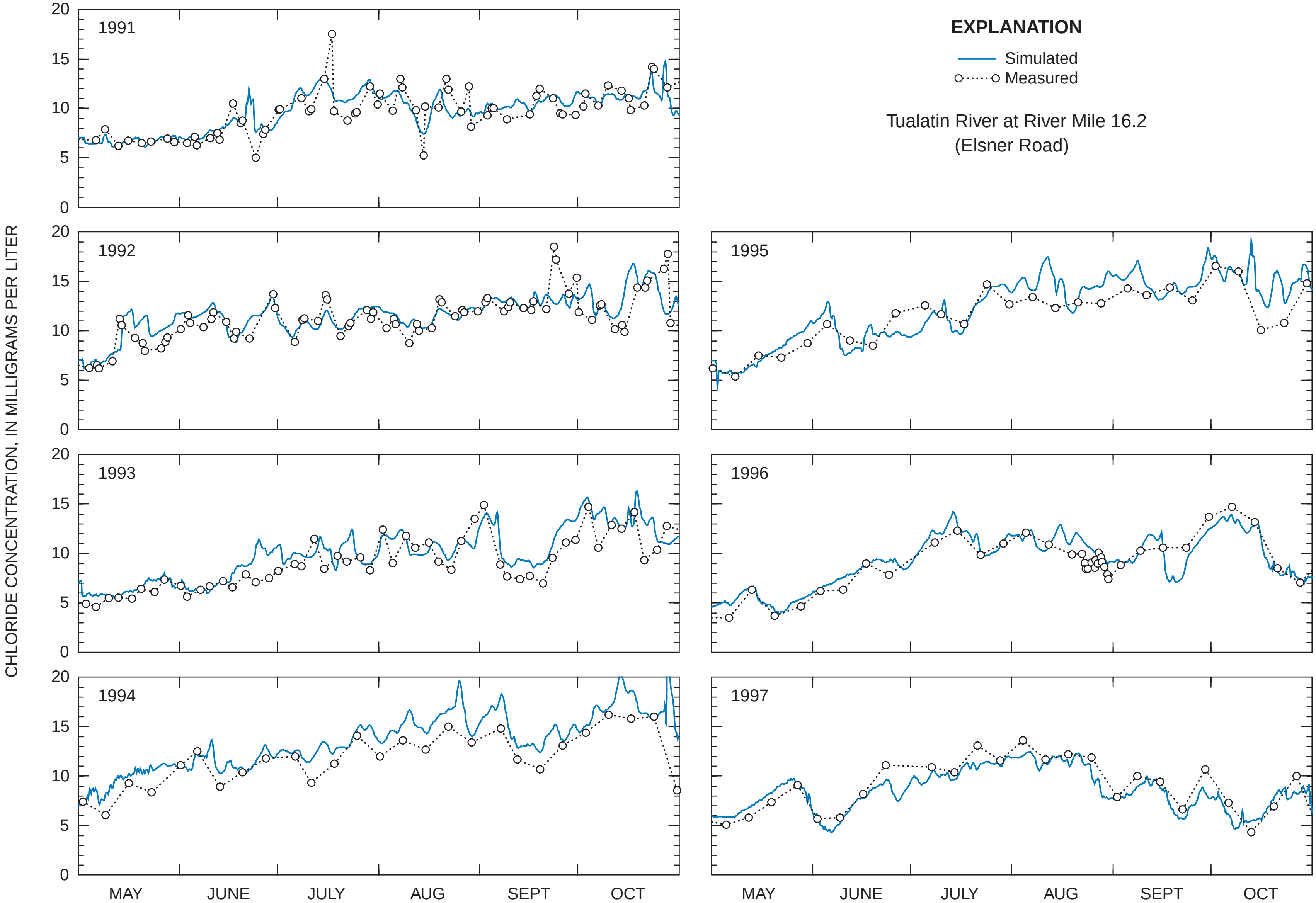

Figure 12. Simulated and measured chloride concentration at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

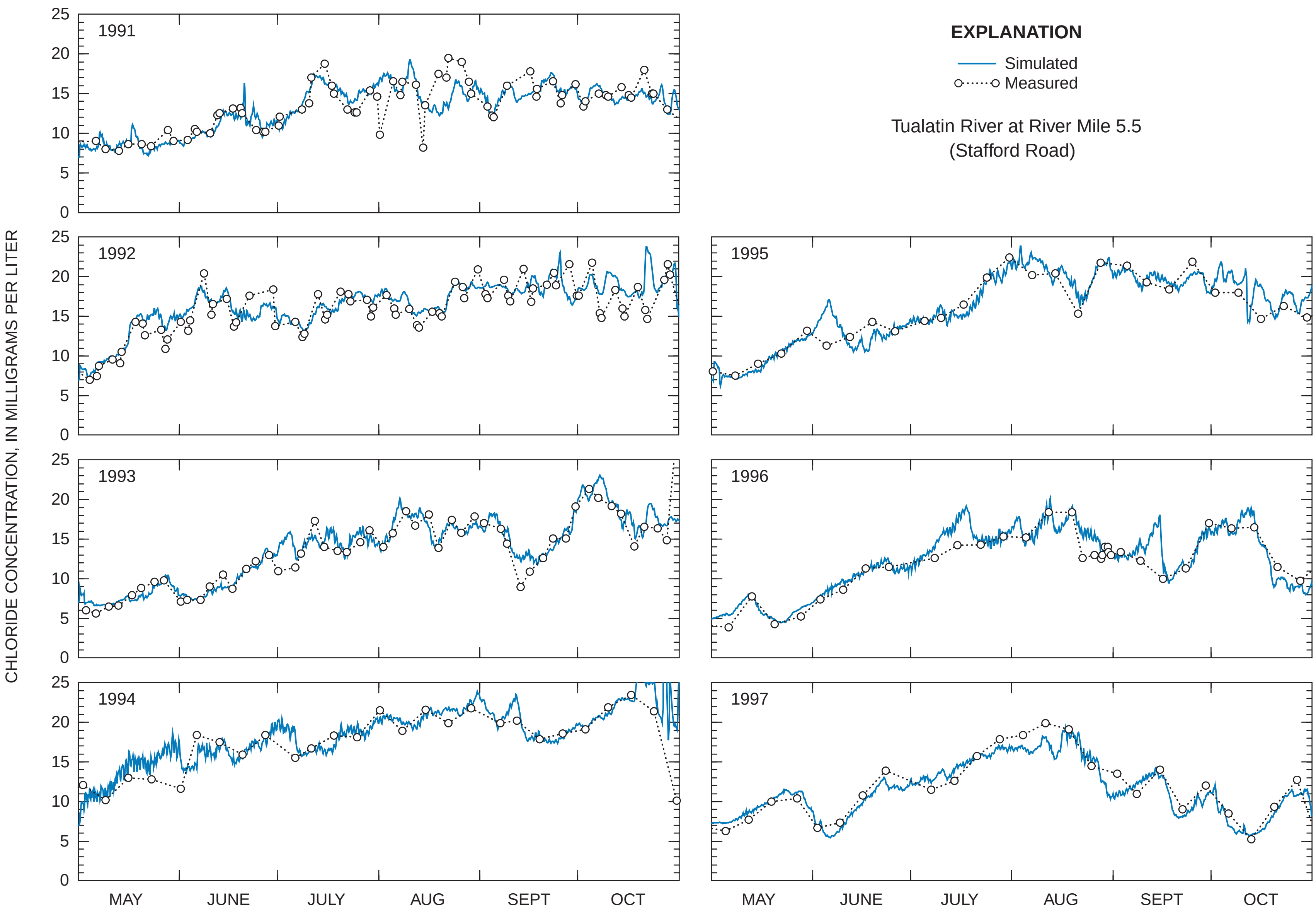

Figure 13. Simulated and measured chloride concentration at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

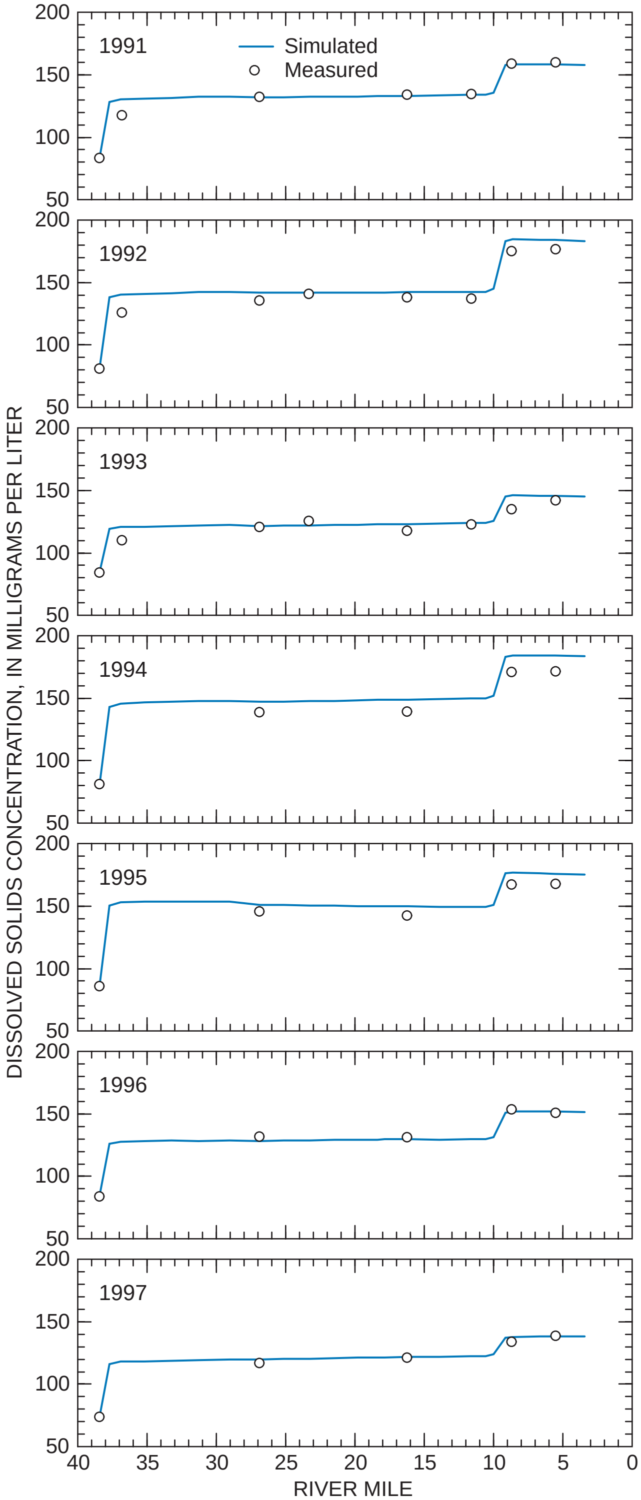

Figure 14. Simulated and measured mean dissolved solids concentrations as a function of river mile for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations are volume-weighted means, also from the top 10 feet of the water column.)

Nutrients, Algae, and Dissolved Oxygen

Many of the reactive constituents simulated by the model influence each other through chemical and biological reactions and therefore are difficult to discuss separately. In the Tualatin River, algal growth is a primary influence on the orthophosphate concentration; the reverse is also true. Algal growth is a secondary factor in determining ammonia concentrations, but a primary influence on dissolved oxygen concentrations. High ammonia concentrations can deleteriously affect dissolved oxygen concentrations through nitrification. In the sections that follow, the model's ability to simulate concentrations of ammonia, nitrate, total phosphorus, orthophosphate, algae (phytoplankton), and dissolved oxygen are analyzed. In each case, the model's ability to simulate one constituent relies to some degree on the model's ability to simulate another; such dependences are highlighted.

Ammonia

For most of the modeled period, ammonia concentrations were low enough that ammonia did not significantly affect water quality in the Tualatin River. This was the case for almost all of 1992, 1994, and 1997, about half of 1991 and 1993, and most of 1995 and 1996. If ammonia concentrations were high during the summer, however, they tended to be very important because ammonia nitrification, the oxidative conversion of ammonia to nitrate, can quickly drive the dissolved oxygen concentration down to dangerously low levels. This is the reason that ammonia is regulated with a TMDL. The largest influence on measured ammonia concentrations is WWTP performance with regard to in-plant nitrification. When WWTP nitrification was optimal, instream ammonia concentrations were low or insignificant. When instream ammonia concentrations became high (> 0.2 mg N/L), the largest contributor more often than not was one or both of the WWTPs. Algae are a secondary influence on the ammonia concentration. Ammonia is a preferred source of nitrogen for algal growth, but this preference for ammonia nitrogen only becomes apparent during large algal blooms.

The longitudinal profile of mean measured and simulated ammonia concentrations illustrates the importance of the WWTPs on this constituent (fig. 17). During 1992 and 1997, WWTP removal of ammonia was optimal for both plants, with effluent concentra-

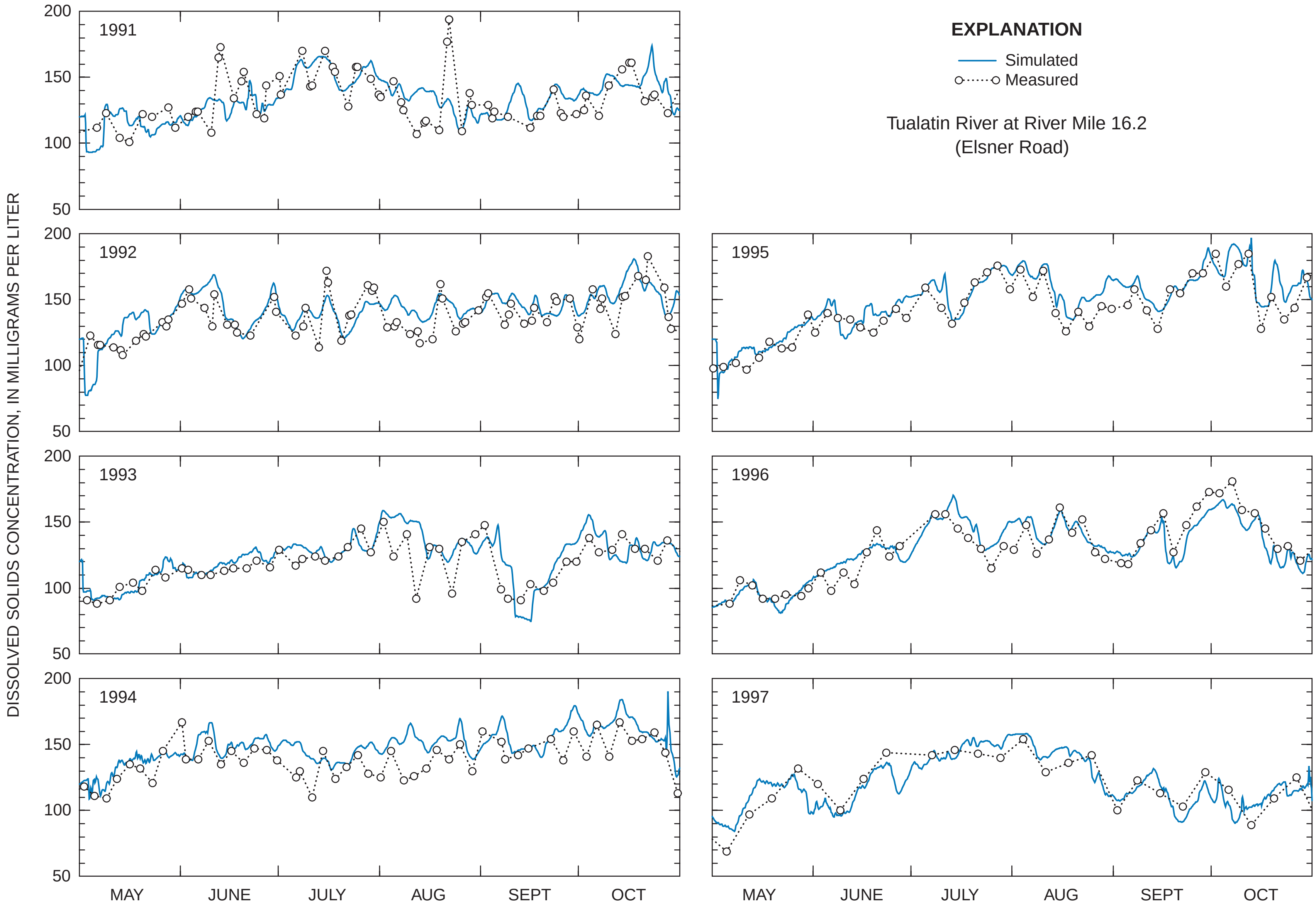

Figure 15. Simulated and measured dissolved solids concentration at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

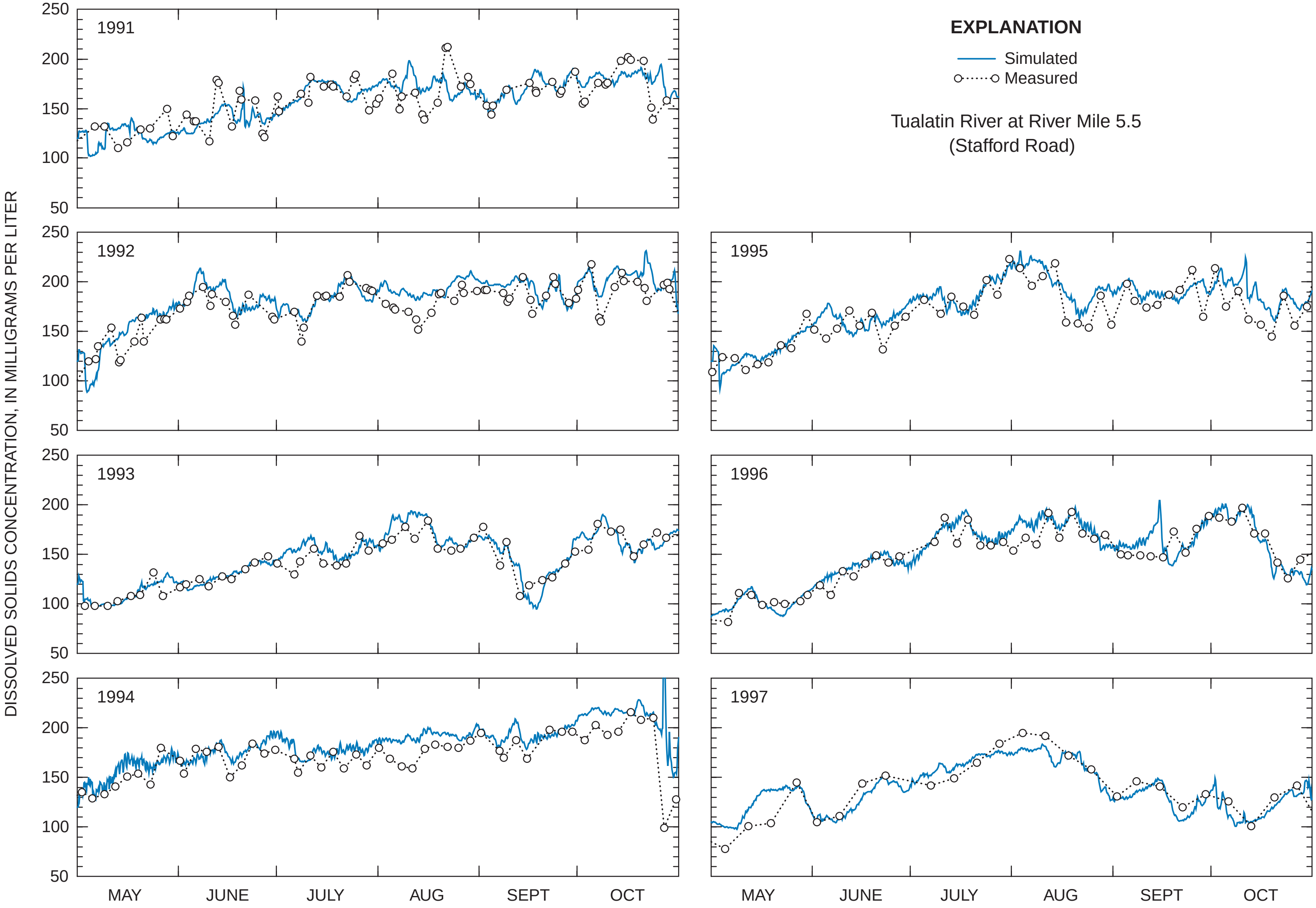

Figure 16. Simulated and measured dissolved solids concentration at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

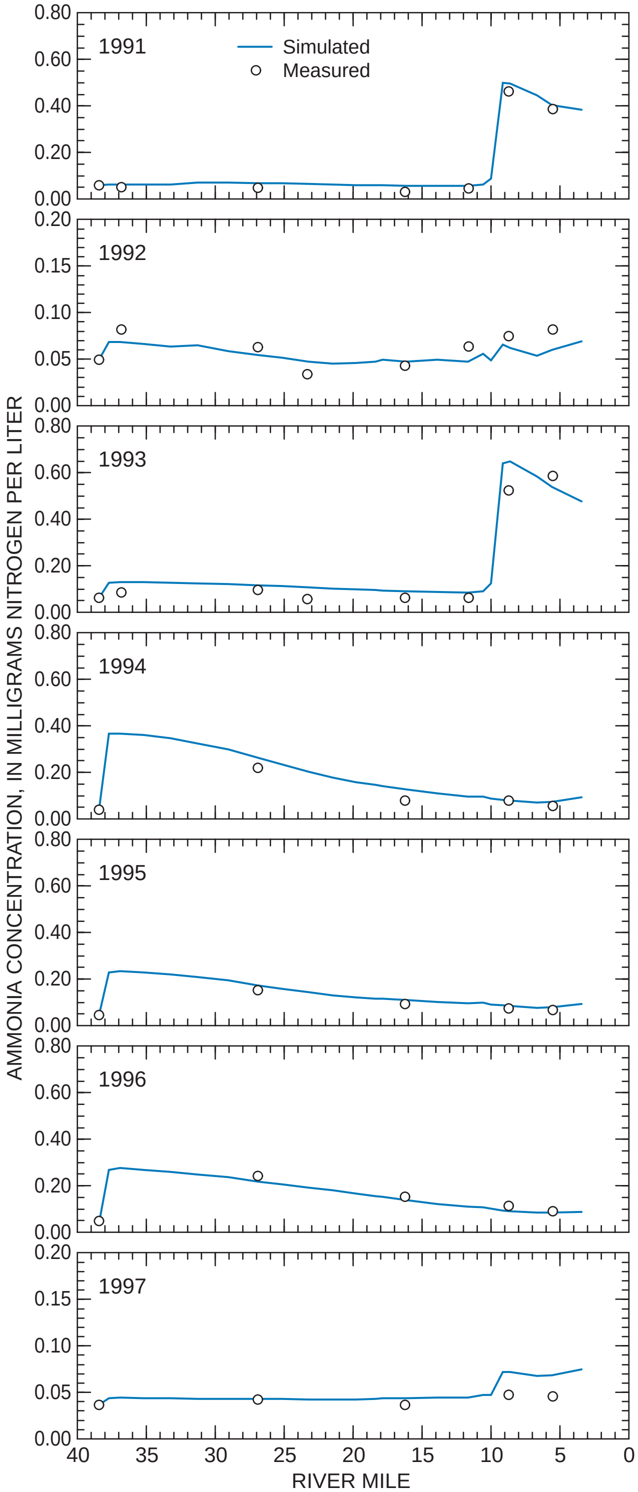

Figure 17. Simulated and measured mean ammonia concentrations as a function of river mile for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations are volume-weighted means, also from the top 10 feet of the water column. Note differences in axis scaling.)

tions of ammonia typically less than 0.1 mg/L and frequently less than 0.05 mg/L. In each of the other summers, at least one ammonia release from one of the WWTPs occurred, either due to plant construction/ expansion or an unplanned, temporary loss of in-plant nitrification. In 1991 and 1993, the release was from the Durham WWTP at RM 9.3 and was due to plant construction and upgrades. In 1994, the problem was minor and occurred in May when streamflow was still somewhat elevated. The releases in 1995 and 1996 from the Rock Creek WWTP (RM 38.1) were caused by a temporary loss of the population of nitrifying bacteria in the plant. For these releases in 1994–1996, the longitudinal profile plots show agreement between the measured and simulated ammonia concentrations, indicating that the rate of instream nitrification used by the model was accurate. These plots also illustrate that the model includes all of the important sources and sinks of ammonia; the fit is good for a wide range of concentrations.

A comparison of measured and simulated ammonia concentrations at RMs 16.2 (Elsner Road, fig. 18) and 5.5 (Stafford Road, fig. 19) further illustrates the ability of the model to simulate the transport and fate of this constituent. The model output matches the data closely, with RMSEs generally less than 0.1 mg N/L (table 5), which is good considering the range and the scatter in some of the measured data. The values of r^2 and MRAE are not as useful for this constituent because the magnitude and range of the data during several summers was small. The duration and peak concentrations during the large ammonia releases of 1995 and 1996 are simulated closely by the model, which will prove to be critically important when the effects of these events on dissolved oxygen is evaluated. Note that the simulated data are plotted with a frequency of six points per day (every 4 hours), so it is possible to discern the simulated diurnal effect of algal uptake and respiration on the ammonia concentration. Uptake of ammonia for algal growth can be an important influence on the ammonia concentration during a bloom; the simulated peak ammonia concentration at RM 5.5 during the release event in 1996 was reduced 5 to 10 percent by algal uptake.

Nitrate

The sum of nitrate and nitrite concentrations (henceforth referred to as “nitrate” due to the low

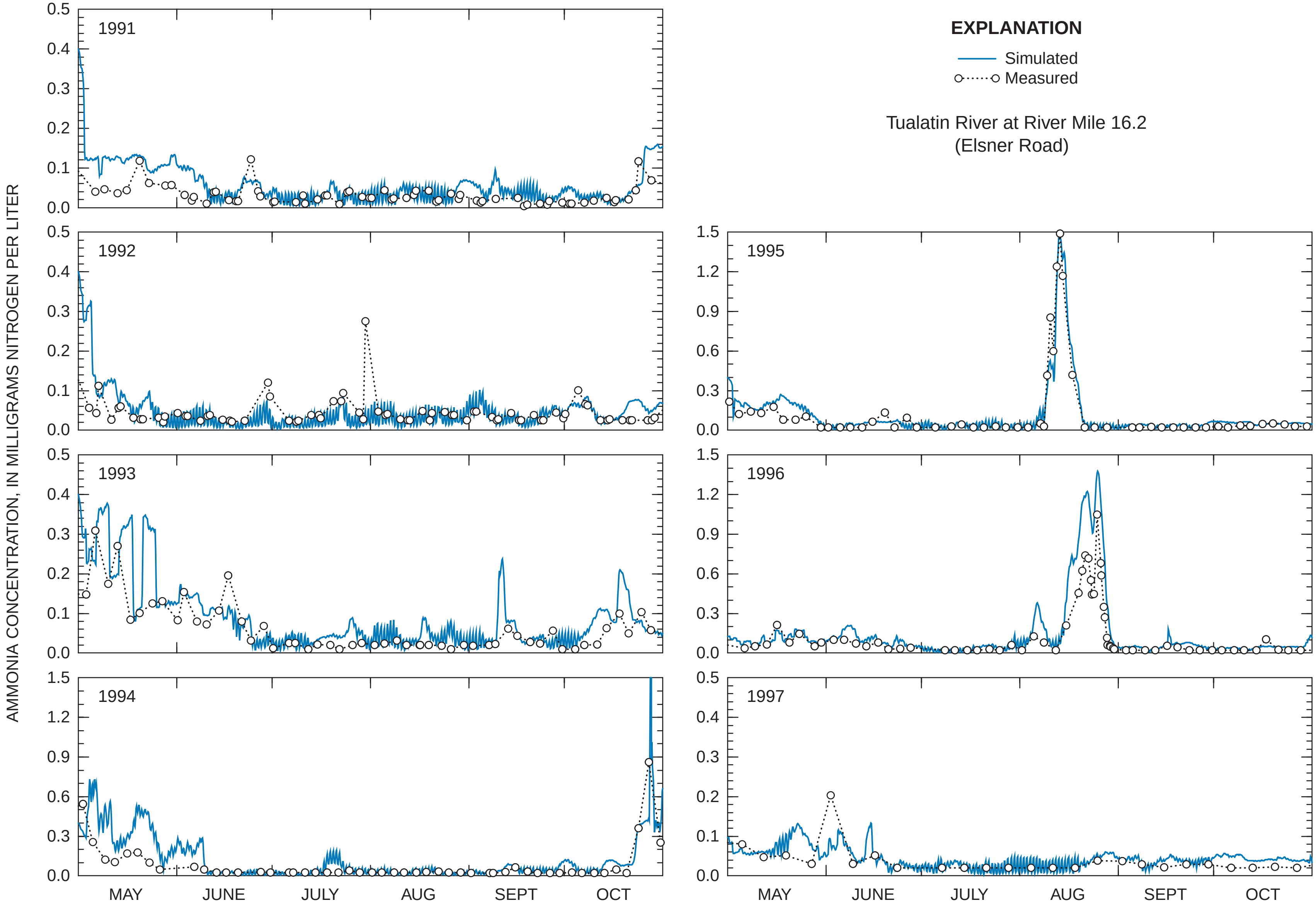

Figure 18. Simulated and measured ammonia concentration at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column. Note differences in axis scaling.)

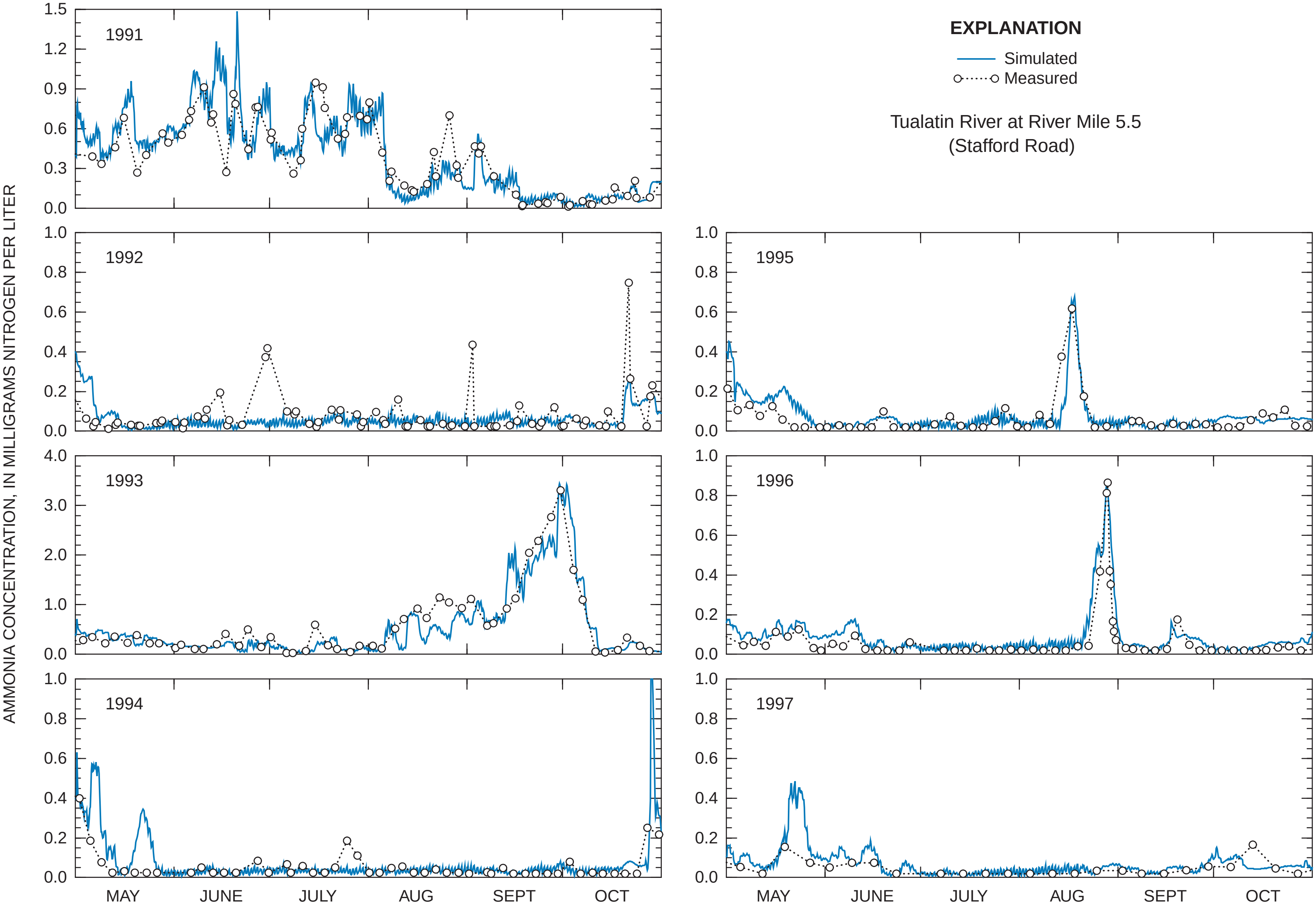

Figure 19. Simulated and measured ammonia concentration at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column. Note differences in axis scaling.)

concentrations of nitrite) in the Tualatin River is influenced mainly by the loads of nitrate from the WWTPs. The longitudinal profile of mean measured and simulated nitrate concentrations shows the effects of the WWTP loads at RMs 38.1 and 9.3 (fig. 20). Nitrate is a source of nitrogen for algal growth, but the concentration of nitrate in this river is high enough that the algae do not significantly affect the instream concentration. Nitrogen is not a growth-limiting nutrient for phytoplankton in this system (Rounds and others, 1999). The main influences on the nitrate concentration are the changing loads from the major sources and the simple advective transport of this constituent downstream. The simulated concentrations of nitrate are not bad, with RMSEs near 0.4 mg N/L, but a significant positive bias is discernible during midsummer of 1992, 1993, and 1994, especially in the time series plots at RMs 16.2 and 5.5 (figs. 21 and 22). As discussed in the original analysis by Rounds and others (1999), this bias may be evidence of a missing sink in the model; indeed, denitrification of nitrate to nitrogen gas may be an important loss mechanism for nitrate during some warm summer periods, and this process is not included in the model. Other than this discrepancy, however, the fit for nitrate is good, and any bias in the simulated nitrate concentration will not affect the other modeled constituents.

Total Phosphorus

Although not a conservative constituent, total phosphorus is pseudoconservative. Because the measurement of total phosphorus includes the phosphorus contained in dissolved organic matter and algae as well as dissolved orthophosphate, the concentration of total phosphorus is unaffected by algal uptake, respiration, and various decomposition processes taking place in the water column. Processes that affect total phosphorus include settling and sediment release as well as changes in the characteristics of its upstream sources. If settling and sediment releases are either minor or they offset, then the concentration of total phosphorus is mainly a reflection of its upstream sources. Indeed, the seasonally averaged longitudinal profile plot for total phosphorus somewhat resembles that of a conservative constituent (fig. 23). Large loads of phosphorus from the Durham WWTP produced obvious increases in the concentration of total phosphorus at RM 9.3 in 1991 and 1993. In the other summer seasons, phosphorus removal

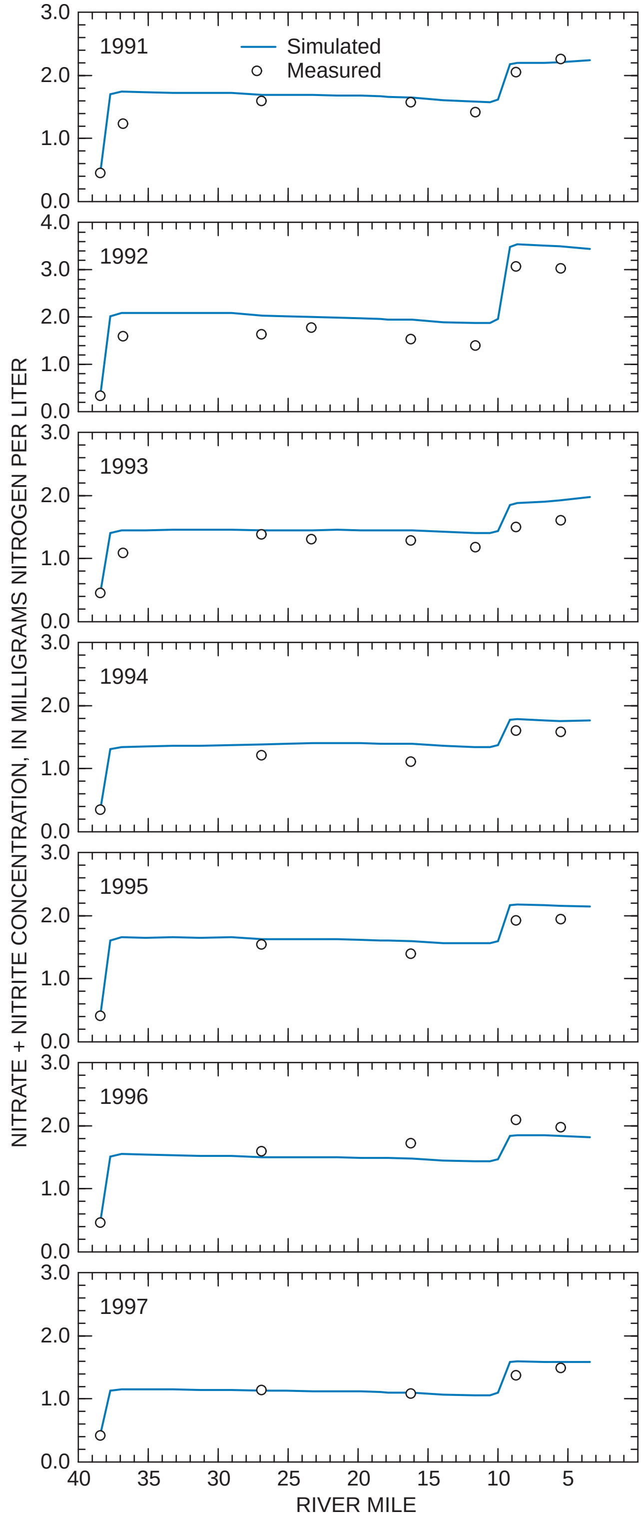

Figure 20. Simulated and measured mean nitrate concentrations as a function of river mile for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations are volume-weighted means, also from the top 10 feet of the water column.)

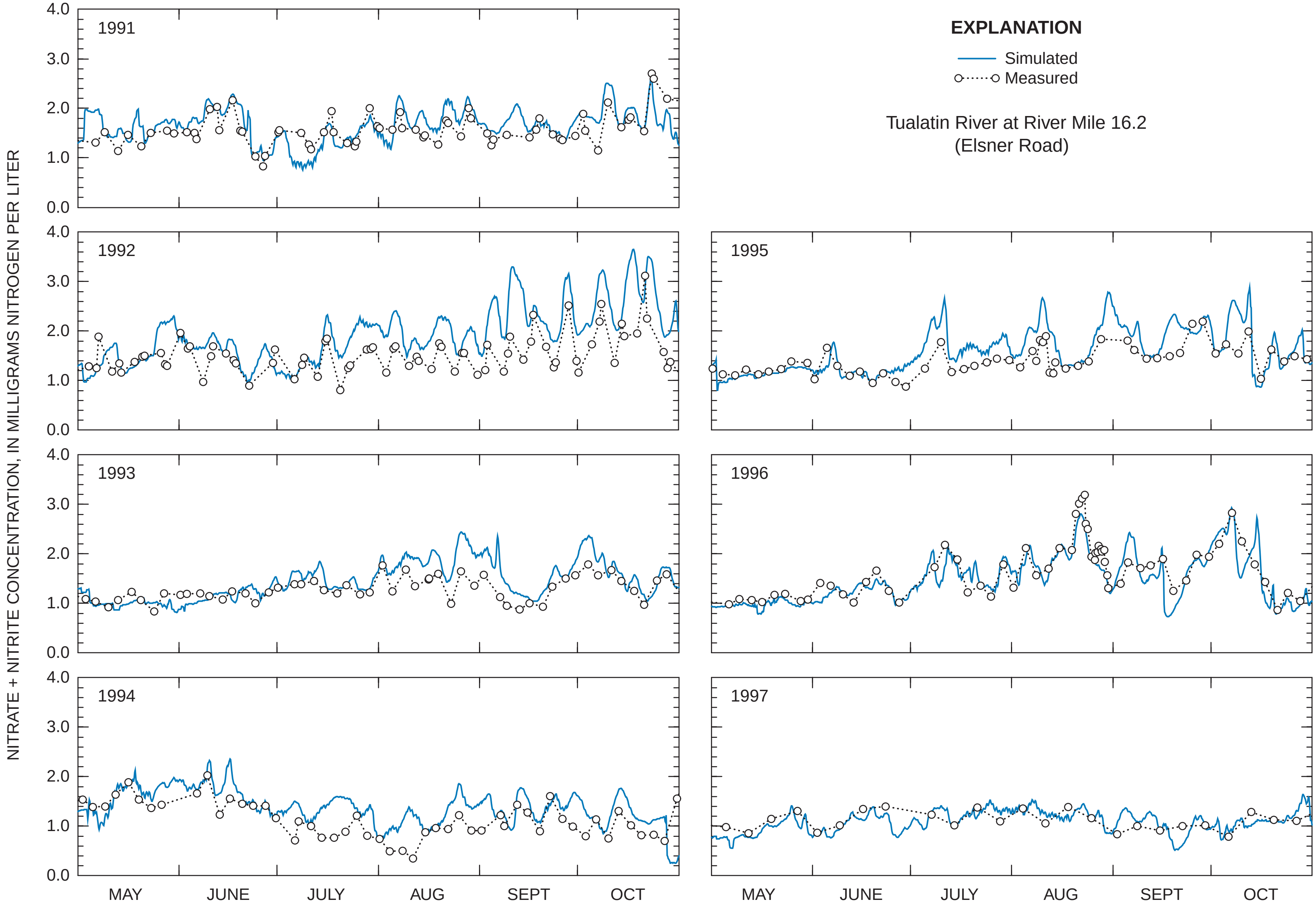

Figure 21. Simulated and measured nitrate concentration at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

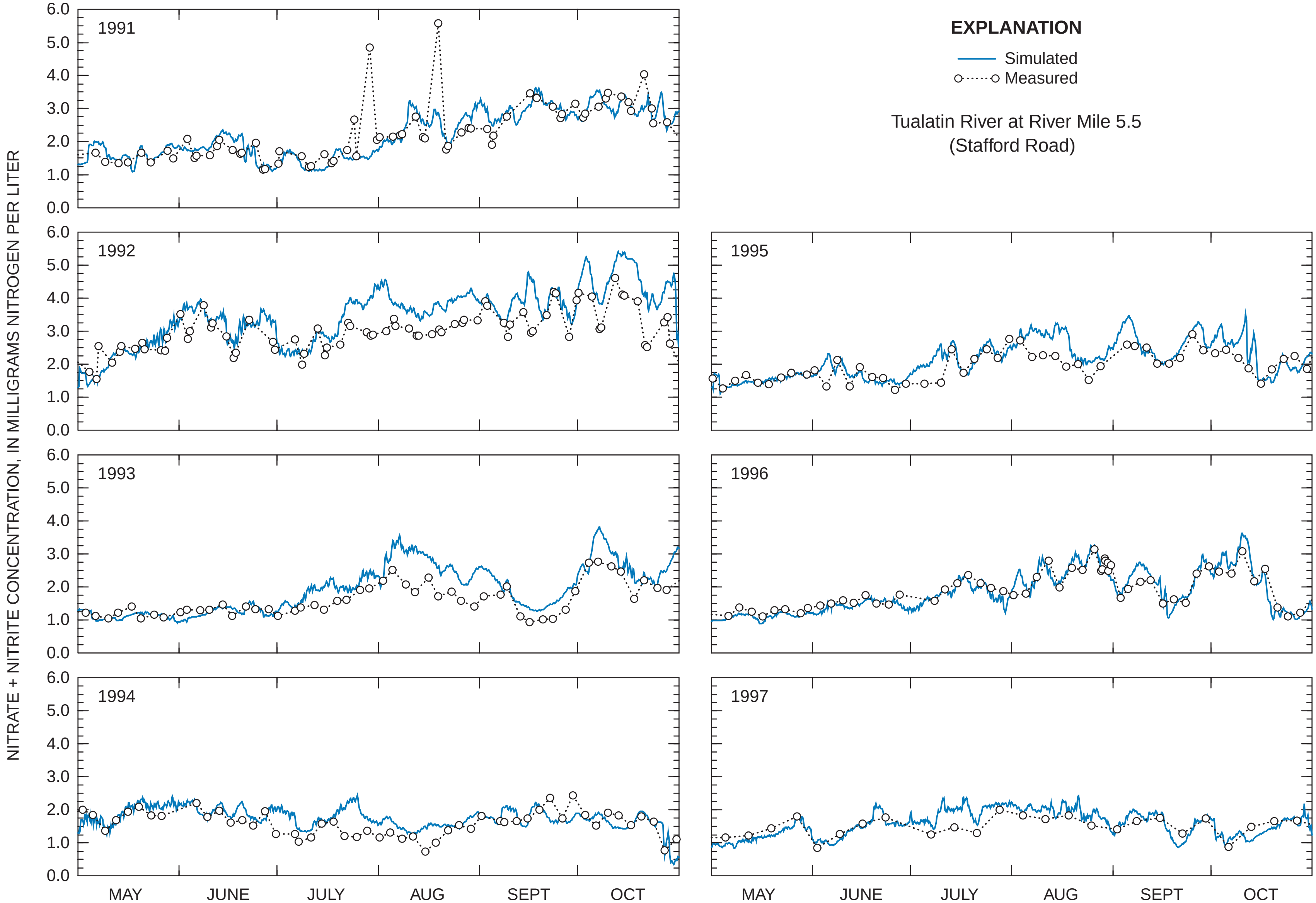

Figure 22. Simulated and measured nitrate concentration at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

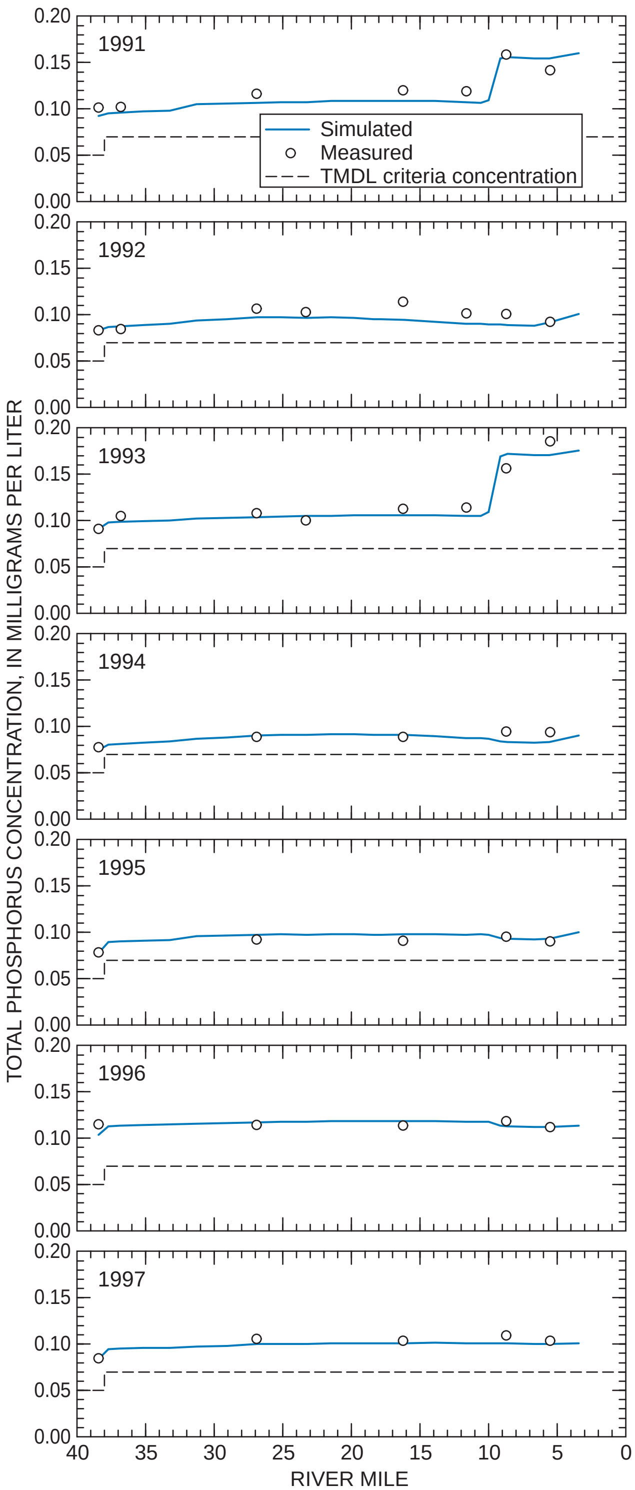

Figure 23. Simulated and measured mean total phosphorus concentrations as a function of river mile for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations are volume-weighted means, also from the top 10 feet of the water column.)

in both WWTPs was optimal and no obvious signature from the WWTPs is visible in the longitudinal profile plots. In fact, in-plant phosphorus removal was so efficient during those summers that the concentration of phosphorus in the effluent typically was less than that in the receiving water, often by a factor of two or more (Unified Sewerage Agency, 1999). Still, the measured instream total phosphorus concentrations were higher than the TMDL concen-tration limit for the entire length of the model reach (fig. 23).

For most of the summers during 1991–1997, the simulated total phosphorus concentrations track the measured concentrations closely, with RMSEs as low as 0.011 mg P/L and usually no more than 0.03 mg P/L (figs. 24 and 25). The calculated r^2 values are not always useful fit statistics for total phosphorus because the measured data range can be small. For example, the r^2 value for the correlation of measured and simulated concentrations at RM 5.5 in 1995 is only 0.44, yet the fit is visibly good, with an RMSE of only 0.016 mg P/L, which is within the range of analytical error for this constituent (see fig. 25 and table 5). The MRAEs for total phosphorus of about 17% also are within the expected range of analytical error of 20% (Rounds and others, 1999). When the WWTP phosphorus-removal operations were optimal, the total phosphorus concentration in the river was nearly constant, although greater than the TMDL criteria concentration of 0.07 mg P/L at these two sites. For more information on the nature of the sources of phosphorus to the Tualatin River, see the analysis published by Kelly and others (1999).

Orthophosphate

Unlike total phosphorus, the concentration of dissolved orthophosphate is greatly affected by instream chemical and biological reactions such as algal uptake, respiration, and decomposition as well as release from the sediments. Orthophosphate is one of the several more challenging constituents to model in the Tualatin River. Taken as a seasonal average, the model simulates the longitudinal trends in the ortho-phosphate concentration rather well (fig. 26). As was the case for total phosphorus, the influence of the Durham WWTP is obvious in these longitudinal profile plots at RM 9.3 during 1991 and 1993. From the upstream boundary to about 10 or 20 miles down-

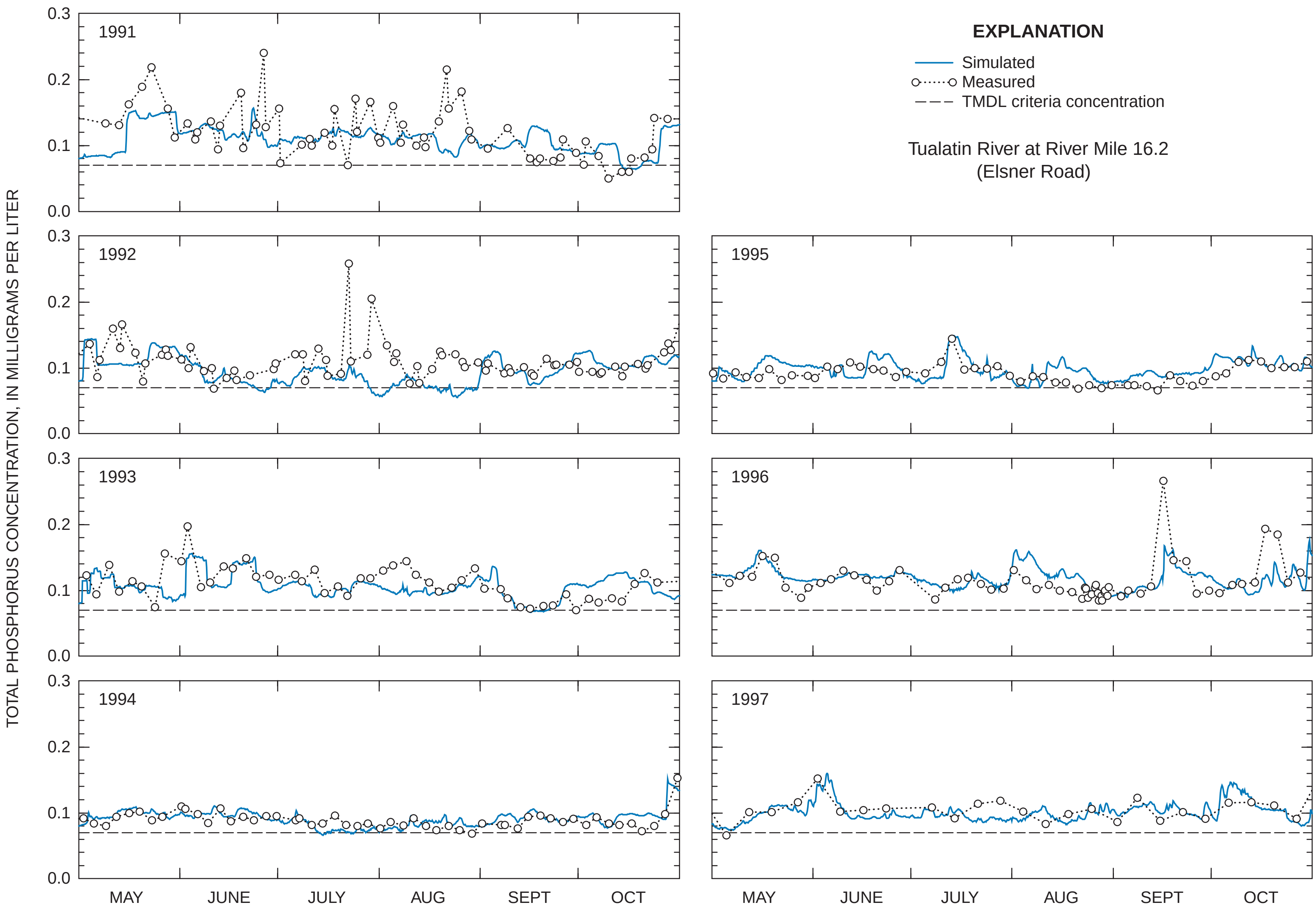

Figure 24. Simulated and measured total phosphorus concentration at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

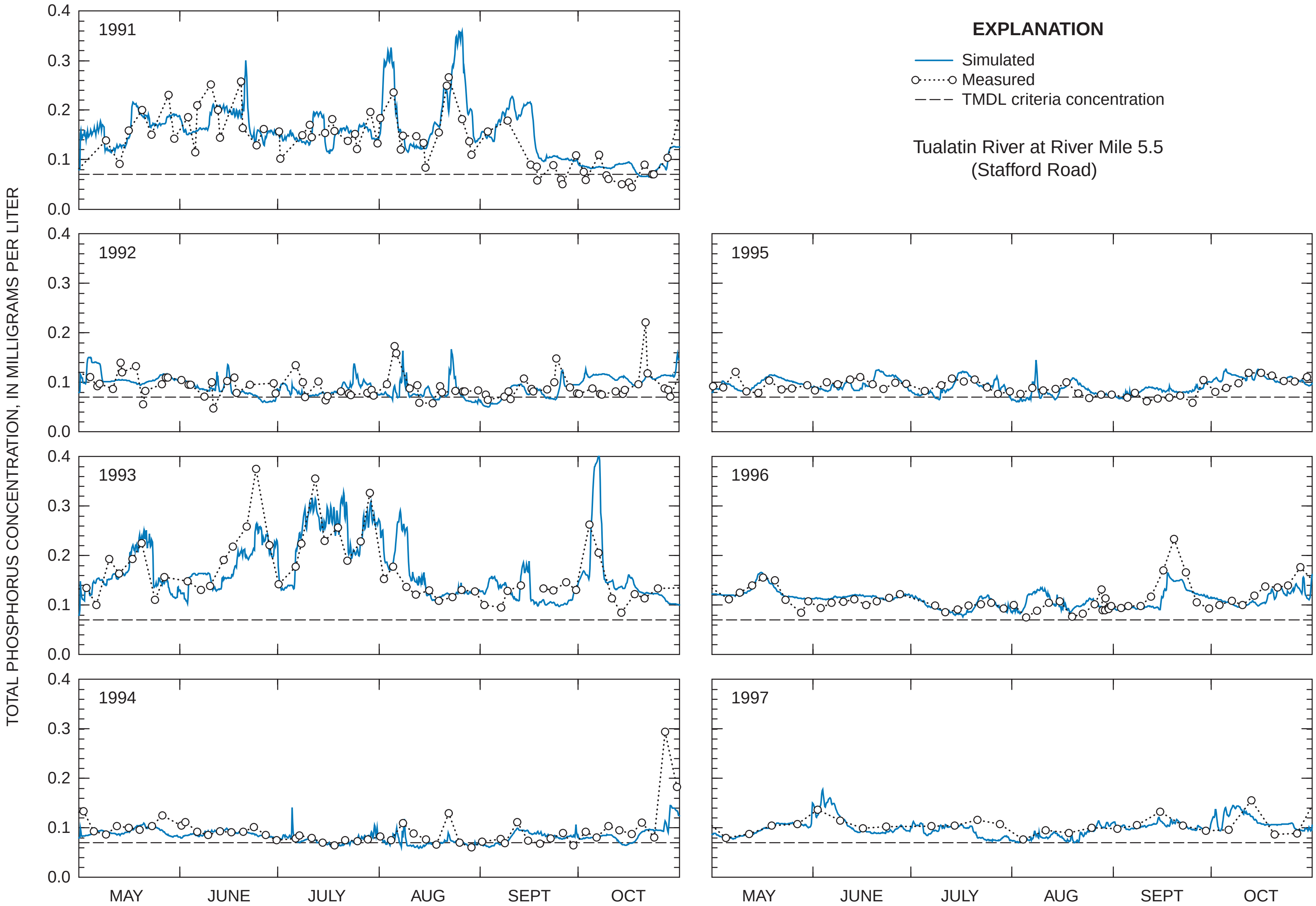

Figure 25. Simulated and measured total phosphorus concentration at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

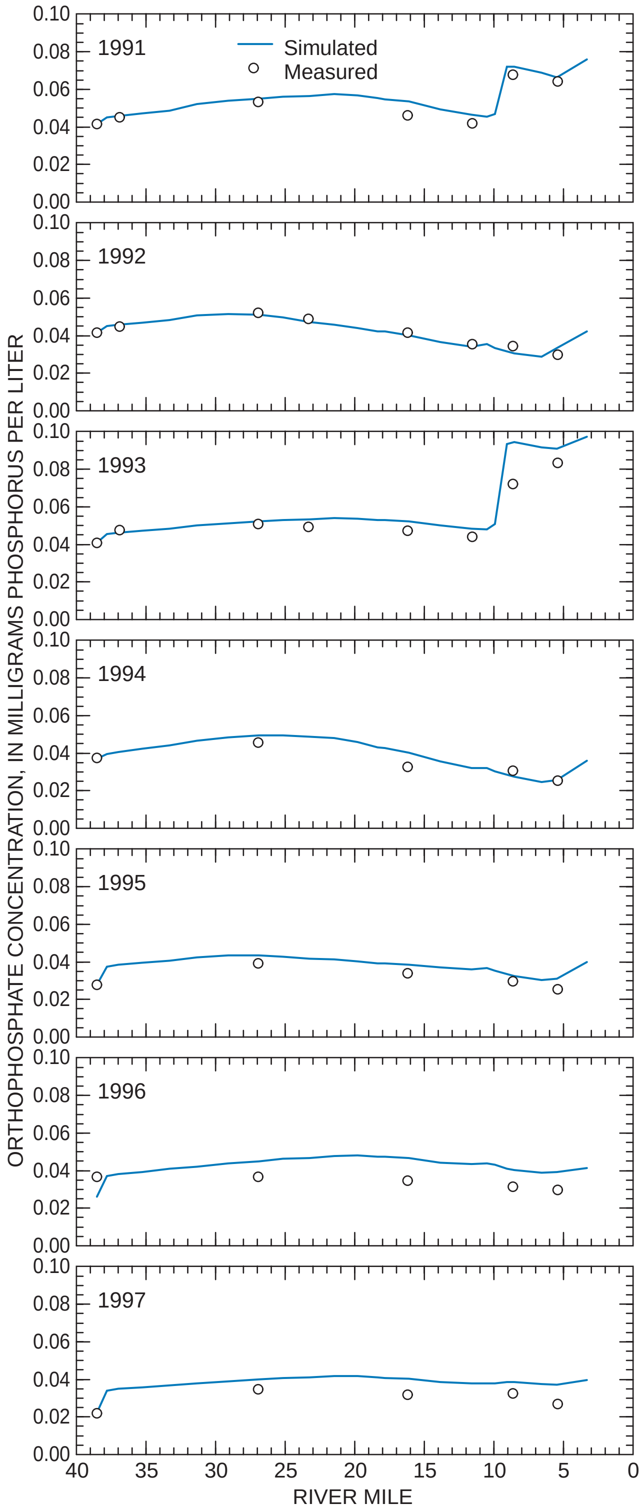

Figure 26. Simulated and measured mean orthophosphate concentrations as a function of river mile for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations are volume-weighted means, also from the top 10 feet of the water column.)

stream depending on the year, the orthophosphate concentration increases slightly due to inputs from small tributaries, ground water, and sediment decomposition. Further downstream, algal uptake tends to decrease the mean orthophosphate concentration. The model simulates these trends well, although a small positive bias is evident for 1996 and 1997.

Comparisons of measured and simulated orthophosphate concentrations at RMs 16.2 (Elsner Road, fig. 27) and 5.5 (Stafford Road, fig. 28) are not as favorable as they are for the seasonal averages. The general trends in orthophosphate concentration at these sites are captured by the model, but the error in the simulated concentration is high for some of the samples. The RMSEs range from 0.013 to 0.037 mg P/L, which is on the high end of the acceptable range. These errors are reflected in the values of the r^2 and MRAE as well (tables 5 and 6). Despite this imprecision, many subsets of these time series are simulated closely. These subsets correspond to periods during which the algal population also was simulated accurately. In fact, inaccuracies in the simulation of the orthophosphate concentration typically are negatively correlated with inaccuracies in the simulation of the algal population (table 7). For example, if the model simulates too little algae at a particular time and place, it is likely that it also simulates an orthophosphate concentration that is too high. The correlations in table 7 indicate that, on a seasonal basis, the errors in the simulation of the algal population can account for up to 50 percent of the errors in the simulated orthophosphate concentrations. Only for 1997, when fewer algal blooms occurred than in any other summer, does this correlation break down. In that year, then, the errors in the simulated orthophosphate concentrations must be due to other causes such as unaccounted-for sorption or algal uptake; the sorption issue is addressed in the context of data from 1996 in a later section of this report.

In the original USGS Tualatin River modeling study (Rounds and others, 1999), no provision was made for the release of sequestered phosphorus from the sediments during anoxic conditions. Because this process was not included, that version of the model could not simulate the buildup of orthophosphate concentrations in isolated pockets of anoxic hypolimnetic water. By extension, the model also could not simulate the transport of orthophosphate from these anoxic waters to the overlying and downstream oxic water. This injection of orthophosphate can sometimes

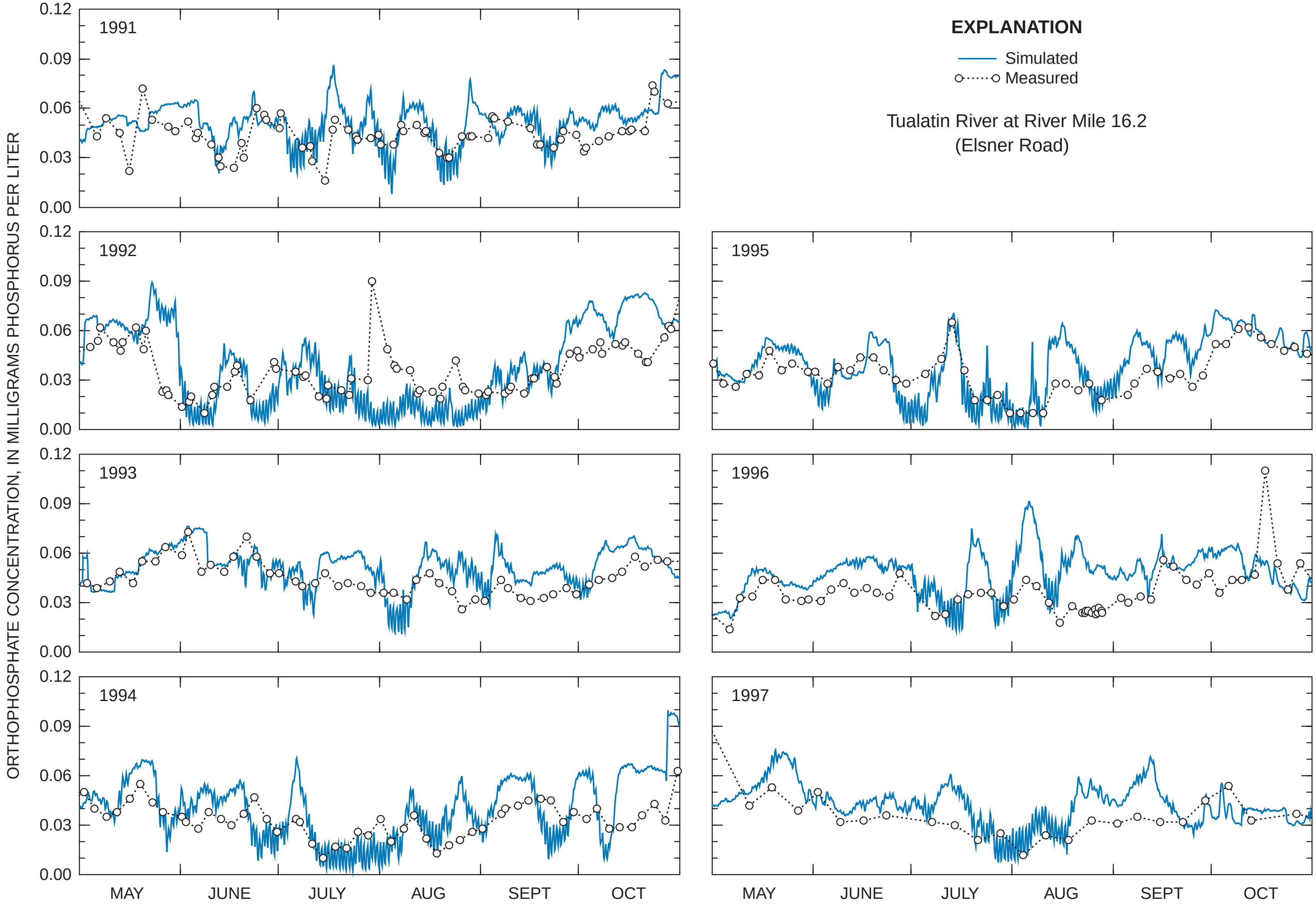

Figure 27. Simulated and measured orthophosphate concentration at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

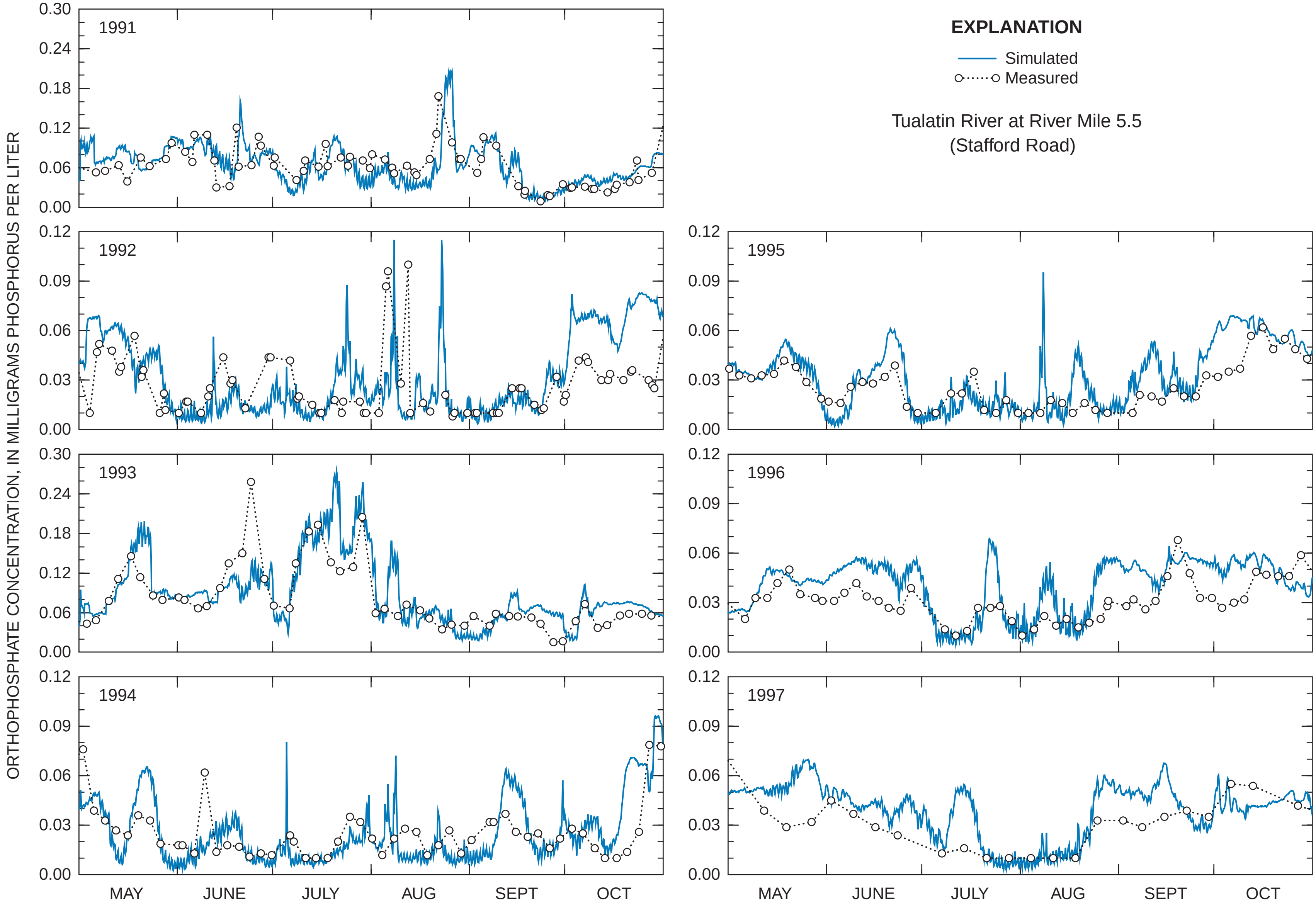

Figure 28. Simulated and measured orthophosphate concentration at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column. Note differences in axis scaling.)

Table 7. Correlation coefficient for the residual in orthophosphate concentration against the residual in chlorophyll-*a* concentration

[RM, river mile; †, one point was removed from this data set; ✓✓, indicates statistical significance at the 99 percent confidence level; ✓, indicates statistical significance at the 95 percent confidence level.]

Year	RM 16.2 (Elsner Road)	Statistical significance	RM 5.5 (Stafford Road)	Statistical significance
1991	-0.40	✓✓	-0.41	✓✓
1992	-.46	✓✓	-.28	✓
1993	-.63	✓✓	-.47	✓✓
1994	-.71	✓✓	-.54	✓✓
1995	-.65	✓✓	-.42	✓✓
1996	-.35†	✓	-.48	✓✓
1997	-.25		-.25	

be important in providing phosphorus to an algal bloom that is or is becoming phosphorus limited. This updated version of the USGS Tualatin River model includes such a sediment phosphorus release process. The effects of this process are not important at RM 16.2 because the reaches upstream of that point do not thermally stratify appreciably and therefore rarely create a layer of anoxic water at the sediment/water interface. At RM 5.5, however, the effects of this process are easily discernible (fig. 28). The spikes in the simulated orthophosphate concentration at RM 5.5 during late July and August of 1992 are attributable to this process. The same is true for the orthophosphate spikes in August of 1993 and those in July and August of 1994 and 1995. This process was not significant or did not occur at all in 1996 or 1997 because very little thermal stratification ever developed during those periods (see fig. 10). These injections of ortho-phosphate, generally during algal blooms, were important in enabling the model to more accurately simulate the size of the algal population. The addition of this algorithm improved the model fit for both orthophosphate and chlorophyll-*a* for 1992 but had little effect on the original calibrations for 1991 and 1993.

Algae

Modeling the dynamics of a population of phytoplankton is a challenging task, especially when trying to capture the essence of those dynamics over a 42-month period spanning parts of 7 hydrologically distinct years. Compromises must be made. As in the original USGS Tualatin River modeling study, the decision was made to sacrifice some short term accuracy in favor of preserving the model's ability to simulate the timing of algal blooms and their general size over a wide range of conditions. In other words, some errors in the simulation of day-to-day fluctuations in water quality would be accepted if the long-term predictive capabilities of the model were enhanced by the compromise. The model could simulate any one algal bloom perfectly if it were calibrated only to the short time period of that bloom, but such a model would fail quickly for a wider range of conditions. A longer view, therefore, was necessary to maximize both the utility of the model and the insights that might be gained from it.

The version of CE-QUAL-W2 used in this study simulates only one algal type. The entire algal community must be given only one growth rate, one respiration rate, etc. Fortunately, no clear species succession is normally observed in Tualatin River; the algal assemblage is dominated by several diatom species. As discussed by Rounds and others (1999), however, changes in temperature, light conditions, streamflow, or nutrient concentrations can cause individuals in the algal community to adapt, resulting in measurable physiological changes at the community level. In this and the original modeling study, a simplified seasonal variation in the light- and nutrient-saturated algal growth rate was permitted in the model to account for seasonal changes in the measured algal primary productivity rate.

Measured primary productivity rates from the summers of 1991–1993 generally were higher when streamflow was high at the start of the summer season, lower during the middle of the summer, and high again late in the summer when the days were shorter. The original study found that the algal growth rate could be adequately represented with a simple step function in each of the summers of 1991–1993, shifted forward or backward in time according to streamflow conditions. The step function used in 1991–1993 was retained in this study, but some additional flexibility was allowed for 1994–1997. In 1991–1993, the growth rate step function always started and ended at 6.0 day^{-1} and the midsummer rate was always 4.5 day^{-1} and lasted 65–70 days. In the absence of measured primary productivity data for 1994–1997, it was recognized that different hydrologic conditions, especially the wetter summers of 1996 and 1997, could alter this simple step function in small but important ways. Use of the original 70-day step function with a range of 4.5 to 6.0 day^{-1} provided a good first estimate for algal growth during 1994–1997, but it quickly became clear that a slightly modified function would provide better results. The light- and nutrient-saturated algal growth rate for 1994–1997 was allowed to be as high as 6.5 day^{-1} and as low as 4.0 day^{-1}, and the period of slower algal growth was allowed to be as long as 95 days or absent altogether. The algal growth rate functions used in this study are depicted in figure 29. The functions used for 1996 and 1997 fit well with the hydrologic conditions during those summers. For 1995, the rate never exceeds 5.0 day^{-1}; this was done to account for the fact that the model often simulated water temperatures that were too high during algal blooms in 1995. As mentioned earlier, a water temperature error of 2˚C can cause an error of as much as 15 percent in other reaction rates. The early and late parts of the 1995 algal growth rate step function were decreased from 6.0 day^{-1} to 5.0 day^{-1} to account for this water temperature problem. Other peculiarities of the function used for 1995 are addressed later in this report.

Algae in the Tualatin River are mostly phytoplankton that move with the current, reproduce while the growing conditions are favorable, and leave the system with the water that transports them. The reservoir reach of the river, from RM 30 to 3.4, is generally deep enough and turbid enough to prevent any significant growth of periphyton or benthic algae. This reach supports the highest concentrations of phytoplankton and most of the algal growth, in part because the river is wide enough to allow sufficient sunlight to reach the river surface. Populations of phytoplankton entering the upper end of the reservoir reach are generally insignificant (< 6 µg/L as chlorophyll-*a*), but can build to high levels (80 µg/L chlorophyll-*a* or more) near the downstream end of the reach if conditions are favorable.

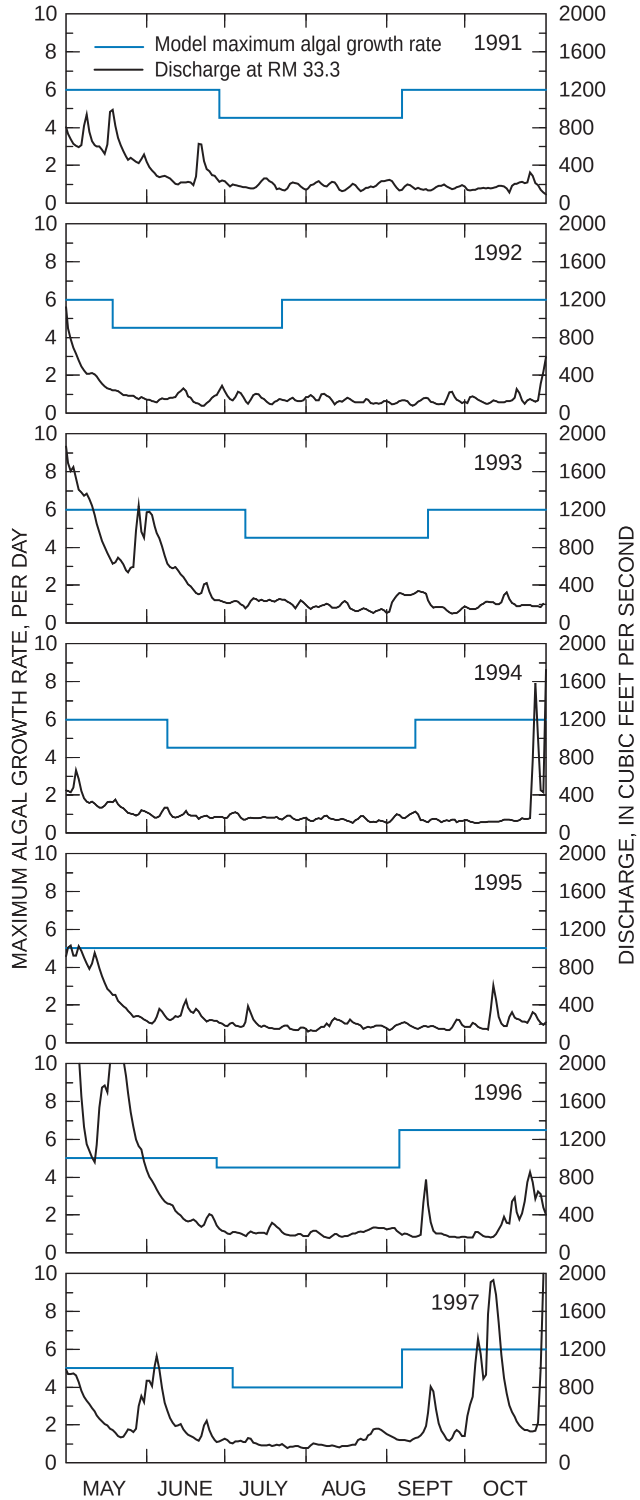

Figure 29. Maximum light- and nutrient-saturated algal growth rate function used in model simulations of the Tualatin River, with river discharge at river mile 33.3, May-October of 1991–1997.

To accurately simulate the algal population, the model must first simulate the residence time, water temperature, light penetration, and nutrient concentrations with sufficient accuracy. In addition, the simplified algorithms used by the model must be sufficiently representative of the algal dynamics in the river. Averaged over the summer season, the model simulates the longitudinal profile of algal populations well, as measured by chlorophyll-*a* (fig. 30). These plots illustrate several important characteristics of the measured and simulated algal population. First, the profiles clearly show the growth of the algal population as the river transports the algae through the modeled reach. Second, the model accurately simulates the location in the reach where algal growth is initiated. Third, the mean size of the algal population is larger in low-flow years (1992 and 1994) than it is in high-flow years (1996 and 1997). This makes sense, because the low-flow years had more sunny days corresponding with residence times long enough to support significant levels of algal growth, and the high-flow years had more cloudy days and more days with short residence times. Finally, the model does reasonably well in simulating the mean size of the algal population, although it simulates a population that is slightly too large in 1991, 1995, and 1996 and perhaps too small in 1997. The discrepancy in 1996 may be related to the higher-than-normal turbidity during that summer; that topic will be explored later.

Zooplankton use both algae and detritus as a food source. Under certain conditions, zooplankton grazing can be a significant loss process for algae. Zooplankton population data were available only for 1991–1993. During that period, grazing was significant for just a few specific periods, most notably for August of 1991. The zooplankton growth curve lags the phytoplankton growth curve such that the zooplankton can be important only in the most downstream 6 to 8 miles of the reservoir reach, if at all. The effects of grazing on the mean algal population are visible in figure 30 as a decreasing concentration of chlorophyll-*a* downstream of RM 5.5 in 1991 and perhaps in 1994. Calibration of the model for 1994–1997 indicated that the influence of zooplankton on the algal population generally was minor. Without data for the actual size of the zooplankton population, this conclusion cannot be verified; however, the model did not appear to require a significant loss process for algae (such as zooplankton grazing) during this period.

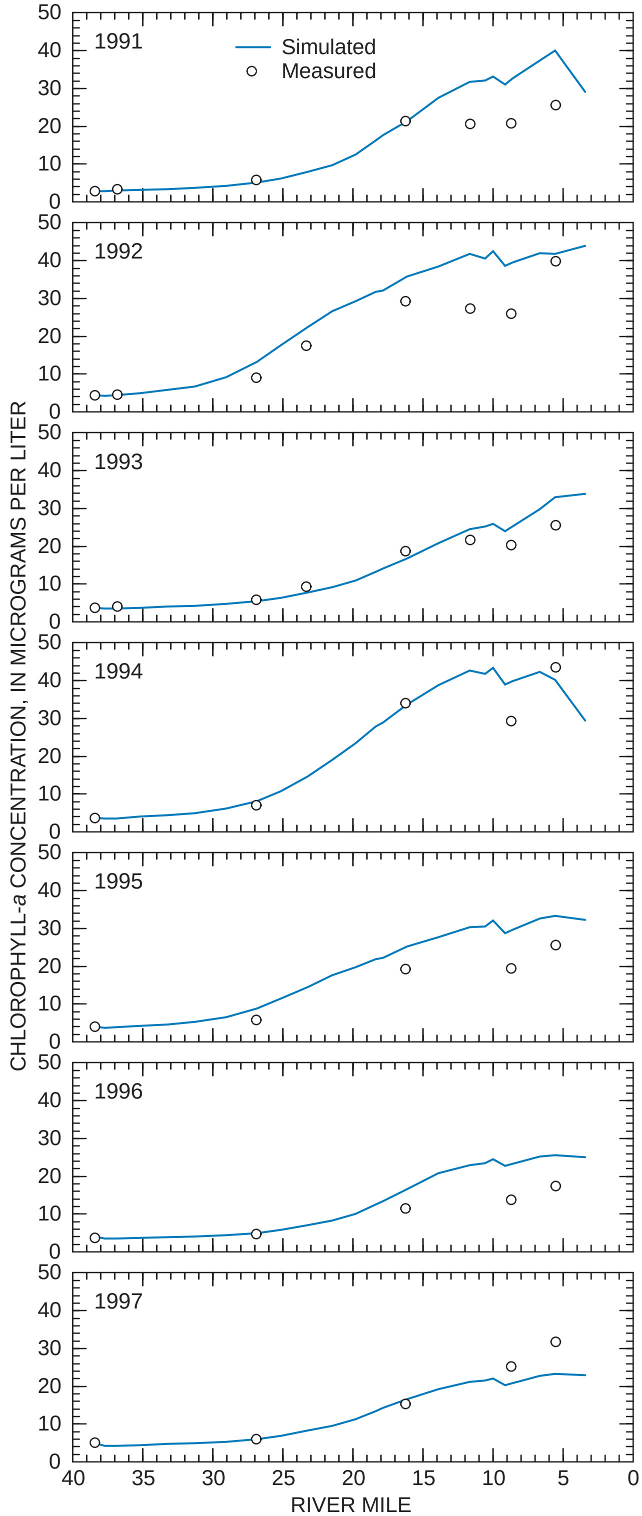

Figure 30. Simulated and measured mean chlorophyll-*a* concentrations as a function of river mile for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations are volume-weighted means, also from the top 10 feet of the water column.)

A comparison of measured and simulated concentrations of chlorophyll-*a* at RMs 16.2 (Elsner Road, fig. 31) and 5.5 (Stafford Road, fig. 32) shows that the model captures the basic trends in the Tualatin River's algal population. The model simulates the *initiation* and *duration* of the algal blooms with reasonable accuracy, an indication that the model algorithms linking algal growth to light conditions, residence time, and water temperature are valid. Most of the time, the size of the simulated blooms is also a good match for the measured data. The r^2 values indicate that the model accounts for most of the variability in the measured chlorophyll-*a* concentrations, especially at RM 16.2 where the zooplankton do not play a significant role (table 5). The mean RMSE at that site is near 12 µg/L, which is good considering that the data typically range as high as 80 µg/L or more. The MRAE is not particularly useful for chlorophyll because small errors during nonbloom periods make the MRAE artificially large for the season. Values of r^2 less than 0.6 at RM 5.5 in 1991 and 1992 are due to the inability of the model to simulate phytoplankton/zooplankton interactions with much accuracy during August of 1991 and June/July of 1992. This short term loss of accuracy is acceptable, however, when compared to the long term predictive capability that would have been lost if the zooplankton mortality rate had been adjusted seasonally in the absence of supporting data. Tests showed that these periods could have been simulated with good accuracy if the zooplankton mortality rate had been different, but then other periods of the simulation suffered. Fortunately, these periods of significant zooplankton grazing are limited, especially in more recent summers. During periods of higher streamflow such as in 1996 and 1997, the zooplankton population is unable to grow to levels that are significant.

Between blooms, algal growth is limited by short residence times, cool water temperatures, poor light conditions, or a combination of all three, but not by low nutrient concentrations. Below a depth of 10 feet or so, the turbidity of the river causes low light conditions to limit algal growth all the time. During a bloom, the model never simulated conditions in which nitrogen concentrations (ammonia and nitrate) limited algal growth. It was common during blooms at RM 5.5 in 1992 and in 1994–1997, however, for the growth of the simulated algal population to be limited by low concentrations of phosphorus. This is illustrated in figure 32, especially for 1996 and 1997, by the peaks of chlorophyll-*a* that appear to be truncated—the blooms level off at a particular concentration and are unable to grow further. The phosphorus limitation also is apparent in plots of orthophosphate for the same time periods (fig. 28); concentrations are driven down to levels near 0.01 mg P/L by algal uptake. Phosphorus also limited algal growth at RM 16.2, but this occurred slightly less often because the algal population had not necessarily grown to high enough levels to be limited at that location. Still, phosphorus control has proved to be an effective means of controlling the size of algal blooms in the Tualatin River. This phosphorus control, in concert with increased minimum streamflow through flow management, has effectively eliminated violations of the State of Oregon maximum pH standard of 8.5 in recent years. Despite this success, phosphorus limits on algal growth have yet to reduce the size of the algal population to the State of Oregon nuisance phytoplankton growth goal of 15 µg/L chlorophyll-*a*.

Although the model successfully simulated the initiation, duration, and general size of most of the algal blooms in the Tualatin River during the 1991–1997 period, its performance probably would have improved if more than one type of algae had been simulated. Having only one algal type makes the simulated algal community unable to respond appropriately to certain changes in streamflow, light, or temperature conditions. For example, during early August of 1995 when the Rock Creek WWTP lost in-plant nitrification and started releasing large ammonia loads, river managers increased the flow in the river. At the same time, light conditions for algal growth became slightly less favorable. This caused a significant decrease in both the measured and simulated algal populations (fig. 31). In order for the model to track the measured chlorophyll-*a* and dissolved oxygen concentrations during this period, however, it was necessary to increase the light- and nutrient-saturated algal growth rate to 5.0 day^{-1}; normally the algal growth step function might be at 4.0 or 4.5 day^{-1} at that time. If the model had two algal types, though, with one growing at a slower rate but able to thrive under lower light conditions, the model might have simulated this period of changing conditions with less difficulty. Having two or three algal types also could eliminate the need for the seasonally variable algal growth rate implemented in this model. Furthermore, the tendency of the simulated algal population to bloom and crash a bit too quickly might be cured with more than one algal type. The primary productivity data available for the

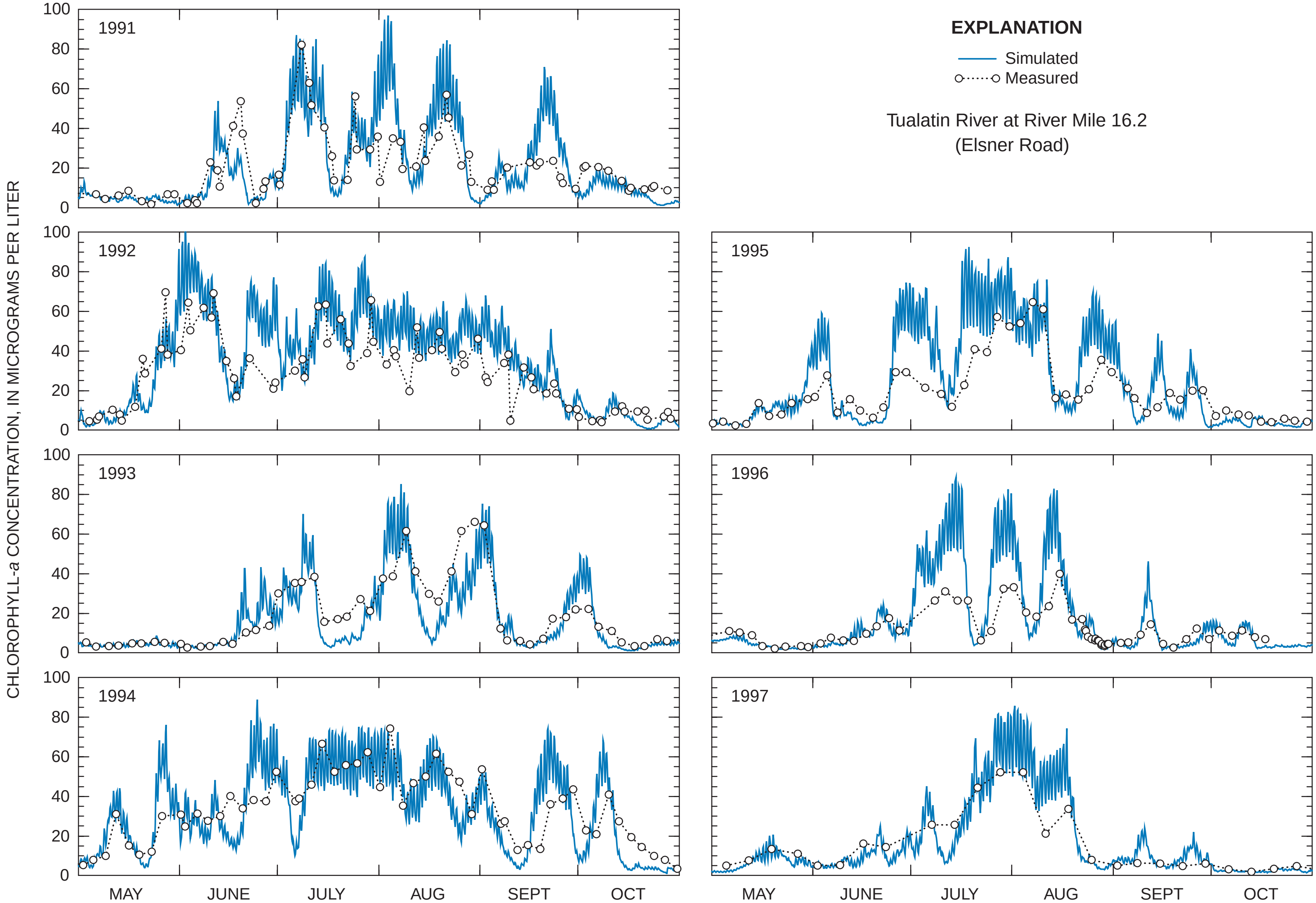

Figure 31. Simulated and measured chlorophyll-*a* concentration at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

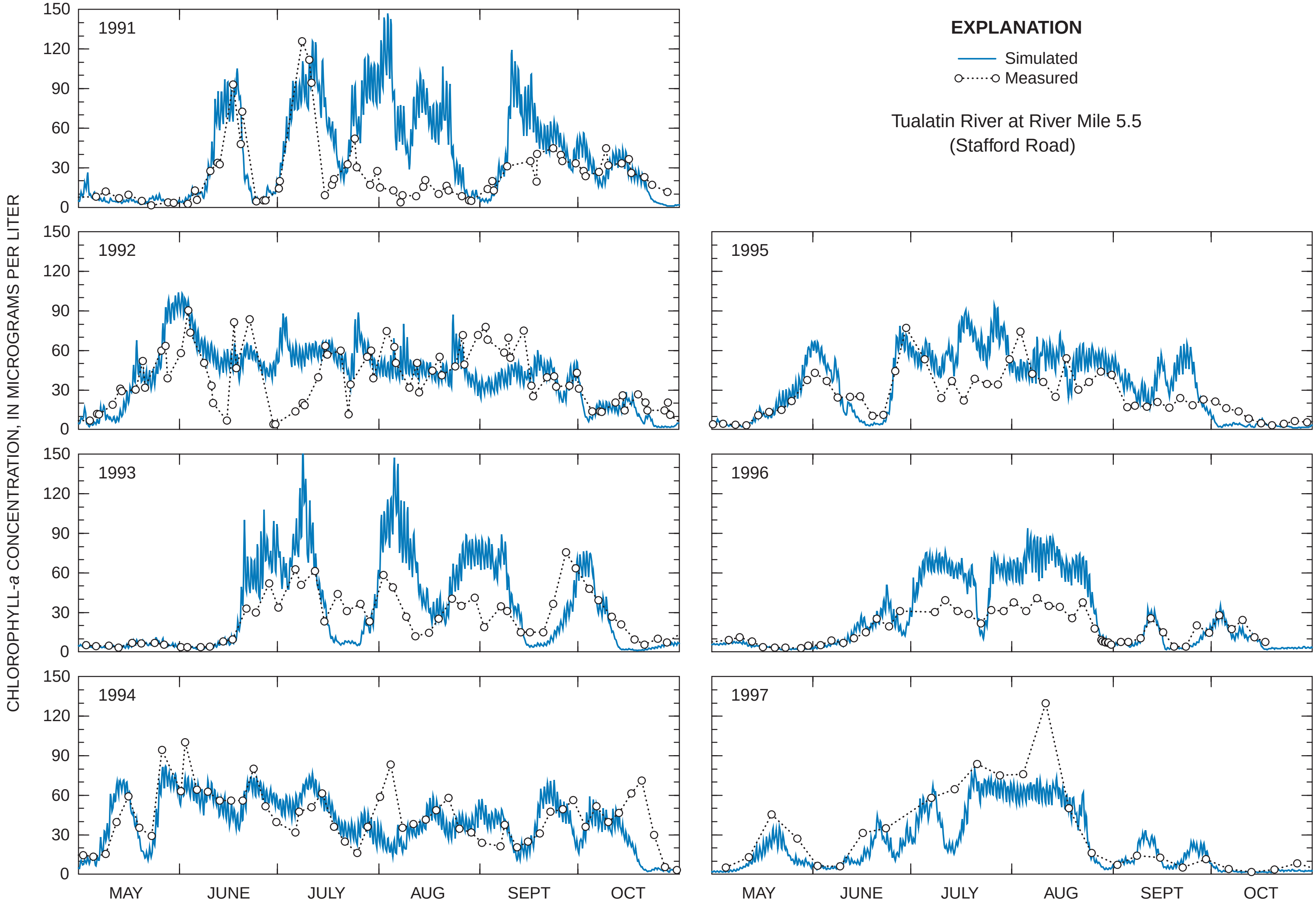

Figure 32. Simulated and measured chlorophyll-*a* concentration at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured concentrations are from composite samples taken from the top 10 feet of the water column. Simulated concentrations (every 4 hours) are volume-weighted means, also from the top 10 feet of the water column.)

Tualatin River, however, are limited to measurements for the algal community as a whole. It would be difficult, therefore, to separate these data and assign them to different algal groups. To prevent the introduction of complexity in the absence of data, it was decided to retain only one algal type in the USGS Tualatin River model and simply recognize the implications of that decision. Perhaps a future revision of the model, combined with further productivity analyses, will allow these complexities of the algal community to be captured.

Dissolved Oxygen

Dissolved oxygen concentrations in the Tualatin River are the result of a combination of many physical, chemical, and biological processes. The concentration is affected by water temperature through its solubility and the effect of temperature on the rates of reactions. Nitrification, respiration, and the decomposition of organic matter in both the water column and the sediment all consume dissolved oxygen. Reaeration is slow in this river most of the time, so photosynthesis typically is the only significant instream source of dissolved oxygen. Finally, well-aerated tributaries or effluent inputs contribute to the river's dissolved oxygen budget. Errors in the magnitude of any of these processes or inputs can translate into errors in the simulated dissolved oxygen concentration.

The seasonally averaged longitudinal profile plots for dissolved oxygen do not show an obvious effect from any individual process (fig. 33). The sources and sinks of dissolved oxygen in the reservoir reach of the Tualatin River tend to offset over the course of a 6-month summer season.

The measured and simulated time series of dissolved oxygen concentrations are much more variable than the seasonally averaged concentration and provide better feedback on model performance. Simulated and measured time series of dissolved oxygen concentration are shown in figures 34, 35, and 36 for RMs 16.2, 5.5, and 3.4 (Elsner Road, Stafford Road, and Oswego diversion dam), respectively. For RMs 16.2 and 5.5, the measured data are means of all individual measurements at less than 10 feet depth, typically at 3, 6, and 9 feet; simulated concentrations are volume-weighted means from the top 10 feet of the model grid. At RM 3.4, the measured data are from a continuous monitor installed next to the fish ladder; the simulated concentrations are from the surface layer, which is the water that flows over the dam and through the fish ladder.

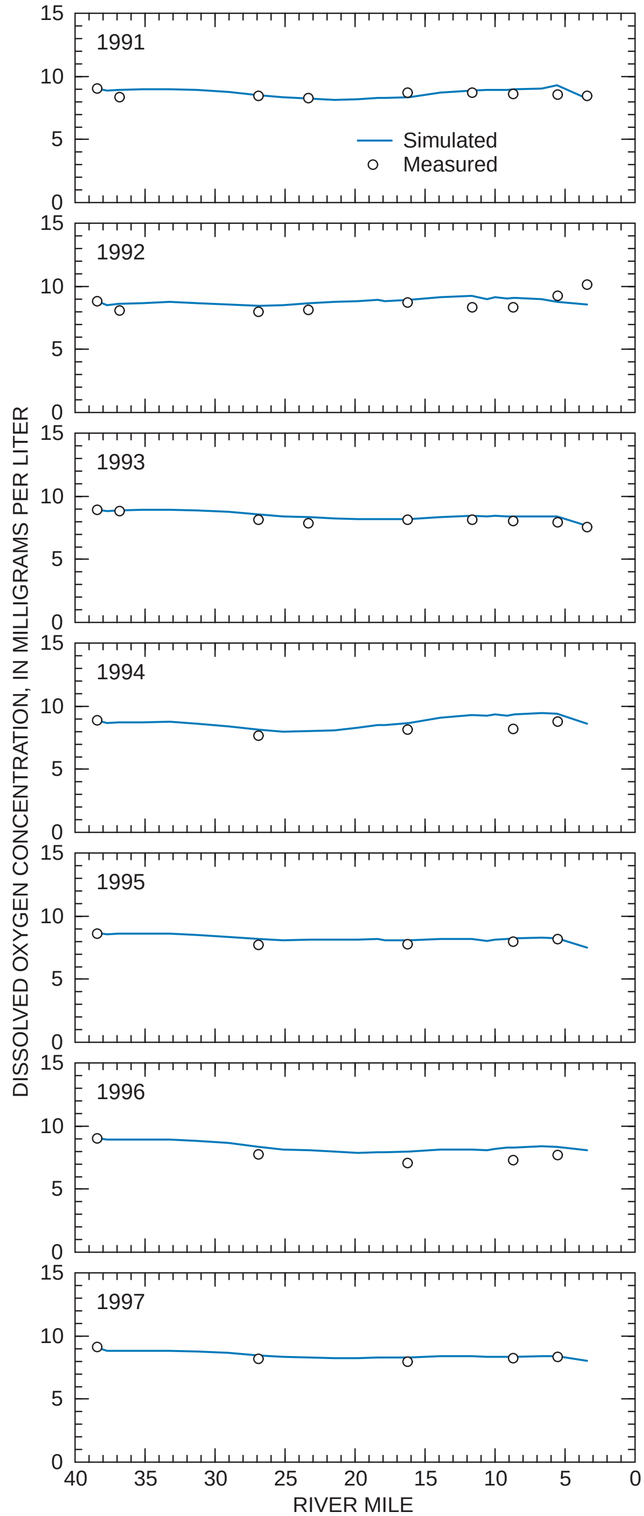

Figure 33. Simulated and measured mean dissolved oxygen concentrations as a function of river mile for May-October of 1991–1997. (Measured concentrations are the mean of all discrete measurements at less than 10 feet depth (typically 3, 6, and 9 feet). Simulated concentrations are volume-weighted means, also from the top 10 feet of the water column.)

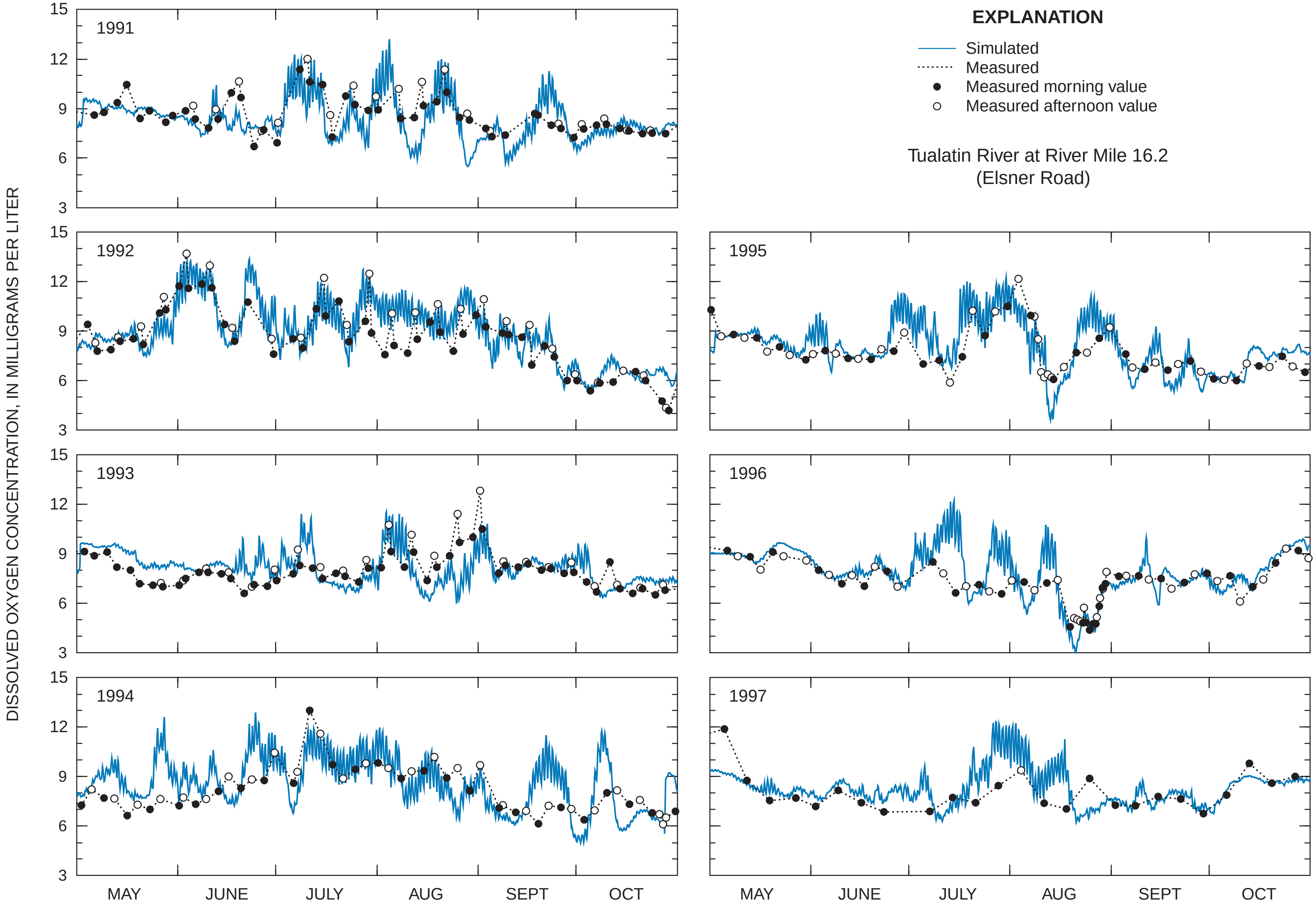

Figure 34. Simulated and measured dissolved oxygen concentration at river mile 16.2 (Elsner Road) for May-October of 1991–1997. (Measured concentrations are the mean of all discrete measurements at less than 10 feet depth (typically 3, 6, and 9 feet). Filled symbols are from morning measurements; open symbols are from afternoon measurements. Simulated concentrations (every 4 hours) are volume-weighted means from the top 10 feet of the water column.)

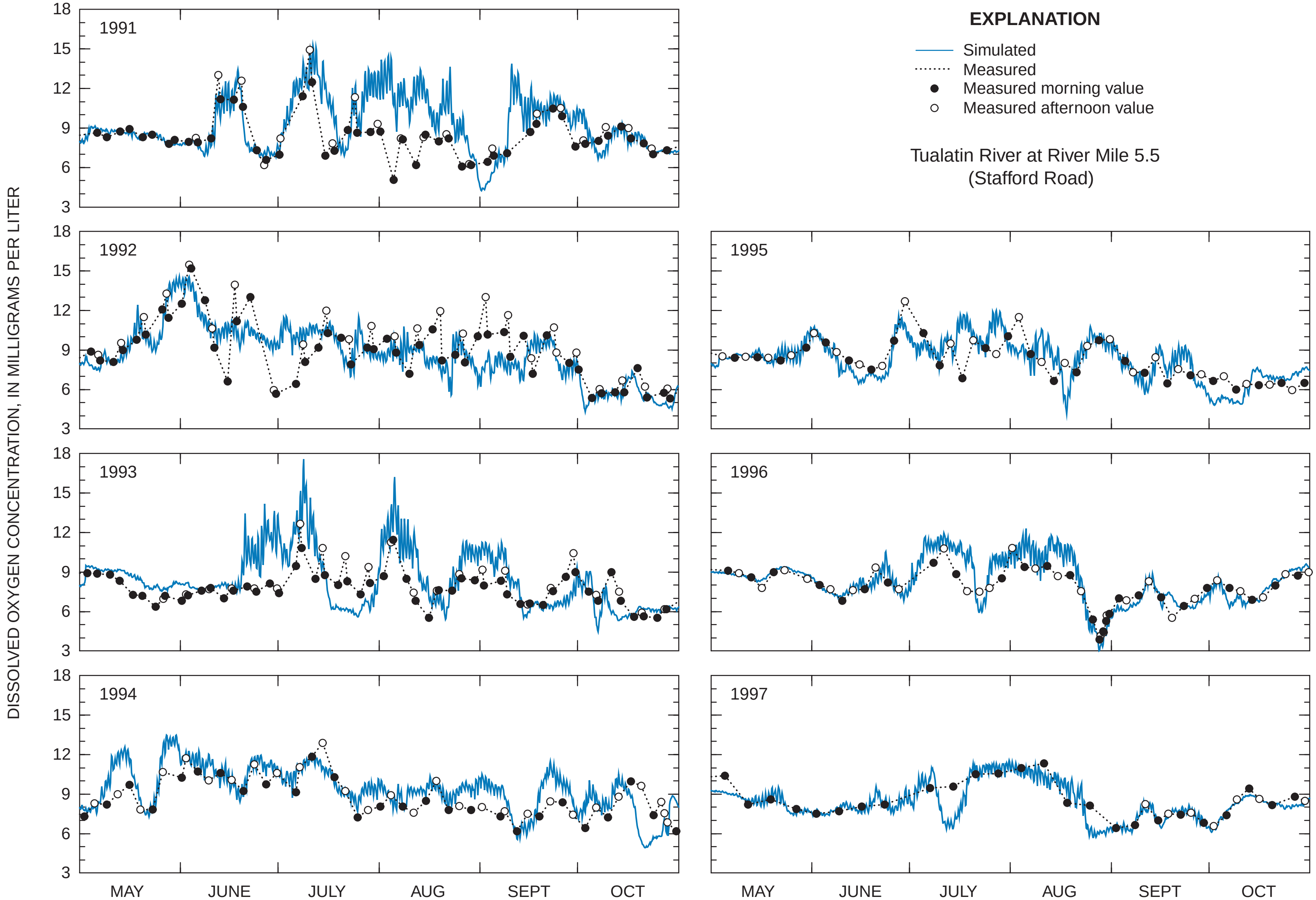

Figure 35. Simulated and measured dissolved oxygen concentration at river mile 5.5 (Stafford Road) for May-October of 1991–1997. (Measured concentrations are the mean of all discrete measurements at less than 10 feet depth (typically 3, 6, and 9 feet). Filled symbols are from morning measurements; open symbols are from afternoon measurements. Simulated concentrations (every 4 hours) are volume-weighted means from the top 10 feet of the water column.)

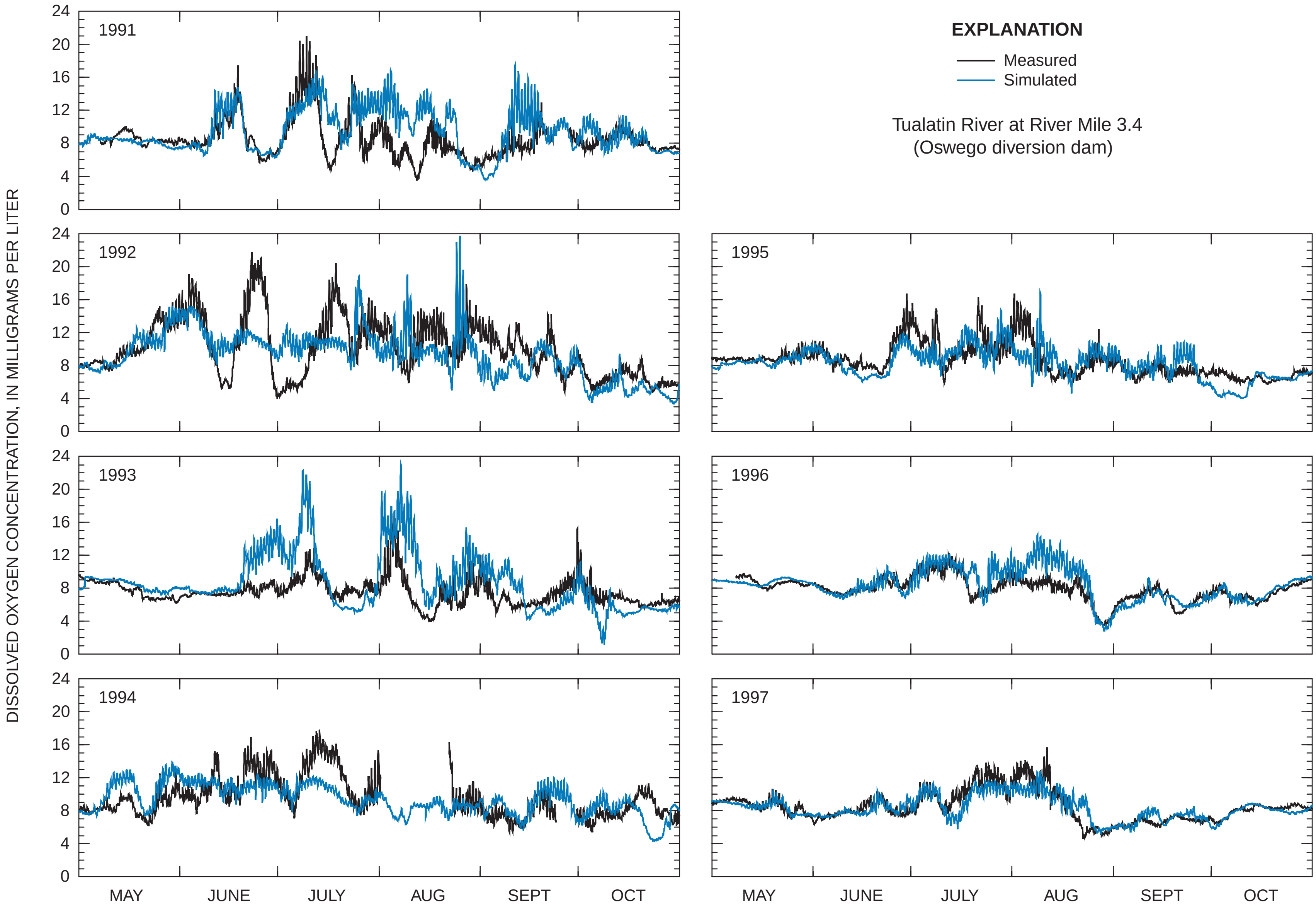

Figure 36. Simulated and measured hourly dissolved oxygen concentrations at river mile 3.4 (Oswego diversion dam) for May-October of 1991–1997.

The ability of the model to accurately simulate dissolved oxygen concentrations in the Tualatin River at these sites varies from excellent to simply acceptable, with a few limited periods where model performance is poor. Values of RMSE are no lower than 0.85 mg/L (table 5), showing how difficult it is to simulate a constituent like dissolved oxygen which is the product of so many complex processes. The fit statistics, however, do not quantify some of the model's more important accomplishments. In particular, the model does a good job simulating dissolved oxygen concentrations in October, when concentrations often are in danger of violating the State standard of 6.5 mg/L. In addition, the model usually simulates the increases in dissolved oxygen associated with algal blooms relatively well; model performance for some blooms is better than for others. Finally, the simulated dissolved oxygen concentrations during the periods of ammonia release from the Rock Creek WWTP in August of 1995 and 1996 compares very well with the measured concentrations.

When the model does not simulate the dissolved oxygen concentration well, the poor performance typically is due to inaccuracies in the size of the simulated algal population. During some periods, like August of 1991 or June and July of 1992, the poor fit is due to known problems in simulating phytoplankton/zooplankton interactions, as discussed previously. Many of the inaccuracies in dissolved oxygen at RM 3.4 probably stem from this same source, as no calibration data were available for phytoplankton or zooplankton downstream of RM 5.5. Whatever the cause, a high correlation exists between the errors in simulating the algal population and the errors in simulating the dissolved oxygen concentration (table 8). All but one of these correlations at RMs 16.2 and 5.5 is statistically significant at the 99 percent confidence level; the other one is significant at the 95 percent confidence level. These correlations indicate that between 14 and 64 percent (mean = 48 percent) of the error in simulated dissolved oxygen concentrations is directly attributable to error in the simulated algal population. Clearly, to do a better job with dissolved oxygen, the focus must be on doing a better job with the phytoplankton. This is evident in the fit statistics as well; the best fits were achieved for 1996 and 1997, summers in which the algal population was the smallest and the most accurately simulated.

WWTP Ammonia Discharge Events

One of the more important uses of the USGS Tualatin River model has been to quantify the effects of discharging various hypothetical loads of ammonia from each of the USA WWTPs (Rounds and Wood, 1998). These model scenarios quantified the ability of the river to assimilate and transport ammonia loads under a wide range of conditions and are the foundation upon which a revised TMDL for ammonia is being built (Oregon Department of Environmental Quality, 2001). The original ammonia TMDL was not sufficiently protective of dissolved oxygen conditions in the river. All of these model scenarios were based on the calibrations from 1991–1993, but with an updated nitrification rate.

Because the model is being used as the basis for such important regulatory and management strategies, it is absolutely critical that the model accurately simulates the effects of small and large loads of ammonia. The ammonia release events from the Rock Creek WWTP in August of 1995 and 1996 provide an

Table 8. Correlation coefficient for the residual in dissolved oxygen concentration against the residual in chlorophyll-*a* concentration

[RM, river mile; ✓✓, indicates statistical significance at the 99 percent confidence level; ✓, indicates statistical significance at the 95 percent confidence level.]

Year	RM 16.2 (Elsner Road)	Statistical significance	RM 5.5 (Stafford Road)	Statistical significance
1991	0.57	✓✓	0.78	✓✓
1992	.38	✓✓	.80	✓✓
1993	.80	✓✓	.79	✓✓
1994	.79	✓✓	.57	✓✓
1995	.70	✓✓	.78	✓✓
1996	.79	✓✓	.71	✓✓
1997	.48	✓	.55	✓✓

excellent data set to evaluate model performance. An examination of figures 18 and 19 for ammonia, and figures 34-36 for dissolved oxygen, show that the model did an excellent job simulating the effects of those ammonia loads as they were transported through the reservoir reach. The nitrification rate of 0.11 day^{-1} used by the model, therefore, is accurate. For 1995, as mentioned during the discussion for algae, the algal growth rates had to be adjusted in order to retain good fits for both algae and dissolved oxygen during the period of the ammonia release; this adjustment was necessary more for the algae than for the ammonia. For 1996, the model predicted the ammonia and dissolved oxygen concentrations closely without any special attention to the period of the ammonia release. The model results, therefore, are sufficiently accurate to be used as the basis for the revised ammonia TMDL.

Suppression of Algal Growth in 1996

During February of 1996, the Pacific Northwest experienced a period of major flooding. The Tualatin River overflowed its banks and flooded many low-lying areas, including many populated areas in and around Tualatin, Oregon. The rain events that contributed to this flooding also caused many landslides in the Coast Range mountains. These landslides contributed large quantities of solids to nearby streams, which continued to be more turbid than normal for the rest of the year. Even the water drawn from Henry Hagg Lake for purposes of flow augmentation was more turbid than normal, and such turbidity persisted for the entire summer of 1996.

Figures 30-32 show that the model overestimates the chlorophyll-*a* concentration during 1996 and that the measured levels of chlorophyll-*a* in 1996 are lower than in all the other summers. The increased turbidity may have suppressed algal growth in the reservoir reach of the Tualatin River that summer. Turbidity can suppress algal growth in at least two ways. First, decreased light penetration tends to suppress algal growth. Increased light absorption by more dissolved, colloidal, and particulate material in the water decreases the amount of solar energy available for photosynthesis. Some, but probably not all, of that effect was included in the model simulations, as light extinction is a function of the suspended solids concentration. The baseline light extinction in the model, however, is based on data from 1991–1993 and may not have been large enough to represent 1996 conditions. Second, increased turbidity and suspended solids concentrations can affect algal growth by sorbing or coprecipitating some of the orthophosphate and making that phosphorus less available for algal growth. Recent research has shown that some fraction of the phosphorus associated with colloidal particles in the Tualatin River and its tributaries is a coprecipitate (Mayer, 1995). Normally, the influence of phosphorus sorption is minor in the Tualatin River. Because of that, and some problems with the sorption code in the model, no phosphorus sorption was included in these simulations (Rounds and others, 1999). If some of the orthophosphate were bound up in colloidal materials, algal growth could have been suppressed. Indeed, simulated levels of orthophosphate were generally too high during 1996. Lower levels would have decreased the algal population, which was simulated to be too large.

SUMMARY

The USGS Tualatin River model, previously calibrated for the May through October periods of 1991, 1992, and 1993, was modified slightly and extended to simulate streamflow, water temperature, and water quality for the May through October periods of 1991–1997. This 42-month time frame includes a wide range of hydrologic and climatic conditions, even wider than the original 1991–1993 period. The summers of 1996 and 1997, which were “wetter” than normal and produced higher flow conditions than the previous 5 years, may represent conditions that are more typical of the next 20 years, as the Pacific Northwest may be entering a period of higher-than-normal precipitation. These more recent summers also are more representative in terms of wastewater treatment plant operations (size and efficiency) than were the summers of 1991–1993 when these plants were undergoing expansions and state-of-the-art upgrades. To retain the value of the USGS Tualatin River model as a management tool and build upon the knowledge gained from the original modeling study, the model was extended to cover the summers of 1994–1997.

The model continues to simulate the flow and water temperature of the river with high accuracy. These are important factors that influence the river’s water quality. Conservative tracers such as chloride and dissolved solids also are simulated well, indicating that all significant sources for these constituents are included in the model. The simulation of ammonia and nitrate matches the measured data very closely. As was the case in the original study, the simulation of nitrate concentrations during periods of very warm weather and low streamflow might be improved by adding a denitrification algorithm to the model. The revised ammonia nitrification rate of 0.11 day^{-1} used in this study allowed accurate simulations of both ammonia and dissolved oxygen during periods when the instream ammonia concentration was high. This demonstrates the utility of the model, as model scenarios form the basis for a revision of existing Total Maximum Daily Loads of ammonia for the Tualatin River.

Model results (and measured nitrogen concentrations) indicate that nitrogen never was a limiting nutrient for algal growth during this period. Light conditions limit algal growth between blooms and at points deeper than about 10 feet depth. Phosphorus was found to limit algal growth, effectively placing a cap on the size of algal blooms, during periods when light, temperature, and travel-time constraints favored algal blooms. Measured and simulated total phosphorus concentrations compared favorably. The model was able to simulate the timing, duration, and relative size of algal blooms with sufficient accuracy to lend insight into the algal dynamics of the river. Some error in the simulated algal population size is due to the decision not to introduce complexities into the model that were not based on measured data. Indeed, the model might have simulated the algal population more closely if more than one algal type were simulated. Roughly half of the error in the simulated orthophosphate and dissolved oxygen concentrations is directly attributable to error in the simulated algal population size. Despite these errors, the simulated dissolved oxygen concentration often matched the measured data well, especially during the sensitive late-summer period and when the phytoplankton population was relatively small in 1996 and 1997. Improvements to the model's ability to simulate phytoplankton, however, would further enhance the model's accuracy with respect to other constituents such as dissolved oxygen.

Future work with this model will include enhancements to the algorithms that describe algae and sorption. The inclusion of several algal types will probably improve the accuracy of the simulation and allow the seasonal variation in the algal growth rate used by this version to be discarded. Improvements to the algorithms that describe phosphorus sorption are necessary before this process can be adequately simulated. The inclusion of an accurate phosphorus sorption mechanism might improve the model's ability to simulate nutrient limitations to algal growth by better describing the amount of bioavailable phosphorus. Through such enhancements and updates, the USGS Tualatin River model will continue to provide river managers and regulators with the information they need to protect the values inherent to this river system.

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