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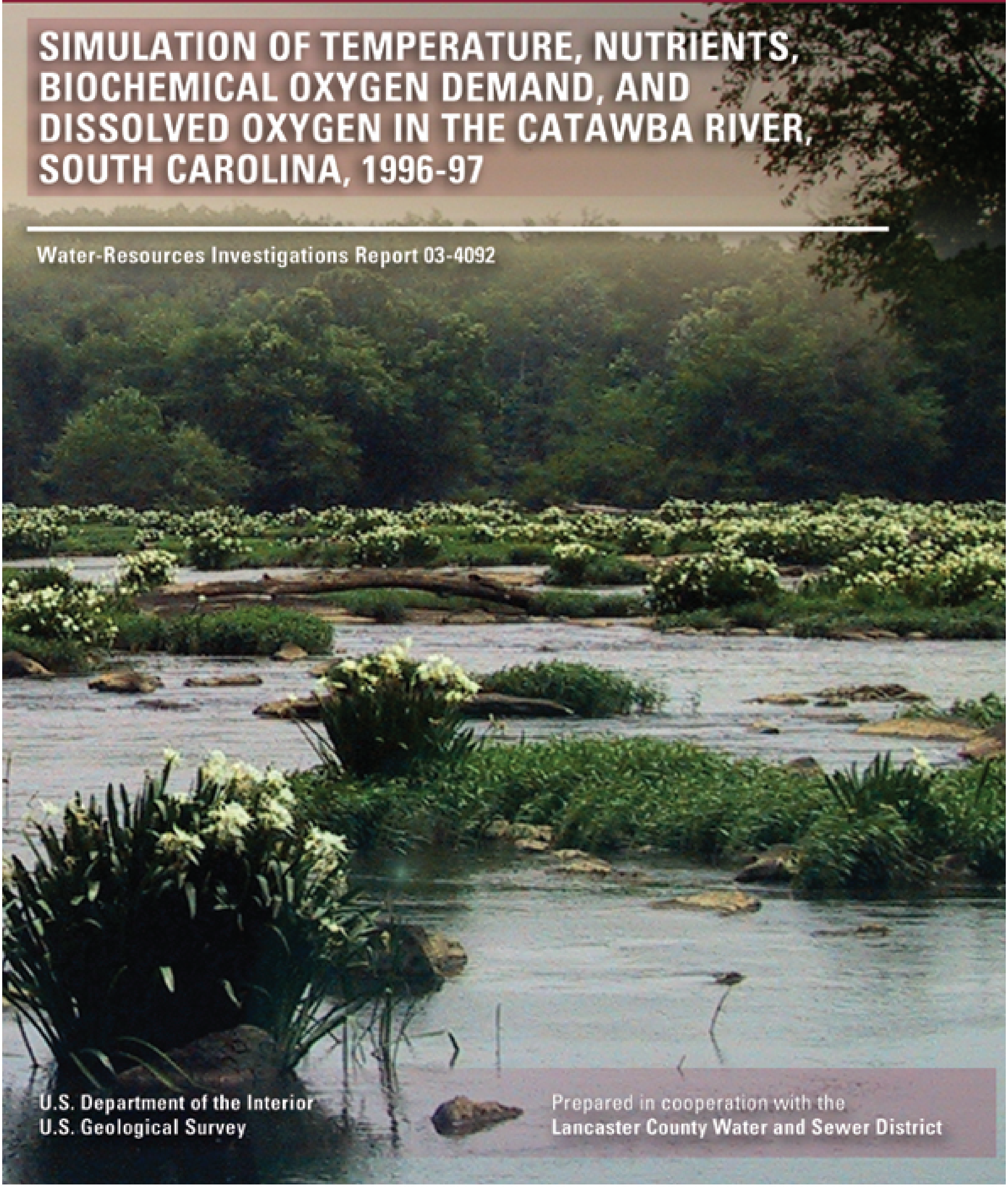
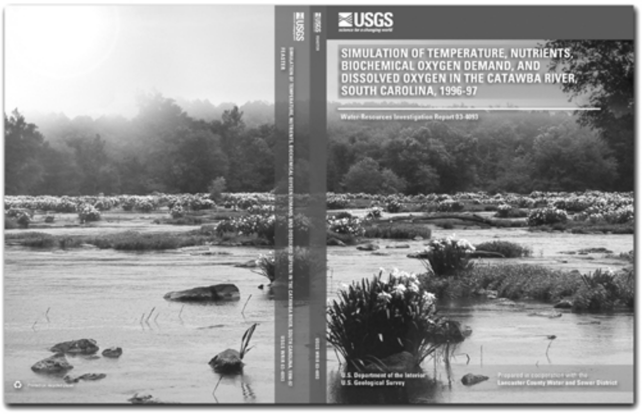
SIMULATION OF TEMPERATURE, NUTRIENTS, BIOCHEMICAL OXYGEN DEMAND, AND DISSOLVED OXYGEN IN THE CATAWBA RIVER, SOUTH CAROLINA, 1996-97

Water-Resources Investigations Report 03-4092



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

Prepared in cooperation with the
Lancaster County Water and Sewer District



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COVER PHOTOGRAPH: The Catawba River at Landsford Canal State Park, Chester County, South Carolina, May 31, 2001.
(*Photograph by Yvon H. Feaster*)

Simulation of Temperature, Nutrients, Biochemical Oxygen Demand, and Dissolved Oxygen in the Catawba River, South Carolina, 1996–97

By Toby D. Feaster, Paul A. Conrads, Wladmir B. Guimaraes, Curtis L. Sanders, Jr., and Jerad D. Bales

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U.S. GEOLOGICAL SURVEY

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Lancaster County Water and Sewer District

Columbia, South Carolina
2003

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, Secretary

U.S. GEOLOGICAL SURVEY
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CONVERSION FACTORS, TEMPERATURE, AND VERTICAL DATUM

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeters (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	4,047	square meter (m^2)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
cubic foot (ft^3)	0.02832	cubic meter (m^3)
gallon (gal)	3.785	liter (L)
acre-foot (acre-ft)	1.233×10^3	cubic meter (m^3)
	Flow Rate	
million gallons per day (Mgal/d)	1.547	cubic feet per second (ft^3/s)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)

Temperature can be converted to degrees Fahrenheit (°F) or degrees Celsius (°C) by using the following equations:

$$°F = (1.8 \times °C) + 32$$
$$°C = (°F - 32) / 1.8$$

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Abbreviations and Acronyms

BLTM	Branched Lagrangian Transport Model
BOD	biochemical oxygen demand
BOD_5	5-day biochemical oxygen demand
$CBOD_u$	ultimate carbonaceous biochemical oxygen demand
CSOD	community substrate oxygen demand
DEC	Duke Energy Corporation
DO	dissolved oxygen
DSS	decision support system
FW	freshwater
ft/d	foot per day
ft/s	foot per second
$(gO_2/m^2)/d$	grams of oxygen per square meter per day
GPP	gross primary production
GPS	global positioning system
HCR	hydrograph control release
lb/d	pound per day
mi/hr	mile per hour
mL/min	milliliter per minute
NAWQA	National Water-Quality Assessment Program
NPDES	National Pollutant Discharge Elimination System
ppm	part per million
RM	river mile
SCDHEC	South Carolina Department of Health and Environmental Control
SOD	sediment oxygen demand
TKN	total Kjeldahl nitrogen

TMDL	total maximum daily load
ton/d	ton per day
UOD	ultimate oxygen demand
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey

Abbreviated water-quality units:

mg/L, milligram per liter	A unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water.
µg/L, microgram per liter	1,000 micrograms per liter is equivalent to 1 milligram per liter.
kilopascal	1,000 units of pressure
Langley	A unit of illumination used to measure temperature, equal to 1 gram calorie per square centimeter of irradiated surface.

Simulation of Temperature, Nutrients, Biochemical Oxygen Demand, and Dissolved Oxygen in the Catawba River, South Carolina, 1996–97

By Toby D. Feaster, Paul A. Conrads, Wladmir B. Guimaraes, Curtis L. Sanders, Jr., and Jerad D. Bales

ABSTRACT

Time-series plots of dissolved-oxygen concentrations were determined for various simulated hydrologic and point-source loading conditions along a free-flowing section of the Catawba River from Lake Wylie Dam to the headwaters of Fishing Creek Reservoir in South Carolina. The U.S. Geological Survey one-dimensional dynamic-flow model, BRANCH, was used to simulate hydrodynamic data for the Branched Lagrangian Transport Model. Water-quality data were used to calibrate the Branched Lagrangian Transport Model and included concentrations of nutrients, chlorophyll *a*, and biochemical oxygen demand in water samples collected during two synoptic sampling surveys at 10 sites along the main stem of the Catawba River and at 3 tributaries; and continuous water temperature and dissolved-oxygen concentrations measured at 5 locations along the main stem of the Catawba River.

A sensitivity analysis of the simulated dissolved-oxygen concentrations to model coefficients and data inputs indicated that the simulated dissolved-oxygen concentrations were most sensitive to water-temperature boundary data due to the effect of temperature on reaction kinetics and the solubility of dissolved oxygen. Of the model coefficients, the simulated dissolved-oxygen concentration was most sensitive to the biological oxidation rate of nitrite to nitrate.

To demonstrate the utility of the Branched Lagrangian Transport Model for the Catawba River, the model was used to simulate several water-quality scenarios to evaluate the effect on the 24-hour mean dissolved-oxygen concentrations at selected sites for August 24, 1996, as simulated during the model calibration period of August 23–27, 1996. The first scenario included three loading conditions of the major effluent discharges along the main stem of the Catawba River—(1) current load (as sampled in August 1996); (2) no load (all point-source loads were removed from the main stem of the Catawba River; loads from the main tributaries were not removed); and (3) fully loaded (in accordance with South Carolina Department of Health and Environmental Control National Discharge Elimination System permits). Results indicate that the 24-hour mean and minimum dissolved-oxygen concentrations for August 24, 1996, changed from the no-load condition within a range of -0.33 to 0.02 milligram per liter and -0.48 to 0.00 milligram per liter, respectively. Fully permitted loading conditions changed the 24-hour mean and minimum dissolved-oxygen concentrations from -0.88 to 0.04 milligram per liter and -1.04 to 0.00 milligram per liter, respectively. A second scenario included the addition of a point-source discharge of 25 million gallons per day to the August 1996 calibration conditions. The discharge was added at S.C. Highway 5 or at a location near Culp Island (about 4 miles downstream from S.C. Highway 5) and had no significant effect on the daily mean and minimum dissolved-oxygen concentration.

A third scenario evaluated the phosphorus loading into Fishing Creek Reservoir; four loading conditions of phosphorus into Catawba River were simulated. The four conditions included fully permitted

and actual loading conditions, removal of all point sources from the Catawba River, and removal of all point and nonpoint sources from Sugar Creek. Removing the point-source inputs on the Catawba River and the point and nonpoint sources in Sugar Creek reduced the organic phosphorus and orthophosphate loadings to Fishing Creek Reservoir by 78 and 85 percent, respectively.

INTRODUCTION

The Catawba River Basin is a valuable resource for North and South Carolina. The headwaters of the Catawba-Wateree River begin in western North Carolina at the foot of the Blue Ridge Mountains (fig. 1). The Catawba River flows into central South Carolina and becomes the Wateree River at Lake Wateree Dam in Kershaw County. The river and its tributaries provide water for many uses, such as hydroelectric-power generation, recreation, water supply, wildlife and fish habitats, and wastewater assimilation. In recent years, a significant increase in industrial and residential development has occurred in the Catawba River Basin in the Charlotte-Rock Hill area of North and South Carolina. Along with the economic benefits that accompany increased development are increased pressures on the area’s natural resources. Streams and reservoirs in the Charlotte-Rock Hill area are increasingly stressed because of the rapid population and commercial growth in the area (Braun, 2000). As a result, there is increased concern about the effects of point- and nonpoint-source pollutant loadings to the Catawba River and its tributaries downstream from the Lake Wylie Dam.

In 1996, seven point sources discharged treated industrial and municipal wastewater effluents into the Catawba River between Lake Wylie Dam and the headwaters of Fishing Creek Reservoir (South Carolina Department of Health and Environmental Control, 1996). Six of the seven point sources were classified as major discharges of flows in excess of 1.0 million gallons per day (Mgal/d; fig. 2). Most wastewater discharges, although treated, still cause an increased demand for dissolved oxygen (DO) and a corresponding decrease of DO concentrations in the river. To protect ecological health, the South Carolina Department of Health and Environmental Control (SCDHEC) regulates these point-source discharges. Currently, the quantity and quality of the discharges allowed in the Catawba River are based on results from a steady-state flow and water-quality model developed by Davis and Floyd, Inc. (1984), which the SCDHEC uses to simulate the water-quality conditions in the Catawba River from the Lake Wylie Dam to the headwaters of Fishing Creek Reservoir (Borders, 1985).

Numerical models that can simulate the unsteady flows, such as those released from the Lake Wylie Dam, are needed to more accurately estimate the dynamic conditions in the Catawba River. Consequently, in 1995 the U.S. Geological Survey (USGS), in cooperation with the Lancaster County Water and Sewer District, initiated a study to (1) develop a hydraulic model to compute the unsteady hydraulic properties and (2) develop a model to simulate water temperature, mass transport, and fate of nutrients, biochemical oxygen demand (BOD), and DO. The results from these simulations will assist water-resource managers in estimating the effects of selected inflows and point-source effluent loadings on DO concentrations in the Catawba River.

Purpose and Scope

This report presents the results of a study to simulate the transport and chemical transformation of DO concentrations in a free-flowing segment of the Catawba River from Lake Wylie Dam to the headwaters of Fishing Creek Reservoir, and organic and inorganic (ortho) phosphorus loadings to Fishing Creek Reservoir in South Carolina. Analyses of the hydrologic, meteorologic, and water-quality data collected during this investigation are discussed, and documentation of the calibration, validation, and application of the one-dimensional, unsteady-flow model and water-quality model used in the simulation are presented.

The modeling effort was completed in three phases. The scope of the first phase included data collection during August 1996 and July 1997. The scope of the second phase was to calibrate and validate the unsteady-flow model, BRANCH (Schaffranek and others, 1981), and the mass-transport model, the enhanced Branched Lagrangian Transport Model (BLTM) (Jobson and Schoellhamer, 1993; Jobson, 1997), and to simulate the movement of a conservative constituent in the system. The scope of the third phase was to calibrate and validate the water-quality model, BLTM, to simulate the fate and transport of nonconservative constituents, such as nutrients, BOD, and DO.

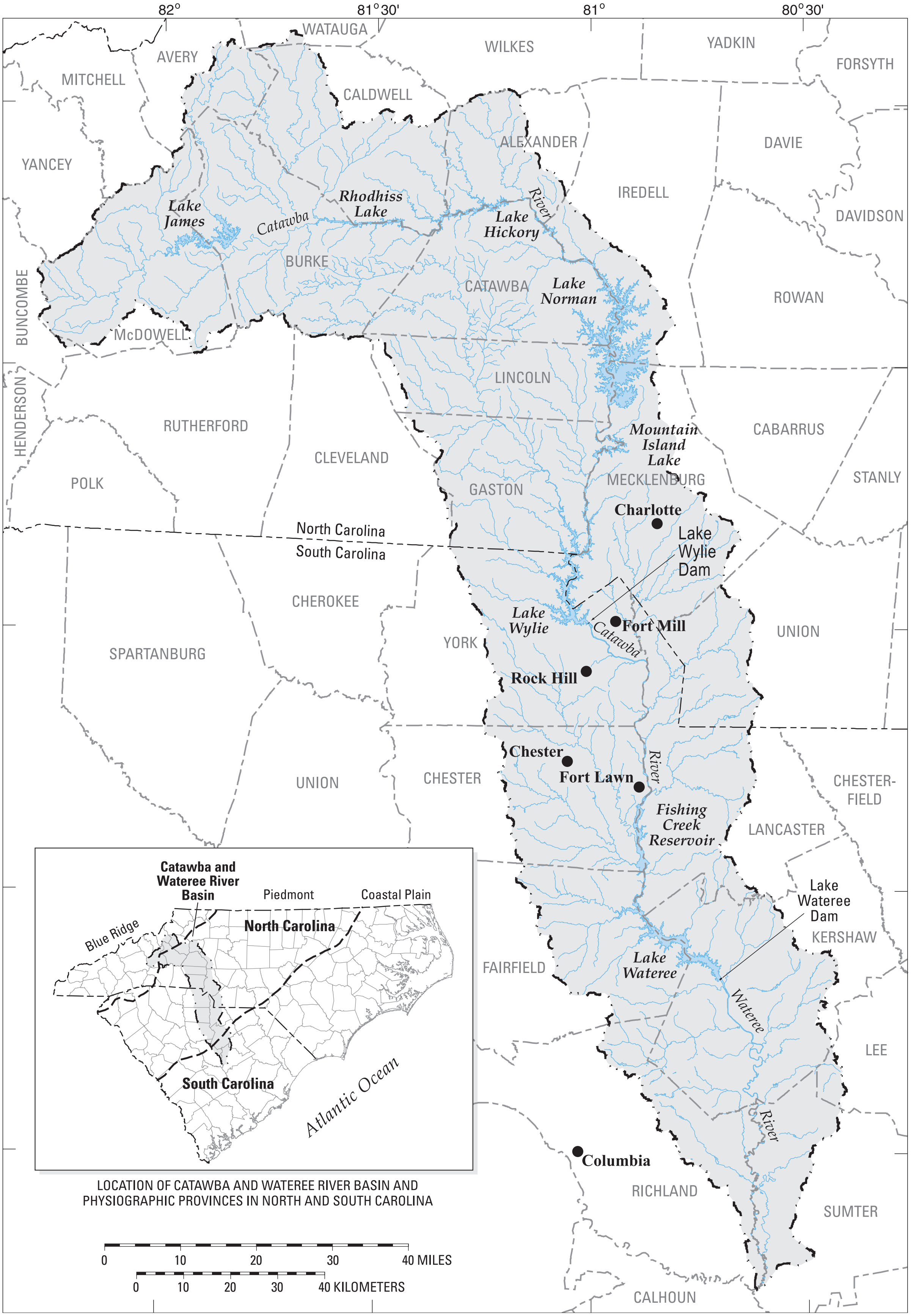
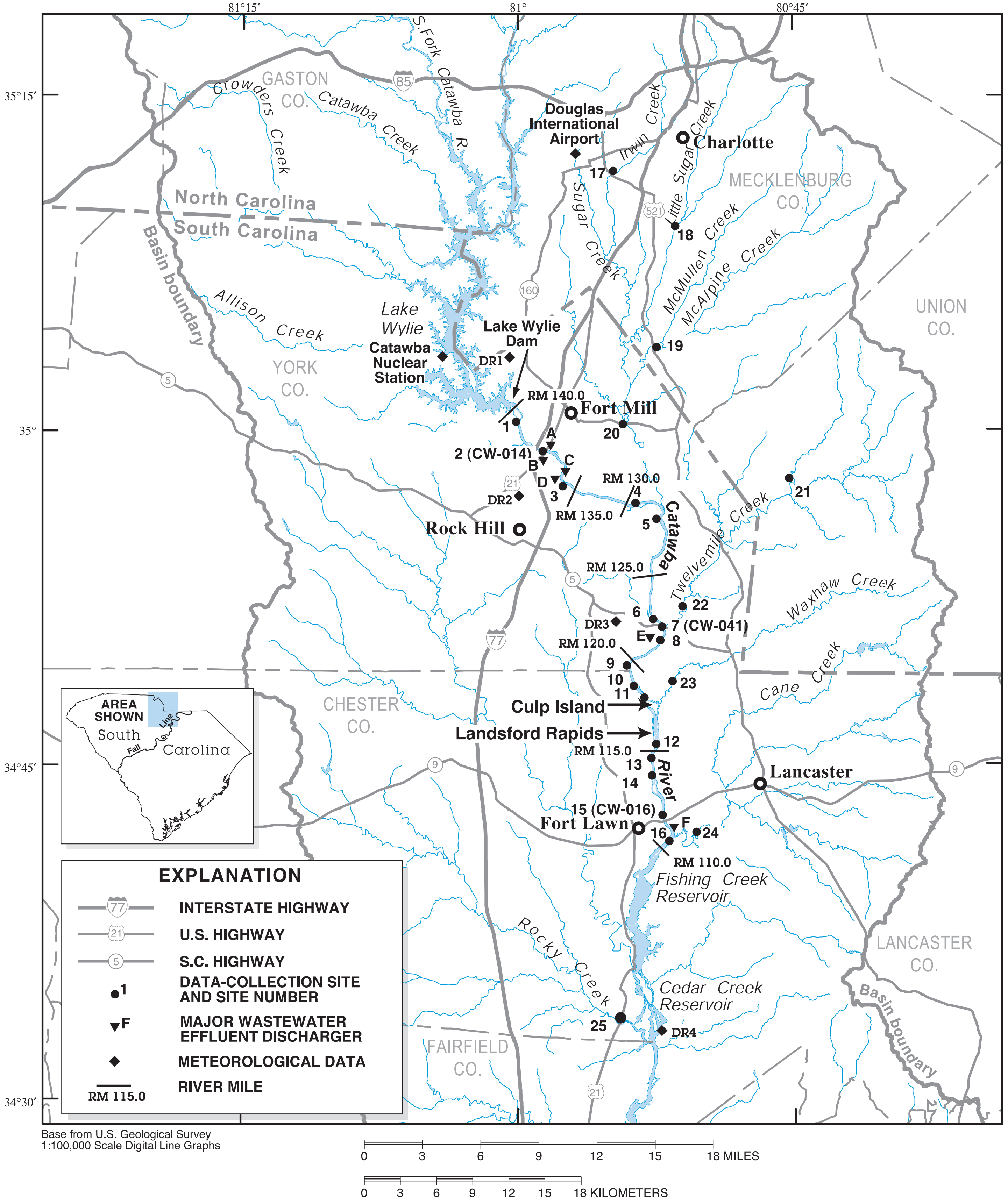


Figure 1. Catawba and Wateree River Basins, North and South Carolina.



Base from U.S. Geological Survey
1:100,000 Scale Digital Line Graphs

Figure 2. Locations of data-collection sites and point-source discharges in the Catawba River Basin, North and South Carolina.

The model was used to simulate water-quality conditions in the study reach for selected scenarios. These scenarios included (1) the effects of changes in loads from existing point sources on DO concentrations in the river, (2) the effect of additional major point-source discharges to the river on DO concentrations in the river, and (3) the effects of various loading scenarios on phosphorus loading to Fishing Creek Reservoir at the downstream end of the study reach.

Description of Study Area

The Catawba-Wateree River Basin begins in the Blue Ridge Province of North Carolina and flows through a series of lakes in the Piedmont Province of North and South Carolina before flowing into the Coastal Plain Province of South Carolina (fig. 1). The study area includes about 31 miles (mi) of the Catawba River from Lake Wylie Dam (river mile [RM] 141.0) to the headwaters of Fishing Creek Reservoir (RM 110.3; fig. 2), which is the longest free-flowing reach of the river. The annual mean flow at USGS streamgaging station 02146000, Catawba River near Rock Hill, S.C. (RM 137.6), is 4,350 cubic feet per second (ft^3/s) and varies from a minimum daily mean streamflow of 227 ft^3/s to a maximum daily mean streamflow of 127,000 ft^3/s, based on 105 years of streamflow record from water years[1] 1896 to 2000 (table 1; Cooney and others, 2001).

The river-channel geometry (width and depth) varies throughout the study area. The width of the Catawba River at the Lake Wylie Dam tailrace is about 1,200 feet (ft). Over the next approximately 1,000 ft, the river narrows to about 500 ft. The channel remains approximately 500 ft wide for the next 23 mi to the head of Landsford Rapids just downstream from the confluence with Waxhaw Creek. Upstream from the confluence with Waxhaw Creek, the Catawba River consists of a rocky bottom with a series of pools and riffles. Downstream from Waxhaw Creek, the width of the river increases to about 1,500 ft through the Landsford Rapids and then constricts to an average width of about 600 ft at the foot of the rapids. Below the rapids, the Catawba River varies in width from about 400 to 600 ft to the bridge at S.C. Highway 9, and then increases to a width of about 1,700 ft in the headwaters of Fishing Creek Reservoir about 1 mi downstream from S.C. Highway 9 (fig. 2). Backwater effects of Fishing Creek Reservoir occur just downstream from Landsford Rapids during low river stages.

Although there are 18 tributaries in this reach of the river, only the 4 tributaries that contribute most of the intervening flow were included in this study (fig. 2; table 2). In downstream order, these tributaries are Sugar Creek, Twelvemile Creek, Waxhaw Creek, and Cane Creek. Of these four major tributaries, the USGS has collected continuous streamflow data for Sugar and Twelvemile Creeks. Flows in Sugar and Twelvemile Creeks may vary considerably, and it has not been uncommon for flows in Sugar Creek to exceed 10,000 ft^3/s (U.S. Geological Survey, 1980; table 1; fig. 2). Sugar Creek is an urbanized basin that drains most of Charlotte, N.C., whereas Twelvemile and Waxhaw Creeks are rural basins. Twelvemile and Waxhaw Creeks have similar physiography, but flows

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[1] Water year is the period October 1 through September 30 and is identified by the year in which it ends.

Table 1. Maximum and mean streamflow, period of record, and drainage area for selected U.S. Geological Survey streamflow gaging stations in the Catawba-Wateree Basin, North and South Carolina

[mi^2, square mile; ft^3/s, cubic foot per second]

USGS station number (fig. 2)	Station name	Period of record	Drainage area (mi^2)	Maximum water-year[a] peak streamflow of record (ft^3/s)	Annual mean streamflow for period of record (ft^3/s)
02146000[b]	Catawba River near Rock Hill, S.C.	1896–2000	3,050	151,000	4,350
02146800[c]	Sugar Creek near Fort Mill, S.C.	1974–79	262	22,700	431
02146900[d]	Twelvemile Creek near Waxhaw, N.C.	1960–2000	76.5	9,970	74.2

[a] Water year is the period October 1 through September 30 and is identified by the year in which it ends.

[b] Cooney and others, 2001.

[c] U.S. Geological Survey Water-Data Report SC-79-1, 1980.

[d] Ragland and others, 2001.

Table 2. Drainage areas at selected locations in the Catawba River Basin, S.C.

Location on Catawba River (fig. 2)	Description	River mile[a]	Drainage area[b] (mi^2)
Lake Wylie Dam	Upstream study limit	141.0	3,020
Sugar Creek	Tributary	129.8	275
Twelvemile Creek	Tributary	122.8	170
Waxhaw Creek	Tributary	117.5	58
Cane Creek	Tributary	110.4	155
Headwaters of Fishing Creek Reservoir	Downstream study limit	110.3	3,850

[a] River mile locations of tributaries are at the confluence of the tributary and the Catawba River.

[b] Drainage areas were delineated and digitized on U.S. Geological Survey 7.5-minute topographic quadrangle maps (U.S. Geological Survey 1968a–e, 1969a–b, 1973). Drainage areas for tributaries were calculated at the confluence with the Catawba River.

are less on Waxhaw Creek because its drainage area is smaller. No streamflow data are available for Waxhaw Creek. The drainage area of the Catawba River increases about 830 square miles (mi^2) through the study reach. These four tributaries account for 658 mi^2 (table 2) or about 80 percent of the intervening drainage of the study area.

The land use and land cover of the Catawba River watershed from Lake Wylie Dam to the Fishing Creek Reservoir Dam (Hydrologic Unit Code 03050103-010; Bower and others, 1999) consists of 68.1 percent forested land, 12.9 percent urban land, 7.0 percent agricultural land, 6.8 percent water, and 5.2 percent scrub and(or) barren land (South Carolina Department of Health and Environmental Control, 1996). Urban land use is concentrated in the upper part of the basin.

The SCDHEC has classified the Catawba River from Lake Wylie to Fishing Creek Reservoir as freshwater (FW) with a daily average DO water-quality standard of not less than 5.0 milligrams per liter (mg/L) with an instantaneous minimum of 4.0 mg/L (South Carolina Department of Health and Environmental Control, 1998). As previously stated, seven point sources discharge treated industrial and municipal wastewater effluents into the Catawba River between Lake Wylie Dam and the headwaters of Fishing Creek Reservoir (South Carolina Department of Health and Environmental Control, 1996). Six of the seven point sources are classified as major dischargers of flows in excess of 1.0 Mgal/d.

Wastewater effluent contains many oxygen-consuming constituents, primarily ammonia and biodegradable organic substances. In terms of water-resource management, the assimilative capacity of a receiving stream is often expressed as pounds per day of ultimate oxygen demand (UOD) that can be assimilated without exceeding the State water-quality standard for DO concentrations. The UOD is the total, theoretical demand for oxygen from carbonaceous and nitrogenous sources. The SCDHEC National Pollutant Discharge Elimination System (NPDES) permit limits for wastewater-treatment facilities located along the study reach (fig. 2) are listed in table 3 (Svetlana Sindler, South Carolina Department of Health and Environmental Control, written commun., 2001). One of the dischargers (E, table 3) on the Catawba River has a hydrograph control release (HCR) permit. Under an HCR permit, dischargers are able to use the increased assimilative capacity during periods of higher streamflow. For example, for a streamflow of 700 ft^3/s, the discharger would be permitted to release 2,052 pounds per day (lbs/d) of UOD. For a streamflow of 2,500 ft^3/s, the discharger would be permitted to release 86,544 lbs/d of UOD.

Previous Studies

Many water-quality studies of the Catawba Basin have been conducted. In 1994, the USGS initiated a study of the Santee River Basin and coastal drainages, including the Catawba River Basin, as part of the National Water-Quality Assessment (NAWQA) Program. As part of the NAWQA Program, Maluk and others (1998) documented a retrospective analysis of data from 90 stations in the Santee River Basin and coastal drainages from 1973 to 1993. The Santee River Basin and coastal drainages study area was divided into four subbasins—the Broad, Catawba, Cooper, and Edisto River subbasins—for analysis. Nutrient concentrations were significantly higher in the Broad and Catawba River subbasins than in the other subbasins. Among 16 sites in the Catawba River subbasin, median ammonia nitrogen, nitrite-plus-nitrate, total Kjeldahl nitrogen (TKN), and total phosphorus concentrations were highest in Sugar Creek. In the late 1980's and 1990's, temporal trends indicated decreasing ammonia nitrogen and TKN concentrations in the Catawba River and Sugar Creek, probably resulting from improvements in wastewater-treatment plant effluent quality and a decrease in

Table 3. National Pollutant Discharge Elimination System permit limits for wastewater-treatment facilities on the Catawba River, S.C.

[Mgal/d, million gallons per day; BOD_5, 5-day biochemical oxygen demand; lbs/d, pounds per day; NH_3-N, ammonia nitrogen; mg/L, milligram per liter; DO, dissolved oxygen; UOD, ultimate oxygen demand; MR, monitor and report; —, not applicable; <, less than; HCR, hydrograph control release; ft^3/s, cubic feet per second; >, greater than]

Wastewater effluent discharger (fig. 2)	Pipe designation	Location (river mile)	Effluent amount (Mgal/d)	BOD_5 (lbs/d)	NH_3-N (mg/L)	DO (mg/L)	UOD (lbs/d)
A	001	137.3	3.65[a]	981	2.0	5.0	2,522
	002		1.97[a]	MR	MR	1.0	—
	003		41.2[a]	1,112[b]	—	4.0	—
B	001	137.0	1.5	375.3	20.0	2.0	1,710
	001[c]		2[c]	500.4[c]	20.0[c]	6.0[c]	2,275[c]
C	001	135.5	0.72[a]	5.0[a, d]	<1.0[a]	—	67.6
	002		.0045[a]	1.72	<1.0[a]	1.0	5.9
D	001	134.5	20.0	3,403	13.6	6.0	15,500
E	001	120.9	HCR[e]	26,082[a]	5.0[f]	1.0	—
	01A[g]		.188[a]	—	—	—	—
	01B[g]		—	—	—	—	—
F	101A	111.1	MR	—	—	—	—
	001		17.4[a]	2,920	310[h]	5.0	10,177
	002		0.23[a]	208	—	—	—

[a] Indicates an average submitted value not a permit limit.

[b] Maximum increase above intake background loading.

[c] New limits for pipe 001 began on March 1, 2001.

[d] BOD_5 concentration is in milligrams per liter.

[e] HCR is based on streamflow of the Catawba River (Q_r)
For streamflow of >815 ft^3/s, $UOD = 50.12 \times Q_r - 38{,}756.5$
For streamflow of <815 ft^3/s, $UOD = 0.34 \times Q_r + 1{,}814.3$

[f] When effluent (Q_d) is >20.0 Mgal/d, the NH_3-N limit is 5.0 mg/L; when Q_d is <20.0 Mgal/d, the NH_3-N limit is 794 lbs/d.

[g] Pipes 01A and 01B are internal discharges.

[h] Ammonia (NH_3-N) concentration is in pounds per day.

agricultural activities in the Sugar Creek subbasin. An analysis of the data indicated that the most significant decreases in ammonia nitrogen and TKN concentrations occurred after 1987. Increasing trends in nitrite-plus-nitrate concentrations occurred in Sugar Creek, which may be a result of changes in land use and(or) wastewater-treatment processes. Trends in phosphorus concentrations appear to be decreasing in the Catawba River and in Lake Wylie, probably as a result of the ban on phosphate detergents instituted in 1988 in North Carolina and in 1992 in South Carolina (Maluk and others, 1998).

The SCDHEC published a watershed water-quality management plan for the Catawba-Santee Basin that describes water-quality-related activities in the watershed and analyzed water-quality data collected in the watershed (South Carolina Department of Health and Environmental Control, 1996). Three SCDHEC monitoring stations (CW-014, CW-041, and CW-016) are located along the Catawba River below Lake Wylie Dam and upstream from Fishing Creek Reservoir at U.S. Highway 21 (RM 137.6), S.C. Highway 5 (RM 122.8), and S.C. Highway 9 (RM 111.4; fig. 2). Results of this basinwide study indicate that recreational uses are fully supported at all three SCDHEC stations. Aquatic life is only partially supported at SCDHEC station CW-014 due to intermittently low DO concentrations, probably as a result of hypolimnetic releases from Lake Wylie Dam. Aquatic life at SCDHEC station CW-041 is fully supported, and decreasing trends in 5-day biochemical oxygen demand (BOD_5), total phosphorus, and total nitrogen concentrations suggest improving conditions. Increasing trends in pH at SCDHEC station CW-041 also were noted. Farther downstream at SCDHEC station CW-016, although aquatic life is fully supported, the water quality may be threatened because of an observed decreasing trend in DO concentrations and an increasing trend in pH. However, decreasing trends in BOD_5, total phosphorus, total nitrogen, and fecal coliform bacteria concentrations suggest improving conditions at SCDHEC station CW-016 (South Carolina Department of Health and Environmental Control, 1996).

The North Carolina Department of Environment, Health, and Natural Resources (NCDEHNR) and the SCDHEC cooperated in a study of the water quality of Lake Wylie (North Carolina Department of Environment, Health, and Natural Resources and South Carolina Department of Health and Environmental Control, 1992). The objectives of this study were to identify patterns of nutrient loadings in the watershed, evaluate the assimilative capacity of Lake Wylie, and determine effective control strategies for protecting Lake Wylie as a water resource. Results of a chemical and physical survey indicated that the lake was threatened by eutrophic conditions resulting from high nutrient loads, especially in the embayments. The tributary arms and embayments consistently exhibited problematic biological responses to the nutrient loading. The South Fork Catawba River was identified as the major source of nutrients to Lake Wylie (fig. 2). The Catawba Creek arm of Lake Wylie consistently exhibited eutrophic conditions during the study. Catawba Creek and Crowders Creek had algal blooms and elevated nutrient and chlorophyll *a* concentrations. Eutrophic conditions also occurred in Allison Creek but not to the extent of those in the South Fork Catawba River, Catawba Creek, and Crowders Creek (North Carolina Department of Environment, Health, and Natural Resources and South Carolina Department of Health and Environmental Control, 1992).

The NCDEHNR also used numerical models to determine the assimilative capacity of Lake Wylie. The results of the modeling simulations indicated that control of point- and nonpoint-source loadings would be required to limit nutrient loads entering Lake Wylie. The eutrophic arms of Lake Wylie—the South Fork Catawba River and Catawba and Crowders Creeks—are nitrogen limited during the algal growing season because of elevated phosphorus loads from point and nonpoint sources. Catawba and Crowders Creeks are dominated by point-source nutrient loadings; whereas, point- and nonpoint-source loadings occur in the South Fork Catawba River. The Catawba River upstream from Lake Wylie has numerous point-source discharges, but the nutrient loads from the Catawba River are trapped in the upstream reservoirs. The loads from the South Fork Catawba River and Catawba and Crowders Creeks result in high concentrations of nutrients in Lake Wylie. From the results of this study, management strategies for Lake Wylie were developed (North Carolina Department of Environment, Health, and Natural Resources and South Carolina Department of Health and Environmental Control, 1992).

The effluent-discharge limits in the Catawba River Basin currently (2003) are based on a steady-state water-quality model from Lake Wylie Dam to the headwaters of Fishing Creek Reservoir (Davis and Floyd, Inc., 1984). The SEMCOG/QUAL II model (Brown and Barnwell, 1987) was calibrated with data collected by Davis and Floyd, Inc., (1984) in September and October 1982 and in October 1983 for their river study plan. Davis and Floyd, Inc., (1984) found that DO concentrations averaged 5.0 mg/L at the head of Landsford Rapids and in the headwaters to Fishing Creek Reservoir, whereas DO concentrations ranged from 6.0 to 8.0 mg/L throughout the remainder of the reach. The SCDHEC reviewed the Davis and Floyd, Inc., (1984) findings and developed wasteload allocations based on a steady-state flow of 702 ft^3/s in the Catawba River (Borders, 1985). Model simulations for fully permitted effluent limits predicted that the DO concentrations would decrease to 5.0 mg/L just upstream from Landsford Rapids, increase to about 7.0 mg/L through the rapids, and decrease to about 5.0 mg/L near the confluence of Cane Creek (Borders, 1985).

Chen and others (1999) developed a decision support system (DSS) to calculate total maximum daily loads (TMDL) of various pollutants for water-quality-limited sections of a river basin. The DSS included a dynamic watershed model, a database, and a consensus-building module. The DSS was applied to the Sugar Creek subbasin to compute a hypothetical TMDL. The TMDL for BOD was computed for upper and lower Sugar Creek and McAlpine Creek. A DO concentration greater than 5.0 mg/L was used as a criterion and BOD loading was the control parameter. Results indicated that for the upper Sugar Creek reach, 30-percent and 86-percent reductions in the nonpoint- and point-source BOD loads, respectively, are required to meet the water-quality criterion. For McAlpine Creek, 9-percent reduction in the point-source BOD load is adequate to meet the water-quality criterion. For the lower Sugar Creek reach, a 30-percent reduction in point-source BOD load is required to meet the water-quality criterion. The upper Sugar Creek reach appears to be the most water-quality-sensitive area in the Sugar Creek subbasin. Without some type of nonpoint-source control, the water-quality standard of 5.0 mg/L for DO will not be met. Chen and others (1999) explained that multiple solutions are possible for different combinations of point- and nonpoint-source loads. Combined total TMDLs for point- and nonpoint-source allocations may differ for various acceptable solutions.

During 1993–97, the USGS conducted an investigation of the relation between land use and stormwater quality in the city of Charlotte and Mecklenburg County, North Carolina. Samples were collected during four storm events per year at each site during the study and were analyzed for a wide range of constituents (Robinson and others, 1996; 1998). Annual loads and yields of selected nutrients and metals were computed for each of nine study basins (Bales and others, 1999).

Approach

An unsteady, one-dimensional water-quality model was developed for the 29.6-mile study reach between Lake Wylie Dam (RM 139.9) to the headwaters of Fishing Creek Reservoir (RM 110.3; fig. 2). Data were collected to construct, calibrate, and test the water-quality model. Most of the data were collected during two separate 5-day study periods—August 23–27, 1996, and July 11–15, 1997. Because of the influence of backwater at RM 110.3, RM 111.4 (Catawba River near Fort Lawn) was chosen as the downstream boundary for comparisons of simulated and measured water-quality data.

The four primary components of the one-dimensional water-quality model are flow, mass transport, water temperature, and water quality. Data required for the flow component include channel geometry (top width, cross-sectional area, stream length, and slope), inflows from major tributaries and point-source dischargers, flow at Lake Wylie Dam and at selected locations within the study reach, and water level at the downstream boundary of the study reach. For the mass-transport component, measurements of conservative dye concentrations through the study reach are needed for model calibration and testing. Data for the water-temperature component include continuous water temperature at the upstream and downstream study reach boundaries and at selected locations within the study reach; and meteorological data (daily high and low air temperatures, solar radiation, dewpoint temperature, wind speed and direction, and percentage of cloud cover). Data needed for the water-quality component of the model include estimates of reaeration, and concentrations of BOD, DO, ammonia, nitrate, organic nitrogen, orthophosphate, total phosphorus, and algal biomass (estimated from chlorophyll *a*) at the upstream boundary, at selected locations along the study reach, at the mouths of major tributaries to the study reach, and at major point-source discharges.

Study-reach channel geometry was field measured on several separate occasions during October 1995–May 1996, and river-reach lengths were obtained from USGS 7.5-minute topographic maps. These data were used to construct the computational grid for the model. The computational grid for the flow component was somewhat more detailed than for the other model components.

The model was calibrated and validated by using data collected during two intensive data-collection periods (August 23–27, 1996—calibration period, and July 11–15, 1997—validation period). Model calibration is accomplished by adjusting model parameters within a reasonable range until model results agree, to the extent possible, with observations. The model is considered to be validated if simulations agree with observations for a period distinct from those used for calibration without further adjustment of model parameters (Ditmars and others, 1987). The data-collection periods were chosen to meet specific criteria: (1) water temperature at or near the typical annual maximum value, (2) the occurrence of both steady and dynamic flows, and (3) the availability of effluent discharge data. The calibration and validation periods included 2 days of steady flow (for calibration and testing of the mass transport component) and 3 days of dynamic flows.

The flow component of the model was calibrated and validated by using measured discharge data from Lake Wylie Dam and water-level (stage) data from the headwaters of Fishing Creek Reservoir. Simulated flows were compared to streamflow data from two gaging stations in the study reach, and simulated water levels were compared to measurements at three stations in the study reach. Simulated flows were used in the simulation of mass transport, water temperature, and water quality.

The mass-transport component of the model was calibrated and validated by simulating the movement of a conservative constituent (rhodamine dye) through the study reach and comparing simulations with measurements of dye transport. Calibration of the mass-transport component required some slight adjustments to the flow model. The water-temperature component of the model was calibrated and validated by adjusting two wind-related parameters and by subsequent adjustment of calculated equilibrium temperatures. Water-quality simulations included the effects of reaeration, biological degradation, and chemical reaction, as well as physical dilution in response to varying streamflows. The reaction-kinetic parameters were adjusted in the calibration of the water-quality model to account for these processes.

Sensitivity analysis is the determination of the effects on model simulations of changes in model parameters and data. The sensitivity of simulated streamflow, time of travel, DO concentrations, and phosphorus concentrations to changes in selected model parameters was determined for the calibrated water-quality model.

Acknowledgments

The complexity of this study required interagency cooperation in addition to individual contributions. The authors thank Gerald White of Lancaster County Water and Sewer District and Henry Forrest (retired) of the Charlotte-Mecklenburg Utility Department for providing support and coordination of local interests in the Catawba River Basin area; the permitted dischargers for providing effluent data during the synoptic sampling surveys; the landowners along the Catawba River for allowing access to the river during the synoptic sampling; and Dave Lyles, Robert Langley, and Michael and Mary McAdams for allowing gages to be located on their respective properties during the data-collection phase of the study. In addition, thanks are extended to Philip Murphy, Mark Koenig, and the staff of the Science and Ecosystem Support Division, Region IV, U.S. Environmental Protection Agency, for determining time-of-travel, reaeration, and substrate and sediment oxygen demand for the lower Catawba River.

DATA-COLLECTION METHODS AND DATA ANALYSIS

A fairly extensive data set of continuous measurements, discrete measurements and sample analyses, and estimates of conditions was assembled to construct, calibrate, and validate the water-quality model (table 4). Data were collected during August 1996 and July 1997. The data collected included (1) continuous measurements of water level, streamflow, water temperature, and DO concentrations; (2) discrete measurements of discharge in four major tributaries and in the Catawba River; (3) estimates of streamflow for the four major tributaries and the intervening drainage area between Lake Wylie Dam and Sugar Creek (fig. 2); (4) time-of-travel information; (5) stream reaeration rates; (6) water chemistry; (7) substrate- and sediment-oxygen demand rates; (8) point-source discharge rates and chemistry; (9) meteorological data; and (10) channel geometry. The data-collection program is described in this section, and the results obtained during the study are summarized. All data-collection stations are referred to by river mile (RM) throughout the remainder of the report.

Continuous Measurements of Water Level, Streamflow, Water Temperature, and Dissolved-Oxygen Concentration

Water level (stage) was measured at 15-minute intervals; water temperature and DO concentrations were measured at 60-minute intervals at RMs 139.9, 137.6, 122.0, 115.5, and 110.3 (fig. 2). These data were collected for various periods, including the two intensive data-collection periods (table 5). Sensors for measuring water temperature and DO concentration were serviced at approximately 2-week intervals and were calibrated in the field by using a precalibrated field meter.

During the two intensive-sampling periods, the greatest range in water level was recorded at RM 139.9 during the July 1997 sampling period (6.28 ft, table 6). Water temperatures were slightly higher and DO concentrations were slightly lower during the July 1997 sampling period than during the August 1996 sampling period (table 6).

Streamflow data have been collected continuously at RM 137.6 (station 02146000) and RM 122.0 (station 02147020) since 1942 and 1992, respectively. Discharge rating curves were not developed for water-level measurement stations at RM 115.5 and RM 110.3 because flow at these sites is affected by backwater from Fishing Creek Reservoir, nor at RM 139.9 because flow data from Lake Wylie

Table 4. Summary of data collected for use in calibrating and validating the Catawba River, S.C., water-quality model

[—, no station number assigned; Q, continuous streamflow; WL, water level; T, water temperature; DO, dissolved oxygen; TOT, time-of-travel data; SQW, synoptic water-quality samples; FM, field measurements, including water temperature, dissolved-oxygen concentration, and pH; R, stream reaeration; CSOD, community substrate oxygen demand; SOD, sediment oxygen demand; MQ, discrete discharge measurements; EI, site only used for estimation of subbasin inflow hydrographs; EQ estimated streamflow; WW, wastewater-effluent flow and chemical concentrations]

Site number (fig. 2)	USGS station number	Station description	Location (river mile)	Data collected
		CATAWBA RIVER STATIONS		
	—	Lake Wylie Dam	141.0	Q
1	02145905	Catawba River at Fort Mill, S.C.	139.9	WL, T, DO, TOT, SQW, FM
2	02146000	Catawba River near Rock Hill, S.C.	137.6	Q, WL, T, DO, SQW, FM
3	02146010	Catawba River near Fort Mill, S.C.	134.6	TOT, R, SQW, FM
4	02146015	Catawba River near Hancock, S.C.	131.1	TOT, R
5	02146821	Catawba River below Fort Mill, S.C.	126.9	SQW, FM
6	021468705	Catawba River above Catawba, S.C.	123.0	SQW, FM
7	021468706	Catawba River near Van Wyck, S.C.	122.8	TOT, R
8	02147020	Catawba River below Catawba, S.C.	122.0	Q, WL, T, DO, TOT, R, SQW, FM
9	02147050	Catawba River below Springdell, S.C.	119.5	CSOD, SOD
10	02147060	Catawba River above Rowell, S.C.	118.5	TOT, R
11	02147070	Catawba River near Rowell, S.C.	117.9	SQW, FM
12	02147187	Catawba River near Landsford, S.C.	115.5	WL, T, DO, SQW, FM
13	02147188	Catawba River below Landsford, S.C.	114.3	TOT, R, CSOD, SOD
14	02147195	Catawba River above Fort Lawn, S.C.	113.6	SOD
15	02147200	Catawba River near Fort Lawn, S.C.	111.4	TOT, R, SQW, FM, MQ
16	02147259	Catawba River at Fort Lawn, S.C.	110.3	WL, T, DO, FM
		TRIBUTARY STATIONS		
17	02146300	Irwin Creek near Charlotte, N.C.	EI	WL, Q
18	02146507	Little Sugar Creek at Archdale Drive at Charlotte, N.C.	EI	WL, Q
19	02146750	McAlpine Creek below McMullen Creek near Pineville, N.C.	EI	WL, Q
20	02146800	Sugar Creek near Fort Mill, S.C.	129.8[a]	SQW, MQ, EQ, FM
21	02146900	Twelvemile Creek near Waxhaw, N.C.	EI	WL, Q
22	02146995	Twelvemile Creek at Van Wyck, S.C.	122.8[a]	SQW, MQ, EQ, FM
23	021471001	Waxhaw Creek near Van Wyck, S.C.	117.5[a]	SQW, MQ, EQ, FM
24	02147258	Cane Creek near Fort Lawn, S.C.	110.4[a]	SQW, MQ, EQ, FM
		POINT SOURCES		
A	—	Discharger A	137.3	WW
B	—	Discharger B	137.0	WW
C	—	Discharger C	135.5	WW
D	—	Discharger D	134.5	WW
E	—	Discharger E	120.9	WW
F	—	Discharger F	111.1	WW

[a] River mile locations are at the confluences of the tributary and the Catawba River.

Table 5. Period of record for continuous water level, water temperature, and dissolved-oxygen concentration monitoring sites on the Catawba River, S.C.

Location (river mile)	Period of record	
	Water-level data	Water temperature and dissolved-oxygen data
139.9	June 5–October 26, 1996; July 8–23, 1997	April 9–October 26, 1996; July 8–23, 1997
137.6	1942 to current date	January 30–November 6, 1996; July 1–14, 1997
122.0	1992 to current date	February 14–October 31, 1996; June 17–July 22, 1997
115.5	June 5–October 24, 1996; July 7–27, 1997	June 5–October 24, 1996; June 11–30, 1997; July 8–27, 1997
110.3	June 5–October 24, 1996; July 10–22, 1997	June 5–October 24, 1996; July 10–22, 1997

Table 6. Summary of water level, water temperature, and dissolved-oxygen data at five sites on the Catawba River, S.C.

[°C, degrees Celsius; mg/L, milligram per liter; ft, foot]

Location (river mile)	Data Type						
	Water level	Water temperature (°C)			Dissolved-oxygen concentration (mg/L)		
	Range (ft)	Maximum	Minimum	Mean	Maximum	Minimum	Mean
August 23–27, 1996 sampling period							
139.9	5.95	28.9	27.8	28.3	5.9	4.1	4.9
137.6	3.83	29.1	26.9	27.9	5.6	3.2	4.3
122.0	4.65	30.6	26.3	27.7	7.6	3.7	4.7
115.5	4.50	29.8	26.6	27.8	10.7	5.3	7.7
110.3	0.64	30.0	26.7	27.9	6.9	4.6	5.8
July 11–15, 1997 sampling period							
139.9	6.28	28.6	23.8	27.0	5.2	1.9	4.3
137.6	4.13	27.5	24.5	26.0	6.0	3.6	4.8
122.0	4.30	31.9	26.3	26.2	8.6	4.7	6.1
115.5	4.42	31.4	26.1	28.4	7.3	4.6	5.8
110.3	2.78	31.8	27.3	29.2	8.0	4.3	6.0

Dam (RM 141.0) were available from Duke Energy Corporation (DEC).

The long-term median streamflow at RM 137.6 (1942–1997) is 3,630 ft^3/s. Median streamflow at this site was 3,380 ft^3/s and 3,490 ft^3/s during 1996 and 1997, respectively. Mean streamflow at RM 137.6 was 4,240 ft^3/s and 4,160 ft^3/s during 1996 and 1997, respectively, compared with the long-term mean of 4,350 ft^3/s (Cooney and others, 2001). The highest water-year peak flow measured at this site was 151,000 ft^3/s in May 1901 before the construction of Lake Wylie Dam.

Continuous records of discharge from Lake Wylie Dam were obtained from DEC. Flows are computed by DEC at 5-minute intervals as the sum of flows through the turbines (based on a rating between power generated and flow), flows over the spillways (based on forebay elevation and spillway gate position), and leakage through the turbines when no power is being generated (80 ft^3/s; J. Denning, Duke Energy Corporation, written commun., June 1998). Hourly mean flows, which were provided by DEC, are computed from the 5-minute flows. For the model application, these hourly mean flows were assumed to occur at the midpoint of the hour (in other words, a flow reported as the hourly mean flow between 0800 and 0900 was assumed to occur at 0830). Flows at the smaller time steps required by the model were linearly interpolated from these hourly values.

Flows from Lake Wylie Dam (RM 141.0) during the period October 7, 1993, to September 30, 1995, were compared to flows measured at the gaging station at RM 137.6. Taking into account the inflow from the intervening drainage area between the two locations (the drainage area at Lake Wylie Dam is 3,020 mi^2, and the drainage area at RM 137.6 is 3,050 mi^2), the analysis indicated that, on average, flows at RM 137.6 were about 94 ft^3/s higher than flows from Lake Wylie Dam. The DEC estimates that there is about 80 ft^3/s of leakage through the dam, in addition to the turbine leakage (T. Ziegler, Duke Energy Corporation, oral commun., January 1997). This leakage, which was not included in the hourly mean flows reported by DEC, is comparable to the 94 ft^3/s mean difference in flows between Lake Wylie Dam and the streamgaging station at RM 137.6. Therefore, 94 ft^3/s was added to all reported flows from Lake Wylie Dam.

Accurate estimates of flow at the upstream boundary of the model are required for model calibration and validation. Therefore, flows reported for Lake Wylie Dam and measured at the gaging station at RM 137.6 were further compared for the model calibration (August 24–30, 1996) and validation (July 11–16, 1997) periods. The analysis indicated that reported flows from Lake Wylie Dam (including the 94 ft^3/s leakage) should be multiplied by 1.09. This adjustment may be necessary because flows from the Lake Wylie Dam are computed from generated power, rather than generated power and the head difference across the turbines as recommended by Vennard (1962).

Discrete Measurements of Discharge

Discharge was measured near the mouth of each of the four major tributaries to the study reach. The tributaries are Sugar, Twelvemile, Waxhaw, and Cane Creeks (fig. 2; table 4). Measurements were made during each of the two intensive data-collection periods in order to verify estimates of tributary inflow (described in the next section) that were used in the water-quality models. Rainfall during the August 1996 data-collection period resulted in a large range in the measured streamflow (table 7). Conditions were much

Table 7. Summary of streamflow data collected at four tributaries to the Catawba River, S.C., during August 23–27, 1996, and July 11–15, 1997

Site number (fig. 2)	USGS station number	Location	River mile[a]	Measured streamflow, in cubic feet per second: Minimum	Measured streamflow, in cubic feet per second: Maximum
		August 23–27, 1996			
20	02146800	Sugar Creek near Fort Mill, S.C.	129.8	125	1,560
22	02146995	Twelvemile Creek at Van Wyck, S.C.	122.8	10.4	196
23	021471001	Waxhaw Creek near Van Wyck, S.C.	117.9	1.17	15.7
24	02147258	Cane Creek near Fort Lawn, S.C.	110.4	-29.6[b]	15.4
		July 11–15, 1997			
20	02146800	Sugar Creek near Fort Mill, S.C.	129.8	105	166
22	02146995	Twelvemile Creek at Van Wyck, S.C.	122.8	9.29	26.1
23	021471001	Waxhaw Creek near Van Wyck, S.C.	117.5	2.00	5.73
24	02147258	Cane Creek near Fort Lawn, S.C.	110.4	-65.7[b]	-23.7[b]

[a] River mile locations are the confluences of the tributary and the Catawba River.

[b] A negative value indicates flow in the upstream direction caused by backwater from the Catawba River.

less variable during the July 1997 measurements. Negative (upstream) flows were measured in Cane Creek during periods of high flow in the Catawba River because of backwater from the river. Streamflow measurements also were made in the Catawba River at RM 111.4 (fig. 2; table 4) simultaneously with water-quality sampling and were used to calibrate and validate the streamflow model.

Estimation of Subbasin Inflow Hydrographs

To calibrate the BRANCH model, it is necessary to account for the water entering the Catawba River between the Lake Wylie Dam tailrace and the headwaters of Fishing Creek Reservoir. The majority of the tributaries to the Catawba River are ungaged; therefore, two estimation methods were used to determine inflow hydrographs for the intervening subbasins in the study area for the calibration and validation flow periods. One method was to use the unit flow—which is the flow at the station, in cubic feet per second, divided by the drainage area at the station, in square miles—from a nearby gaging station and the area of the subbasin to estimate the inflow. The other method was to use the difference between simulated flows at RM 122.0 and measured flows at RM 122.0 to estimate inflows.

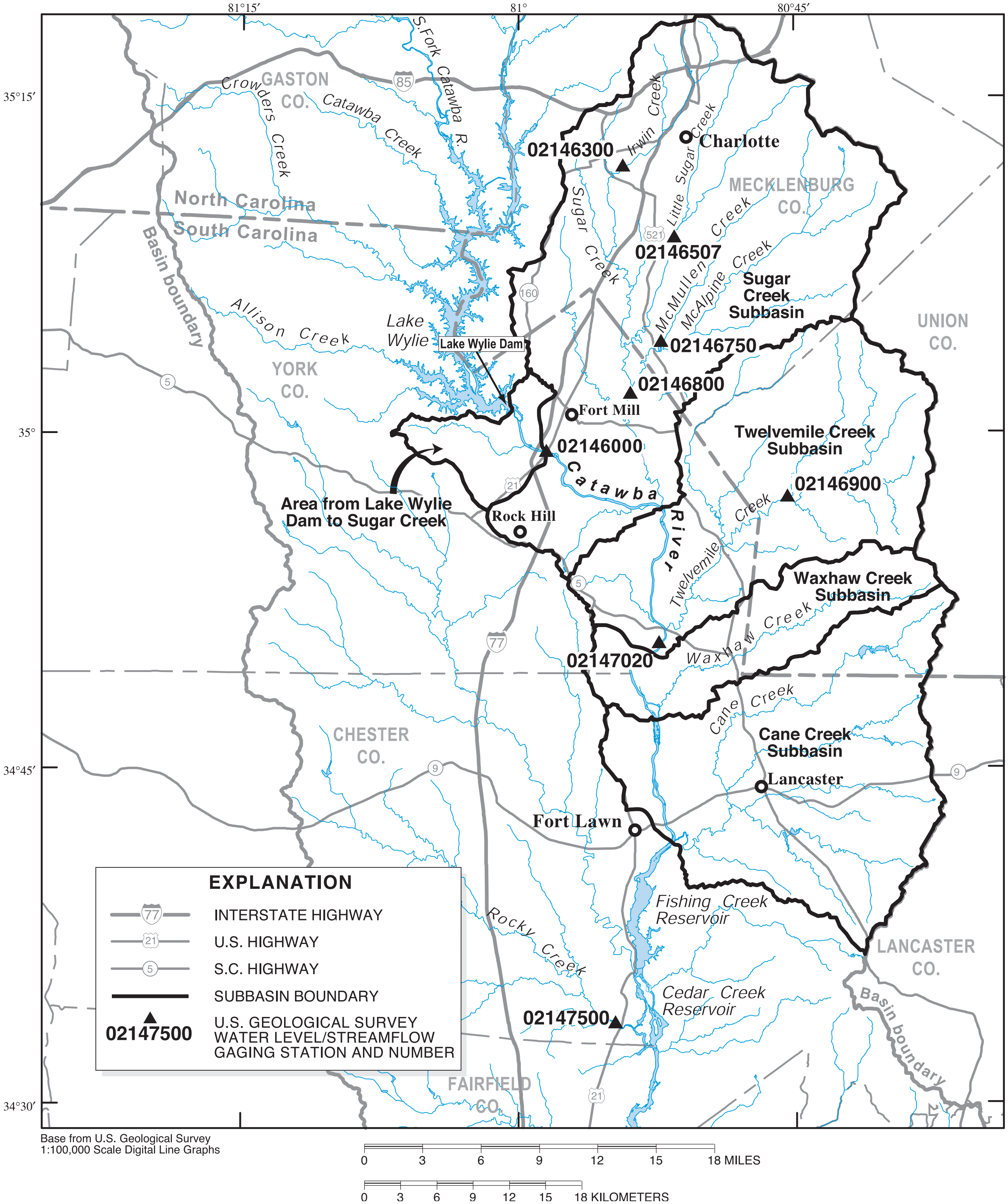
The flows in the tributaries of the Catawba River are characteristic of a combination of influencing factors—runoff from rural land, runoff from urban land, and discharge of wastewater-treatment effluent. Five inflow subbasins to the Catawba River were delineated—the Sugar, Twelvemile, Waxhaw, and Cane Creek subbasins, and the area between Lake Wylie Dam and the Sugar Creek subbasin (fig. 3). Of the five subbasins, only Sugar Creek tributary flow is a combination of runoff from rural land, runoff from urban land, and effluent discharge from wastewater-treatment plants. Streamflow data for the urbanized subbasins of Sugar Creek were collected at three USGS stations (fig. 3)—Little Sugar Creek (USGS station 02146507), McAlpine Creek (USGS station 02146750) and Irwin Creek (USGS station 02146300). An urbanized watershed is one in which at least 15 percent of the drainage area is covered with commercial, industrial, and (or) residential development land use (Sauer and others, 1983). The area of the Sugar Creek subbasin downstream from these three stations is mostly rural. The other four subbasins have solely rural-dominated flows. To simplify the tributary-flow computations, the drainage-area delineations include small drainage areas near the Catawba River that are not actually in the drainage areas of these creeks (table 8).

Table 8. Dominant factors influencing subbasin flow along with the drainage areas used in the subbasin flow estimates to the Catawba River, S.C.

[mi^2, square mile; WWTP, wastewater-treatment plant effluent]

Subbasin	Dominant influence on flow	Drainage area[a] (mi^2)
Area from Lake Wylie Dam to Sugar Creek	Rural runoff	24
Sugar Creek	Urban runoff, rural runoff, WWTP	311
Twelvemile Creek	Rural runoff	178
Waxhaw Creek	Rural runoff	65
Cane Creek	Rural runoff	188

[a] Drainage areas include small drainage areas to the Catawba River and are only for flow estimates; they do not represent the actual drainage area for each watershed.



Base from U.S. Geological Survey
1:100,000 Scale Digital Line Graphs

Figure 3. Locations of U.S. Geological Survey streamgaging stations and the subbasins used for inflow estimates to the Catawba River, S.C.

Unit-Flow Method

To compute the inflow hydrographs for the rural subbasins and the rural area of Sugar Creek, the unit flow at Rocky Creek at Great Falls, S.C. (station 02147500, fig. 3), in cubic feet per second per square mile [(ft^3/s)/mi^2], was multiplied by the drainage areas of the subbasins. This method is more accurate when applied to low-flow regimes with no storm runoff, which is the condition for which the model is anticipated to be used. This method can result in over or under prediction of tributary flows on a long-term basis. Therefore, the flows are adjusted by a monthly adjustment factor (*K*), which is the ratio of the monthly mean unit rural flow, in cubic feet per second per square mile, between RM 137.6 (station 02146000) and RM 122.0 (station 02147020) (fig. 3; table 4) to the monthly mean unit flow at station 02147500. This forces the monthly volumes of flow estimated for the Catawba River to be conserved.

For the rural subbasins and the rural area of the Sugar Creek subbasin, streamflow is the product of the area of the subbasin, the unit flows of Rocky Creek (station 02147500), and the monthly adjustment factor (*K*).

$$Q_t = KU_{ruv}A_t , \qquad (1)$$

where

Q_t is the flow for the rural subbasin, in cubic feet per second;

U_{ruv} is the unit-value flow for Rocky Creek (station 02147500), in cubic feet per second per square mile; and

A_t is the drainage area of the subbasin, in square miles.

The monthly adjustment factor, *K*, is the ratio of the monthly mean unit rural flows for the Catawba River between RM 137.6 and RM 122.0 to the monthly mean rural unit flows for Rocky Creek. The monthly mean rural flow between RM 137.6 and RM 122.0, Q_{mrc}, is computed by subtracting the upstream station (RM 137.6) flows from the downstream station (RM 122.0) flows and subtracting the urban flows from Sugar Creek. The unit monthly mean rural flow (U_{mrc}) is computed by dividing Q_{mrc} by the rural area of the Catawba River between RM 137.6 and RM 122.0 (table 9). The unit rural flow for Rocky Creek is computed by dividing the mean monthly flow by the drainage area of the Rocky Creek Basin (table 9).

Table 9. Drainage area of U.S. Geological Survey stations used to estimate subbasin inflows for the Catawba River, S.C.

USGS station number (fig. 3)	Station name	Drainage area, in square miles
02146000	Catawba River near Rock Hill, S.C.	3,050[a]
02146300	Irwin Creek near Charlotte, N.C.	30.7[b]
02146507	Little Sugar Creek at Archdale Drive at Charlotte, N.C.	42.6[b]
02147020	Catawba River below Catawba, S.C.	3,540[a]
02146750	McAlpine Creek below McMullen Creek near Pineville, N.C.	92.4[b]
02147500	Rocky Creek near Great Falls, S.C.	194[a]

[a] Cooney and others, 2001.

[b] Ragland and others, 1999.

$$K = U_{mrc} / U_{mr} , \qquad (2)$$

where

U_{mrc} is the unit monthly mean rural flow between RM 137.6 and RM 122.0, in cubic feet per second per square mile, and is defined as

$$U_{mrc} = (Q_{mrc} / (A_c - A_{02146300} - A_{02146507} - A_{02146750})), \qquad (3)$$

and

$$Q_{mrc} = Q_{m122.0} - Q_{m137.6} - Q_{m02146300} - Q_{m02146507} - Q_{m02146750} - E_I - E_M , \qquad (4)$$

where

Q_{mrc} is the monthly mean rural flow between RM 137.6 and RM 122.0, in cubic feet per second (ft^3/s);

A_c is the drainage area between RM 137.6 and RM 122.0 (Sugar Creek and Twelvemile Creek subbasins), in square miles (mi^2);

$A_{02146300}$ is the drainage area at USGS station 02146300, in mi^2;

$A_{02146507}$ is the drainage area at USGS station 02146507, in mi^2;

$A_{02146750}$ is the drainage area at USGS station 02146750 in mi^2;

$Q_{m122.0}$ is the monthly mean flow for RM 122.0, in ft^3/s;

$Q_{m137.6}$ is the monthly mean flow for RM 137.6, in ft^3/s;

$Q_{m02146300}$ is the monthly mean flow for USGS station 02146300, in ft^3/s;

$Q_{m02146507}$ is the monthly mean flow for USGS station 02146507, in ft^3/s;

$Q_{m02146750}$ is the monthly mean flow for USGS station 02146750, in ft^3/s,

E_I is the monthly mean effluent flow into Irwin Creek, in ft^3/s; and

E_M is the monthly mean effluent flow into McAlpine Creek, in ft^3/s.

The unit monthly mean rural flow (U_{mr}) for Rocky Creek (station 02147500), in cubic feet per second per square mile, is defined as:

$$U_{mr} = Q_{m02147500} / A_{02147500}, \qquad (5)$$

where

$Q_{m02147500}$ is the monthly mean flow for USGS station 02147500, in cubic feet per second; and

$A_{02147500}$ is the drainage area of USGS station 02147500, in square miles.

Sugar Creek streamflow (Q_s) is a combination of rural and urban flows and wastewater-treatment plant effluent. The rural portion of the flow was estimated as described above. The urban portion of the flow is the sum of the streamflow recorded at the three USGS stations in the basin. The wastewater-treatment effluent amounts were obtained from the facilities on Irwin Creek and McAlpine Creek.

$$Q_s = A_s K U_{ruv} + Q_{02146300} + Q_{02146507} + Q_{02146750} + E_I + E_M, \qquad (6)$$

where

Q_s is the flow for Sugar Creek, in cubic feet per second;

A_s is the drainage area of Sugar Creek downstream from USGS stations 02146300, 02146507, and 02146750, in square miles;

and all remaining variables have been defined previously.

Maximum, mean, and minimum flows for selected periods are listed in table 10, and inflow hydrographs are shown in figure 4.

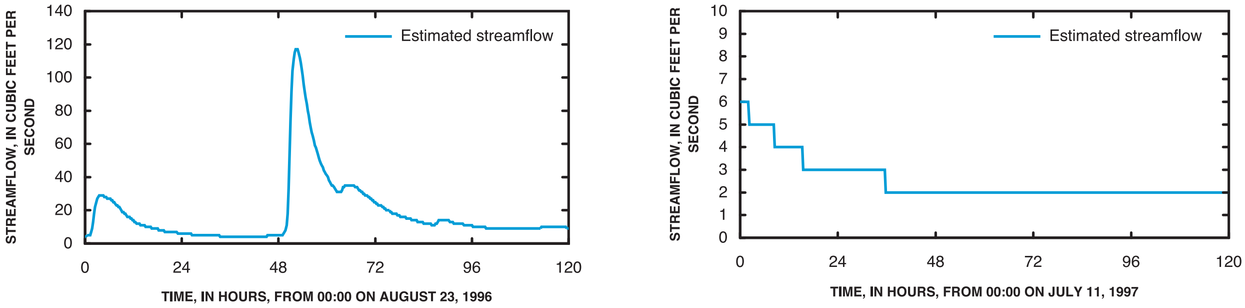
Flow-Simulation Method

Comparison of inflow hydrographs using the unit-flow estimation method and measured streamflow indicated that the method technique did not correctly estimate the high streamflow from the rainfall of August 25, 1996, for the Sugar Creek subbasin. Therefore, the BRANCH model was used to estimate inflows for the rural area of Sugar Creek for the August 23–27, 1996, sampling period. For the BRANCH model simulations, the tributary flows from Sugar Creek were set to zero while all remaining tributary flows were estimated. An inflow hydrograph for the Sugar Creek subbasin was then computed at RM 122.0 by subtracting the simulated flow at RM 137.6 from the measured flows at RM 122.0. The measured flows at RM 122.0 were delayed 54 minutes to better align the hydrographs before subtraction. The resulting hydrograph was smoothed by hand and delayed by 2 hours to transfer the hydrograph to the mouth of Sugar Creek. The 2-hour lag time was determined by BRANCH model simulations.

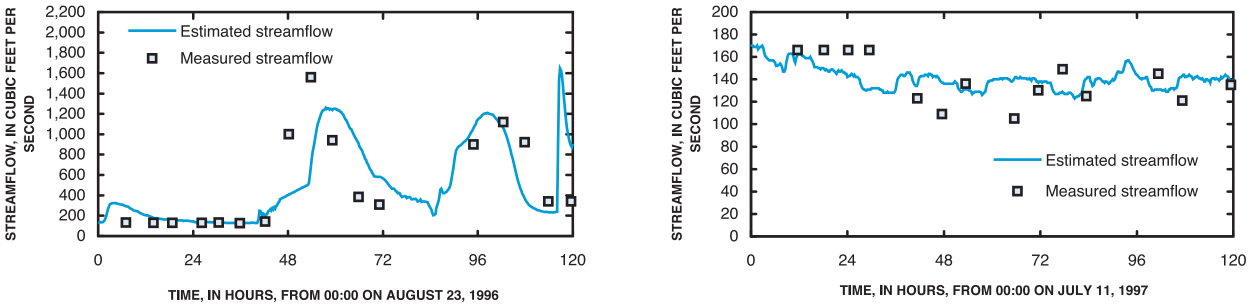
Estimated subbasin mean inflows for August 23–27, 1996, and July 11–15, 1997, were

A. AUGUST 23-27, 1996 **B. JULY 11-15, 1997**

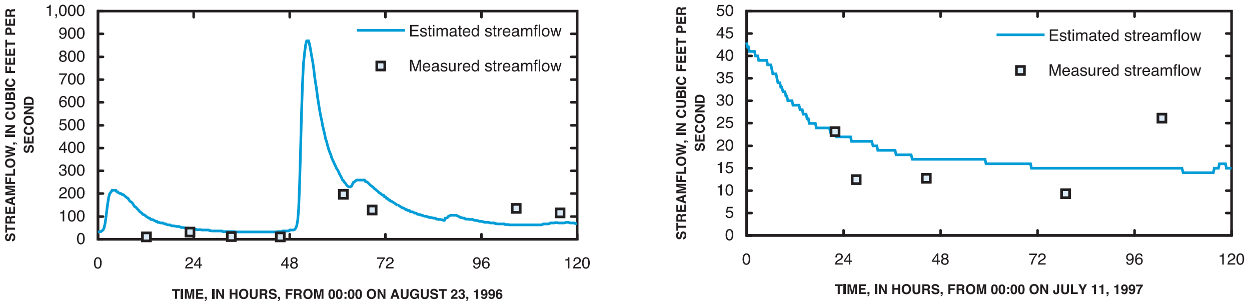
Lake Wylie Dam to Sugar Creek Subbasin *Lake Wylie Dam to Sugar Creek Subbasin*



Sugar Creek Subbasin *Sugar Creek Subbasin*



Twelvemile Creek Subbasin *Twelvemile Creek Subbasin*



Waxhaw Creek Subbasin *Waxhaw Creek Subbasin*

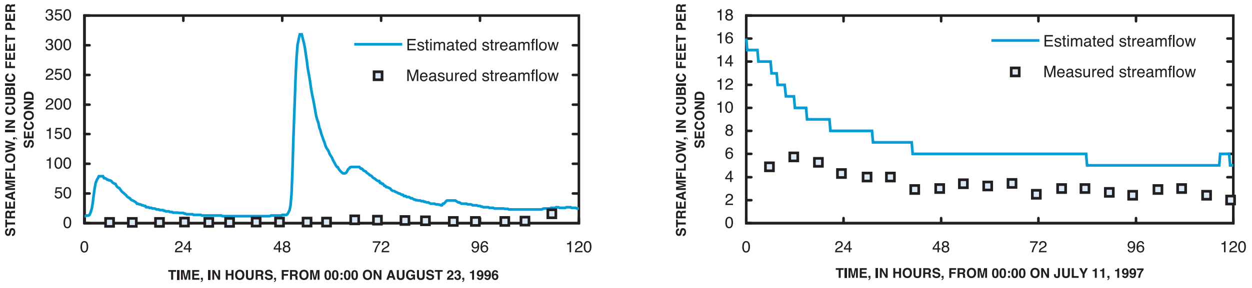


Figure 4. Estimated and measured streamflows for Lake Wylie to Sugar Creek subbasin, and the Sugar Creek, Twelvemile Creek, and Waxhaw Creek subbasins for (A) August 23–27, 1996, and (B) July 11–15, 1997.

Table 10. Maximum, mean, and minimum streamflows for August 23–27, 1996, and July 11–15, 1997, at selected sites in the Catawba River Basin, S.C.

[ft^3/s, cubic foot per second; —, no data]

Location (fig. 3)	River mile[a]	August 23–27, 1996, streamflow (ft^3/s)			July 11–15, 1997, streamflow (ft^3/s)		
		Maximum	Mean	Minimum	Maximum	Mean	Minimum
Measured streamflow							
Outflow from Lake Wylie Dam	141.0	9,390	3,530	190	9,550	3,560	190
Catawba River at USGS station 02146000	137.6	9,400	3,560	214	9,630	3,390	81.0
Catawba River at USGS station 02147020	122.0	9,470	4,010	997	8,870	3,380	589
Estimated streamflow							
Lake Wylie Dam to Sugar Creek subbasin[b]	139.0	117	18.2	4.00	6.00	2.52	2.00
Sugar Creek subbasin	129.8	1,650	508	128	171	141	123
Twelvemile Creek subbasin	122.8	870	135	32.0	43.0	19.1	14.0
Waxhaw Creek subbasin	117.5	318	49.1	12.0	16.0	6.94	5.00
Total inflow into the Catawba River[c]	—	2,960	710	176	236	170	144

[a] River mile 139.0 is the location at which estimated inflow was input into the streamflow model.

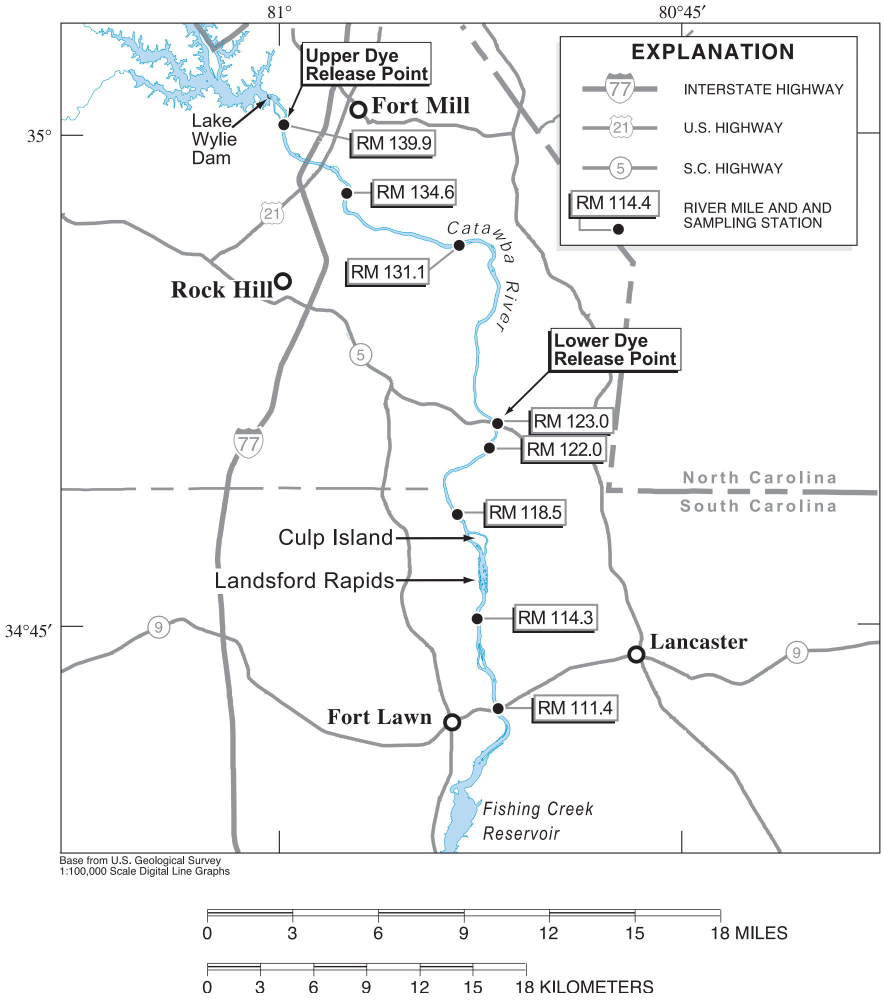
[b] The river mile location refers to the location of the confluence of the tributary and the Catawba River.

[c] Inflows include the intervening drainage area from Lake Wylie Dam to Sugar Creek subbasin, and the subbasins of Sugar, Twelvemile, and Waxhaw Creeks.

20 and 5 percent, respectively, of measured mean flows at RM 137.6. Sugar Creek also accounted for 72 and 83 percent of the total estimated mean inflow from the tributaries for August 23–27, 1996, and July 11–15, 1997, respectively (table 10).

Time of Travel

Two time-of-travel studies were conducted (August 24–25, 1996, and July 12–13, 1997) during the periods of intensive data collection. For each study, rhodamine dye was released at two locations—RM 139.9 (upper release) and RM 123.0 (lower release)—and samples were collected at seven locations (fig. 5; table 11). Dye was released instantaneously at RM 139.9, and sampling and analysis were based on the methods described by Kilpatrick and Wilson (1989). Dye at RM 123.0 was released at a continuous rate of 74.9 milliliters per minute (mL/min) for 9 hours on August 24, 1996, and at a continuous rate of 48.6 mL/min for 12 hours on July 12, 1997. Dye



Base from U.S. Geological Survey
1:100,000 Scale Digital Line Graphs

Figure 5. Locations of dye releases and sampling sites for the time-of-travel studies on the Catawba River, S.C., August 1996 and July 1997.

Table 11. Time-of-travel data collected on the Catawba River, S.C., for August 24–25, 1996, and July 12–13, 1997

[µg/L, microgram per liter; mi/hr, mile per hour; ft^3/s, cubic foot per second; N/A, not applicable; —, no data]

River mile	Peak concentration (µg/L)	Time of travel of centroid of dye cloud, (hours)	Mean velocity (mi/hr)	Dye cloud base width (hours)	Mean streamflow (ft^3/s)
August 24–25, 1996					
139.9	Upper release point	N/A	N/A	N/A	—
134.6	18.1	—	—	6.00	2,300
131.1	—	—	—	—	2,330
111.4	3.62	27.6	0.84	19.0	3,470
123.0	Lower release point	N/A	N/A	N/A	—
122.0	11.5[a]	—	—	11.0	3,150
118.5	8.45[a]	5.4	.65	13.0	3,260
114.3	7.10[a]	13.5	.57	15.0	3,440
111.4	7.40[a]	19.7	.54	17.0	3,470
July 12–13, 1997					
139.9	Upper release point	N/A	N/A	N/A	—
134.6	4.60	—	—	3.00	3,380
131.1	3.30	3.4	1.03	8.00	3,510
111.4	6.37	29.1	.80	22.0	4,080
123.0	Lower release point	N/A	N/A	N/A	—
122.0	11.9[a]	—	—	15.0	3,980
118.5	9.16[a]	3.7	.95	17.0	4,010
114.3	7.78[a]	9.6	.80	19.0	4,080
111.4	7.38[a]	14.4	.74	21.0	4,080

[a] Peak concentration was determined as the average plateau concentration.

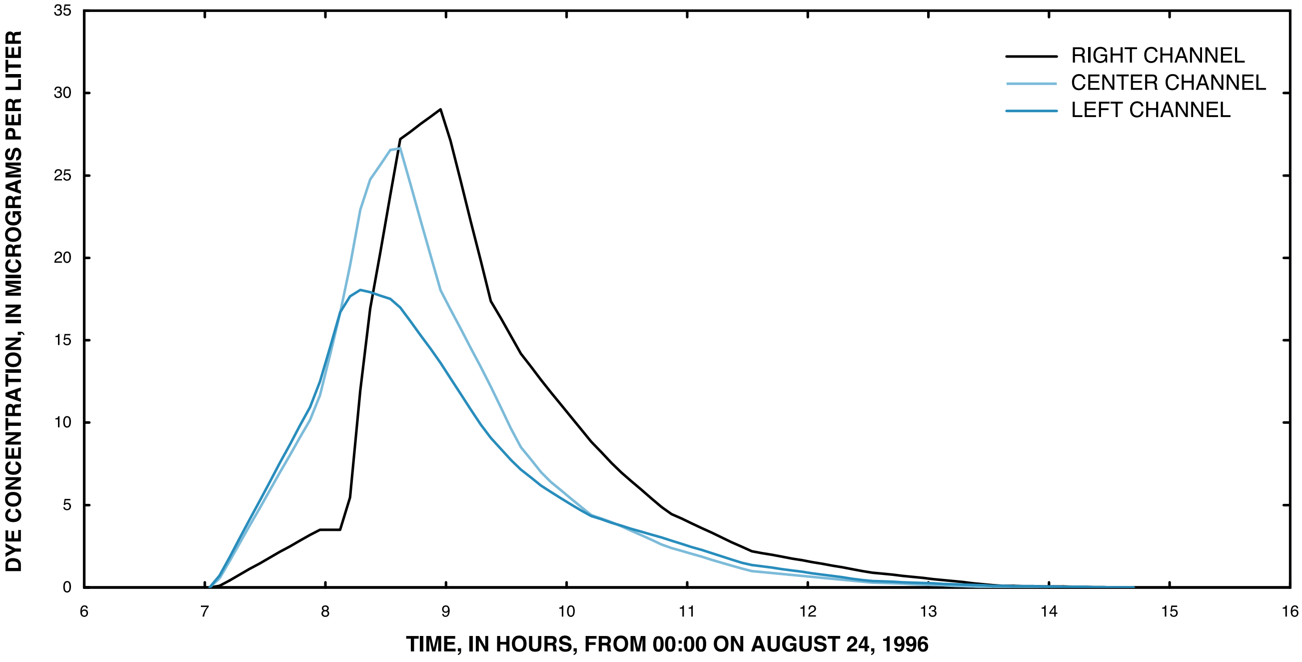
from the upper release was traced from RM 134.6 to RM 111.4, and dye from the lower release was traced from RM 122.0 to RM 111.4 (table 11).

During the second and third days of the sampling periods, releases from Lake Wylie Dam were held approximately steady at about 2,300 and 4,100 ft^3/s in 1996 and 1997, respectively. During the 1996 study, tributary flow into the study reach increased because of rainfall on August 24 and 25, whereas tributary flow remained relatively steady through the reach during the 1997 study period.

For the upper release, dye did not completely mix across the river at RM 134.6 and RM 131.1 (fig. 5). Therefore, the channel at these two sampling locations was divided into three sections (flow tubes) of equal flow, and dye samples were collected at the center of flow for each section according to methods described by Yotsukura and others (1984). The dye response curve (temporal distribution of dye concentration at the point) was determined for each flow tube (Kilpatrick and others, 1989). The three dye-tracer response curves at each sampling location (for example, fig. 6A) were combined by averaging the dye concentrations at each time step to produce a composite response curve (fig. 6B). The composite response curve was used to represent the laterally averaged dye concentration at the sampling location. Dye released at the upper release point was laterally mixed at RM 111.4, so a single sample collected near the center of the channel represented the cross-sectional mean concentration.

Fluorometer readings are relative values of fluorescence intensity. Fluorometers are calibrated using standards, or prepared solutions of known concentrations (Wilson and others, 1986). Dye concentrations from the upper releases were measured at RM 111.4 by the USGS using a fluorometer calibrated with different dye standards than the fluorometer used by the U.S. Environmental Protection Agency (USEPA) at RM 134.6 and RM 131.1. In order to compare the concentrations measured from the USGS

A. RIVER MILE 134.6



B. RIVER MILE 134.6

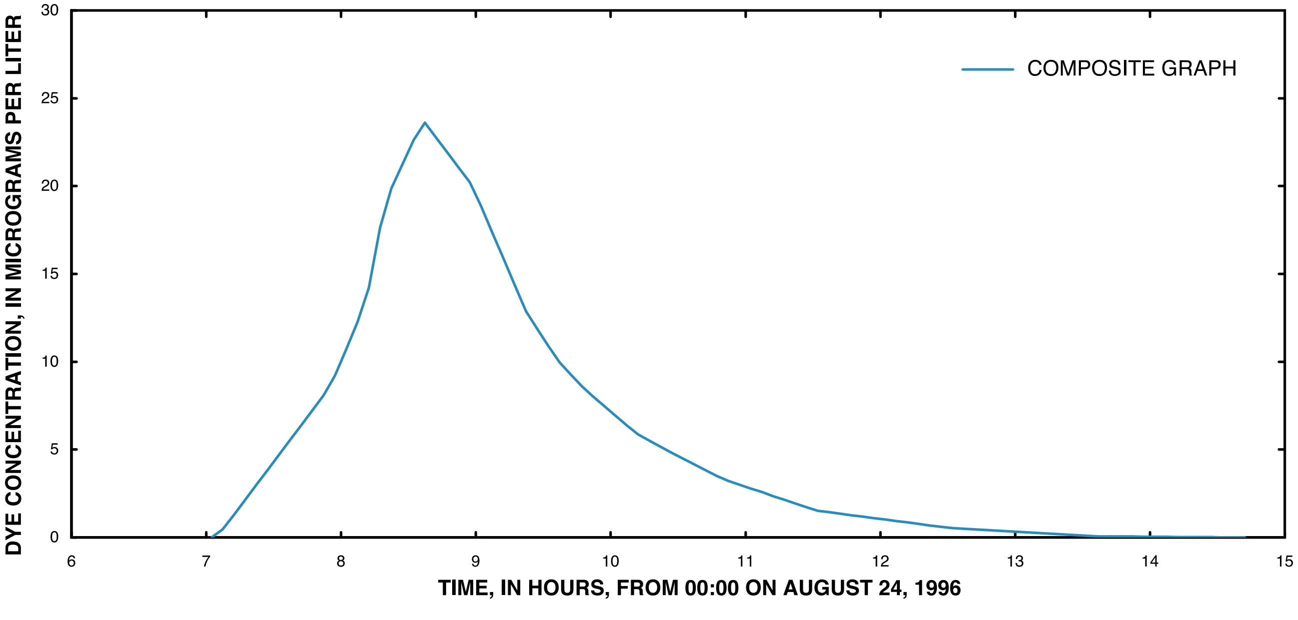
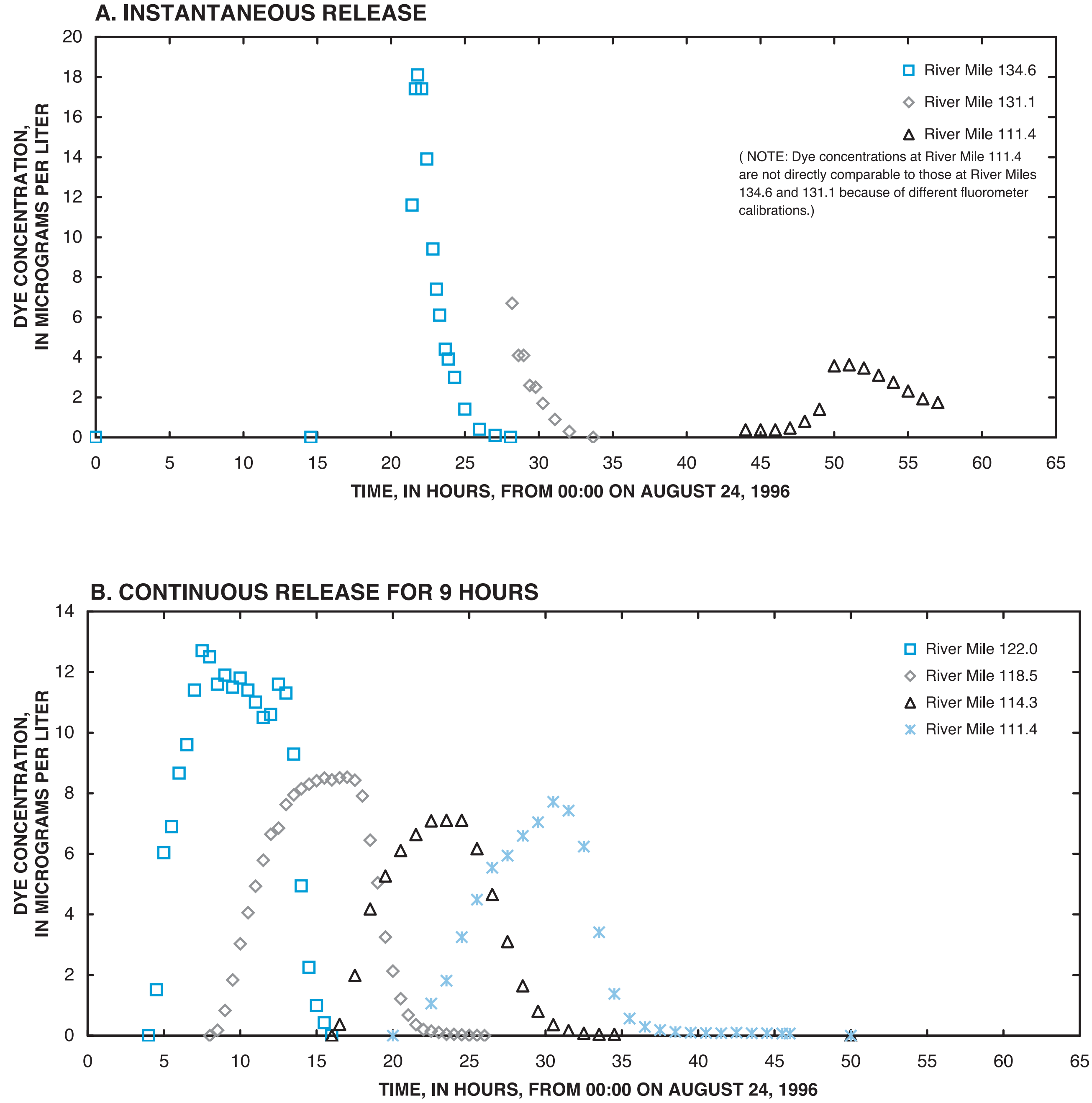


Figure 6. (A) Measured dye concentrations from the left-, center-, and right-channel flow tubes, and (B) the composite dye concentration at river mile 134.6 on the Catawba River, S.C., August 1996.

fluorometer with those measured from the USEPA fluorometer, the exact relation between the two fluorometers would have to be known and all readings would have to be converted to a common scale. No such relation was determined during this study; therefore, the dye concentrations reported at RM 111.4 for the upper release are not directly comparable to the measured dye concentrations at RM 134.6 and RM 131.1 (figs. 7, 8). The upper release dye concentrations at RM 111.4 were used to determine travel time of the dye cloud, but were not be used to estimate dilution and dispersion between RM 131.1 and RM 111.4.

Travel times were faster during the July 1997 study than during the August 1996 study because of the increased streamflow during July 1997 (table 11). The mean dye transport velocity for the lower reach



(NOTE: Dye concentrations at River Mile 111.4 are not directly comparable to those at River Miles 134.6 and 131.1 because of different fluorometer calibrations.)

Figure 7. Measured concentrations for (A) instantaneous dye releases in the upper and (B) continuous dye releases in the lower points on the Catawba River, S.C., August 24–25, 1996.

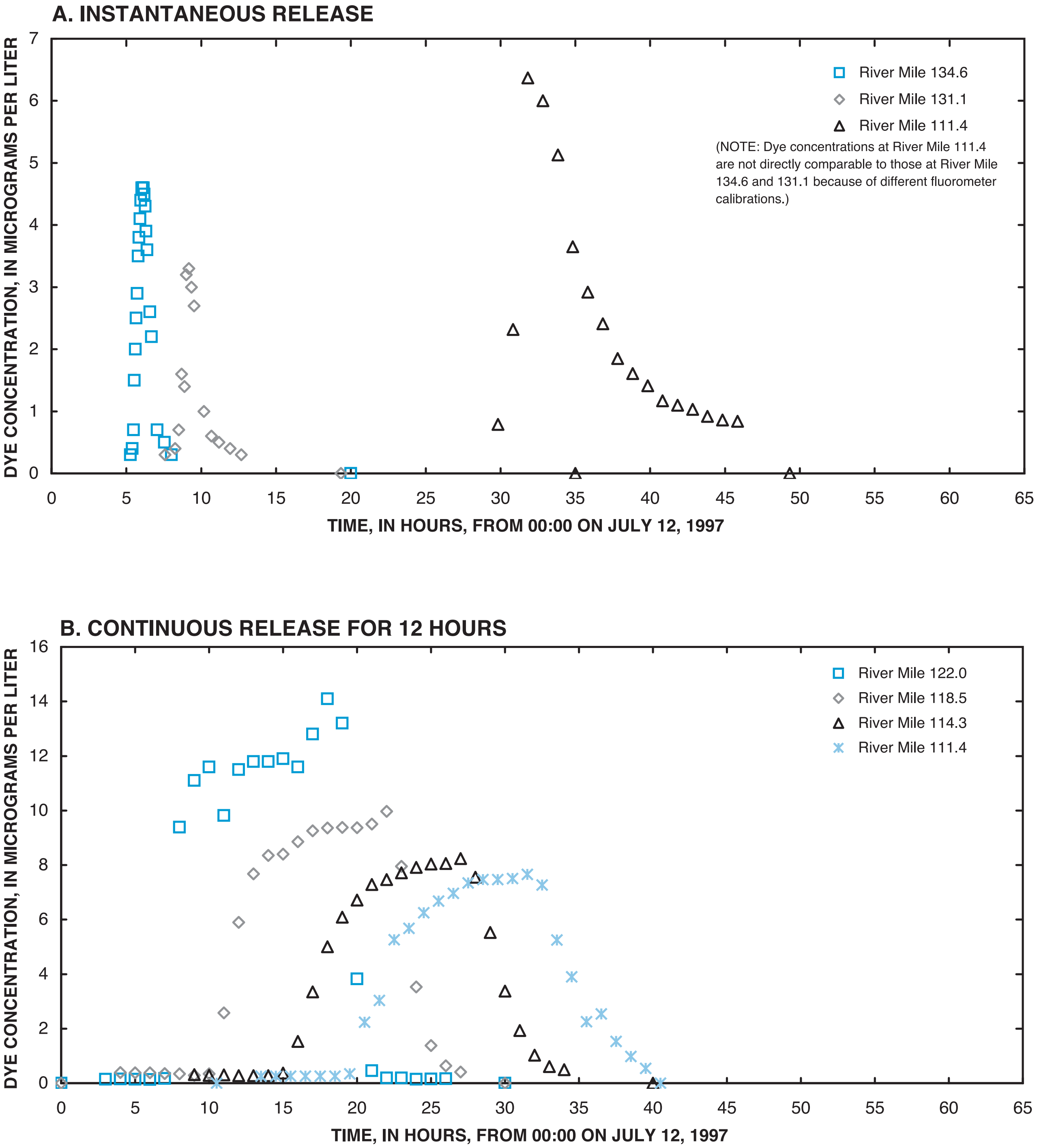


Figure 8. Measured concentrations for (A) instantaneous dye releases in the upper and (B) continuous dye releases in the lower points on the Catawba River, S.C., July 12–13, 1997.

(RM 123.0–111.4) increased from 0.54 mile per hour (mi/hr) during the 1996 study to 0.74 mi/hr during the 1997 study. As previously stated, flows from the Lake Wylie Dam were held steady for the time-of-travel studies. During the 1997 study, steady flows at the dam were approximately 50 percent greater than flows during the 1996 study. The mean dye transport velocity for the entire sampled reach (RM 134.6–RM 111.4) was only slightly affected by the increased flow, decreasing from 0.84 mi/hr in August 1996 to 0.80 mi/hr in July 1997 (table 11).

Dilution and dispersion result in attenuation of the peak dye concentrations as the dye travels downstream (figs. 7, 8). In the lower reach (RM 123.0–RM 111.4), the largest percentage of reduction in peak dye concentrations occurred between stations RM 122.0 and RM 118.5, which were pool and riffle sections of the reach. In this reach, the peak concentration was reduced about 25 percent during both studies (figs. 7, 8; table 11). During the 1997 study, the peak concentration was reduced about 28 percent in a 3.5-mi pool and riffle reach of the river (RM 134.6–RM 131.1). Reduction in the peak dye concentration through the rapids section of the river (RM 118.5–RM 114.3) was about 15 percent in both studies (fig. 5; table 11). Downstream from the rapids, where flow is influenced by Fishing Creek Reservoir backwater, there was little reduction in the peak concentration. (The apparent increase in peak dye concentration between RM 114.3 and RM 111.4 in 1996 was probably due to measurement error.)

The change in the width (time base) of the dye cloud from station to station is a measure of dispersion. The time base of the dye cloud increased from 0 (instantaneous release) at RM 139.9 to 19.0 hours at RM 111.4 during the 1996 study, and to 22.0 hours during the 1997 study (table 11). Higher streamflow in 1997 contributed to greater dispersion of the dye cloud.

Although the dye is theoretically conservative, some dye loss occurred through the study reach. Dye losses may be caused by storage, adherence to materials, dilution below detection levels, or incomplete sampling of the response curve. The dye loss in the lower reach (RM 123.0–RM 111.4) was about 35 percent during both study periods. During the 1997 study, an apparent dye loss of about 11 percent occurred between RM 134.6 and RM 131.1.

Reaeration Rates

Reaeration is the transfer of oxygen from the atmosphere to the river across the air-water interface. The reaeration rate is a function of water temperature and mechanical mixing within the river. The reaeration rates calculated during the field studies were used to compare with computed reaeration rates in the water-quality model simulations, but were not used directly in the simulations because the field reaeration rates were measured during steady-flow conditions.

Data for the calculation of reaeration rates were collected August 24–25, 1996, and July 12–13, 1997, simultaneously with time-of-travel data. Krypton gas was injected continuously into the river at RM 123.0 (fig. 5). The methods are based on the assumption that the krypton gas is desorbed from the water at a rate proportional to the rate at which oxygen is absorbed by the water (Kilpatrick and others, 1989). Samples were collected near the center of the channel at RMs 122.0, 118.5, 114.3, and 111.4 and were analyzed for krypton gas concentrations. Reaeration rates were calculated for each of the three subreaches bounded by these sampling points (table 12). Reaeration rates from some

Table 12. Measured and calculated reaeration rate coefficients, reach length, and travel times for the lower Catawba River, S.C., August 24–25, 1996, and July 12–13, 1997

[ft/s, foot per second; ft^3/s, cubic foot per second; °C, degrees Celsius]

River mile (fig. 2)	Reach distance (miles)	Mean stream depth (feet)	Mean stream velocity (ft/s)	Streamflow (ft^3/s)	Travel time (hours)	Measured reaeration rate coefficient (K_a) (per day at 20 °C)	Calculated reaeration rate coefficient (K_a) (per day at 20 °C)			
							O'Connor and Dobbins (1958)	Churchill and others (1962)	Owens and others (1964)	Langbein and Durum (1967)
August 24–25, 1996										
122.0–118.5	3.5	9.0	0.72	2,830	5.7	0.36	0.41	0.21	0.30	0.29
118.5–114.3	4.2	3.0	.78	2,950	7.7	3.45	2.19	1.45	2.41	1.38
114.3–111.4	2.9	6.5	.68	3,010	6.5	.66	.64	.35	.53	.43
July 12–13, 1997										
122.0–118.5	3.5	9.3	1.05	3,860	3.9	0.43	0.47	0.29	0.36	0.41
118.5–114.3	4.2	3.5	1.02	3,890	5.9	4.55	1.99	1.45	2.17	1.46
114.3–111.4	2.9	7.0	.88	3,890	5.0	.80	.65	.40	.54	.50

commonly used rate-calculation equations were computed for comparison with the measured rates.

Reaeration rates were similar in 1996 and 1997. The rates were lowest in the pool and riffle (RM 122.0–RM 118.5) reach of the river and highest through the Landsford Rapids (RM 118.5–RM 114.3) reach (table 12). The reach downstream from Landsford Rapids (RM 114.3–RM 111.4) had higher reaeration rates than the pool and riffle reach because the flow is more turbulent through this reach of the river. With two exceptions, reaeration rates computed from four empirical equations available in the BLTM underestimated the reaeration rate coefficients from 3 to 68 percent, but the spatial distribution of rate coefficients was similar to the observed (table 12). The O'Connor and Dobbins (1958) equation overestimated measured reaeration rates for the reach between RM 122.0 and RM 118.5 for both the 1996 and the 1997 measurements by 14 and 9 percent, respectively.

Water-Quality Data

The amount of biologically available nutrients, mainly phosphorus and nitrogen, contributes to algal production in a river system. The amount of algal production (algal biomass) often is estimated by measuring the level of chlorophyll *a* (a pigment found in green plants, such as algae) in the water. Excessive algal production in nutrient-enriched water causes a variety of associated water-quality problems, including increased BOD, extreme vertical and diel changes in DO concentrations (from oversaturated to depleted), and high turbidity. In order to simulate nutrient, algal, and DO dynamics in flowing waters, data sets of diel changes in nutrient concentrations and the associated response in chlorophyll *a*, DO, and BOD concentrations are required.

Different forms of nitrogen can be present in natural waters and can change dynamically from one form to another depending on certain water characteristics and associated reaction kinetics. Organic nitrogen is a component in organic compounds (amino acids, amines, proteins, and humic substances). Inorganic forms of nitrogen include ammonia, nitrite, and nitrate. Total Kjeldahl nitrogen (TKN) is the sum of organic nitrogen and ammonia forms. Sources of TKN include the decay of organic material (such as decaying algal biomass in bed sediments), animal wastes, and wastewater. Under oxygen-rich conditions and certain biological activity in streams and rivers, TKN can be converted to the extremely mobile, inorganic forms of nitrate and nitrite, consuming DO in the process. This process contributes to the overall BOD of the water. Inorganic nitrogen is the preferred form for aquatic plants. The most important form of inorganic phosphorus is orthophosphate, which is the preferred form of phosphorus for uptake by aquatic plants.

Water-quality data were collected at approximately 6-hour intervals at 10 sites along the study reach of the Catawba River and at 4 sites on tributaries to the study reach during the 5-day intensive sampling periods (August 23–27, 1996, and July 11–15, 1997). Daily mean flow rates and constituent concentrations were obtained from the six major wastewater dischargers in the study reach (table 3). Each stream sample was collected as a single grab at a wadeable location near the bank. Field measurements of water temperature and DO concentrations were made at the time of sample collection. Water samples were analyzed for ammonia, TKN (organic nitrogen plus ammonia), nitrite plus nitrate, nitrite, total phosphorus, dissolved phosphorus, dissolved orthophosphorus, chlorophyll *a*, and BOD_5 concentrations. During the second and third days of the sampling periods, flows from the Lake Wylie Dam were held steady at about 2,300 and 4,100 ft^3/s in 1996 and 1997, respectively. During the remaining 3 days of the sampling period, flows ranged from 685 to 10,500 ft^3/s in 1996 and from 94.5 to 9,460 ft^3/s in 1997.

Some constituent concentrations were computed from these measured values. Nitrate concentrations were computed by subtracting the nitrite from the nitrite-plus-nitrate concentrations. Organic nitrogen concentrations were computed by subtracting ammonia concentrations from TKN, and organic phosphorus concentrations were computed by subtracting dissolved orthophosphate from total phosphorus concentrations.

Many water samples had measured constituent concentrations less than the detection limit, which were reported as censored values. The rationale that was used to address censored values of constituents was to apply more consistent approaches that provided the highest possible concentration values for model input. When determining a computed constituent concentration from a censored concentration, a censored concentration was replaced by zero producing the highest possible computed concentration. For example, given a nitrite-plus-nitrate concentration of 0.81 mg/L and a censored nitrite concentration of less than 0.02 mg/L, the computed nitrate concentration would be 0.81 mg/L because the nitrite concentration was replaced by zero. However, for input of measured constituent concentrations into the water-quality model, censored values were replaced by the detection limit of a selected constituent. For example, if the nitrite concentration was reported as less than 0.02 mg/L, the value of 0.02 mg/L was used in the model.

Water-quality data collected in the study reach at RM 126.9, about 3 mi downstream from the mouth of Sugar Creek, were not used in this study. Point-source samples were collected from the west bank of the river, whereas Sugar Creek enters the river from the east. Subsequent analysis indicated that the inflow from Sugar Creek had not mixed laterally at RM 126.9. Hence, the single grab sample was not representative of water quality in the entire cross section. Samples also were collected at the mouth of Cane Creek (RM 110.4, fig. 2) but were not included in the model. Cane Creek is located approximately 0.1 mi upstream from the downstream boundary, which is not enough distance for sufficient mixing of the water-quality constituents. Consequently, RM 111.4 was used as the most downstream location for comparing simulated and observed water-quality constituents.

A private laboratory certified by the SCDHEC provided the chemical analyses for the water-quality samples. The contract laboratory provided sample bottles, which were pre-cleaned and prepared with preservatives as needed. Samples were kept on ice in coolers also provided by the laboratory, and were picked up several times per day by a representative of the contract laboratory in order to assure that samples with limited holding times were processed within the required timeframe. Analytical results were reviewed for completeness and accuracy as part of the USGS, South Carolina District, quality-assurance and quality-control procedures (W. Wang, U.S. Geological Survey, written commun., 1999).

The contract laboratory analyzed samples collected at each sampling site for ultimate carbonaceous biochemical oxygen demand ($CBOD_u$) and BOD_5. A review of the laboratory data indicated that most of the results from the $CBOD_u$ analyses were unacceptable. The $CBOD_u$ results were to have been used to compute site-specific f-ratios (ratio of $CBOD_u$ to BOD_5) and decay coefficients. Because site-specific f-ratios could not be computed from the Catawba River samples, f-ratios currently used by the SCDHEC were used in the water-quality model (M. Carswell, South Carolina Department of Health and Environmental Control, oral commun., October 1998). An f-ratio of 1.5:1 was used to convert measured instream BOD_5 to $CBOD_u$, and the f-ratios used for the wastewater effluents were as follows: sites A and D, 1.5:1; sites B, C, and F, 3:1; and site E, 4:1.

The method used to determine chlorophyll *a* concentrations involves obtaining a sample extract by concentrating the sample through several processes using acetone acid. The sample extract is then placed in a cuvette and the optical density is read at 750, 664, 647, and 630 nanometers. The chlorophyll *a* concentration of the original sample is computed by multiplying the extract concentration by a corrected optical density factor. For this study, instead of reporting the chlorophyll *a* sample concentrations, the contract laboratory inadvertently reported the chlorophyll *a* extract concentrations. The extract concentrations, in milligram per liter, were converted to sample concentrations, in microgram per liter (µg/L), by multiplying the extract concentrations by 75 (Michael Woodrum, Shealy Environmental Laboratory, oral commun., 1999). The detection limit used by the laboratory for the chlorophyll *a* extract concentrations was 0.001 mg/L. Consequently, the detection limit for the chlorophyll *a* sample concentrations was 0.075 µg/L.

Laboratory Quality Assurance and Quality Control

A summary of the quality assurance and quality control for the laboratory analyses was compiled. Environmental samples were analyzed in five sets by the laboratory. The total number of individual analytical runs ranged from 60 to 118 in each laboratory set. Quality assurance data were provided by the laboratory in a compiled format for TKN and ammonia analyses and by laboratory set for the remaining analyses.

For the nitrite-plus-nitrate analysis, each laboratory set included 9 to17 method blanks, 9 to 17 standards with concentrations of 0.02, 0.80, and 2.0 mg/L, and 6 to 8 matrix spikes and matrix spike duplicates. All method blanks were below the detection limit of 0.020 mg/L. Percent differences between the standard concentrations and the measured values were less than 10 percent. The percent recoveries of the matrix spikes and matrix duplicate spikes ranged from 76 to 121 percent. Relative percent difference between matrix spikes and duplicates was less than 10 percent except for one spike/duplicate with a percent difference of 11 percent.

For nitrite analysis, 8 to15 method blanks, 10 to 13 standards with concentrations of 0.02, 0.80, and 2.0 mg/L, and 6 to 8 matrix spikes and matrix spike duplicates were analyzed for each laboratory set. All but one method blank were below the detection limit of 0.020 mg/L; the one detection had a concentration of 0.032 mg/L. The percent difference between the standard concentration and the measured value was less than 10 percent for analyzed standards. The percent recoveries ranged from 70 to 135 percent of the matrix spikes and duplicate spikes except for one matrix spike and duplicate spike that had an unacceptably low recovery of 49 percent. Relative percent difference between matrix spikes and duplicates was less than or equal to 2 percent.

For TKN analysis, a total of 32 method blanks, 33 standards with concentrations of 1.0 and 10.0 mg/L, and 28 matrix spikes and matrix spike duplicates were analyzed. Detections of TKN in the method blanks ranged from 0.11 to 0.46 mg/L with a median value of 0.23 mg/L. To adjust for the detections in the method blanks, blank values were subtracted from standard and environmental sample measurements. Percent differences between the standard concentrations and the measured values were less than 8 percent. Percent recoveries of matrix spikes and matrix spike replicates ranged from 85 to 106 percent, with median relative percent difference between spikes and duplicates of 2.5 percent (ranged between 0 and 16 percent).

For ammonia analysis, a total of 42 method blanks, 34 standards with concentrations of 0.1, 1.0, and 10.0 mg/L, and 28 matrix spikes and matrix spike duplicates were analyzed. Detections in the method blanks ranged from 0.014 to 0.92 mg/L with a median value of 0.099 mg/L. Percent differences between the standard concentrations and the measured values were less than 9 percent. Percent recoveries of matrix spikes and duplicate spikes ranged from 78 to 115 percent. The median relative percent difference between spikes and duplicates was 2.4 percent (ranged between 0 and 13 percent).

For dissolved phosphorus analysis, 7 to 8 method blanks, 9 to 10 standards with concentrations of 0.01, 0.5, and 3.0 mg/L, and 5 to 7 matrix spikes and matrix spike duplicates were analyzed for each laboratory set. Results of the method blanks were all below the detection limit of 0.01 mg/L. Percent differences between the standard concentrations and the measured values were less than 10 percent. Percent recoveries ranged from 80 to 120 percent for 25 of the 28 matrix spikes; 3 matrix spikes had unacceptably low recoveries of 3, 41, and 44 percent.

For orthophosphate analysis, 7 to 8 method blanks, 9 to 10 standards with concentrations of 0.01, 0.5, and 3.0 mg/L, and 5 to 7 matrix spikes and matrix spike duplicates were analyzed for each laboratory set. The method blanks were less than the detection limit of 0.010 mg/L except for one blank with a detection of 0.15 mg/L. Percent differences between the standard concentrations and the measured values were less than 10 percent. The percent recoveries ranged from 75 to 126 percent for 27 of the 28 matrix spikes and duplicate spikes. One matrix spike and duplicate spike had an unacceptably low recovery of 36 percent.

For total phosphorus analysis, 7 to 8 method blanks, 9 to 10 standards with concentrations of 0.01, 0.5, and 3.0 mg/L, and 5 to 7 matrix spikes and matrix spike duplicates were analyzed for each laboratory set. All method blanks were below the detection limit of 0.01 mg/L. Percent differences between the standard concentrations and the measured values were less than 10 percent. The percent recoveries ranged from 80 to 120 percent for 27 of the 28 matrix spikes: 1 matrix spike had an unacceptably low recovery of 56 percent.

For 5-day biochemical oxygen demand analysis, one method blank, two standards, and five to seven duplicates were analyzed for each laboratory set. Only one blank was below the detection limit of 0.5 mg/L; the one detection had a concentration of 1.3 mg/L. The standard was a 2-percent glucose-glutamic acid standard with a concentration of 198 mg/L. The measured values had to be within ±30.5 mg/L of the 198 mg/L standard concentration. All but one of the measured values were within ±30.5 mg/L of the standard concentration; one measured value was +46 mg/L (the same laboratory set as the detection). Relative percent differences between duplicates and environmental samples ranged from 0 to 59 percent.

For chlorophyll *a* analysis, one method blank and five to seven duplicates were analyzed for each laboratory set. All method blanks were below the detection limit of 0.075 g/L. Relative percent differences between duplicates and environmental samples were zero except for two analyses that had -10 and -20 percent differences.

Summary Statistics

The water-quality data are summarized statistically, including mean and median values, in table 13. For this analysis, all concentrations reported as less than the detection limit were set equal to the detection limit. The mean is an average of the data set and is influenced strongly by extremely high or low values, commonly called outliers. The median represents the 50th percentile of the data, which indicates that 50 percent of the data are either greater or less than the value. The median is not affected as strongly by outliers and is often used to compare ranges of data in different data sets (Helsel and Hirsch, 1992). Additional statistics were used to construct box plots of concentration data for constituents of interest. Box plots serve as graphical summaries of the data distribution, displaying the median, range (from the upper 75th to lower 25th percentile representing 50 percent of the data), skewness, and outliers (Helsel, 1987).

Median organic nitrogen concentrations in the Catawba River ranged from 0.07 to 0.16 mg/L in August 1996 and from 0.10 to 0.21 mg/L in July 1997 (figs. 9, 10; table 13). The highest median organic nitrogen concentrations (0.16 and 0.21 mg/L) were measured at RM 123.0 in 1996 and at RM 117.9 in 1997, respectively. Median ammonia concentrations were high (0.19 mg/L) in both 1996 and 1997 at RM 117.9 relative to most other main stem sites (figs. 9, 10; table 13). The highest median ammonia concentrations (0.20 and 0.21 mg/L), however, were at the upstream end of the study reach at RM 139.9 because of releases of ammonia-enriched hypolimnetic waters from Lake Wylie.

Median nitrate concentrations were higher in the downstream segment of the study reach, and nitrate concentrations generally increased from 0.16 and 0.13 mg/L upstream to 0.61 and 0.50 mg/L downstream in 1996 and 1997, respectively (figs. 9, 10; table 13). A fairly large increase in median nitrate concentrations occurred between RM 134.6 and RM 122.0 (figs. 9, 10; table 13); these two stations bracket the confluence of Sugar Creek with the Catawba River at RM 129.8. Median nitrate concentrations of 2.75 and 2.92 mg/L in Sugar Creek were an order of magnitude greater than at other locations on the Catawba River in 1996 and 1997, respectively. The highest nitrate concentrations in Sugar Creek occurred during August 1996, when streamflow in Sugar Creek was somewhat elevated following rainfall during August 24–25 (figs. 11, 12; table 13). Nitrate concentrations in the river downstream from Sugar Creek also generally were higher than at upstream sites for both sampling periods.

Median concentrations of organic phosphorus in the Catawba River ranged from 0.02 to 0.09 mg/L in 1996, and from 0.10 to 0.17 mg/L in 1997 (figs. 9, 10; table 13). Median concentrations of orthophosphate ranged from less than the detection limit of 0.01 to 0.17 mg/L in 1996, and from below the detection limit to 0.14 mg/L in 1997 (figs. 9, 10; table 13). The highest median concentrations of orthophosphate were measured at RM 115.5 (0.17 mg/L) in 1996 and at RM 117.9 (0.14 mg/L) in 1997. River miles 117.9 and 115.5 are located just upstream and downstream, respectively, from the Landsford Rapids. There was a general downstream increase in orthophosphate concentrations for both 1996 and 1997 (figs. 9, 10; table 13).

Median BOD_5 concentrations in the river ranged from less than 1.0 to 1.3 mg/L in 1996 and from 1.9 to 4.7 mg/L in 1997 (figs 9, 10; table 13). The highest BOD_5 concentrations during both periods were downstream from RM 123.0. BOD_5 concentrations in water released from Lake Wylie were somewhat greater in 1997 than in 1996. The average daily BOD load from the six point-source dischargers in 1996 (6.86 tons per day (ton/d) during August 23–27, 1996) was slightly more than the average daily load in 1997 (6.46 tons/d during July 11–15, 1997). The highest loads were from dischargers D, E, and F. The BOD_5 concentrations for July 14–15, 1997, were not reported because of laboratory sample-preparation problems. The median BOD_5 concentrations in Sugar Creek were higher in 1997 than in 1996 (figs. 11, 12; table 13). The higher BOD concentrations in the Catawba River during the 1997 study period are probably a result of a combination of (1) higher inputs from Lake Wylie and (2) possible resuspension of bottom organic material as a result of higher flows in 1997 relative to 1996.

Median chlorophyll *a* concentrations measured in the Catawba River in 1996 ranged from 3.2 µg/L at RM 139.9 to 7.6 µg/L at RM 117.9 (table 13). The general pattern of increasing chlorophyll *a*

Table 13. Mean and median concentrations of selected constituents in main-stem sites on and tributary sites to the Catawba River, S.C., for August 23–27, 1996, and July 11–15, 1997

[<, less than detection limit; NC, not computed]

River mile	Number of samples	Constituent concentrations, in milligrams per liter									
		Total Kjeldahl nitrogen		Total organic nitrogen		Total ammonia		Total nitrate		Total phosphorus	
		Mean	Median	Mean	Median	Mean	Median	Mean	Median	Mean	Median
August 23–27, 1996											
Main-stem sites on Catawba River											
139.9	20	0.33	0.30	0.11	0.07	0.22	0.20	0.14	0.16	0.04	0.04
137.6	20	.31	.23	.10	.08	.20	.14	.19	.22	.04	.04
134.6	20	.37	.30	.13	.08	.24	.17	.21	.23	.12	.11
123.0	13	.41	.34	.23	.16	.19	.11	.37	.39	.14	.13
122.0	20	.40	.29	.21	.10	.19	.13	.54	.52	.21	.19
117.9	18	.32	.28	.11	.07	.21	.19	.58	.55	.24	.22
115.5	19	.34	.29	.13	.11	.21	.14	.63	.61	.27	.27
111.4	20	.30	.25	.12	.11	.18	.15	.57	.58	.22	.22
Tributary sites to Catawba River											
Sugar Creek	20	0.41	0.39	0.23	0.14	0.18	0.15	3.48	2.75	1.40	1.16
Twelvemile Creek	20	.40	.37	.26	.18	.14	.13	.56	.56	.18	.15
Waxhaw Creek	20	.33	.28	.18	.15	.15	.13	.12	.12	.09	.09
July 11–15, 1997											
Main-stem sites on Catawba River											
139.9	20	0.37	0.37	0.16	0.12	0.21	0.21	0.13	0.13	0.14	0.13
137.6	20	.34	.32	.17	.12	.17	.18	.14	.16	.18	.19
134.6	20	.36	.32	.21	.20	.16	.16	.19	.19	.23	.23
123.0	20	.34	.27	.20	.10	.15	.13	.33	.27	.22	.20
122.0	18	.38	.33	.25	.17	.15	.14	.43	.35	.25	.26
117.9	17	.47	.41	.28	.21	.18	.19	.50	.45	.26	.25
115.5	19	.38	.34	.22	.20	.16	.16	.55	.50	.28	.27
111.4	17	.38	.29	.25	.18	.18	.17	.52	.49	.30	.29
Tributary sites to Catawba River											
Sugar Creek	20	0.76	0.70	0.52	0.51	0.24	0.16	2.76	2.92	2.42	2.40
Twelvemile Creek	20	.31	.30	.16	.14	.15	.12	.32	.33	.28	.26
Waxhaw Creek	20	.28	.24	.13	.10	.15	.14	.09	.10	.29	.28

Table 13. Mean and median concentrations of selected constituents in main-stem sites on and tributary sites to the Catawba River, S.C., for August 23–27, 1996, and July 11–15, 1997—Continued

[<, less than detection limit; NC, not computed]

River mile	Number of samples	Constituent concentrations, in milligrams per liter									
		Total organic phosphorus		Dissolved orthophosphate		5-day biochemical oxygen demand		Dissolved oxygen		Chlorophyll *a*, in micrograms per liter	
		Mean	Median	Mean	Median	Mean	Median	Mean	Median	Mean	Median
August 23–27, 1996											
Main-stem sites on Catawba River											
139.9	20	0.03	0.02	0.01	<0.01	1.4	<1.0	4.4	4.2	5.4	3.2
137.6	20	.03	.03	.01	<.01	1.1	<1.0	4.8	4.9	4.7	4.8
134.6	20	.05	.05	.07	.07	1.5	<1.0	5.6	5.5	5.1	3.2
123.0	13	.06	.05	.08	.08	1.9	<1.0	6.1	5.8	4.9	4.4
122.0	20	.08	.08	.13	.12	1.3	<1.0	6.0	5.9	6.4	5.8
117.9	18	.10	.09	.14	.14	1.8	1.3	5.4	5.2	8.0	7.6
115.5	19	.10	.08	.17	.17	1.4	1.2	6.5	6.5	7.9	6.4
111.4	20	.08	.08	.15	.16	1.4	<1.0	6.4	6.2	5.7	6.2
Tributary sites to Catawba River											
Sugar Creek	20	0.58	0.33	0.83	0.80	2.7	2.2	NC	NC	12.0	10.4
Twelvemile Creek	20	.12	.09	.06	.06	1.8	1.7	NC	NC	18.5	13.7
Waxhaw Creek	20	.04	.04	.05	.04	1.8	1.2	NC	NC	7.3	5.2
July 11–15, 1997											
Main-stem sites on Catawba River											
139.9	20	0.12	0.10	0.02	<0.01	3.0	2.0	4.1	4.4	<0.075	<0.075
137.6	20	.15	.14	.03	<.01	3.4	2.2	4.5	4.4	<.075	<.075
134.6	20	.18	.17	.05	.04	3.5	2.0	5.6	5.3	.018	<.075
123.0	20	.14	.12	.09	.06	2.5	1.9	6.5	6.3	<.075	<.075
122.0	18	.16	.17	.11	.09	5.0	2.8	6.5	6.3	<.075	<.075
117.9	17	.13	.12	.15	.14	4.1	2.1	6.3	6.3	.083	<.075
115.5	19	.13	.14	.16	.13	5.1	4.5	6.8	6.7	<.075	<.075
111.4	17	.19	.15	.13	.12	5.9	4.7	6.5	6.6	<.075	<.075
Tributary sites to Catawba River											
Sugar Creek	20	0.41	0.14	2.02	2.25	3.8	2.5	NC	NC	<0.075	<0.075
Twelvemile Creek	20	.13	.13	.15	.15	3.8	2.6	NC	NC	<.075	<.075
Waxhaw Creek	20	.12	.09	.17	.17	6.2	5.4	NC	NC	<.075	<.075

NOTE: To compute mean values, concentrations reported as less than detection limit were set equal to the detection limit.

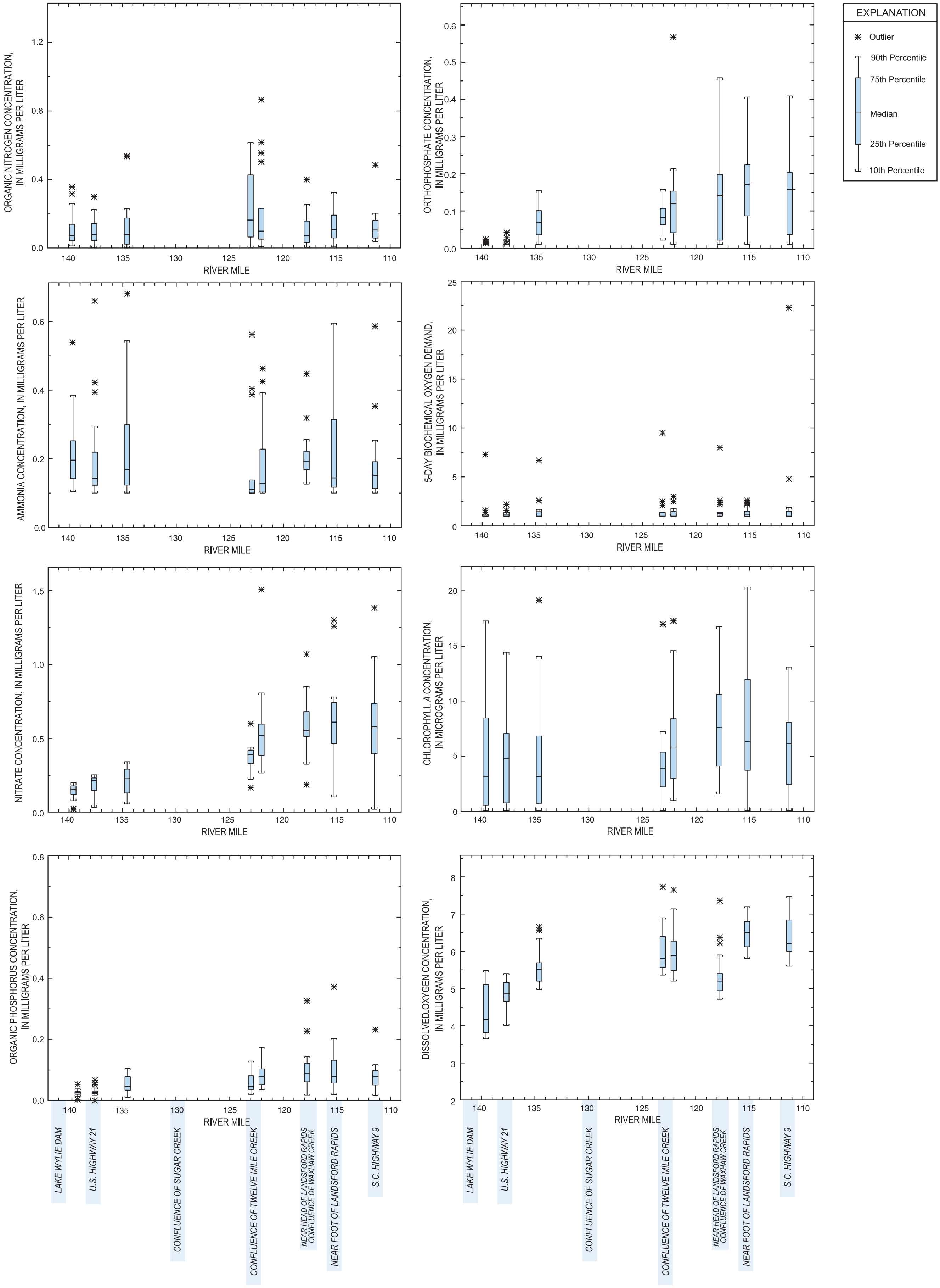


Figure 9. Summary of water-quality data collected at eight stations on the Catawba River, S.C., August 23–27, 1996.

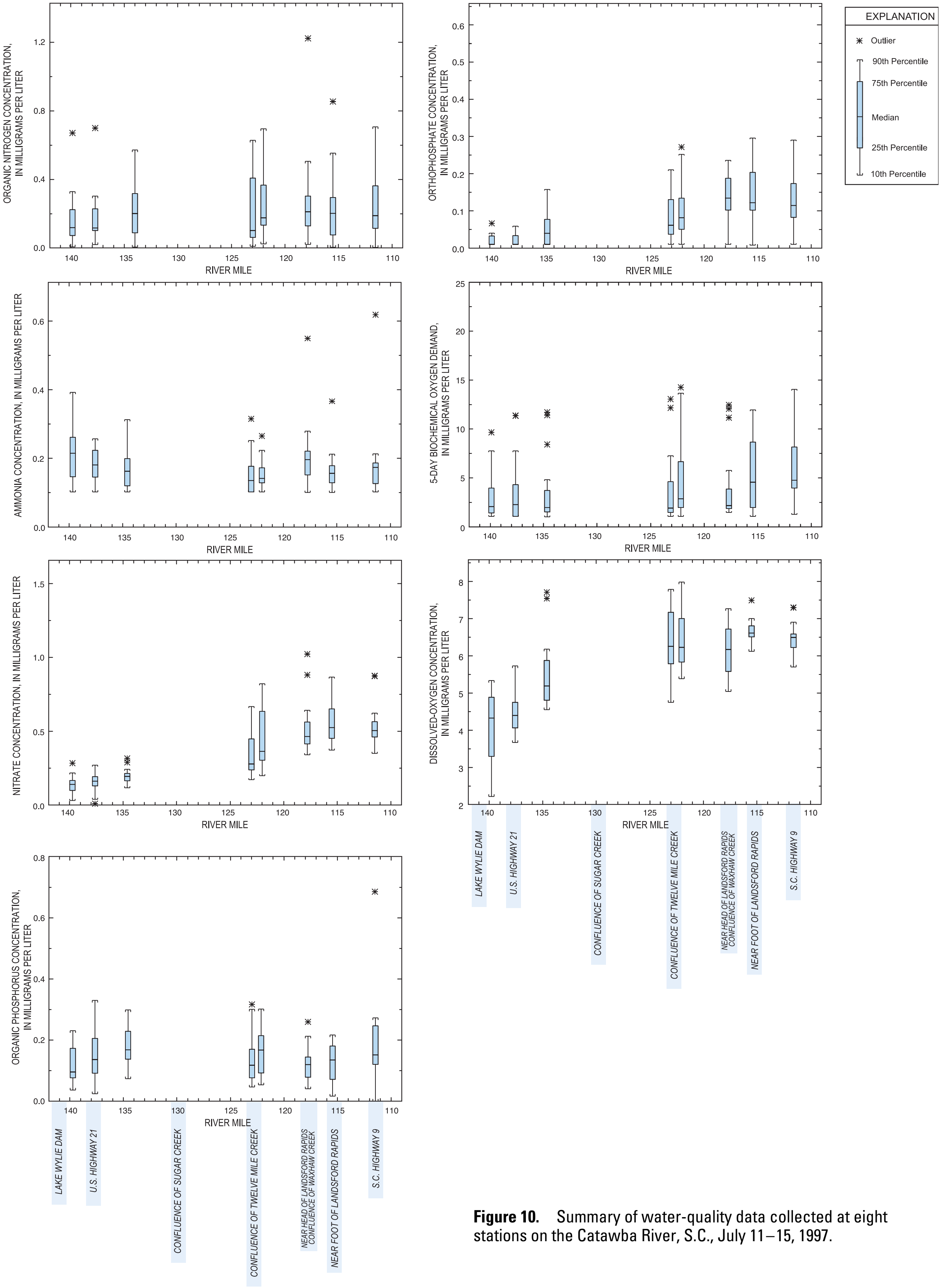


Figure 10. Summary of water-quality data collected at eight stations on the Catawba River, S.C., July 11–15, 1997.

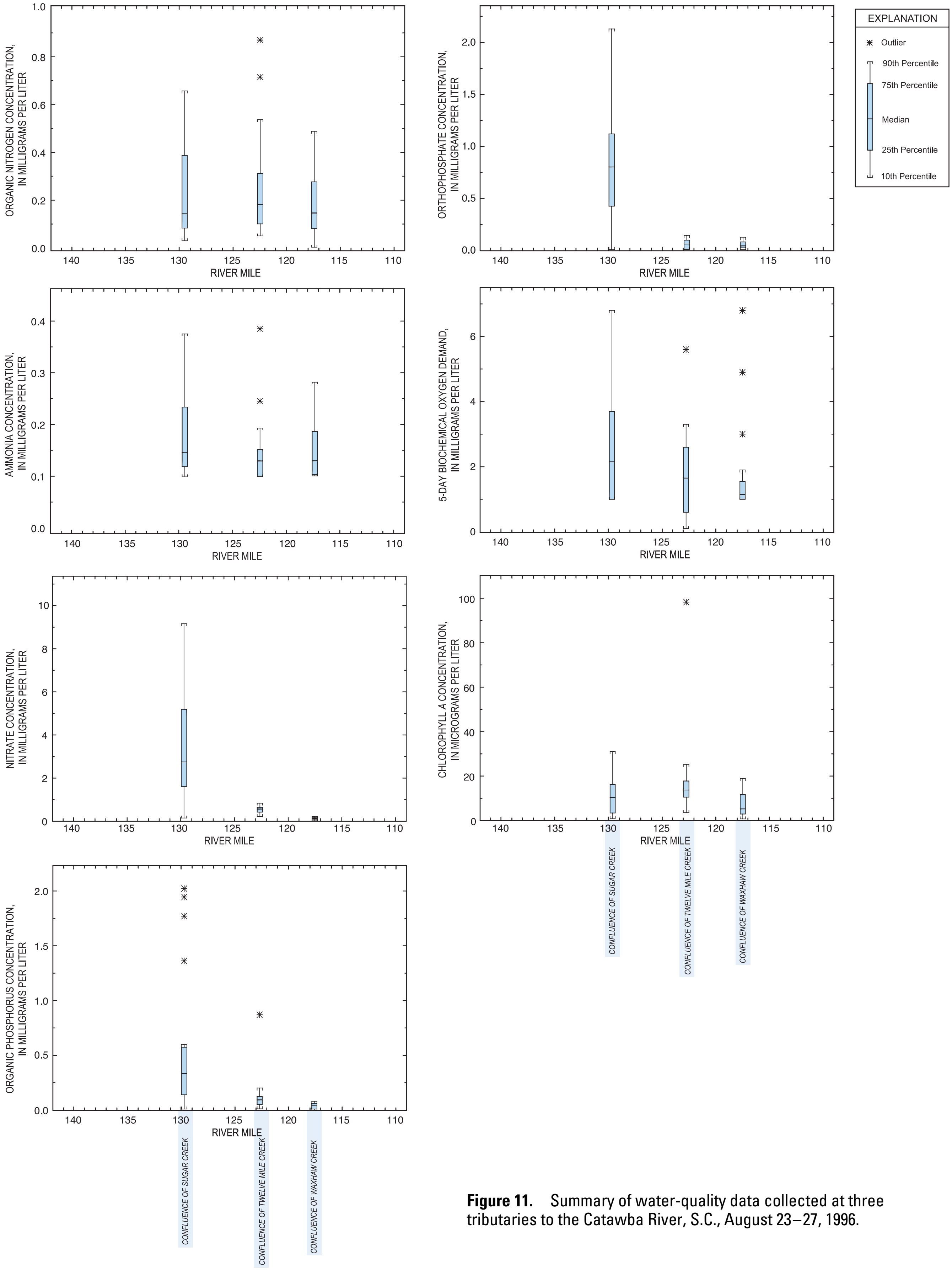


Figure 11. Summary of water-quality data collected at three tributaries to the Catawba River, S.C., August 23–27, 1996.

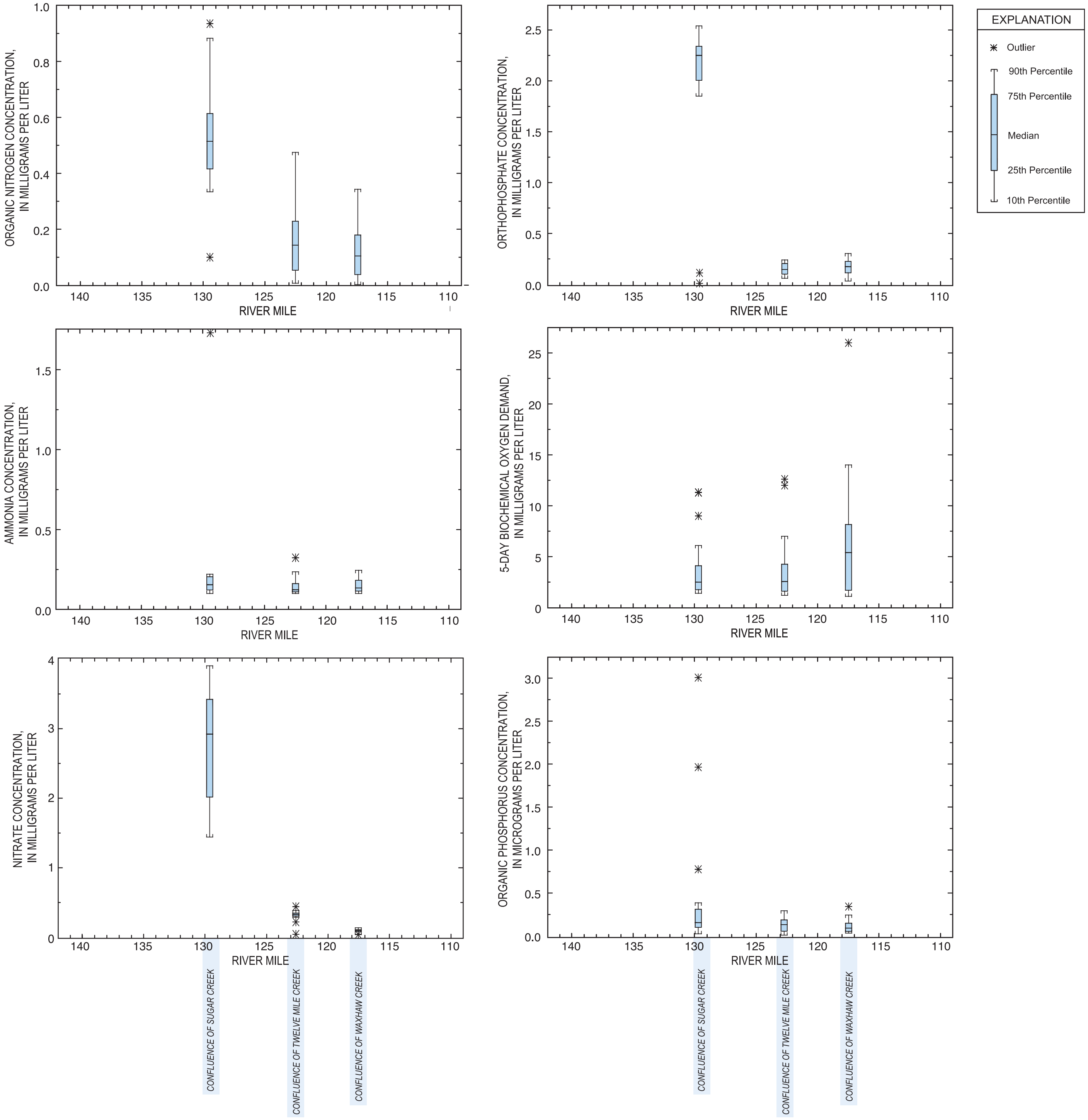


Figure 12. Summary of water-quality data collected at three tributaries to the Catawba River, S.C., July 11–15, 1997.

concentrations in 1996 is similar to the patterns for nitrate and orthophosphate (the most biologically available nutrients) during the same time period. All of the chlorophyll *a* concentrations in the 1997 samples were less than the detection limit of 0.075 µg/L with the exception of two samples, which were 2.18 µg/L at RM 134.6 and 0.22 µg/L at RM 117.9. Because of this, no box plots are included for the 1997 chlorophyll *a* concentrations.

The SCDHEC has collected water-quality samples at three locations in the study reach since 1958. The SCDHEC sites are at RM 137.6 (SCDHEC site number CW-014), RM 122.8 (CW-041), and RM 111.4 (CW-016, fig. 2). The BOD_5, total phosphorus, and TKN (organic nitrogen plus ammonia) data for these three stations for the period 1958–97 were retrieved from the USEPA STORET database. Mean concentrations of these constituents for the periods 1958–97 (entire period) and 1990–97 (recent period) in the database were compared with mean concentrations of the same constituents collected during this study at the respective stations, and the results are shown in table 14.

In addition to the main-stem water-quality samples, samples also were collected near the mouths of the three major tributaries—Sugar, Twelvemile, and Waxhaw Creeks—to the study reach. The median organic nitrogen concentrations were similar at all tributary sites, ranging from 0.10 to 0.18 mg/L for both 1996 and 1997 sampling periods, except for Sugar Creek (0.51 mg/L) in 1997 (figs. 11, 12; table 13). Median ammonia concentrations in the tributaries were similar to the median concentrations in the Catawba River, although one sample collected from Sugar Creek in 1997 had a concentration of 1.73 mg/L and is plotted as an outlier in figure 12. The median nitrate concentrations in Sugar Creek were 2.75 and 2.92 mg/L in 1996 and 1997, respectively; whereas, the median nitrate concentrations in Twelvemile and Waxhaw Creeks ranged from 0.10 to 0.56 mg/L in 1996 and 1997, respectively (figs. 11, 12; table 13). As previously noted, nitrate concentrations in the Catawba River were noticeably higher downstream from the mouth of Sugar Creek than upstream (figs. 9, 10; table 13).

Median organic phosphorus concentrations in Sugar Creek were higher than those in Twelvemile and Waxhaw Creeks in 1996 but concentrations at all three sites were similar in 1997. The median organic phosphorus concentrations for Sugar Creek were 0.33 and 0.14 mg/L in 1996 and 1997, respectively; whereas, median organic phosphorus concentrations were 0.09 and 0.13 mg/L in Twelvemile Creek, and 0.04 and 0.09 mg/L in Waxhaw Creek in 1996 and 1997, respectively (figs. 11, 12; table 13). Median orthophosphate concentrations in 1996 and 1997 were an order of magnitude higher in Sugar Creek (0.80 and 2.25 mg/L, respectively) than in Twelvemile and Waxhaw Creeks. As with nitrate, orthophosphate concentrations in the Catawba River generally were higher downstream from the mouth of Sugar Creek than upstream. Twelvemile and Waxhaw Creeks had median orthophosphate concentrations within the range of the Catawba River sites downstream from Sugar Creek.

Table 14. Mean concentrations of 5-day biochemical oxygen demand, total phosphorus, and total Kjeldahl nitrogen collected by the South Carolina Department of Health and Environmental Control during 1958–97 and by the U.S. Geological Survey during this study at three locations on the Catawba River, S.C.

[BOD_5, 5-day biochemical oxygen demand; mg/L, milligram per liter]

Data-collection period	Mean concentration		
	BOD_5 (mg/L)	Total phosphorus (mg/L)	Total Kjeldahl nitrogen (mg/L)
River mile 137.6			
1958–97 (variable collection intervals)	1.91	0.06	0.42
1990–97 (monthly collection interval)	1.25	0.04	0.38
1996–97	2.30	0.11	0.32
River mile 122.8			
1958–97 (variable collection intervals)	2.66	0.19	0.67
1990–97 (monthly collection interval)	1.23	0.11	0.38
1996–97	2.87	0.16	0.37
River mile 111.4			
1958–97 (variable collection intervals)	2.89	0.19	0.69
1990–97 (monthly collection interval)	2.71	0.14	0.50
1996–97	3.46	0.20	0.32

In 1996, the highest median chlorophyll *a* concentration of 13.7 µg/L was measured in Twelvemile Creek, and the maximum concentration of

98.2 µg/L also was detected in a sample from Twelvemile Creek (fig. 11). The median chlorophyll *a* concentrations in Sugar and Twelvemile Creeks of 10.4 and 13.7 µg/L, respectively, were greater than the median concentrations measured at any of the river sites. All chlorophyll *a* concentrations in the tributaries in 1997 were less than the detection limit of 0.075 µg/L.

Substrate and Sediment Oxygen Demand Data

Sediment-oxygen demand (SOD) can be a major consumer of DO in rivers (Schnoor, 1996). The settling of particulate organic matter results in bottom material with a high organic content, which generates an oxygen demand in the overlying water as bacteria consume the organic material. Organic matter can originate from human sources, such as wastewater discharges or polluted runoff, or from natural sources, such as rooted aquatic plants or natural detritus in runoff (Thomann and Mueller, 1987). In areas of high organic loading, the SOD can be in the range of 5 to 10 grams of oxygen per square meter per day [(gO_2/m^2)/d]; for natural waters, or streams with small wastewater discharges, the range is 0.1 to 1.0 (gO_2/m^2)/d (Schnoor, 1996). Sediment-oxygen demand also is a function of stream velocity—in reaches having high velocities, very little organic matter settles to the bottom or is scoured during high-flow events.

Community metabolism and SOD were determined to provide an estimate of the oxygen-consuming processes of the total substrate and sediment community in the Catawba River (M. Koenig, U.S. Environmental Protection Agency, written commun., 1998). Community metabolism was assessed by using methods described by Odum and Hoskins (1958), and SOD rates were determined at selected locations (fig. 2; table 15) by using the chamber method of Murphy and Hicks (1986). Included in the assessment of metabolic oxygen demand are the total community oxygen gross primary production (GPP), total community respiration, water-column respiration, community substrate-oxygen demand (CSOD), and SOD at selected locations (table 15). The community metabolism approach was used to determine the oxygen demand of the substrate, because the Catawba River is dominated by rocky substrates and (or) scoured bottoms, which preclude the deployment of SOD chambers in most locations.

The SOD chamber method involves placement of an opaque chamber over an isolated area of sediment and volume of water. The DO concentrations are then monitored to determine the rate of change of DO in the chamber. The major difference in the CSOD and the SOD is that the SOD is an on-site measurement of the rate that oxygen is consumed by the sediments, whereas the CSOD is a calculated value obtained by subtracting the water column respiration from the total community respiration. The water-column respiration is measured in an enclosed chamber, and the total community respiration is computed by using the diurnal curve method (Odum and Hoskins, 1958) applied to the DO and water-temperature data collected continuously during the August 24–25, 1996, and July 12–13, 1997, study periods. The diurnal curve

Table 15. Community production and respiration, water-column respiration, and substrate- and sediment-oxygen demand for selected locations on the Catawba River, S.C., August 24–25, 1996, and July 12–13, 1997

[(gO_2/m^3)/d, grams of oxygen per cubic meter per day; (gO_2/m^2)/d, grams of oxygen per square meter per day; °C, degrees Celsius; —, no data]

River mile	Community gross primary production, in (gO_2/m^3)/d		Total community respiration, in (gO_2/m^3)/d		Water column respiration, in (gO_2/m^3)/d		Community substrate-oxygen demand and sediment-oxygen demand, in (gO_2/m^2)/d at 20 °C	
	August 24–25, 1996	July 12–13, 1997	August 24–25, 1996	July 12–13, 1997	August 24–25, 1996	July 12–13, 1997	August 24–25, 1996	July 12–13, 1997
122.8	5.46	1.80	5.04	2.34	1.49	1.38	3.40[a]	1.10[a]
122.0	5.70	2.60	5.28	2.64	1.49	1.38	3.70[a]	1.40[a]
119.5	—	—	—	—	—	—	—	.58[b]
118.5	—	1.40	—	1.92	—	1.38	—	.60[a]
114.3	1.26	—	4.32	—	1.49	—	2.70[a]	2.33[b]
113.6	—	—	—	—	—	—	—	.86[b]

[a] Computed by methods described by Odum and Hoskins, 1958.

[b] Computed by using sediment-oxygen demand chambers designed by Murphy and Hicks, 1986.

method includes the graphical analysis of changes in the concentration of DO over a 24-hour period with appropriate corrections for atmospheric diffusion of oxygen. Rates were adjusted to a water temperature of 20 °C. Diffusion rates were determined by using the results from the measured reaeration rate coefficients.

Dissolved-oxygen metabolism was different during the two study periods, and the difference was probably related to the hydrologic and physical conditions under which the measurements were made. The river flow was higher in 1997 than in 1996, resulting in increased velocities and depths. Gross primary production of benthic and periphytic algal communities was greater in 1996 than in 1997, which is consistent with lower flows and shallower depths, reduced scouring, and less suspended sediment in 1996, all of which allow greater light penetration through the water column. The total community respiration also was greater in 1996 than in 1997 (table 15), which also was consistent with better-developed and active benthic activity. Focus on the total community respiration rates, in comparison with water column respiration, indicated that respiration for this segment of the Catawba River was dominated by the benthic (periphytic algae) communities associated with the shallow bottoms and rock substrates, particularly during the August 1996 study. This condition was not as pronounced in the 1997 study. Increased flow probably contributed to the lower benthic algal activity in 1997. Although water-column respiration was similar in August 1996 and July 1997 (1.49 and 1.38 $(gO_2/m^3)/d$, respectively) suggesting a similar level of oxygen-demanding processes, the total community respiration was considerably less in 1997 than in 1996, suggesting other non-algal community processes contributed to the water-column respiration in 1997 (table 15).

The SOD chambers were deployed at locations along the stream (fig. 2) where the stream bottom consisted of soft (sand/mud) material to assure that a tight seal was formed between the chamber and the stream bottom. SOD measurements were made in 1997 at RM 119.5 (water depth of 4.8 ft), RM 114.3 (depth of 2.5 ft), and RM 113.6 (depth of 8.0 ft). Benthic algae were observed in the mud bottom at RM 114.3. The SOD rates at RM 119.5 and RM 113.6 were 0.58 and 0.86 $(gO_2/m^2)/d$, respectively, and the SOD rate at shallow RM 114.3 was 2.33 $(gO_2/m^2)/d$ (table 15). The higher rate at RM 114.3 was a result of the presence of the benthic algal community on the stream bottom. At stations with greater water depths, such as RM 119.5 and RM 113.6, less light penetrates to the bottom of the stream, which minimizes benthic algal activity. The similarity in the 1996 CSOD value at RM 114.3 (2.70 $(gO_2/m^2)/d$) and the 1997 SOD value (2.33 $(gO_2/m^2)/d$) suggests that the primary production and respiration in the study reach was dominated by contributions and demand of the shallow water benthic algal and (or) periphytic communities.

Point-Source Flow and Water-Quality Data

Water-quality data and flow rates from six major point-source dischargers along the study reach were obtained for the August 1996 and July 1997 sampling periods (table 16). Samples were collected at various intervals by the dischargers and composited to represent a daily mean value. Flow rates were recorded as mean flow for the day. Samples were analyzed for ammonia, nitrite, nitrite plus nitrate, dissolved orthophosphate, dissolved organic phosphorus, total phosphorus, total Kjeldahl nitrogen, and BOD_5.

Meteorological Data

Meteorological data are required to simulate water temperature in the water-quality model. Required data include daily high and low air temperature and the respective times of occurrence, daily mean wind speed, daily mean dew-point temperature, and daily mean cloud-cover conditions (table 17). Air temperature data were obtained from measurements made at the Catawba Nuclear Station (C. Sheryl, Duke Energy Corporation, written commun., 1997); the remainder of the meteorological data were measured at Douglas International Airport, Charlotte, N.C. (W. Tyler, South Carolina Department of Natural Resources, written commun., 1998). These meteorological stations are about 4.25 mi and 12.75 mi, respectively, north of Lake Wylie Dam, and about 25.3 mi and 34.5 mi, respectively, north of the downstream end of the study reach.

Temperatures generally were lower during the August 1996 sampling period than during the July 1997 data-collection period, primarily because of cloudy and rainy conditions during much of the first period (table 17). Temperatures during the July 1997 period were more typical of 1961–90 mean conditions than

Table 16. Wastewater-effluent data from water samples collected by six major dischargers on the Catawba River, S.C., during August 23–27, 1996, and July 11–15, 1997

[Mgal/d, million gallons per day; ft^3/s, cubic foot per second; mg/L, milligram per liter; TKN, total Kjeldahl nitrogen; BOD, biochemical oxygen demand; <, less than; Q, data not included, laboratory results questionable; NR, data not reported, laboratory set-up problem]

Date	Discharge, in Mgal/d (ft^3/s)	Ammonia nitrogen, in mg/L	Nitrite-nitrogen, in mg/L	Nitrite+nitrate, in mg/L	Dissolved orthophosphate, in mg/L	Dissolved phosphorus, in mg/L	Total phosphorus, in mg/L	TKN, in mg/L	5-day BOD in mg/L
Discharger A									
8/23/96	2.24 (3.47)	<0.100	0.068	0.911	Q	4.46	4.46	0.157	20.1
8/24/96	3.01 (4.66)	<.100	0.078	1.36	Q	3.02	3.05	.167	6.3
8/25/96	3.71 (5.74)	.130	0.073	1.27	13.4	13.9	14.0	.608	3.50
8/26/96	4.22 (6.53)	.126	0.021	1.81	Q	3.06	3.08	.367	1.50
8/27/96	3.93 (6.08)	<.100	<.020	3.72	3.51	3.48	4.03	.309	2.30
7/11/97	6.77 (10.5)	0.214	0.022	0.338	<0.010	<0.010	16.9	0.900	39.2
7/12/97	5.02 (7.77)	<.100	.021	.062	6.77	17.7	18.3	.558	10.1
7/13/97	5.60 (8.66)	<.100	<.020	<.020	7.87	Q	Q	.687	5.5
7/14/97	4.75 (7.35)	.133	<.020	.161	<.010	<.010	<.010	.790	Q
7/15/97	5.68 (8.79)	<.100	<.020	.034	<.010	Q	Q	1.17	Q
Discharger B									
8/23/96	0.776 (1.20)	0.163	<0.020	9.78	1.36	1.58	1.58	0.295	4.60
8/24/96	.724 (1.12)	.758	.159	11.9	Q	1.60	1.72	1.53	5.60
8/25/96	.854 (1.32)	.187	.247	7.22	Q	1.28	1.28	1.17	1.90
8/26/96	.819 (1.27)	.485	<.020	2.02	1.30	1.44	1.47	1.01	3.00
8/27/96	1.09 (1.69)	.802	.020	1.74	1.05	1.04	1.12	.987	4.30
7/11/97	0.738 (1.14)	0.302	0.044	7.01	Q	2.53	2.99	2.37	5.40
7/12/97	.753 (1.17)	.361	.031	5.49	2.56	2.64	2.89	1.39	4.90
7/13/97	.785 (1.21)	.448	.022	2.09	2.40	2.71	2.83	2.50	5.10
7/14/97	.792 (1.23)	1.78	.132	3.46	Q	2.05	2.17	4.17	NR
7/15/97	.794 (1.23)	1.39	.260	4.37	Q	2.24	2.72	3.29	NR
Discharger C									
8/23/96	0.005 (.008)	2.19	0.988	14.6	<0.010	0.041	0.408	3.78	12.9
8/24/96	.008 (.012)	2.04	.729	14.0	Q	.038	.314	3.40	10.2
8/25/96	.006 (.009)	1.70	.649	15.7	.023	.048	.257	1.74	8.80
8/26/96	.005 (.020)	1.80	.025	3.11	.040	.063	.320	3.20	<1.0
8/27/96	.008 (.012)	1.41	.321	3.19	.028	.035	.251	1.99	10.0
7/11/97	0.004 (.006)	4.59	0.674	8.95	0.085	0.155	0.341	7.29	66.0
7/12/97	.004 (.006)	3.09	.763	5.72	Q	.184	.378	6.42	8.00
7/13/97	.005 (.008)	2.07	.715	2.05	.021	.147	.180	4.35	10.4
7/14/97	.003 (.005)	1.40	.777	1.62	.083	.131	.133	4.45	NR
7/15/97	.004 (.006)	.821	.020	.184	<.010	.063	.242	3.54	NR

Table 16. Wastewater-effluent data from water samples collected by six major dischargers on the Catawba River, S.C., during August 23–27, 1996, and July 11–15, 1997—Continued

[Mgal/d, million gallons per day; ft^3/s, cubic foot per second; mg/L, milligram per liter; TKN, total Kjeldahl nitrogen; BOD, biochemical oxygen demand; <, less than; Q, data not included, laboratory results questionable; NR, data not reported, laboratory set-up problem]

Date	Discharge, in Mgal/d (ft^3/s)	Ammonia nitrogen, in mg/L	Nitrite-nitrogen, in mg/L	Nitrite+ nitrate, in mg/L	Dissolved orthophosphate, in mg/L	Dissolved phosphorus, in mg/L	Total phosphorus, in mg/L	TKN, in mg/L	5-day BOD in mg/L
Discharger D									
8/23/96	8.97 (13.9)	8.70	0.048	0.100	0.770	1.09	3.34	9.53	53.3
8/24/96	7.96 (12.3)	.111	.057	.057	1.10	1.38	1.68	5.26	25.0
8/25/96	6.06 (9.38)	6.07	Q	Q	Q	1.17	1.20	9.44	16.5
8/26/96	9.95 (15.4)	5.63	.030	.139	.457	.743	1.42	6.84	21.3
8/27/96	8.04 (12.4)	7.91	<.020	.046	2.64	2.84	3.04	9.79	27.7
7/11/97	8.88 (13.7)	0.860	0.246	1.68	Q	0.742	0.843	2.97	41.0
7/12/97	6.85 (10.6)	.608	.198	2.67	0.407	.444	.488	2.23	8.60
7/13/97	5.68 (8.79)	.452	.061	2.02	.435	.465	.484	2.07	3.40
7/14/97	8.38 (13.0)	.308	.072	1.66	.867	.904	.981	1.50	NR
7/15/97	8.64 (13.4)	.489	.174	4.03	1.16	1.19	1.31	2.38	NR
Discharger E									
8/23/96	38.2 (59.1)	3.63	<0.020	0.063	0.798	1.23	1.38	4.77	42.1
8/24/96	24.7 (38.2)	2.93	.029	.108	1.13	1.15	1.34	2.95	28.6
8/25/96	31.6 (48.9)	3.00	.068	.108	Q	Q	1.41	3.54	31.4
8/26/96	31.0 (48.0)	2.22	.052	.117	.780	1.09	1.20	3.00	29.3
8/27/96	63.1 (97.6)	2.07	.074	.088	.482	.777	1.11	2.49	29.0
7/11/97	23.0 (35.6)	5.82	0.199	0.284	0.274	1.05	1.06	7.35	35.1
7/12/97	32.4 (50.1)	5.51	.301	.925	Q	.95	1.21	6.38	44.2
7/13/97	25.8 (39.9)	6.06	Q	.218	Q	.90	1.02	6.78	48.0
7/14/97	20.2 (31.2)	5.84	.236	.426	.982	1.09	1.38	7.27	NR
7/15/97	30.9 (47.8)	.939	.143	3.32	.744	1.08	1.09	7.40	NR
Discharger F									
8/23/96	11.5 (17.8)	0.180	0.054	0.346	0.067	0.098	0.148	0.380	10.4
8/24/96	12.8 (19.8)	.223	.036	.103	Q	.107	.140	2.13	10.6
8/25/96	12.1 (18.7)	.219	.048	.330	.010	.152	.218	1.06	19.4
8/26/96	7.90 (12.2)	.267	Q	Q	.101	.180	.254	.657	26.7
8/27/96	8.00 (12.4)	.392	.037	.134	Q	.123	.278	2.02	16.1
7/11/97	9.60 (14.9)	0.410	0.072	0.568	0.144	0.178	0.219	2.13	5.00
7/12/97	11.2 (17.3)	.302	.039	.538	.225	.290	.312	2.02	12.6
7/13/97	16.3 (25.2)	.293	.048	.410	.140	.232	.207	1.97	22.4
7/14/97	14.1 (21.8)	.223	.048	.321	.025	.184	.190	2.03	NR
7/15/97	16.6 (25.7)	.223	.760	4.29	.055	.137	.234	1.99	NR

Table 17. Meteorological data for August 23–28, 1996, and July 11–16, 1997, and 1961–90 average daily high and low air temperatures in the Catawba River Basin, S.C.

[°C, degrees Celsius; mi/hr, miles per hour; F, fair; C, cloudy; MS, mostly sunny; S, sunny]

Date	Air temperature (°C) and time measured[a]		Daily mean wind speed,[b] (mi/hr)	Daily mean dewpoint temperature[b] (°C)	Cloud cover conditions for daylight hours[b]	1961–90 mean air temperature[c] (°C)	
	High	Low				High	Low
8/23/96	31.7 1600	20.9 0700	5	19.4	F	31.1	20.0
8/24/96	32.1 1700	20.5 0800	5	19.4	F	30.6	20.0
8/25/96	27.1 1800	19.5 0400	4	19.4	C	30.6	20.0
8/26/96	25.2 1200	21.1 1700	3	20.6	C	31.1	20.0
8/27/96	28.3 1700	20.1 0500	3	19.4	C	30.6	20.0
8/28/96	26.6 1700	20.2 0500	3	19.4	C	30.6	20.0
7/11/97	29.3 1800	22.3 0700	6	19.1	MS	32.2	20.6
7/12/97	30.0 1800	18.4 0700	2	15.6	MS	31.7	21.1
7/13/97	32.5 1700	19.8 0700	3	18.3	S	31.7	20.6
7/14/97	33.4 1600	22.4 0600	3	20.6	S	32.2	20.6
7/15/97	34.0 1800	22.9 0700	3	20.6	S	32.2	21.1
7/16/97	32.9 1600	22.4 0700	4	20.6	S	32.2	21.1

[a] Measured at Catawba Nuclear Station.

[b] Measured at Charlotte-Douglas International Airport.

[c] Measured at Winthrop University in Rock Hill, S.C. (H. Mizzell, South Carolina Department of Natural Resources, written commun., 1999).

were temperatures during the August 1996 period (table 17). Rainfall was measured daily during the 1996 study period, and greater amounts occurred on August 25 and 26 than on the other study days (table 18). No rainfall occurred during the July 1997 sampling period. Dew-point temperatures were less variable during the first sampling period, ranging from 19.4 to 20.6 °C, compared to a range of 15.6 to 20.6 °C during the second period (table 17). Winds speeds were low during both data-collection periods.

Table 18. Daily rainfall at selected sites in the Catawba River Basin, August 24–28, 1996

[NR, no rainfall; T, trace]

Site number (fig. 2)	Measurement location	August 1996[a] rainfall, in inches				
		24	25	26	27	28
DR1	Fort Mill	NR	0.12	0.77	NR	NR
DR2	Winthrop University	0.55	T	.06	0.02	0.51
DR3	Catawba	NR	.17	NR	.10	.41
DR4	Great Falls	.32	.84	.01	.37	.03

[a] National Oceanic and Atmospheric Administration, 1996.

Channel Geometry

Channel geometry was measured during field surveys made at various times between October 1995 and May 1996. Lateral bed-elevation profiles were surveyed at all data-collection sites and at other selected locations where significant changes in channel shape and (or) slope occurred. The "wetted" part of each cross section was measured by using a boat-mounted fathometer; flood-plain elevations were surveyed and flood-plain widths were estimated from stadia readings from standard levels. Station datums were determined by using differential global positioning system (GPS) surveys to determine reference elevations. Static observation techniques used in the GPS surveys are expected to have a maximum vertical error of ±2 centimeters (cm) plus 2 parts per million (ppm) multiplied by baseline length, and a maximum horizontal error of ±1 cm plus 2 ppm multiplied by baseline length in the horizontal. In other words, a baseline length of 10 kilometers (km) is expected to have a maximum error of plus or minus 4 cm, or 0.13 ft in the vertical [2 cm + (0.000002)(10,000 meters [m])(100 cm/m) = 4 cm], and 3 cm, or 0.10 ft in the horizontal [1 cm + (0.000002)(10,000 m)(100 cm/m = 3 cm]. The baseline length never exceeded 10 km (D.D. Nagle, U.S. Geological Survey, oral commun., 1999). Cross-section geometry could not be measured throughout the entire length of Landsford Rapids (RM 117.9–RM 115.5). Therefore, conventional levels were used to measure a few hundred feet into the channel and then the cross-section width was extrapolated from USGS 7.5-minute topographic maps (U.S. Geological Survey, 1968a, 1968e).

Channel geometry in the study reach varied considerably in width and depth (fig. 13). At the tailrace of Lake Wylie Dam, the Catawba River is wide and shallow, with an average depth, during data collection, of about 2–3 ft and a width of about 1,200 ft. The river then constricts and diverges around a small island (fig. 2), where nearly all flow is in the left channel. The left channel is about 400 ft wide and 6–8 ft deep. Below the small island, the Catawba River cross sections range from about 400 to 600 ft in width, with depths fluctuating from 7 to 9 ft just upstream from the riffles and less than a foot through the riffles until reaching Landsford Rapids (RM 117.9). The estimated widths of the cross sections through the rapids are about 1,500 ft and the average depths are about 1 ft. The Catawba River then narrows downstream from the Landsford Rapids to a width of about 400–600 ft and average depths of about 10–12 ft. Downstream from RM 111.4 (fig. 2), the river widens to 1,700 ft at RM 110.3. Two section controls are in the study reach (fig. 14), one at RM 137.5 and the other at the head of Landsford Rapids at RM 117.4.

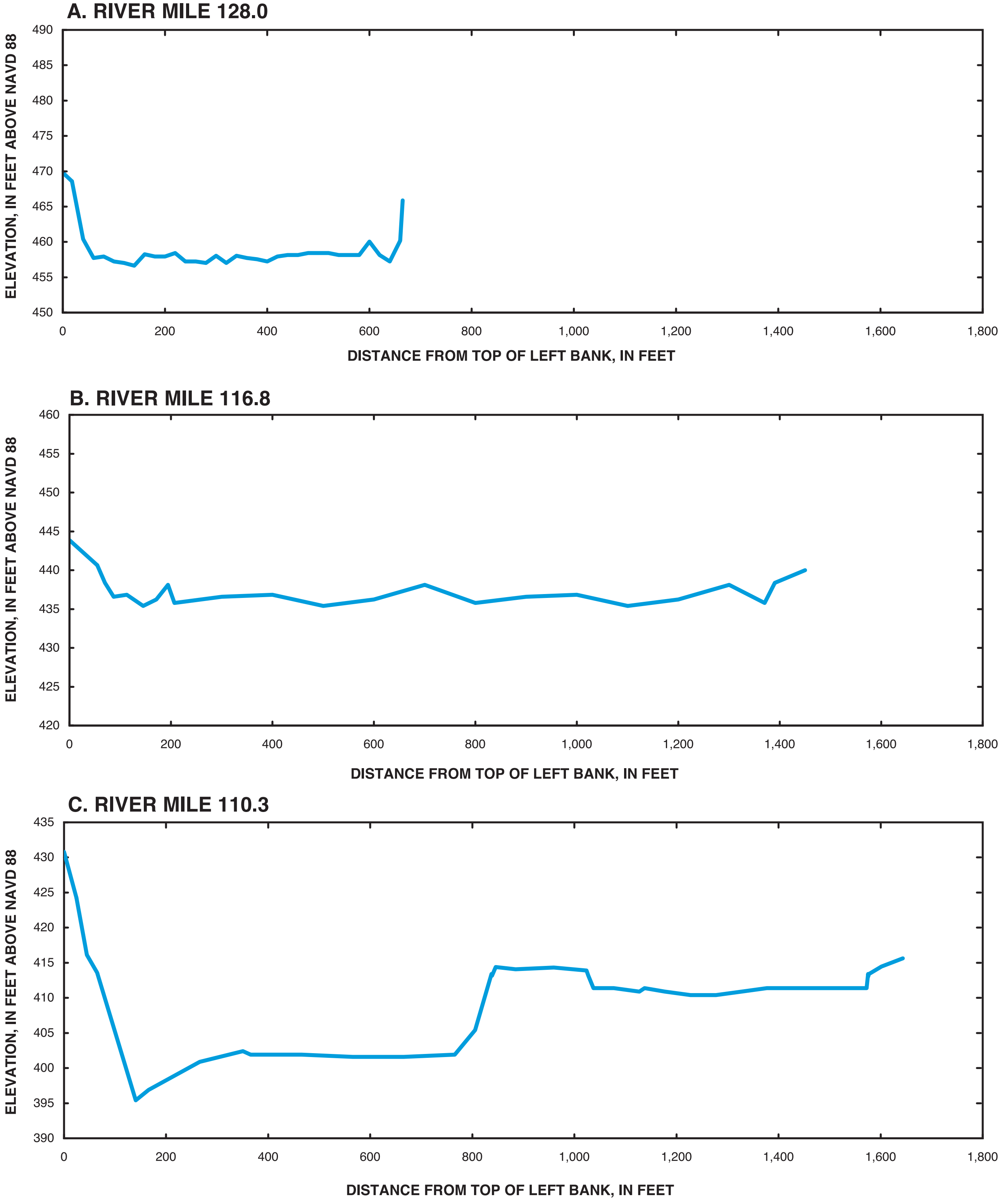


Figure 13. Cross sections of the Catawba River, S.C., at (A) pools and riffles (river mile [RM] 128.0), (B) rapids (RM 116.8), and (C) the headwaters of Fishing Creek Reservoir (RM 110.3).

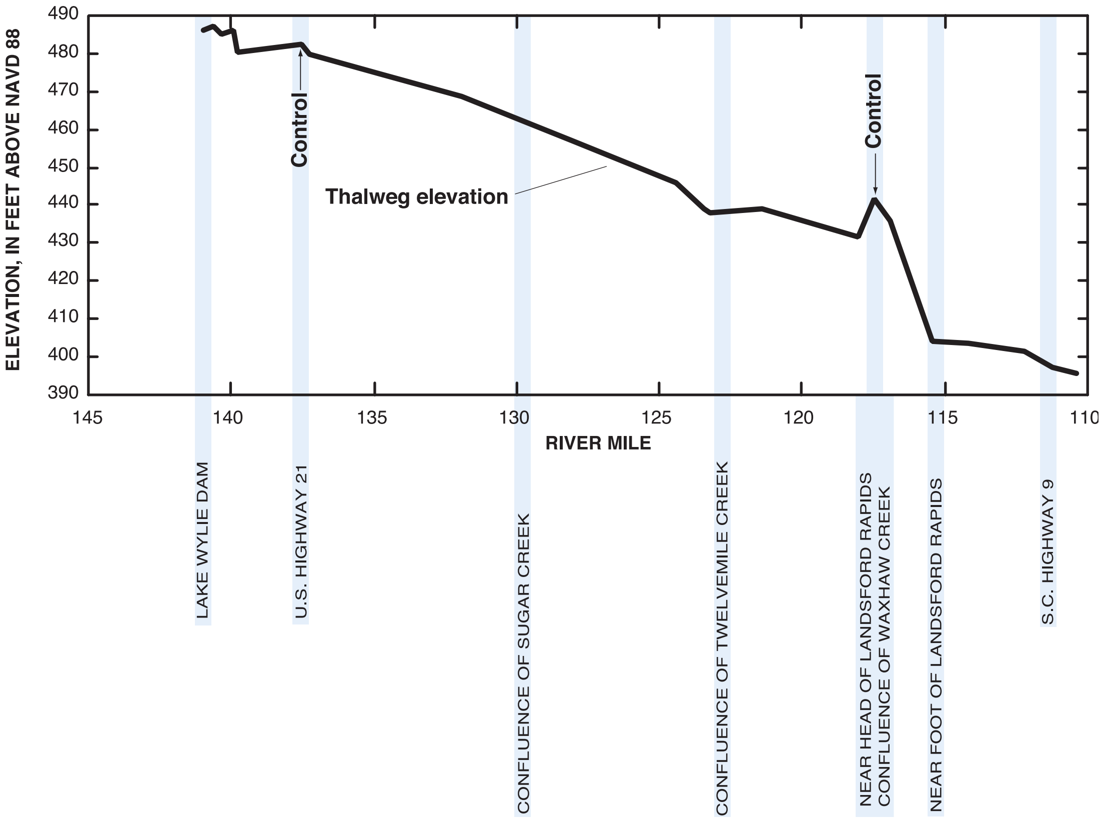


Figure 14. Longitudinal profile of the thalweg elevations of the Catawba River, S.C.

WATER-QUALITY MODEL DESCRIPTION

The one-dimensional, dynamic flow BRANCH model (Schaffranek and others, 1981) was used to simulate the hydraulic properties of the Catawba River. The hydraulic properties simulated by the BRANCH model then were used in the Branched Lagrangian Transport Model, BLTM (Jobson and Schoellhamer, 1993). The BLTM is a one-dimensional, dynamic transport model that is capable of simulating the fate of water-quality parameters, such as water temperature, DO, BOD, and nutrients in a network of open channels.

The BRANCH model is a one-dimensional, unsteady-flow numerical model for simulation of streamflow in interconnected channels (Schaffranek and others, 1981). The model can simulate flow accurately in the backwater conditions that occur downstream from the Landsford Rapids (fig. 2) as well as the unsteady flows resulting from upstream hydropower operations. The model numerically solves the equations of continuity (7) and motion (8).

$$B(\partial Z / \partial t) + (\partial Q / \partial x) - q = 0 \ , \qquad (7)$$

where

B is total channel top width, in feet;
Z is water level, in feet;
t is time, in seconds;
Q is discharge, in cubic feet per second;
x is longitudinal distance along the channel, in feet; and
q is lateral side-channel flow, in cubic feet per second per foot of channel.

$$(\partial Q / \partial t) + \partial(\beta Q^2 / A) / \partial x + gA(\partial Z / \partial x) + (gk / AR^{4/3}) Q|Q| - qu' - \xi B_c U_a^2 \cos a = 0 \ , \qquad (8)$$

where

β is the dimensionless momentum coefficient;
A is cross-sectional area, in square feet;
g is the gravitational acceleration constant, in feet per second per second;
k is a function defining flow resistance;
R is hydraulic radius, in feet;
u' is the x-component of the lateral side-channel flow velocity, in feet per second;

ξ is the dimensionless wind resistance coefficient;
B_c is the top width of the conveyance part of the cross section, in feet; and
U_a is the wind velocity in feet per second, occurring at an angle α from the positive x-axis.

Approximate solutions for the nonlinear partial-differential unsteady-flow equations are obtained by using a weighted four-point finite-difference approximation. The flow-resistance function is expressed as $k = (\eta/1.486)^2$, where η is an unsteady flow-resistance coefficient similar to Manning's *n*, which applies strictly to steady flow.

Assumptions implicit in equations 7 and 8 include homogeneous (uniform density) flow; prismatic channel (relatively straight with simple geometry), and a mild and uniform channel gradient so the flow remains subcritical. Although the Catawba River has irregularities in the channel bottom, especially at the two section controls at RM 137.5 and RM 117.4 (fig. 14), the slope between cross sections is relatively small and the flow is subcritical. The Landsford Rapids section of the Catawba River has a relatively steep slope, but flow is subcritical throughout this section.

The BLTM solves the one-dimensional, unsteady transport equation by using a Lagrangian coordinate system, in which the computational nodes move with the flow (Jobson and Schoellhamer, 1993). The transport equation is solved for each of the constituents included in the simulations. In the Lagrangian coordinate system, the transport equation is:

$$\partial C/\partial t = \partial(D(\partial C)/\partial\xi)/\partial\xi + S + \Phi + K(C - CR)\,, \quad (9)$$

where

C is concentration, in milligrams per liter;
t is time, in seconds;
D is the longitudinal dispersion coefficient, in square feet per second;
ξ is the Lagrangian-distance coordinate, in feet;
S is the rate of production of the concentration, which is independent of the concentration (zero-order production rate), in milligrams per liter per second;
Φ is the rate of change in concentration due to tributary inflow, in milligrams per liter per second;
K is the rate of production of the constituent, per second; and
CR is the equilibrium concentration (that is, the concentration at which the internal production ceases), in milligrams per liter.

The Lagrangian distance coordinate, ξ, is given by

$$\xi = x - x_o - \int_{t_o}^{t} u\,dt\,, \quad (10)$$

where

x is the Eulerian (stationary) distance coordinate along the river, in feet;
x_o is the location of the parcel of water at time t_o; and,
u is the cross-sectional mean stream velocity, in feet per second.

The dispersion factor used in BLTM is related to the dispersion coefficient and is inversely proportional to the square of the stream velocity. The factor is defined as:

$$D_f = D / \Delta t\mu^2\,, \quad (11)$$

where

D_f is dispersion factor, dimensionless;
D is dispersion coefficient, in square feet per second;
Δt is simulation time step, in seconds; and
μ is the representative stream velocity, in feet per second.

The longitudinal and temporal variation in concentrations along the river reach is approximated by solving equation 9 for a series of water parcels, which are assumed to be completely mixed and which have a length of about $\mu\Delta t$. The concentration at a specific point is the concentration of the parcel in which the point is located. The assumption of completely mixed parcels can cause interpolation errors when determining the concentration at a given point if the longitudinal concentration gradient is large, thus violating the assumption of uniform concentration in a parcel. However, no abrupt longitudinal gradients in concentration were observed in the study reach, and the assumptions of BLTM are met in this application.

The BLTM uses the same water-quality reaction kinetics (fig. 15) as the QUAL2E model (Brown and Barnwell, 1987) to simulate the fate and transport of organic nitrogen, ammonia, nitrite plus nitrate, organic phosphorus, orthophosphate, BOD, DO, and chlorophyll *a*. The BLTM can simulate the effects of multiple point-source discharges, withdrawals, tributary flows, and incremental inflows and outflows on instream water quality.

The rates of most chemical and biological reactions are temperature dependent, so accurate simulations of water temperature are required. An equilibrium temperature algorithm is needed for the BLTM to simulate the water temperature (Jobson, 1977, 1980). The equilibrium temperature is defined as the water temperature at which the net surface heat exchange becomes zero. Theoretically, a body of water given enough time reaches an equilibrium temperature and remains at this temperature as long as the meteorological conditions (solar radiation, wind speed and direction, air temperature, cloud cover, etc.) and influent water temperature remain constant.

Applying the principle of conservation of energy to a one-dimensional open channel, the conservation of temperature equation, in the Lagrangian form, is:

$$\begin{aligned} &dT / dt + UdT / dx - [D_x(d^2T / dx^2)] \\ &\quad = (H_tW) / (C_p\rho A), \end{aligned} \tag{12}$$

where

T is the cross-sectional average water temperature;
t is the time;
U is the stream velocity;
x is the longitudinal coordinate;
D_x is the longitudinal dispersion coefficient;
H_t is the flux of thermal energy from the air to the water;
W is the top width of the channel;
C_p is the specific heat of water at constant pressure;
ρ is the density of water; and
A is the cross-sectional area.

The term on the right side of equation 12 represents the rate of water-temperature change resulting from the exchange of energy between the atmosphere and the water.

Rather than measuring all of the meteorological parameters (solar radiation, wind speed and direction, ambient air temperature, cloud cover, etc.) needed to compute the water temperature, the equilibrium temperature for this application was estimated by using the program EQULTMP (Jobson, 1997). Data on daily maximum and minimum ambient air temperature, the respective times of occurrence of these temperatures, and average daily wind speed are needed for this program to compute the equilibrium temperatures for a specified time.

Simulations of phytoplankton growth require data on solar radiation. For this study, solar radiation was estimated by using the program SOLAR (Jobson, 1997), which requires data representing longitude, longitude of the local time meridian, latitude, altitude of sunrise and sunset, atmospheric pressure, coefficients in empirical equations to determine precipitable water content of the atmosphere, cloud cover, and dew-point temperature.

WATER-QUALITY MODEL CALIBRATION AND VALIDATION

This section describes the construction, calibration, and validation of the Catawba River water-quality model, which includes the BRANCH and BLTM components previously described. The model was calibrated using data collected during August 23–27, 1996, and validated by using data collected during July 11–15, 1997. The effects of small changes in the values of calibrated model parameters and boundary conditions on simulation results were evaluated through sensitivity analyses.

Computational Grid

The BRANCH and the BLTM model components require different types of computational grids. In the BRANCH model, the study reach is represented as a series of cross sections, channel segments (defined by an upstream and downstream cross section), junctions, and branches of the river (fig. 16). Data that characterize the conveyance, area, width, and storage capacity at each cross section as a function of water level are required. A time series of water level or streamflow data are provided at the

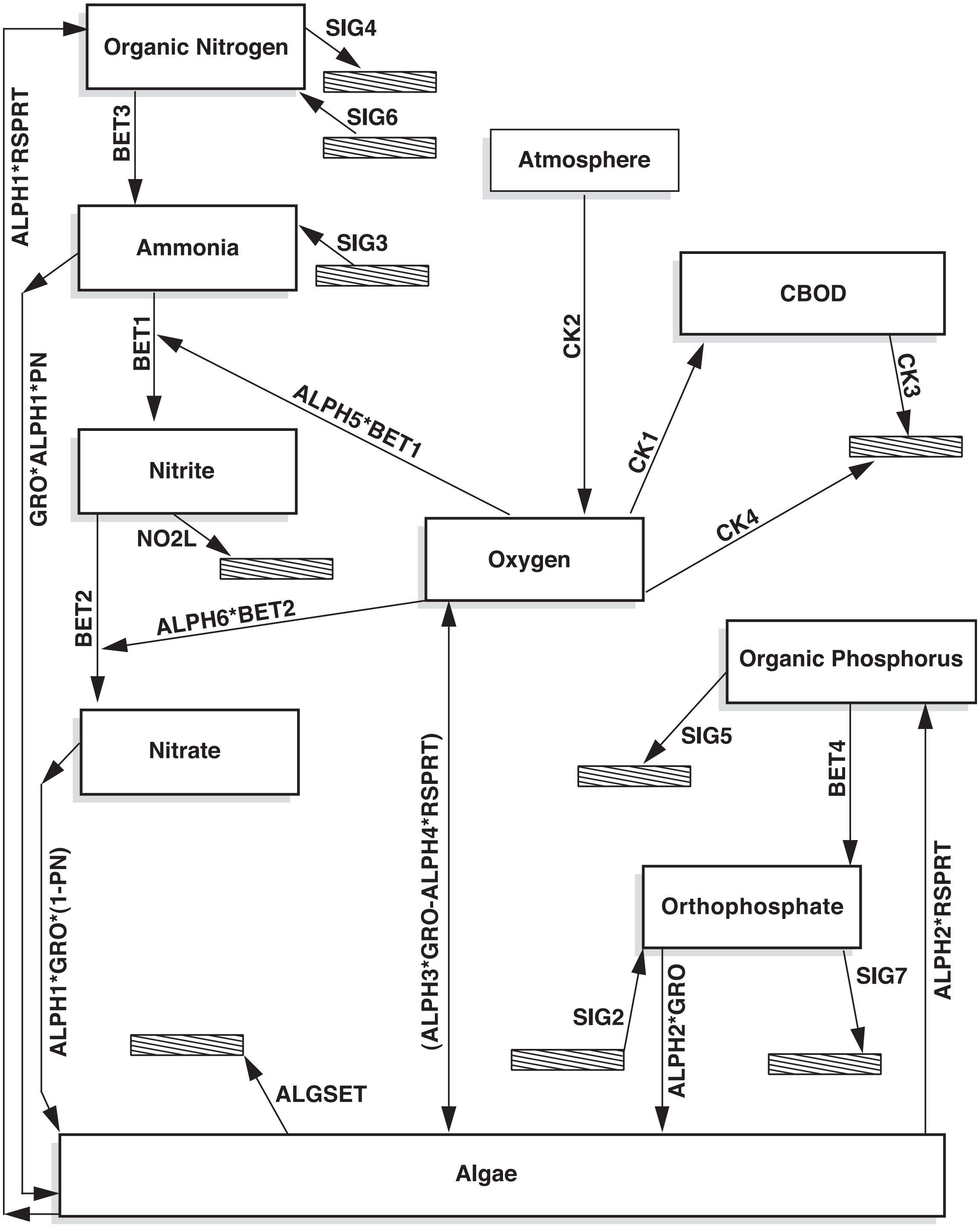


Figure 15. Major constituent interactions in the QUAL2E subroutine of the Branched Lagrangian Transport Model.

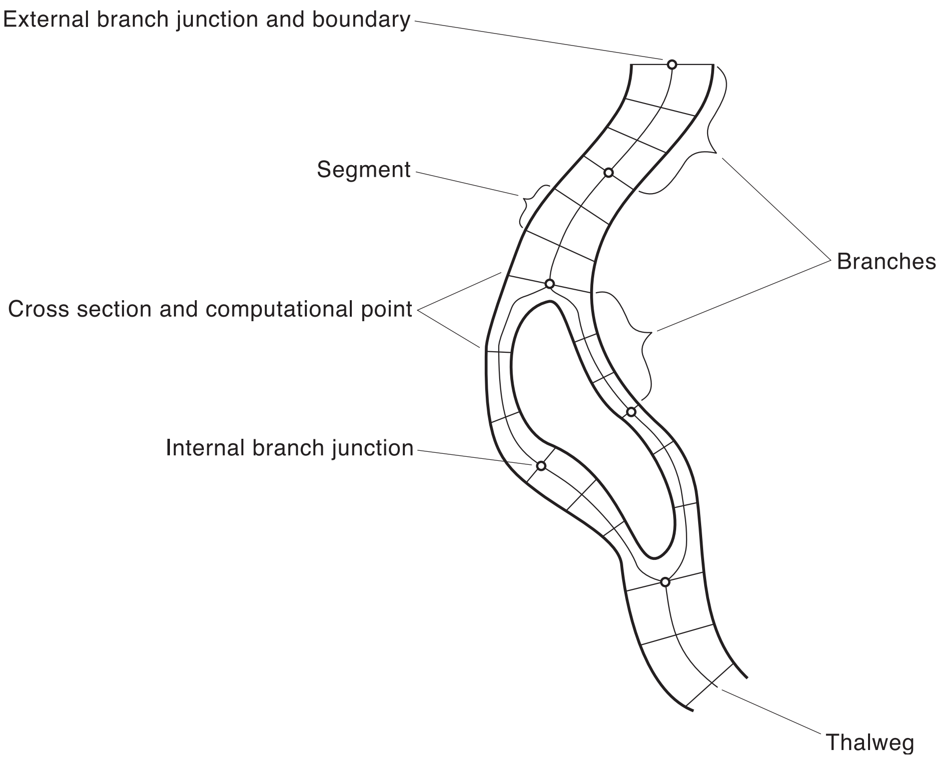


Figure 16. Idealized BRANCH model schematization.

upstream and downstream (external) boundaries of the grid. All other water levels and streamflows are simulated at cross sections within the computational grid.

The BRANCH model grid for the study reach consisted of 32 branches, 30 internal junctions, 33 cross sections, and 2 external boundaries. For the BLTM application, internal junctions were removed to minimize numerical dispersion, and the river was described by four branches, two internal junctions, and two external boundaries (fig. 17).

Water Level and Streamflow

Flow data from Lake Wylie Dam, adjusted as previously described, were used as the upstream boundary data. Water-level data measured at RM 110.3 in the headwaters of Fishing Creek Reservoir were used as the downstream boundary condition. Estimated tributary inflows from Sugar, Twelvemile, and Waxhaw Creeks, and from the six point-source dischargers were added to the main stem flow at the cross section nearest the mouth of the respective tributary or discharge.

Water level at RM 110.3 was measured by using a pressure transducer. Wave activity on Fishing Creek Reservoir resulted in water-level oscillations of about 0.1 ft around the mean value. When the original record that included these small water-level oscillations was used as the downstream boundary for the BRANCH model, large oscillations in simulated flows resulted. Consequently, the measured water levels were smoothed by using a spline function (for example, fig. 18), and the smoothed values subsequently were used as the boundary condition.

As previously discussed, the study reach includes the 2.4 mi Landsford Rapids section. A wide, shallow channel composed of rocks and large boulders characterizes this section of the river. The simulations of flow through these shoals, with an average water depth of about 1 ft, were extremely sensitive to cross-sectional shape, segment length, and time step. In order to avoid numerical instabilities, the distance between cross sections had to be increased to about 5,000 ft, and the computational time step was reduced to 2 minutes. This distance between cross sections made it impossible to accurately simulate the details of the water surface at the head and toe of Landsford Rapids

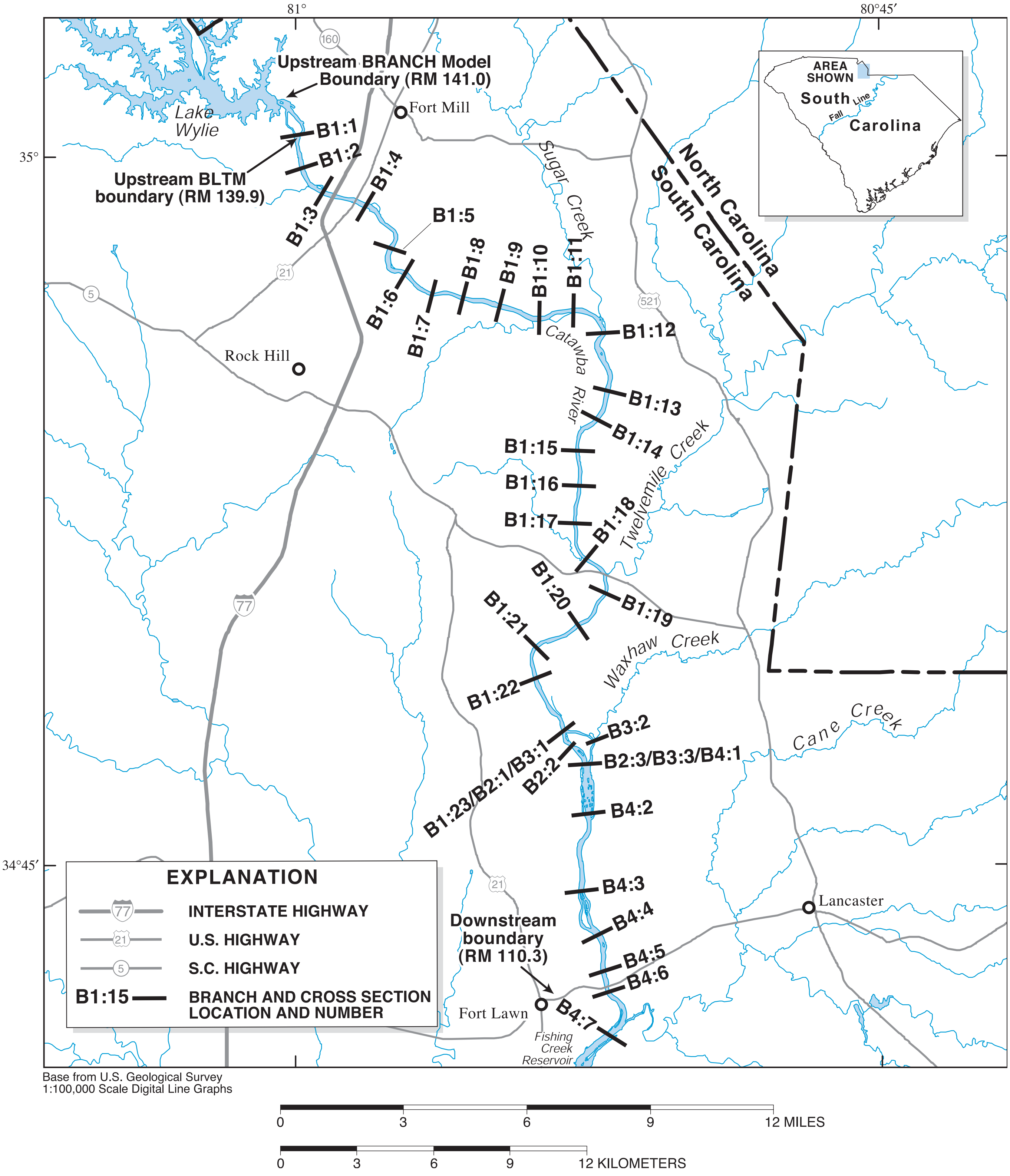


Figure 17. Schematization of the Branched Lagrangian Transport Model cross sections in the study reach of the Catawba River, S.C.

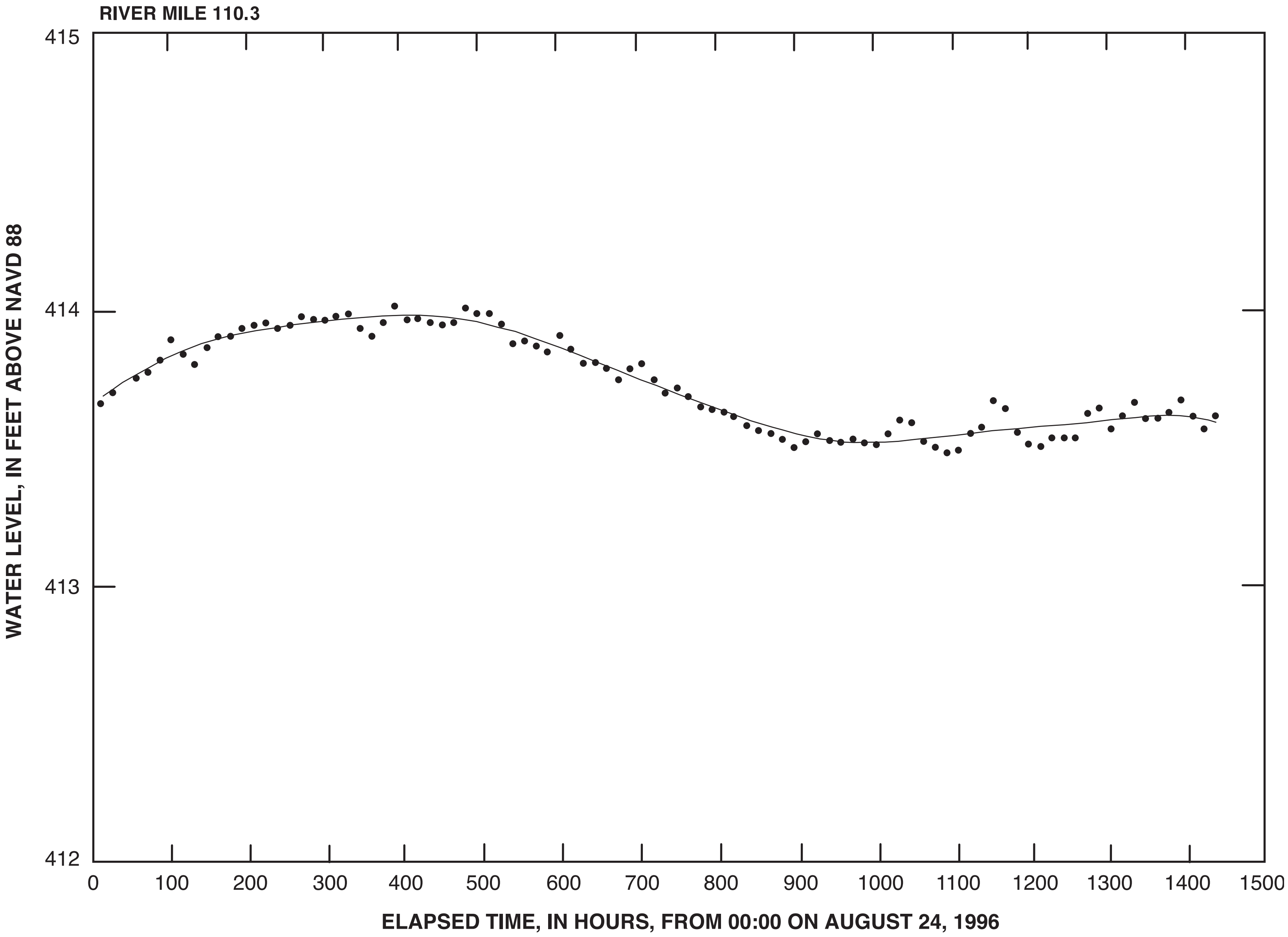


Figure 18. Example of spline smoothing of water levels measured at U.S. Geological Survey streamgaging station 02147259 (river mile 110.3) on the Catawba River at Fort Lawn, S.C.

and at the head of Fishing Creek Reservoir—locations where the water-surface slope had fairly steep gradients. Detailed convergence testing was not performed because simulations demonstrated that the model would fail at larger time steps and at smaller stream-segment lengths.

Model calibration was accomplished by adjusting the datum of the controls within the study reach and then by adjusting the resistance coefficients. The flow-resistance coefficients, which are a function of depth, ranged from 0.034 to 0.050 for the stream section from Lake Wylie Dam to the control at the head of the Landsford Rapids (RM 141.0–RM 118.0), from 0.033 to 0.048 for the first section of the rapids spanning the length of Culp Island (RM 118.0–RM 116.8), from 0.034 to 0.100 through the Landsford Rapids (RM 116.8–RM 115.5), and from 0.034 to 0.040 for the section below Landsford Rapids to S.C. Highway 9 (RM 115.5–RM 111.4) (fig. 2).

Statistics generated for comparison of measured and simulated water level and streamflow (table 19) included the timing error, mean error, and the standard deviation of the errors. Timing errors of streamflow and water level were determined by iteratively (1) calculating the correlation coefficient between measured and simulated values for the entire simulation period, (2) shifting the simulated hydrograph forward (or backward) in time, (3) recalculating the correlation coefficient, (4) and continuing until the best correlation was determined. The timing adjustment required to give the best correlation coefficient was the “timing error.” A positive timing error means that simulated flows occurred later than measured flows. In order to compute the errors in simulated water level and streamflow, the simulated hydrograph was shifted by the timing error, and the streamflow and water-level errors then were computed by subtracting the measured value from the simulated value. The mean of the errors is a measure of the bias of the simulation and is an indication of how much higher or lower the simulated values are in relation to the measured values. The standard deviation of the errors is a measure of the scatter of errors about the mean.

The water-level timing error for the model calibration was -10 minutes at RM 137.6, +30 minutes at RM 122.0, and +50 minutes at RM 115.5 (table 19). Similarly, simulated flows for the validation period were predicted to occur before corresponding measured flows in the upstream part of the reach and after corresponding measured flows in the downstream part of the reach (figs. 19, 20). Water level was underpredicted by an average of 0.32 and 0.27 ft at RM 137.6 and RM 122.0, respectively, and was over predicted by an average of 0.75 ft at RM 115.5 (table 19). The measured water-level range was between about 4 and 5 ft at the measurement sections (fig. 19). Flow was under simulated by an average of 1.5 percent at RM 137.6, and over simulated by an average of 0.8 percent at RM 122.0 (table 19).

Model performance for the validation period was similar to the calibration period (table 19; figs. 21, 22),

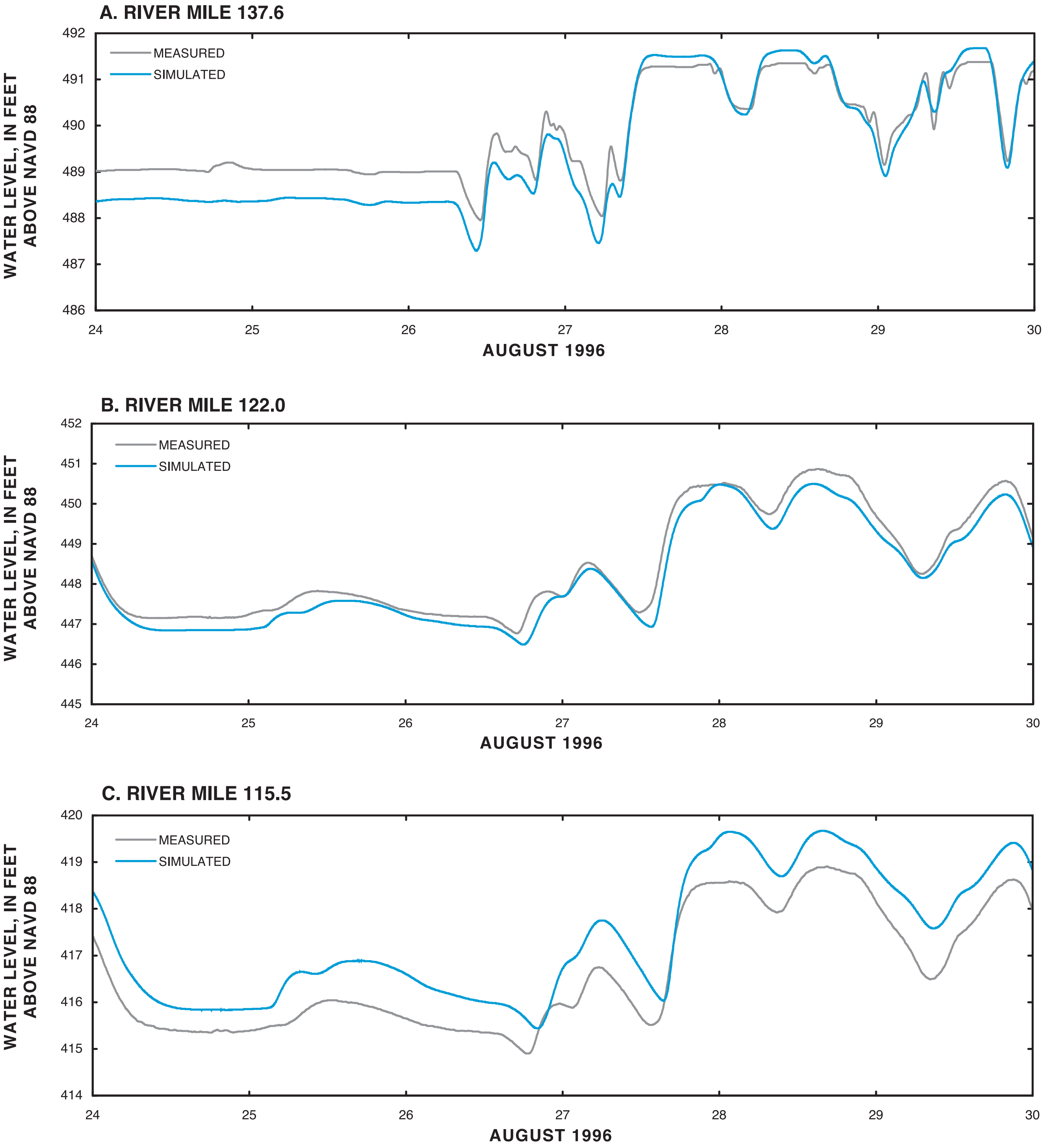
Table 19. Summary water-level and streamflow statistics for BRANCH model calibration and validation

[—, flow not simulated at this location]

River mile (fig. 2)	Timing error,[a] in minutes		Mean error[b]		Standard deviation of errors,[b] in percent	
	Water level	Streamflow	Water level, in feet	Streamflow, in percent	Water level	Streamflow
	Calibration (August 24–30, 1996)					
137.6	-10	-20	-0.32	-1.5	0.37	7.5
122.0	+30	+20	-0.27	+0.8	0.14	5.5
115.5	+50	—	+0.75	—	0.20	—
	Validation (July 11–17, 1997)					
137.6	-30	-30	-0.24	+5.3	0.14	12.8
122.0	+40	+20	-0.24	+4.1	0.14	12.7
115.5	+50	—	+0.51	—	0.31	—

[a] Positive error means that simulated values occurred after measured values.

[b] Hydrographs were shifted by amount equal to timing error before water-level and streamflow errors were calculated. Errors then were calculated by subtracting the measured value from the simulated value.



A. RIVER MILE 137.6

B. RIVER MILE 122.0

C. RIVER MILE 115.5

Figure 19. Measured and simulated water levels used in the calibration of the BRANCH model for U.S. Geological Survey streamgaging stations (A) 02146000, Catawba River near Rock Hill, S.C. (river mile 137.6); (B) 02147020 Catawba River below Catawba, S.C. (river mile 122.0); and (C) 02147187, Catawba River near Landsford, S.C. (river mile 115.5), during August 24–29, 1996.

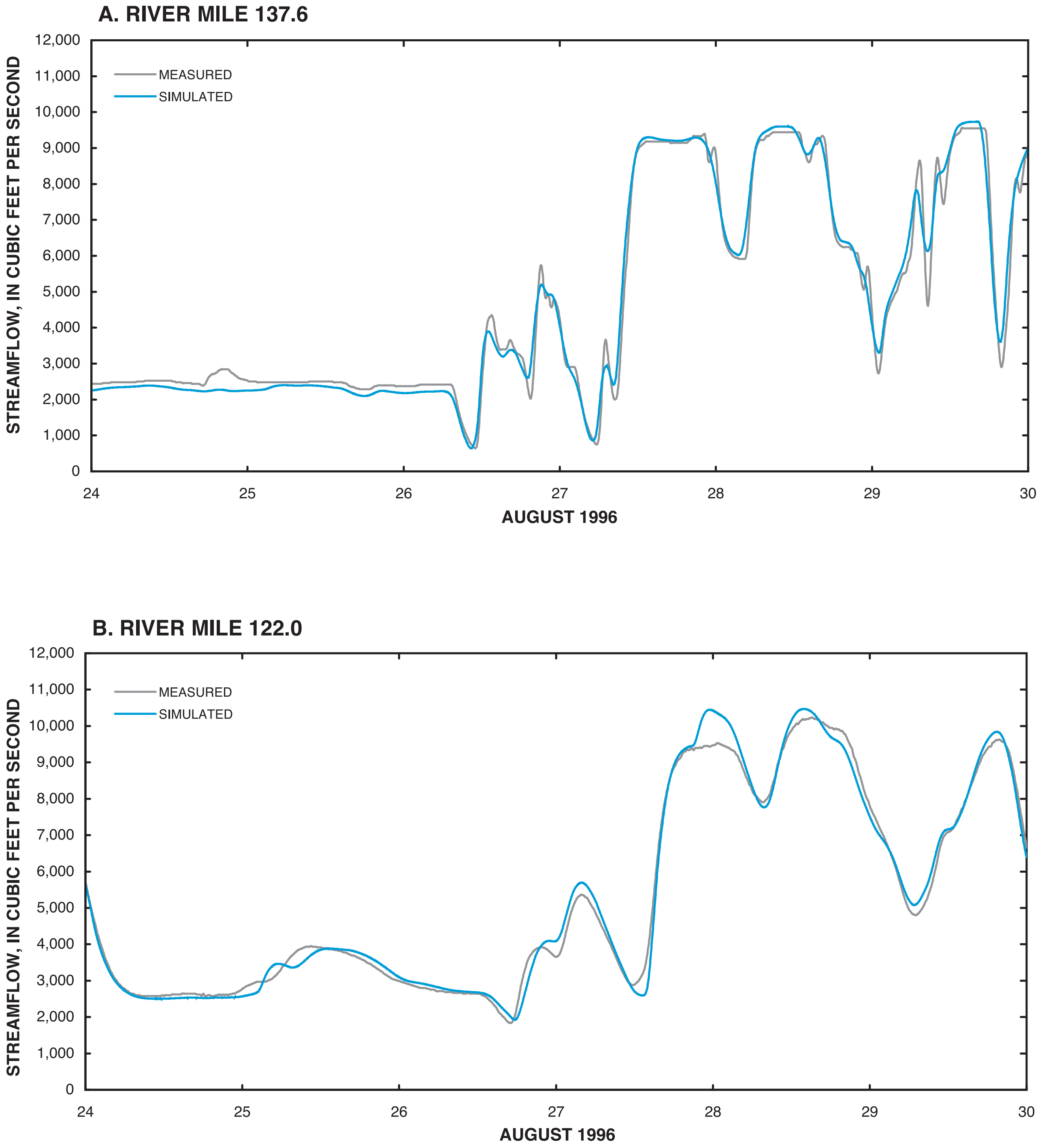


Figure 20. Measured and simulated streamflows used in the calibration of the BRANCH model for U.S. Geological Survey streamgaging stations (A) 02146000, Catawba River near Rock Hill, S.C. (river mile 137.6) and (B) 02147020, Catawba River below Catawba, S.C. (river mile 122.0), during August 24–29, 1996.

with a slightly poorer performance for the validation period. Nevertheless, the streamflow prediction errors are well within the range of the accuracy of measurements, which is typically considered to be about 5 percent. The streamflow model is applicable for the range of flows for which it was tested (190 to 9,740 ft^3/s), but is unstable for flows less than 190 ft^3/s.

Mass Transport

The mass transport component of BLTM was calibrated by using time-of-travel data collected during August 24–26, 1996, (figs. 23, 24) and validated using time-of-travel data collected during July 12–13, 1997 (fig. 25). Calibration was achieved by adjusting the dispersion factor (eq. 11) and adding a storage term to the BRANCH model. Despite the variation in flow conditions in the study reach, the calibrated dispersion factor was constant at 0.20 throughout the study reach. Initially, simulated dye transport rates were much greater than measured. To achieve calibration, a storage term was added at selected cross sections of the flow (BRANCH) model. The storage was added below the observed minimum water-surface elevation in the center of the channel at selected locations to increase the cross-sectional area, resulting in a decreased velocity and, subsequently, mass transport. The addition of this storage term, however, did not affect simulations of streamflow and water level.

Streamflow from the Lake Wylie Dam was steady at about 2,300 ft^3/s during the August 24–26, 1996, time-of-travel study. The measured dye concentrations at RM 134.6 were used as the upstream boundary condition (table 20). At RM 131.1 during August 24–26, 1996, the peak concentration was not sampled, and measured concentrations were limited to the receding limb of the hydrograph (fig. 23A). The measured and simulated dye concentrations at RM 111.4 resulting from the dye release in the upper reach matched fairly well (fig. 23B). For the dye release in the lower reach, the simulated dye concentrations in the pool-and-riffle sections of the river arrived later than the measured dye concentrations (RM 118.5, fig. 24A), but the timing difference was reduced after the dye traveled through the rapids (RM 114.3, fig. 24B). Dye concentrations released in the lower reach were simulated accurately in the backwater reach of the river (fig. 24C).

Table 20. Measured and simulated time of travel of a conservative dye tracer during August 24–26, 1996, and July 12–13, 1997, on the Catawba River, S.C.

[n/a, not applicable; —, not measured]

River mile	August 24–26, 1996 — Time from dye release to occurrence of centroid of dye cloud, in hours — Measured	Simulated	Difference,[a] in hours	July 12–13, 1997 — Time from dye release to occurrence of centroid of dye cloud, in hours — Measured	Simulated	Difference,[a] in hours
Upper reach						
139.9	Dye release location	n/a	n/a	Dye release location	n/a	n/a
134.6	Data used as boundary condition	—	—	—	—	—
131.1	—	—	—	3.4	2.9	0.5
111.4	27.6	27.5	0.1	29.1	23.7	5.4
Lower reach						
123.0	Dye release location	n/a	n/a	Dye release location	n/a	n/a
122.0	Data used as boundary condition	n/a	n/a	n/a	n/a	n/a
118.5	5.4	7.5	-2.1	3.7	4.8	-1.1
114.3	13.5	14.1	-0.6	9.6	9.8	-0.2
111.4	19.7	19.9	-0.2	14.4	14.6	-0.2

[a] Positive difference means that simulated values occurred after measured values.

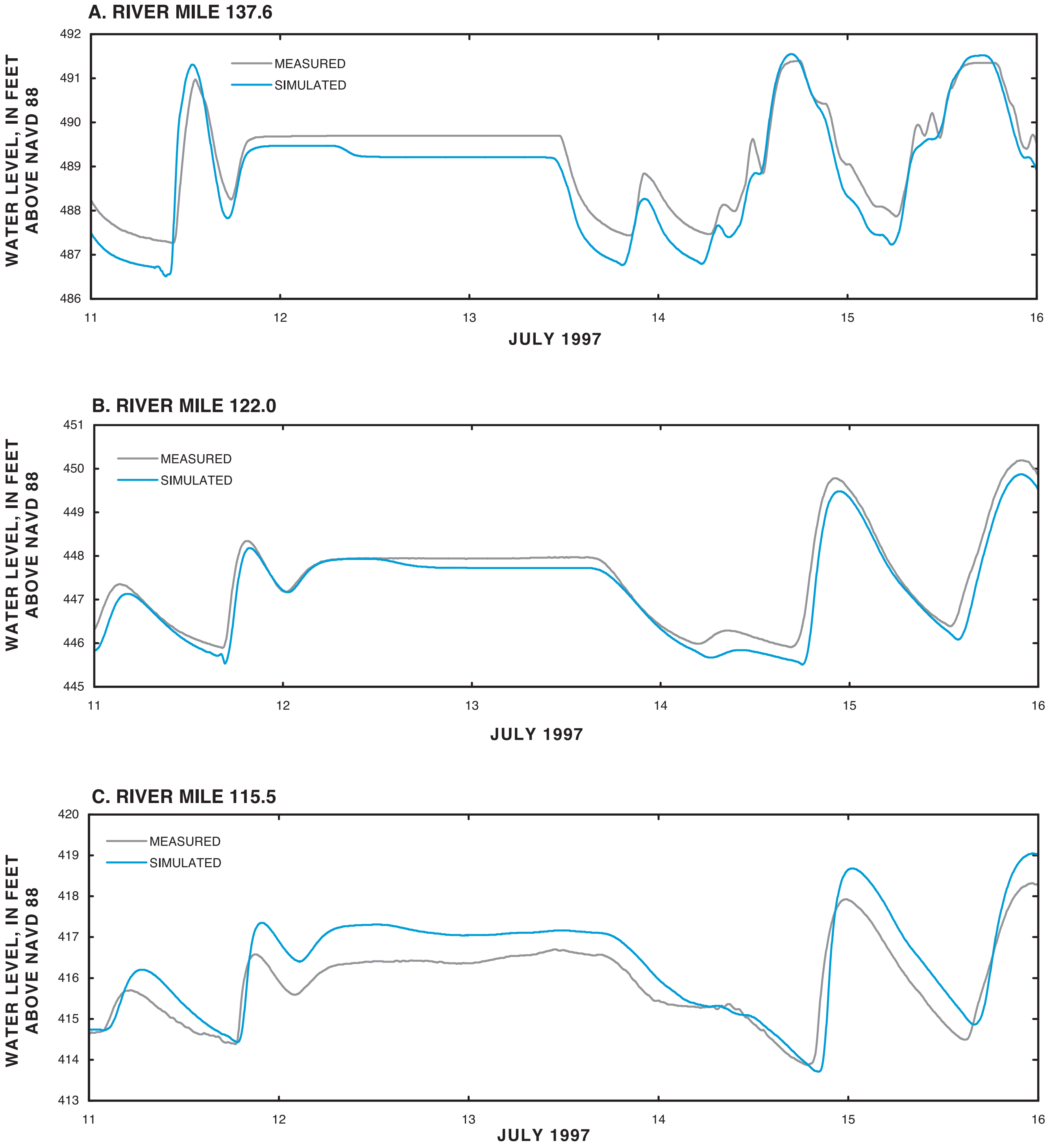
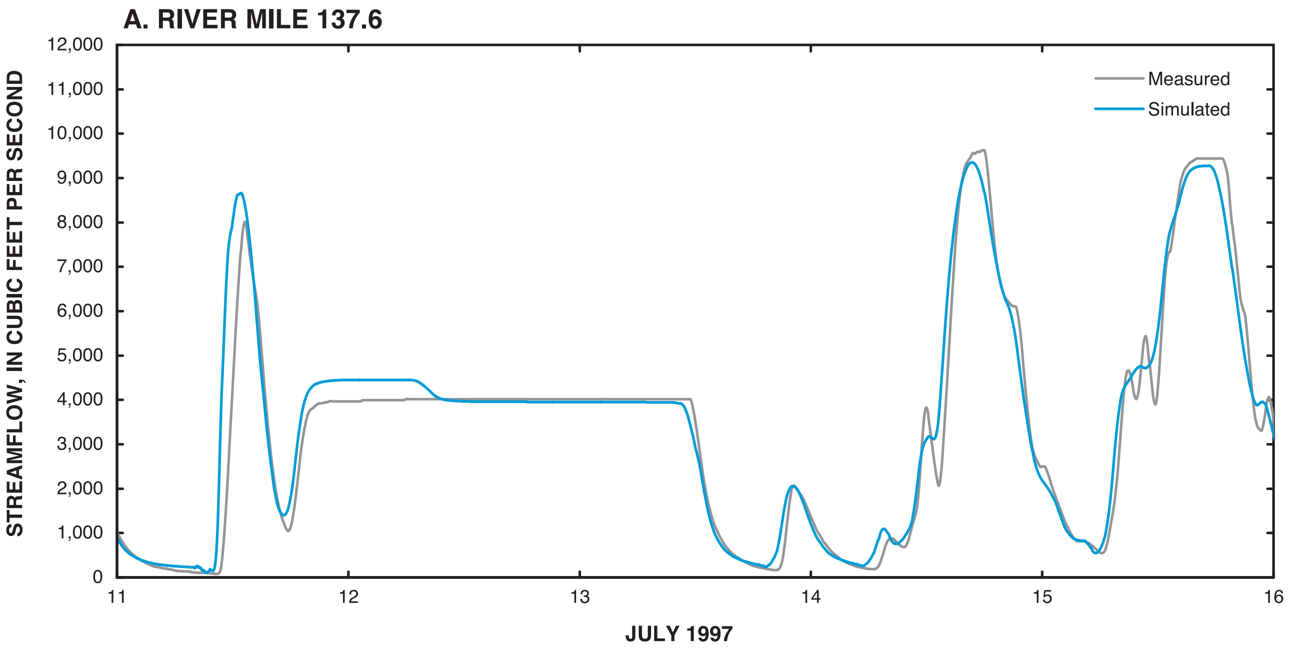


Figure 21. Measured and simulated water levels used in the validation of the BRANCH model for U.S. Geological Survey streamgaging stations (A) 02146000, Catawba River near Rock Hill, S.C. (river mile 137.6); (B) 02147020 Catawba River below Catawba, S.C. (river mile 122.0); and (C) 02147187, Catawba River near Landsford, S.C. (river mile 115.5), during July 11–15, 1997.



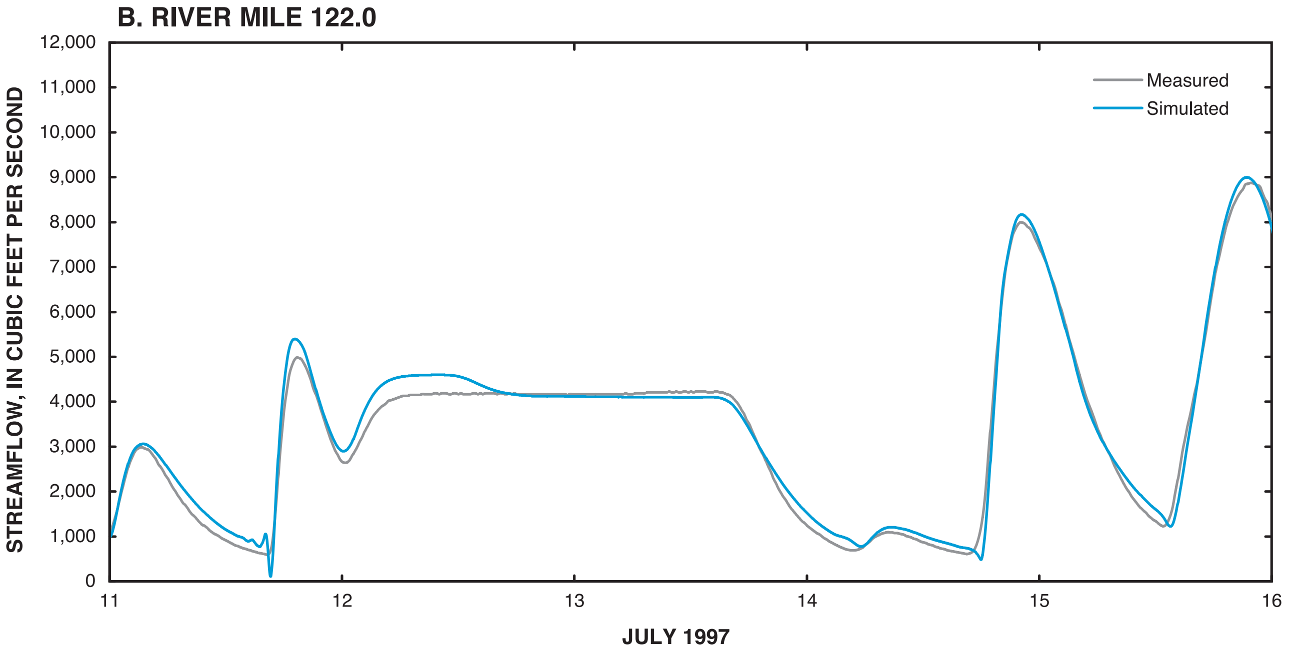


Figure 22. Measured and simulated streamflows used in the validation of the BRANCH model for U.S. Geological Survey streamgaging stations (A) 02146000, Catawba River near Rock Hill, S.C. (river mile 137.6) and (B) 02147020, Catawba River below Catawba, S.C. (river mile 122.0), during July 11–15, 1997.

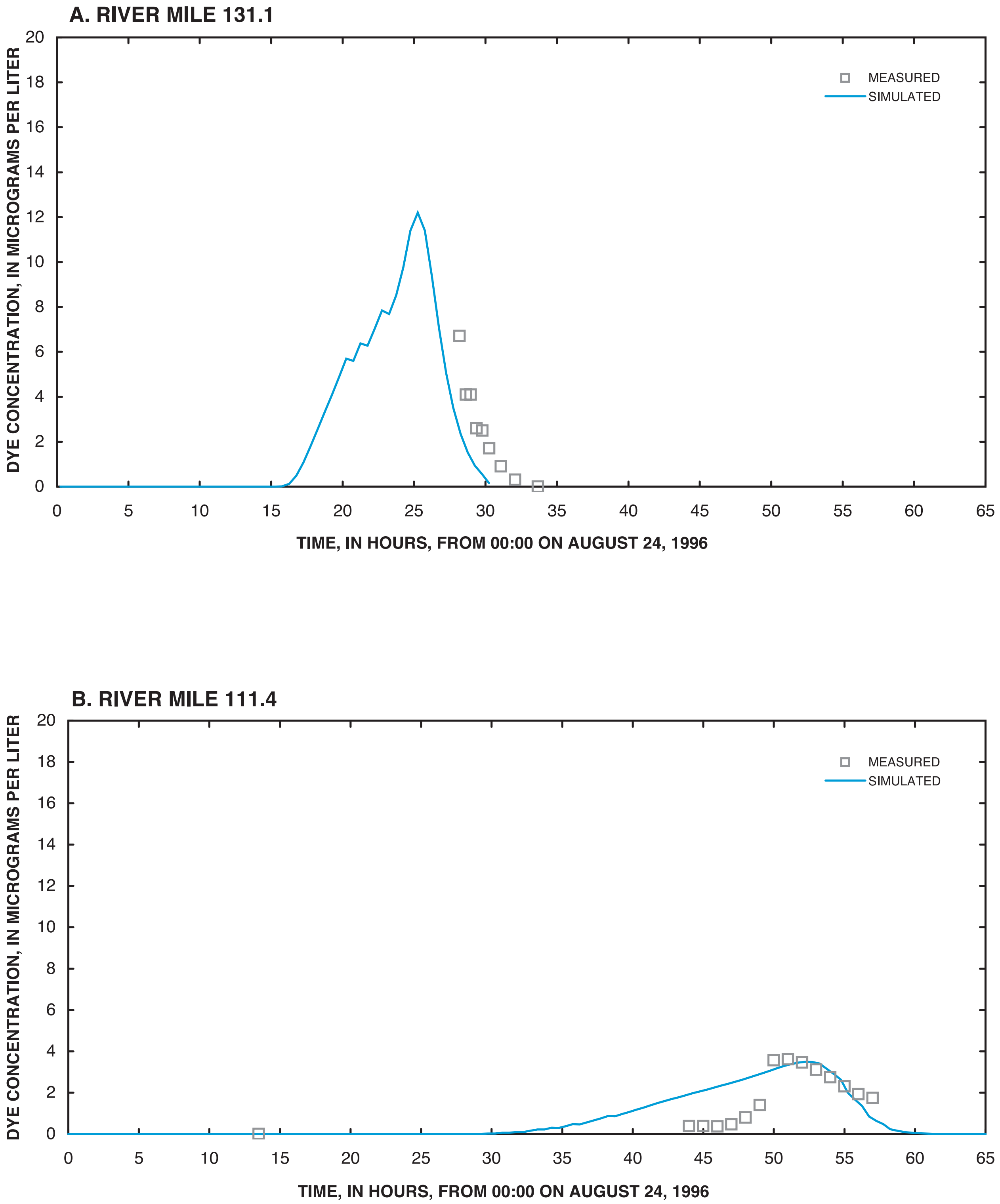


Figure 23. Measured and simulated dye concentrations used in the calibration of the Branched Lagrangian Transport Model for the Catawba River, S.C., at (A) river mile 131.1 and (B) river mile 111.4, resulting from the dye release in the upper reach, August 24–26, 1996.

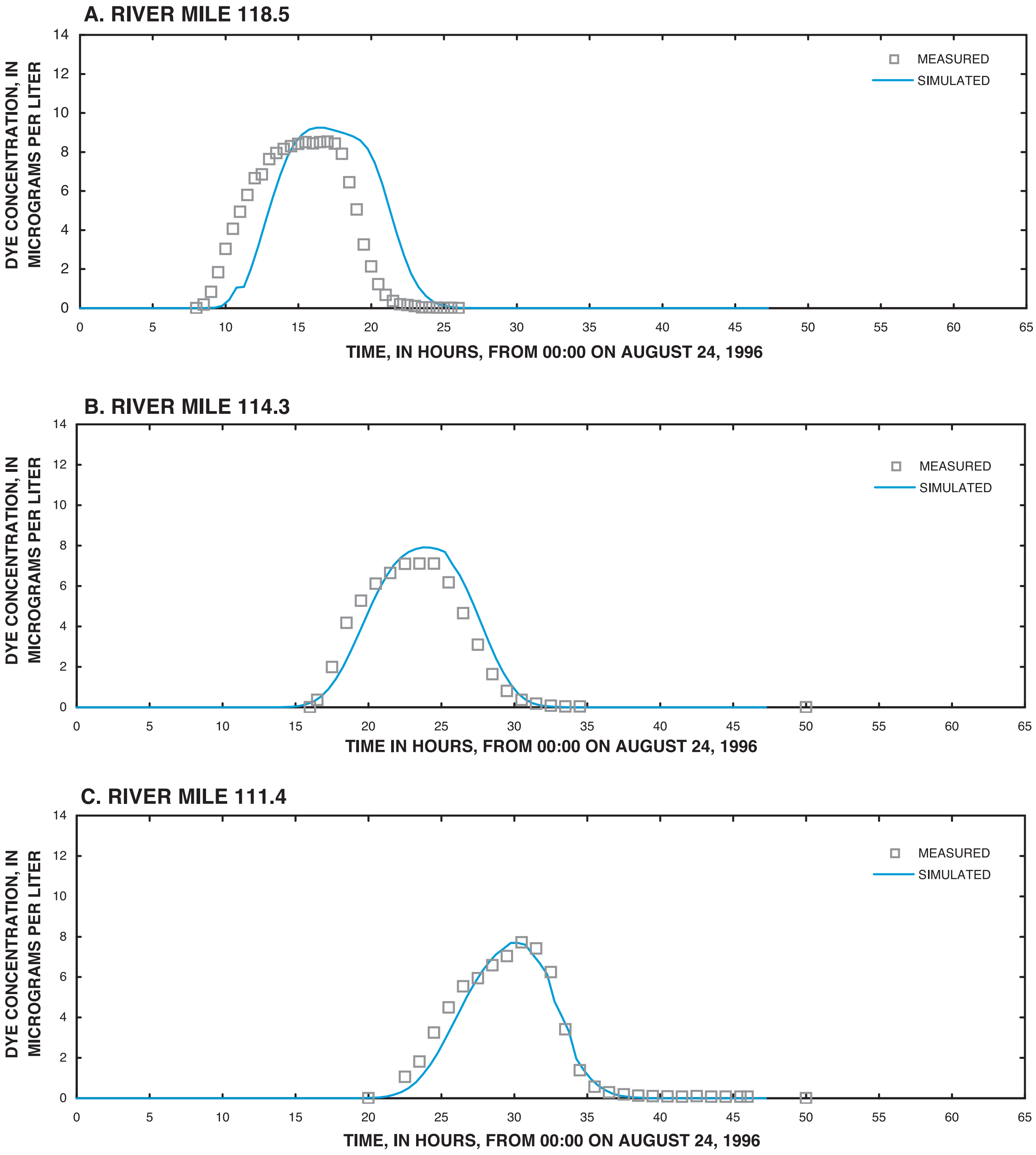


Figure 24. Measured and simulated dye concentrations used in the calibration of the Branched Lagrangian Transport Model for the Catawba River, S.C., at (A) river mile 118.5; (B) river mile 114.3; and (C) river mile 111.4, resulting from the dye release in the lower reach, August 24–26,1996.

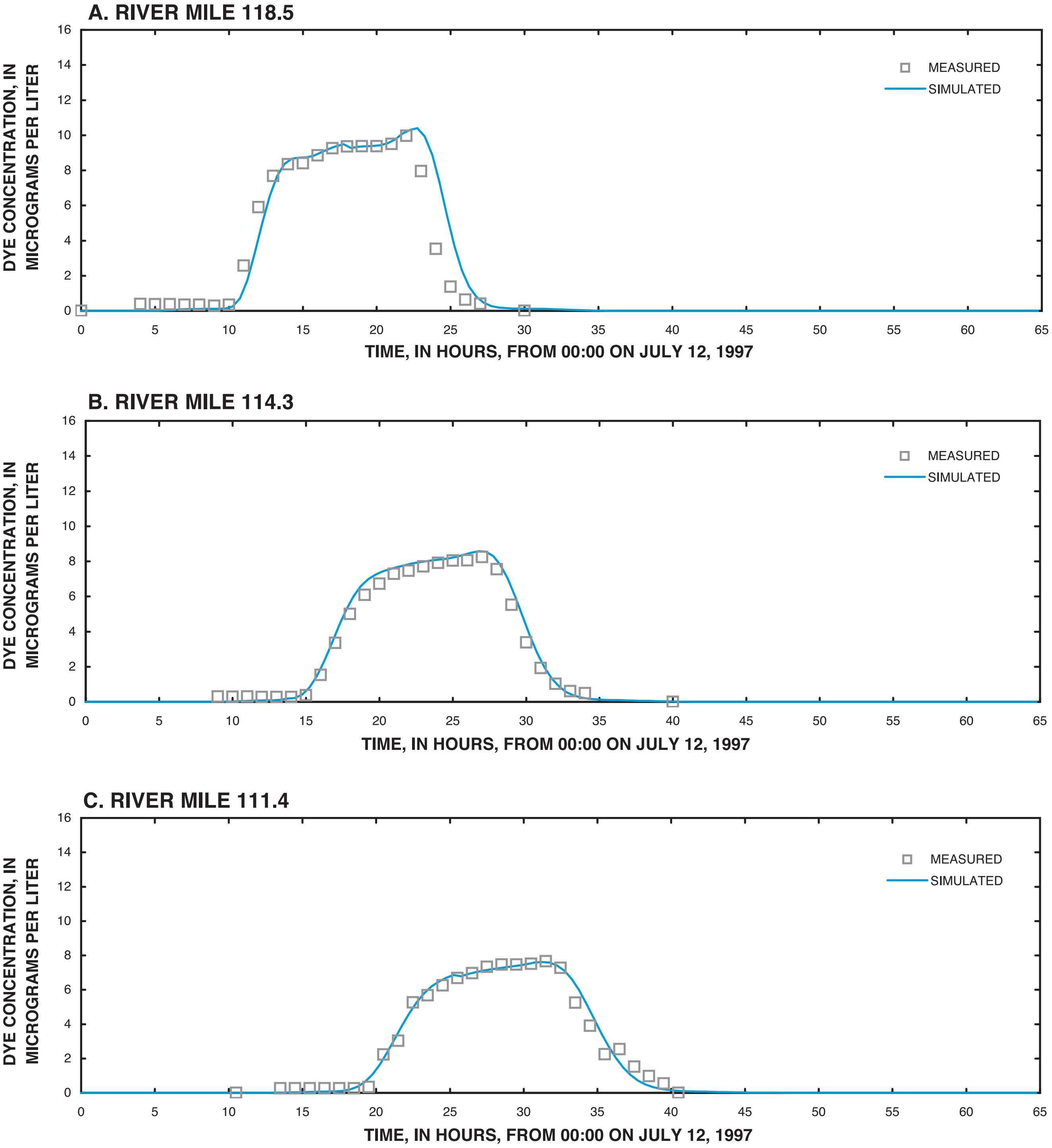


Figure 25. Measured and simulated dye concentrations used in the validation of the Branched Lagrangian Transport Model for the Catawba River, S.C., at (A) river mile 118.5; (B) river mile 114.3; and (C) river mile 111.4, resulting from the dye release in the lower reach, July 12–13,1997.

For the July 1997 validation period, flow from Lake Wylie Dam was steady at about 4,100 ft^3/s for 48 hours. The time of travel of the dye cloud was simulated accurately in the upper reach of the river (table 20), and the simulated and measured dye distributions in the lower reach were in close agreement (fig. 25). The same channel storage and dispersion factors were used for both the August 1996 and July 1997 simulations.

Water Temperature

Hourly measurements of water temperature at RM 139.9 were used as the upstream boundary conditions. Daily average water temperatures at the six point-source discharges were set close to the ambient instream temperature of the river. Water temperature simulations were calibrated by adjusting the free-convection and mass-transfer coefficients in the wind function of the BLTM. The calibration was refined and improved by adjusting the calculated equilibrium temperatures at the upstream boundary by 15 percent. Equilibrium temperatures were determined from the computer program EQULTMP (Jobson, 1997). Future applications of the Catawba River water-quality model to time periods other than those used for calibration and validation should include an increase of 15 percent in calculated equilibrium temperatures before the water-quality simulations are made.

Simulated and measured water temperatures exhibited the same general daily fluctuations during the calibration and validation periods (figs. 26, 27). For the calibration period, the maximum absolute errors ranged from 0.81 to 2.07 °C, and the minimum absolute errors ranged from 0.0 to 0.05 °C (table 21). For the validation period, the maximum absolute errors ranged from 0.75 to 2.37 °C, and the minimum absolute errors ranged from 0.00 to 0.08 °C (table 21).

Water Quality

The transport and chemical transformation of DO, algal biomass (represented as chlorophyll *a)*, organic nitrogen, ammonia nitrogen, nitrate nitrogen, nitrite nitrogen, organic phosphorus, dissolved orthophosphate, and ultimate carbonaceous biochemical oxygen demand ($CBOD_u$) were simulated. Nitrite nitrogen simulations are not discussed further in this report because of the relatively low concentrations of nitrite in natural waters and because nitrite is typically reported in combination with nitrate as nitrite-plus-nitrate nitrogen. Because of the anticipated use of the water-quality model, more emphasis is given to reasonable simulations of DO and phosphorus than to other constituents. Simulations of all constituents, however, are closely related (fig. 15); thus, results for all constituents (other than nitrite) are discussed.

Water-quality boundary conditions were determined by the following methods. Hourly measurements of DO concentrations at RM 139.9 were used as the upstream boundary. Concentrations of chlorophyll *a*, organic nitrogen, ammonia nitrogen, nitrite nitrogen, nitrate nitrogen, organic phosphorus, dissolved orthophosphate, and $CBOD_u$ (determined from measured BOD_5), measured at approximately 6-hour intervals, were interpolated to 30-minute intervals and used at the upstream and downstream boundaries and to represent tributary stream inputs. Daily average concentrations for each of the six point-source dischargers (table 16) were used to represent point-source loadings to the study reach. Nitrite and chlorophyll *a* concentrations were assumed to be zero for all dischargers. Because of the close proximity to each other, the two discharges at RM 137.3 and RM 137.0 (fig. 2; table 3) were combined to represent a single source. The concentration of the combined flow was determined by using the conservative mixing equation:

$$C_{mixed} = (C_A Q_A + C_B Q_B) / (Q_A + Q_B), \qquad (13)$$

where,

C_{mixed} is the concentration after mixing, in milligrams per liter (mg/L);

C_A is the concentration at discharge A, in mg/L;

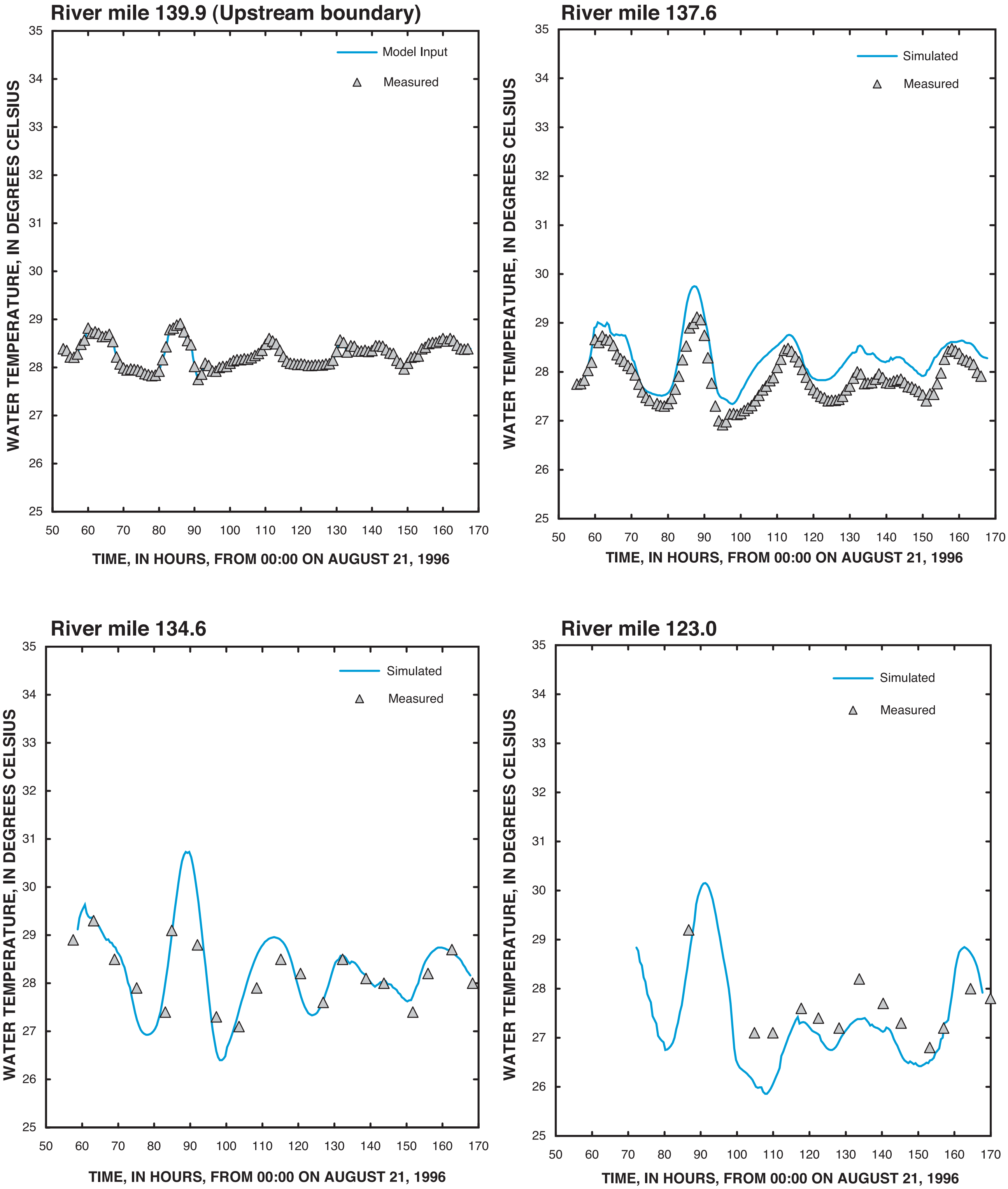
Q_A is the flow at discharge A, in cubic feet per second (ft^3/s);

C_B is the concentration at discharge B, in mg/L; and

Q_B is the flow at discharge B, in ft^3/s.

Point-source discharges were added to the study reach at the BLTM grid location nearest the actual discharge location.

The water-quality model was calibrated by adjusting the constant (global) and variable (local)


River mile 139.9 (Upstream boundary)

River mile 137.6

River mile 134.6

River mile 123.0

Figure 26. Measured and simulated water temperatures used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

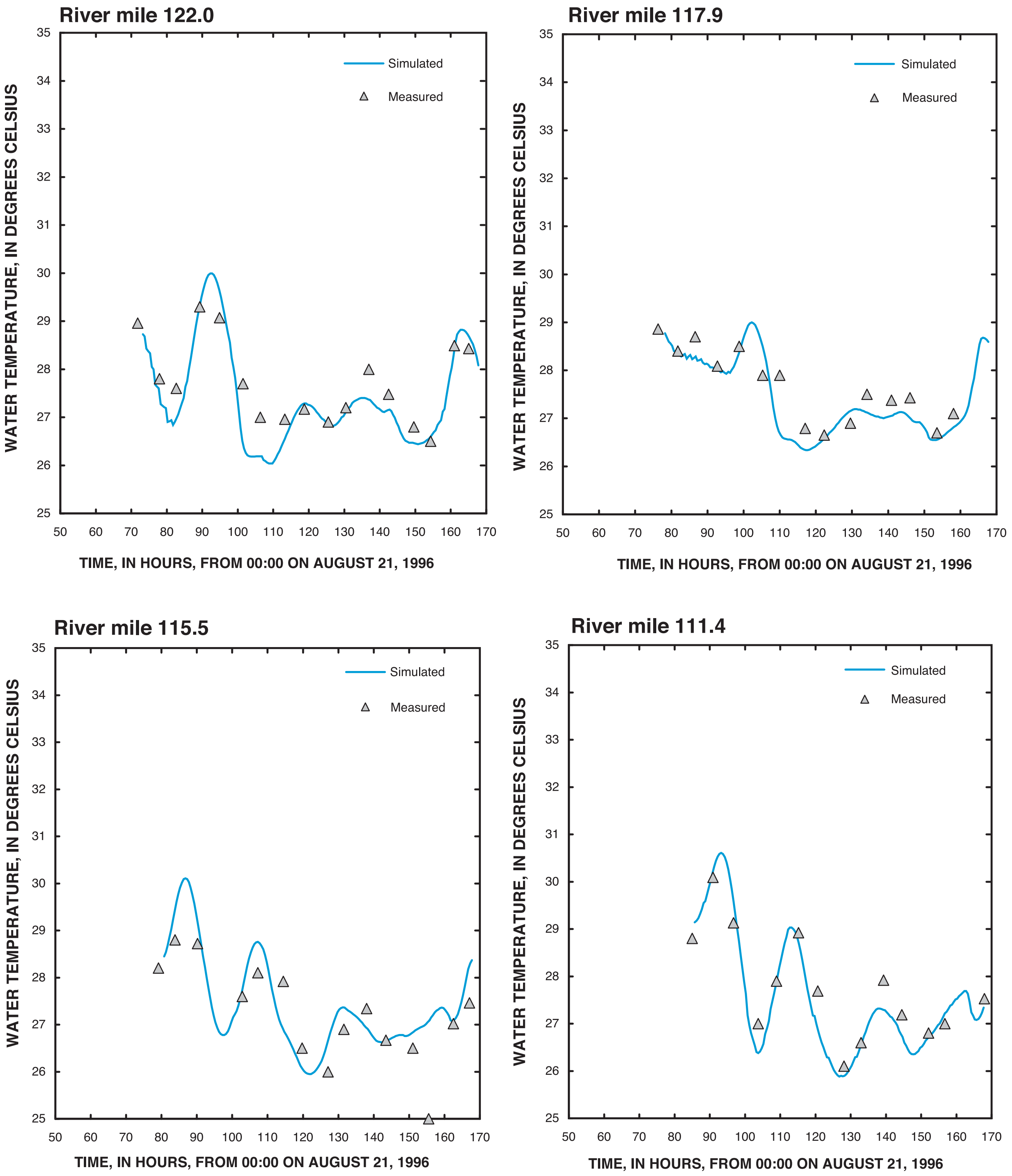


Figure 26 (Continued). Measured and simulated water temperatures used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

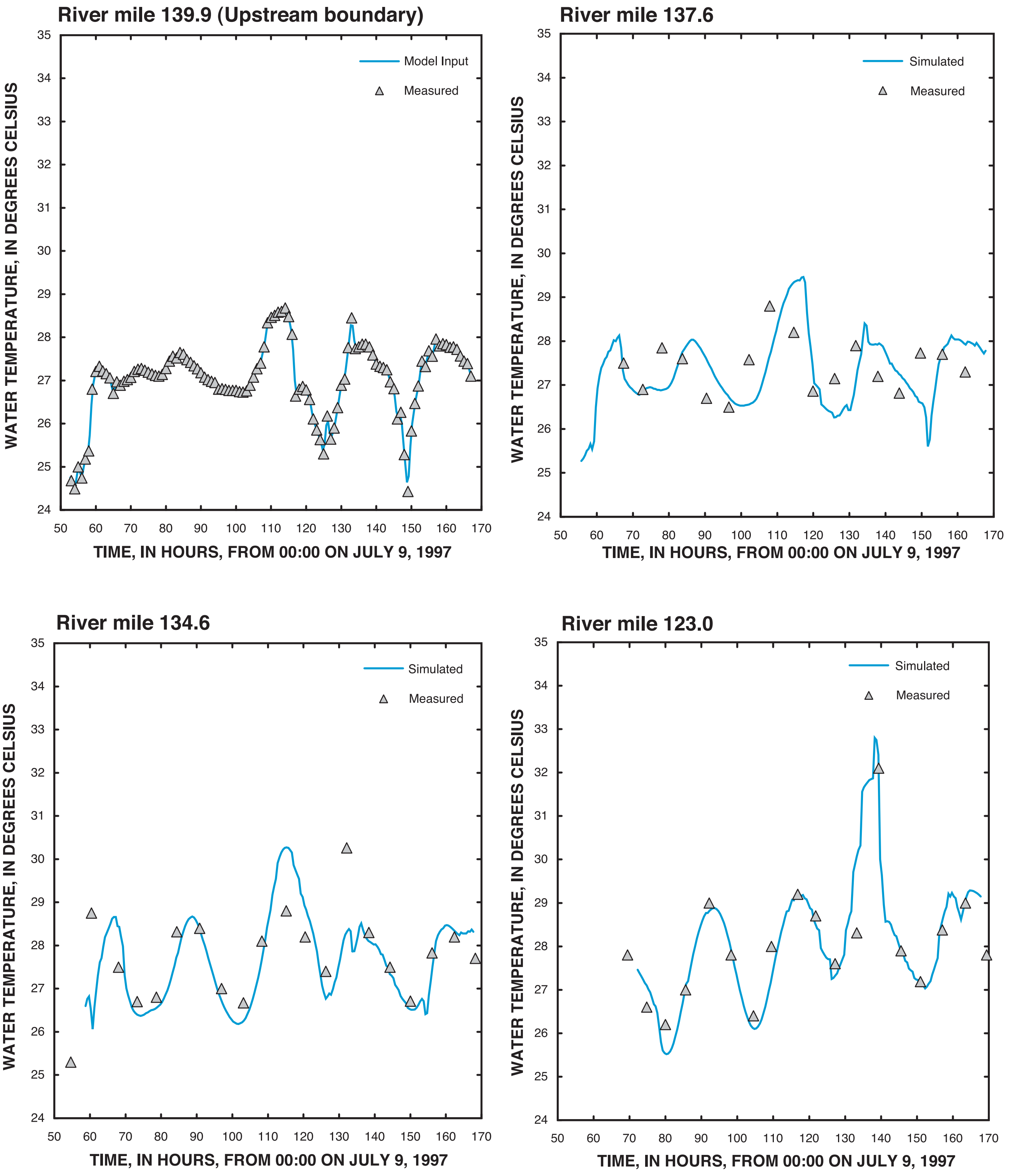


Figure 27. Measured and simulated water temperatures used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

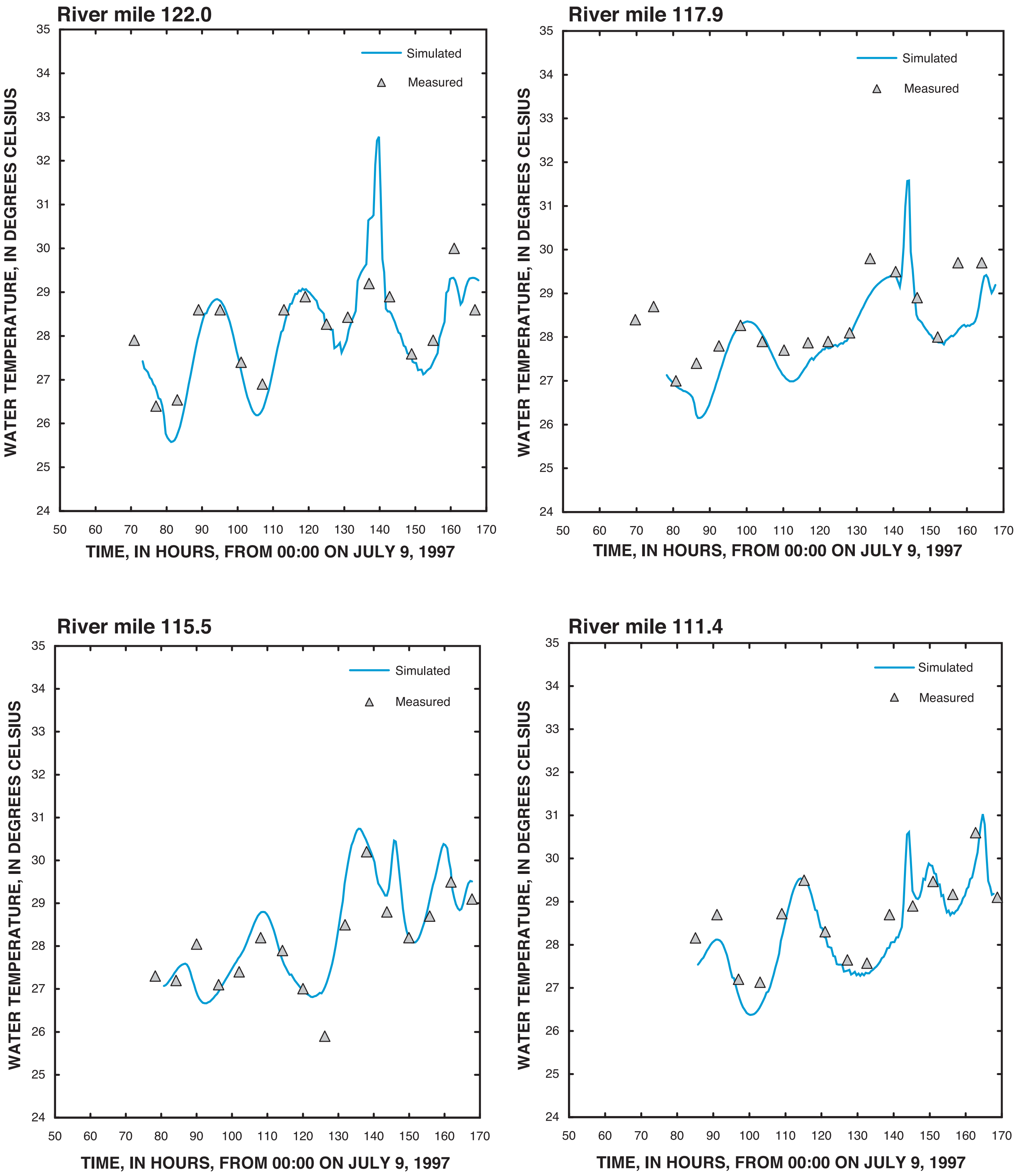


Figure 27 (Continued). Measured and simulated water temperatures used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

Table 21. Summary statistics for measured and simulated water temperatures on the Catawba River, S.C., during August 23–27, 1996, and July 11–15, 1997

[°C, degrees Celsius]

River mile	Number of observations	Mean measured temperature (°C)	Mean simulated temperature (°C)	Root mean square error (°C)	Mean error (°C)	Maximum absolute error (°C)	Minimum absolute error (°C)
August 23–27, 1996							
137.6	110	27.9	28.3	0.43	0.40	0.78	0.00
134.6	19	28.1	28.3	.44	.13	1.04	.03
123.0	13	27.6	27.2	.61	-.39	1.04	.05
122.0	16	27.6	27.4	.50	-.25	1.27	.02
117.9	14	27.6	27.3	.45	-.22	1.21	.02
115.5	14	27.2	27.5	.76	.32	2.07	.02
111.4	12	27.7	27.5	.41	-.19	.81	.03
July 11–15, 1997							
137.6	18	27.4	27.3	0.74	-0.05	1.14	0.03
134.6	19	27.8	27.6	.89	-.23	2.37	.03
123.0	17	28.1	28.2	.62	.09	1.76	.01
122.0	16	28.2	28.1	.58	-.11	1.46	.08
117.9	16	28.4	28.0	.65	-.37	1.55	.01
115.5	15	28.1	28.3	.57	.22	1.14	.00
111.4	13	28.6	28.3	.45	-.30	.75	.01

kinetic rate coefficients (table 22) until the simulated constituent concentrations approximated the measured concentrations. Simulated concentrations were considered acceptable when the average simulated constituent concentrations for the period of measured data were within the range of observed concentrations for a given location. In the final calibration, all rate coefficients were within the suggested ranges described by Bowie and others (1985), Brown and Barnwell (1987), and Jobson and Schoellhamer (1993) except the local settling rate for algae, which was set at 0.20 foot per day (ft/d; recommended values are 0.50 to 6.00), the free-convection wind coefficient, which was set at 1.00 millimeter per day kilopascal (recommended value is 3.01), and the hydrolysis rate of organic nitrogen to ammonia, which was set at 0.10 (recommended values are 0.02 to 0.04).

The general approach to model calibration was to first calibrate the algal dynamics and then the nutrient cycling. The last constituent to be calibrated was DO. The BLTM has eight options for including atmospheric reaeration in the model. Four algorithms were tested in the Catawba River model: O'Connor and Dobbins (1958), Churchill and others (1962), Owens and others (1964), and Langbein and Durum (1967) (table 12). The O'Connor and Dobbins formula was developed by using data from moderately deep to deep channels with depths of 1 to 30 ft and velocities of 0.5 to 1.6 feet per second (ft/s) (Bowie and others, 1985). The Churchill formula was based on observed reaeration rates below dams where oxygen-deficient water was released and depths ranged from 2 to 11 ft and velocities ranged from 1.8 to 5 ft/s (Bowie and others, 1985). The Owens equation was based on oxygen-recovery data monitored in six streams in England following deoxygenation with sodium sulfite and having depths of 0.4 to 11 ft and velocities of 0.1 to 5 ft/s (Bowie and others, 1985). The Langbein and Durum formula was based on synthesis of data from Streeter (1926), O'Connor and Dobbins (1958), Churchill and others (1962), and Krenkel and Orlob (1962) (Bowie and others, 1985). The Krenkel and Orlob formula was based on data from a 1-ft-wide flume with depths between 0.08 and 0.2 ft (Bowie and others, 1985). Based on a comparison of (1) measured and calculated reaeration rates (table 12), (2) measured and simulated DO concentrations (subsequently discussed), and (3) conditions under which data were collected for development of these reaeration equations, the equation of Langbein and Durum (1967)

Table 22. Rate coefficients, recommended values, and values used in the Catawba River water-quality model

[N, nitrogen; mg/L, milligram per liter; P, phosphorus; µg, microgram; mg, milligram; —, dimensionless; BOD, biochemical oxygen demand]

Coefficient	Coefficient description	Recommended values (Jobson and Schoellhamer, 1993)	Values used	Units
CKL	Light half-saturation constant for algae	0.02–0.10	0.02	Langley per minute
CKN	N half-saturation constant	0.01–0.30	0.26	mg/L
CKP	P half-saturation constant for algae	0.001–0.05	0.04	mg/L
GRO	Maximum specific algal growth rate	1.00–3.00	2.50	per day
IGRO	Growth rate option (limiting nutrient)	Option 1, 2	2	—
K20	Reaeration option (Langbien and Durum, 1967)	Option 1–8	6	—
LFO	Light function option (half saturation)	Option 1, 2	1	—
PN	Algal preference factor for ammonia	0–1.00	0.25	—
RSPRT	Algal respiration rate	0.05–0.5	0.15	per day
SHAD0	Light extinction	Variable	0.7	per meter
SHAD1	Linear self shading	0.002–0.02	0.002	per meter per µg chlorophyll *a* per liter
SIG2	Benthos source rate for dissolved P	Variable	2.00	mg P per square foot per day
SIG3	Benthos source rate for ammonia	Variable	2.00	mg N per square foot per day
SIG4	Organic N settling rate	0.001–0.10	0.1	per day
SIG5	Organic P settling rate	0.001–0.10	0.1	per day
SIG 6	Benthos source rate for organic N	—	1.00	per day
SIG 7	Dissolved P settling rate	—	0.10	per day
A1	Free-convection wind	3.01	1.00	millimeters per day kilopascal
ALGSET	Local settling rate for algae	0.50–6.00	0.20	feet per day
ALPH0	Ratio of chlorophyll *a* to algal biomass	10.0–100.0	67.0	mg chlorophyll *a* per mg algae
ALPH1	Fraction of algal biomass that is N	0.07–0.09	0.07	mg N per mg algae
ALPH2	Fraction of algal biomass that is P	0.01–0.02	0.01	mg P per mg algae
ALPH3	Oxygen uptake per unit of algae growth	1.40–1.80	1.80	mg oxygen per mg algae
ALPH4	Oxygen uptake per unit of algae respired	1.60–2.30	1.60	mg oxygen per mg algae
ALPH5	Oxygen uptake per unit of ammonia oxidized	3.00–4.00	3.45	mg oxygen per mg N
ALPH6	Oxygen uptake per unit of nitrite oxidized	1.00–1.14	1.14	mg oxygen per mg N
B1	Mass-transfer coefficient	1.13	1.13	millimeters per day per kilopascal per meter per second
BET1	Biological oxidation rate of ammonia to nitrite	0.1–1.0	0.4	per day
BET2	Biological oxidation rate of nitrite to nitrate	0.20–2.00	2.00	per day
BET3	Hydrolysis rate of organic N to ammonia	0.02–0.04	0.10	per day
BET4	Decay rate of organic P to dissolved P	0.01–0.70	0.02	per day
CK1	Carbonaceous BOD decay rate	0.02–3.40	0.18	per day
CK2	Reaeration rate (for K20 Option 1)	0–100.0	—	per day
CK3	Carbonaceous BOD sink rate	-0.36–0.36	0.00	per day
CK4	Benthos oxygen consumption	Variable	0–75.0	mg oxygen per square foot per day

was selected for the Catawba River model. The reaeration rate coefficients calculated from the field studies (table 12) were not used in the model because these rates were strictly for the steady-flow conditions under which the field studies were conducted.

Results of the calibration (August 23–27, 1996) and validation (July 11–15, 1997) of water-quality simulations are presented as time-series plots showing measured and simulated organic nitrogen, ammonia nitrogen, nitrate nitrogen, algal biomass, organic phosphorus, orthophosphate, $CBOD_u$, and DO. In addition, the calibration and validation statistics are presented in tables 23 and 24, respectively.

Initial conditions at the model boundaries were assumed to be the mean concentrations of the measured data for the sampling period. Model simulations were started 2 days prior to the actual sampling period to allow for flushing of these estimated initial conditions through the study reach. Travel time from the upstream boundary to each sampling station was estimated by using the dye-concentration data (table 11). The first simulated data used at each site to compute the statistics comparing the simulated and measured values were determined based on the travel time of the first sample at the upstream boundary. This procedure was used to ensure that the initial conditions had passed each sampling station before statistics were computed. As a result, the downstream stations generally have fewer simulated and observed values than do the upstream stations.

<u>Organic nitrogen</u>: The mean errors for organic nitrogen at the sampling sites for the 1996 study period ranged from -0.11 to 0.03 mg/L (table 23). Four of the seven sites had a negative mean error, indicating that on average, the constituent was underpredicted. For the 1997 sampling period, the mean errors ranged from -0.07 to -0.02 mg/L, again indicating that on average, the constituent was underpredicted but to a lesser degree than the 1996 period (tables 23, 24; figs. 28, 29). Other observations pertaining to the organic nitrogen simulations include the following.

- The root mean square error of the simulated values was about equal to the mean concentrations at each station (tables 23, 24).
- In 1996, mean measured organic nitrogen concentrations increased between RM 137.6 and RM 122.0. Three point-source dischargers are located between RM 137.6 and RM 134.6. Although organic nitrogen concentrations from these dischargers was slightly higher than the concentrations in the river, flows from these dischargers were small compared to river flow. Therefore, the overall increase in organic nitrogen between RM 137.6 and RM 134.6 is possibly the result of a source that is not included in the model, or analytical error, or failure to sample high organic nitrogen water at the upstream boundary. Measured organic nitrogen also increased between RM 123.0 and RM 122.0. This is probably due to the influence of Twelvemile Creek, which had a median organic nitrogen concentration of 0.21 mg/L and a maximum concentration of 0.87 mg/L during the 1996 sampling period.
- In 1997, the mean organic nitrogen concentrations were relatively the same throughout the river, ranging from 0.16 to 0.22 mg/L. Again, there was a slight increase between RM 137.6 and RM 122.0. As in 1996, the organic nitrogen concentrations from the three dischargers between RM 137.6 and RM 134.6 were slightly higher than the concentrations in the river, but the flows were relatively small. As in 1996, the 1997 simulated results also seem to indicate that some sources of organic nitrogen to the river may be underestimated.

<u>Ammonia nitrogen</u>: For the 1996 calibration period, the mean errors ranged from -0.05 to 0.01 mg/L. The majority of the sites had negative mean errors, indicating that on average, the measured data were underpredicted. For the 1997 validation period, the mean errors ranged from -0.01 to 0.04 mg/L; all but one site had positive mean errors, indicating that on average, the measured data were being overpredicted (tables 23, 24; figs. 30, 31). The wide variability in some of the ammonia measurements caused some of the values to seem questionable. For example, the high value of about 0.53 mg/L at RM 117.9 during the 1997 sampling period (fig. 31) is much higher than the other values observed during this period, and there is no indication of elevated values downstream from this site. Likewise, the elevated values at RM 115.5 during the 1996 study (fig. 30), which are higher than concentrations upstream at RM 117.9, could be explained by high ammonia inflows from Waxhaw Creek, which seems unlikely (fig. 11), or by analytical error.

<u>Nitrate nitrogen</u>: For the 1996 calibration period, the mean errors for the nitrate nitrogen simulations ranged from -0.12 to 0.24 mg/L. Six of the seven sites had negative mean errors, indicating that on average,

Table 23. Summary statistics for the measured and simulated water-quality concentrations for seven sites on the Catawba River, S.C., August 23–27, 1996

[mg/L, milligram per liter]

River mile	Number of observations	Mean measured concentration	Mean simulated concentration	Root mean square error	Mean error	Maximum absolute error	Minimum absolute error
Organic nitrogen (mg/L)							
137.6	20	0.10	0.11	0.08	0.00	0.21	0.01
134.6	19	.14	.11	.18	-.03	.46	.01
123.0	12	.23	.15	.23	-.07	.39	.05
122.0	16	.26	.14	.27	-.11	.73	.02
117.9	14	.11	.15	.15	.03	.30	.00
115.5	13	.16	.16	.11	.00	.18	.03
111.4	12	.15	.13	.15	-.02	.38	.02
Ammonia nitrogen (mg/L)							
137.6	20	0.20	0.21	0.14	0.01	0.32	0.00
134.6	19	.24	.23	.16	-.01	.51	.00
123.0	12	.20	.18	.16	-.02	.41	.05
122.0	16	.20	.19	.13	-.01	.24	.01
117.9	14	.22	.18	.09	-.04	.26	.01
115.5	13	.24	.19	.18	-.05	.41	.00
111.4	12	.20	.15	.14	-.04	.44	.01
Nitrate nitrogen (mg/L)							
137.6	20	0.19	0.15	0.08	-0.04	0.17	0.00
134.6	19	.22	.16	.10	-.06	.21	.01
123.0	12	.37	.61	.36	.24	.77	.00
122.0	16	.59	.54	.32	-.05	.99	.01
117.9	14	.64	.59	.20	-.04	.41	.02
115.5	13	.74	.62	.35	-.12	.77	.02
111.4	12	.68	.63	.42	-.05	.96	.02
Algal biomass (mg/L)							
137.6	20	0.30	0.34	0.19	0.04	0.51	0.01
134.6	19	.33	.32	.30	-.01	.77	.02
123.0	13	.32	.40	.22	.07	.62	.02
122.0	16	.37	.49	.39	.12	.74	.01
117.9	14	.48	.51	.35	.03	.55	.03
115.5	14	.40	.52	.45	.12	1.28	.04
111.4	13	.32	.57	.53	.26	1.26	.04

Table 23. Summary statistics for the measured and simulated water-quality concentrations for seven sites on the Catawba River, S.C., August 23–27, 1996—Continued

[mg/L, milligram per liter]

River mile	Number of observations	Mean measured concentration	Mean simulated concentration	Root mean square error	Mean error	Maximum absolute error	Minimum absolute error
Organic phosphorus (mg/L)							
137.6	20	0.03	0.02	0.02	-0.01	0.05	0.00
134.6	18	.05	.03	.04	-.02	.07	.00
123.0	12	.06	.08	.03	.02	.08	.00
122.0	16	.08	.07	.03	-.01	.05	.00
117.9	14	.08	.08	.06	.00	.17	.01
115.5	13	.09	.08	.06	-.01	.13	.00
111.4	11	.08	.07	.02	.00	.05	.00
Orthophosphate (mg/L)							
137.6	20	0.01	0.01	0.01	0.00	0.03	0.00
134.6	19	.07	.03	.06	-.04	.14	.00
123.0	12	.09	.18	.12	.09	.19	.01
122.0	16	.16	.15	.12	-.01	.42	.00
117.9	14	.17	.16	.10	-.01	.30	.01
115.5	13	.22	.17	.11	-.05	.25	.03
111.4	11	.19	.16	.10	-.03	.24	.01
Ultimate carbonaceous biochemical oxygen demand (mg/L)							
137.6	20	1.63	2.06	1.62	0.43	6.39	0.01
134.6	19	2.34	2.21	2.12	-.13	6.78	.00
123.0	12	1.89	2.87	1.92	.98	4.34	.07
122.0	16	2.00	2.41	1.54	.41	3.85	.03
117.9	14	2.91	3.87	2.25	.96	5.71	.27
115.5	13	2.11	3.73	2.08	1.62	3.98	.88
111.4	12	2.20	3.93	2.03	1.19	4.10	.64
Dissolved oxygen (mg/L)							
137.6	20	4.85	4.65	0.43	-0.20	0.93	0.05
134.6	19	5.53	5.45	.45	-.08	1.04	.05
123.0	13	6.07	6.55	.73	.48	1.10	.02
122.0	16	6.01	6.50	.73	.48	1.38	.11
117.9	14	5.34	6.07	.93	.73	1.59	.00
115.5	13	6.48	6.54	.36	.06	.68	.03
111.4	12	6.44	6.48	.51	.05	.98	.12

Table 24. Summary statistics for the measured and simulated water-quality concentrations for seven sites on the Catawba River, S.C., July 11–15,1997

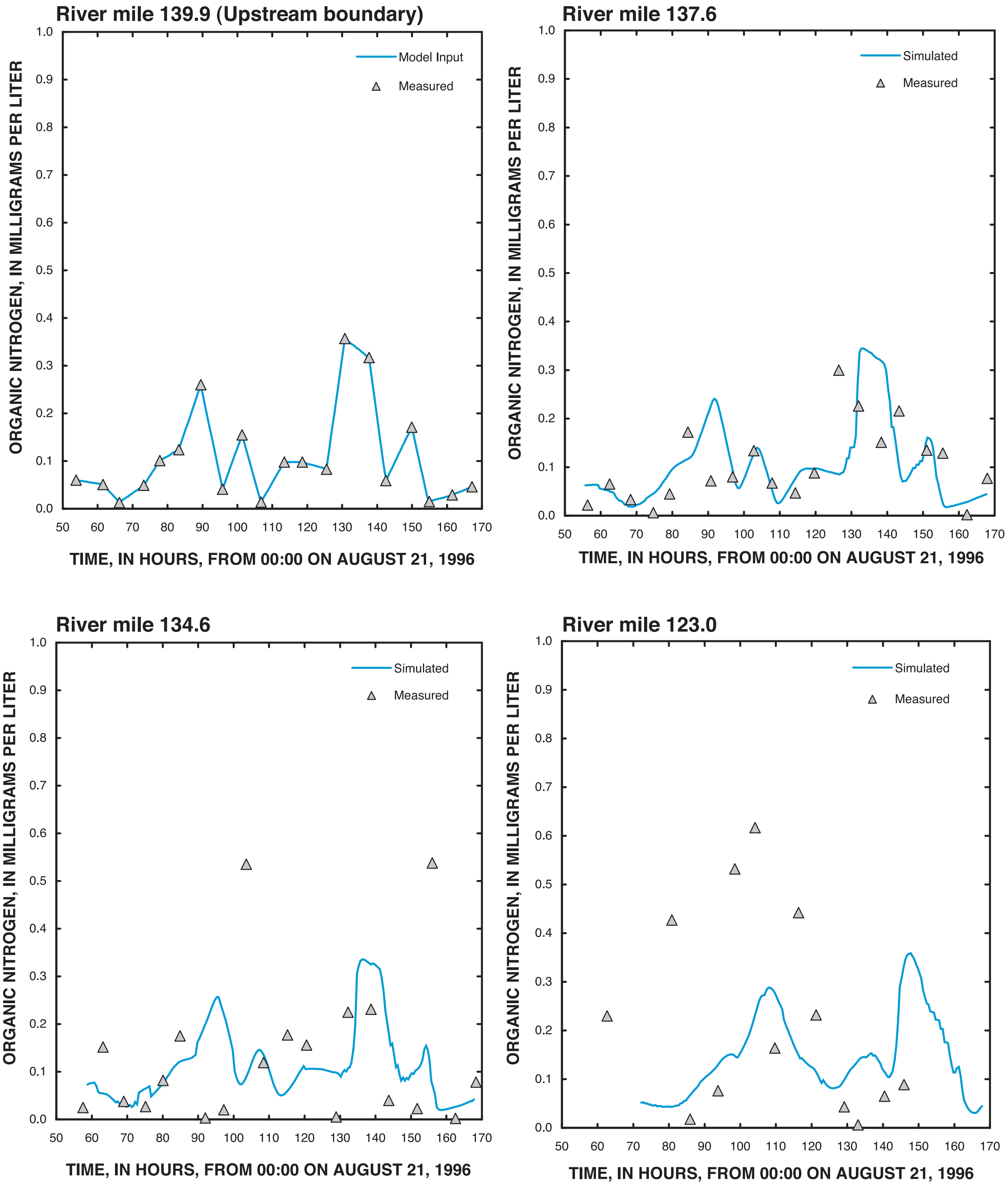
[mg/L, milligram per liter]

River mile	Number of observations	Mean measured concentration	Mean simulated concentration	Root mean square error	Mean error	Maximum absolute error	Minimum absolute error
Organic nitrogen (mg/L)							
137.6	19	0.17	0.15	0.18	-0.02	0.62	0.00
134.6	18	.20	.16	.21	-.03	.45	.03
123.0	17	.19	.16	.20	-.03	.45	.02
122.0	16	.22	.16	.19	-.05	.48	.00
117.9	15	.20	.15	.12	-.04	.29	.00
115.5	15	.16	.15	.13	-.02	.42	.00
111.4	10	.20	.12	.15	-.07	.36	.02
Ammonia nitrogen (mg/L)							
137.6	19	0.17	0.20	0.05	0.02	0.11	0.00
134.6	19	.16	.20	.07	.04	.16	.01
123.0	17	.14	.18	.07	.04	.15	.01
122.0	16	.14	.17	.05	.03	.08	.00
117.9	16	.20	.20	.11	-.01	.39	.01
115.5	15	.16	.19	.08	.03	.18	.00
111.4	13	.14	.17	.04	.03	.09	.00
Nitrate nitrogen (mg/L)							
137.6	19	0.14	0.13	0.06	-0.02	0.11	0.00
134.6	19	.18	.16	.06	-.02	.13	.01
123.0	17	.31	.31	.09	.00	.26	.00
122.0	16	.42	.30	.16	-.12	.35	.00
117.9	16	.40	.32	.11	-.08	.24	.03
115.5	15	.49	.33	.16	-.15	.24	.04
111.4	13	.45	.38	.12	-.07	.28	.01
Algal biomass (mg/L)							
137.6	19	0.005	0.005	0.000	0.000	0.001	0.000
134.6	19	.012	.005	.032	-.007	.141	.000
123.0	17	.005	.006	.002	.001	.005	.000
122.0	16	.005	.007	.003	.002	.007	.000
117.9	16	.005	.007	.002	.002	.006	.000
115.5	15	.005	.007	.003	.002	.008	.000
111.4	13	.005	.008	.003	.003	.005	.001

Table 24. Summary statistics for the measured and simulated water-quality concentrations for seven sites on the Catawba River, S.C., July 11–15,1997—Continued

[mg/L, milligram per liter]

Organic phosphorus (mg/L)							
137.6	19	0.16	0.12	0.10	-0.03	0.20	0.00
134.6	19	.18	.15	.10	-.07	.15	.00
123.0	17	.13	.13	.10	.06	.17	.00
122.0	14	.17	.13	.10	.02	.23	.01
117.9	14	.12	.13	.07	.03	.11	.00
115.5	14	.13	.13	.10	.01	.15	.01
111.4	13	.17	.12	.19	-.01	.56	.00
Orthophosphate (mg/L)							
137.6	19	0.02	0.03	0.02	0.00	0.04	0.00
134.6	19	.05	.05	.05	.00	.13	.00
123.0	17	.08	.17	.11	.09	.25	.00
122.0	16	.09	.16	.09	.07	.23	.01
117.9	16	.11	.12	.05	.01	.09	.00
115.5	15	.12	.13	.06	.01	.16	.01
111.4	13	.12	.15	.09	.02	.21	.00
Ultimate carbonaceous biochemical oxygen demand (mg/L)							
137.6	19	5.29	4.42	2.67	-0.87	6.29	0.18
134.6	19	5.27	4.34	3.47	-.92	11.59	.05
123.0	17	4.77	3.82	4.84	-.95	14.02	.31
122.0	15	5.78	3.68	5.32	-2.10	9.66	.30
117.9	16	5.88	5.29	4.56	-.59	10.33	.23
115.5	15	7.36	5.05	4.52	-2.31	9.10	.35
111.4	13	7.41	4.81	4.58	-2.60	8.84	.10
Dissolved oxygen (mg/L)							
137.6	18	4.56	4.50	0.55	-0.07	1.40	0.05
134.6	19	5.54	5.33	1.05	-.21	3.28	.12
123.0	17	6.45	6.42	.78	-.03	1.32	.18
122.0	16	6.52	6.43	.85	-.09	1.74	.10
117.9	16	6.10	5.76	.85	-.34	1.68	.05
115.5	15	6.72	6.25	.61	-.46	1.19	.07
111.4	13	6.56	6.00	.79	-.56	1.28	.01


River mile 139.9 (Upstream boundary)

River mile 137.6

River mile 134.6

River mile 123.0

Figure 28. Measured and simulated organic nitrogen concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

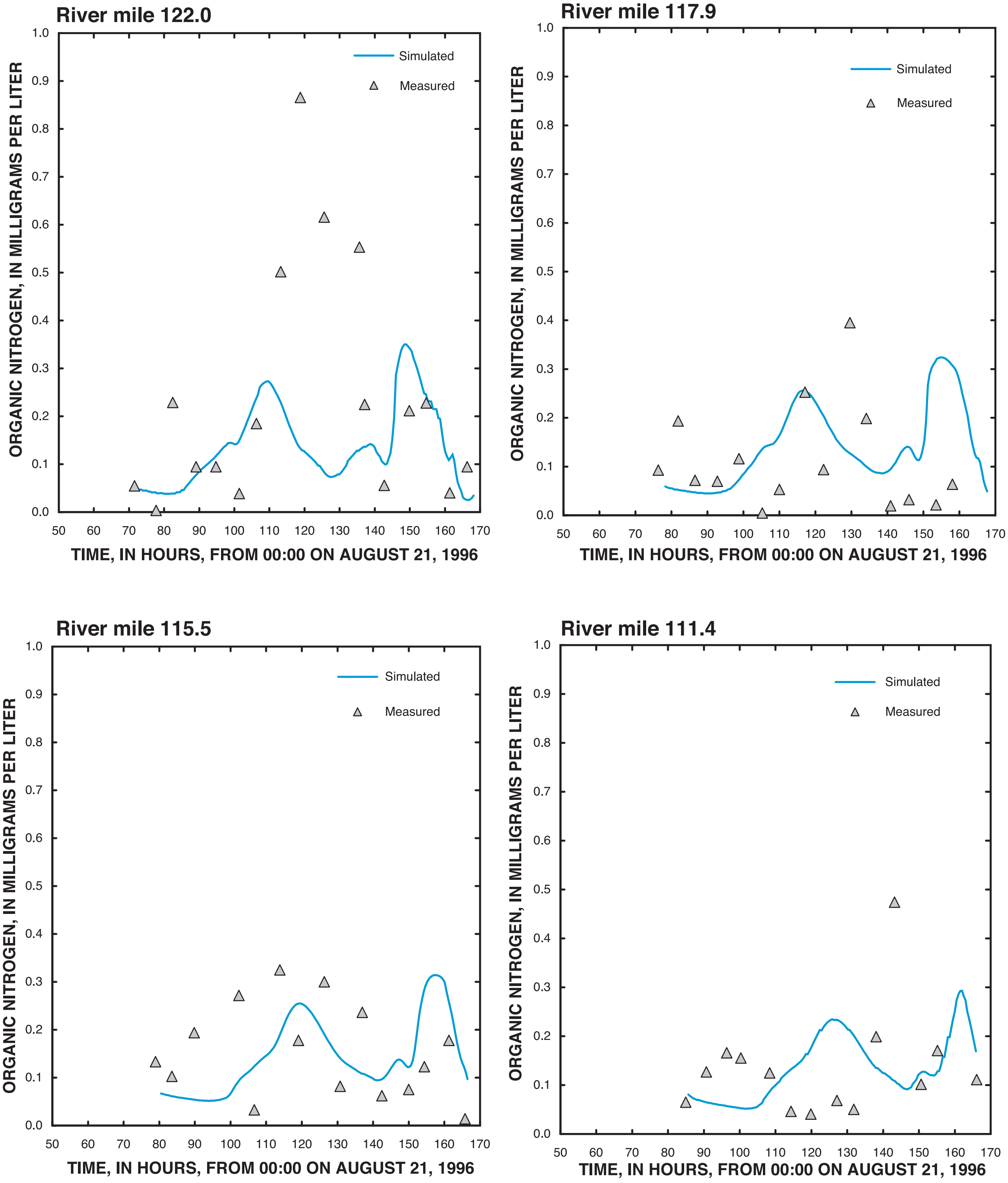
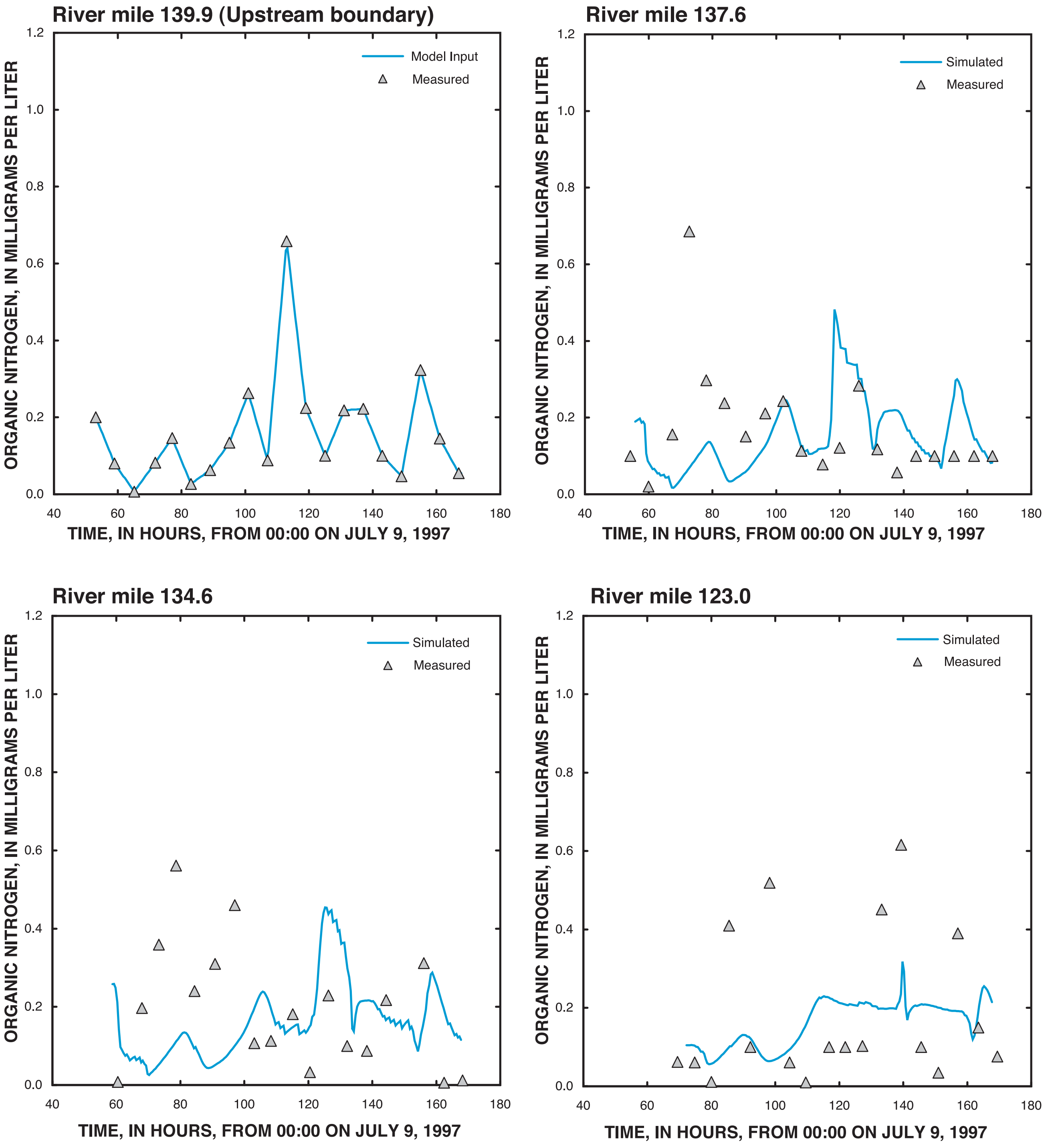


Figure 28 (Continued). Measured and simulated organic nitrogen concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

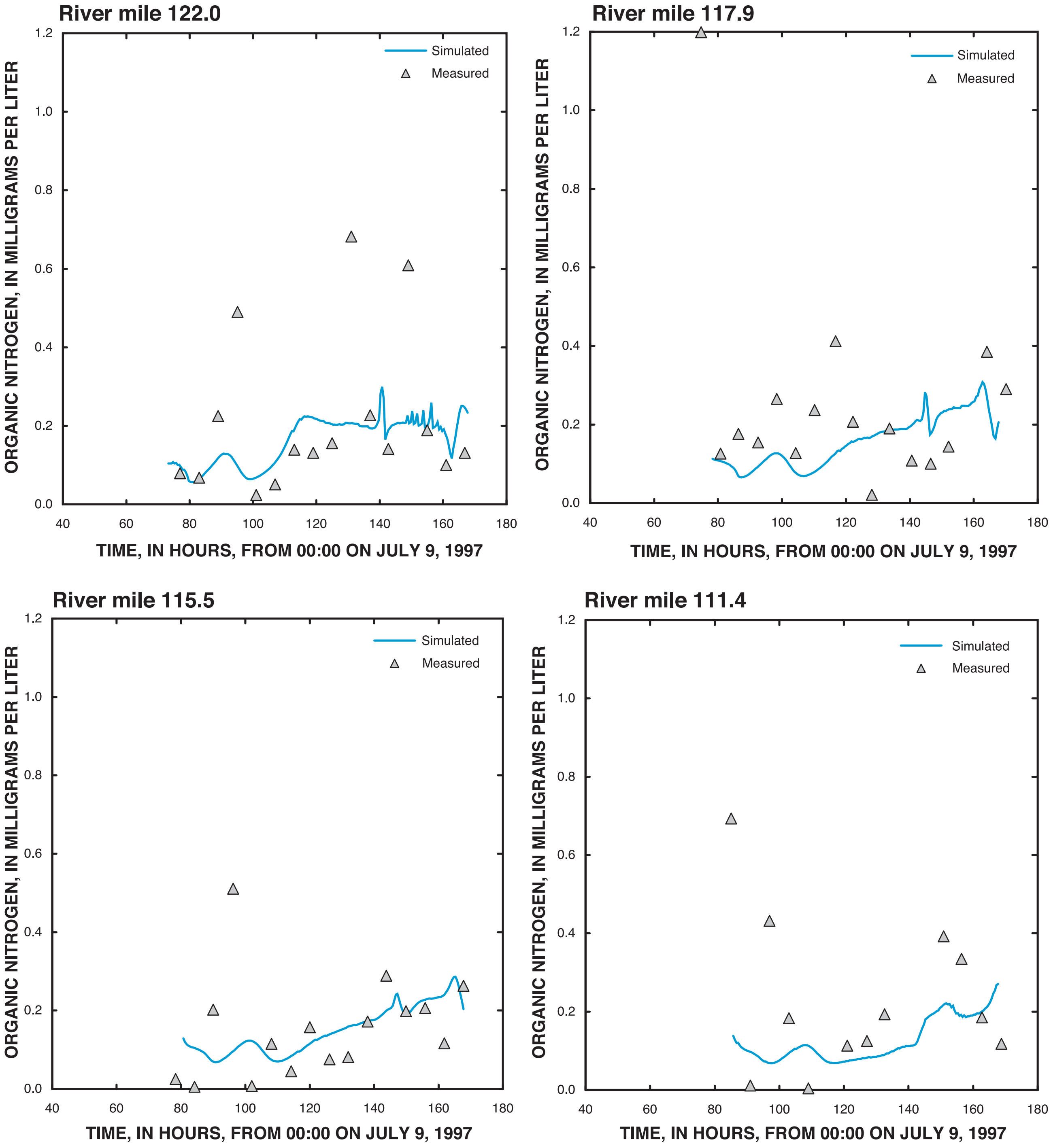

River mile 139.9 (Upstream boundary)

River mile 137.6

River mile 134.6

River mile 123.0

Figure 29. Measured and simulated organic nitrogen concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

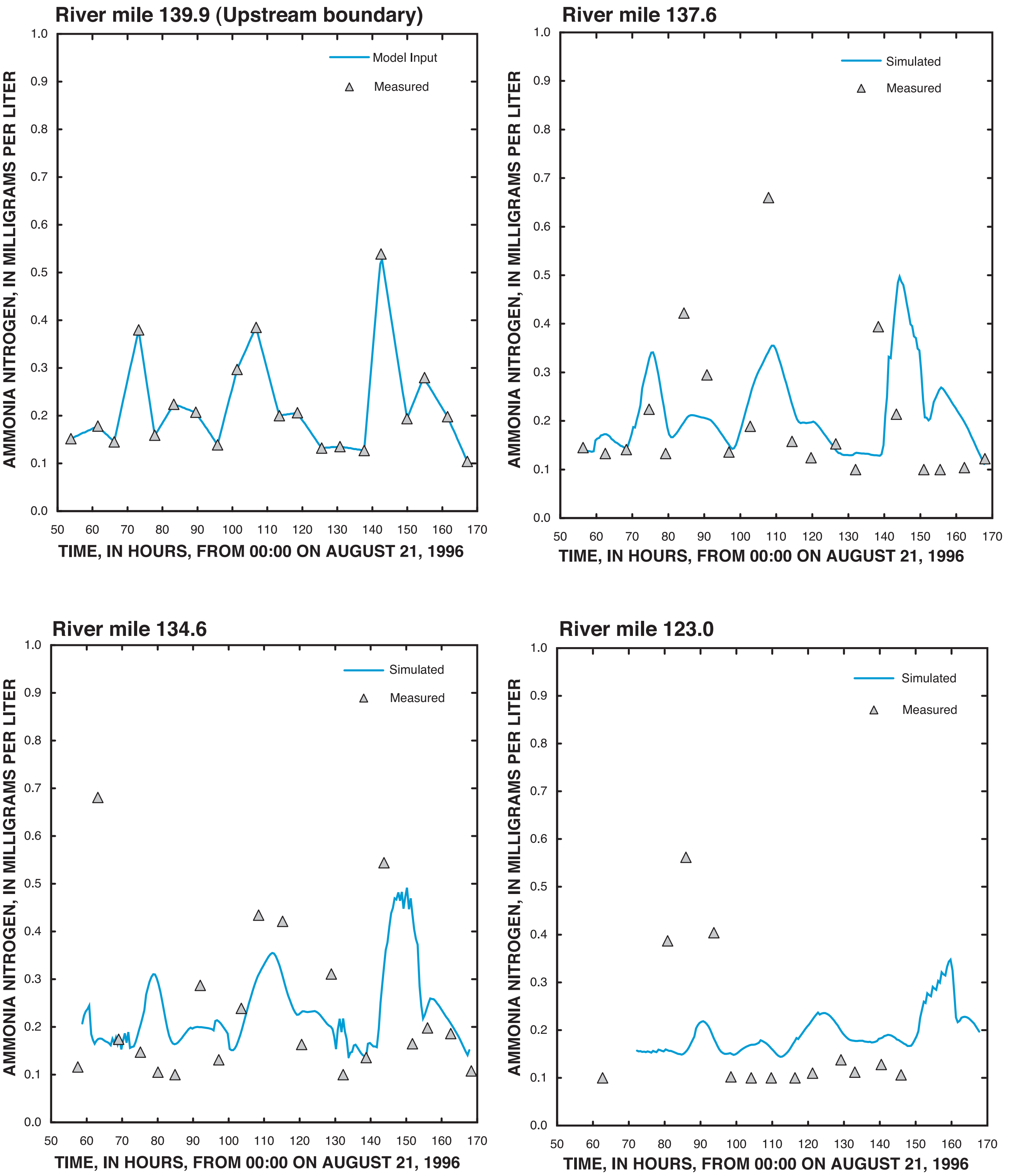

River mile 122.0

River mile 117.9

River mile 115.5

River mile 111.4

Figure 29 (Continued). Measured and simulated organic nitrogen concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.


River mile 139.9 (Upstream boundary)

River mile 137.6

River mile 134.6

River mile 123.0

Figure 30. Measured and simulated ammonia nitrogen concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

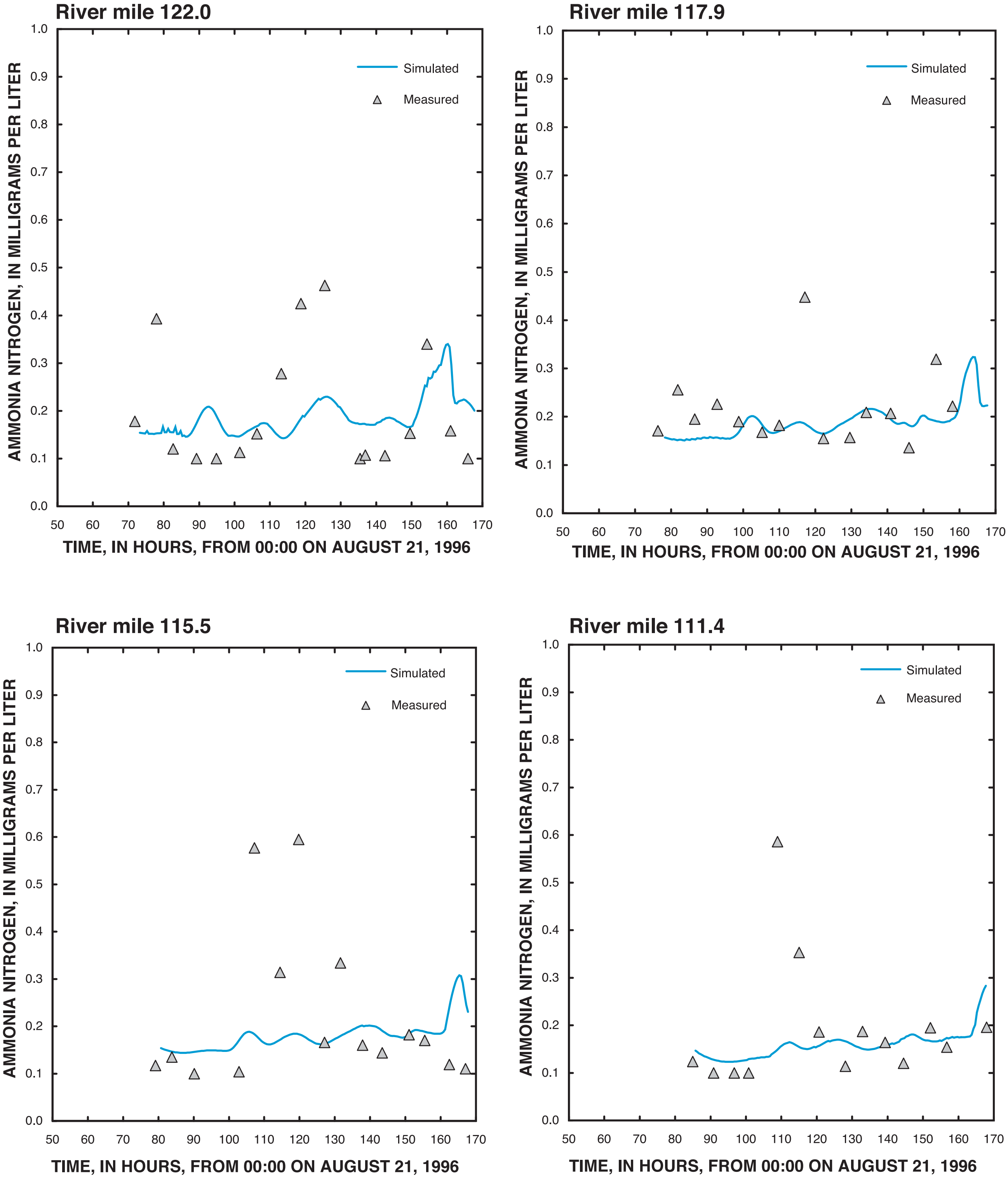


Figure 30 (Continued). Measured and simulated ammonia nitrogen concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

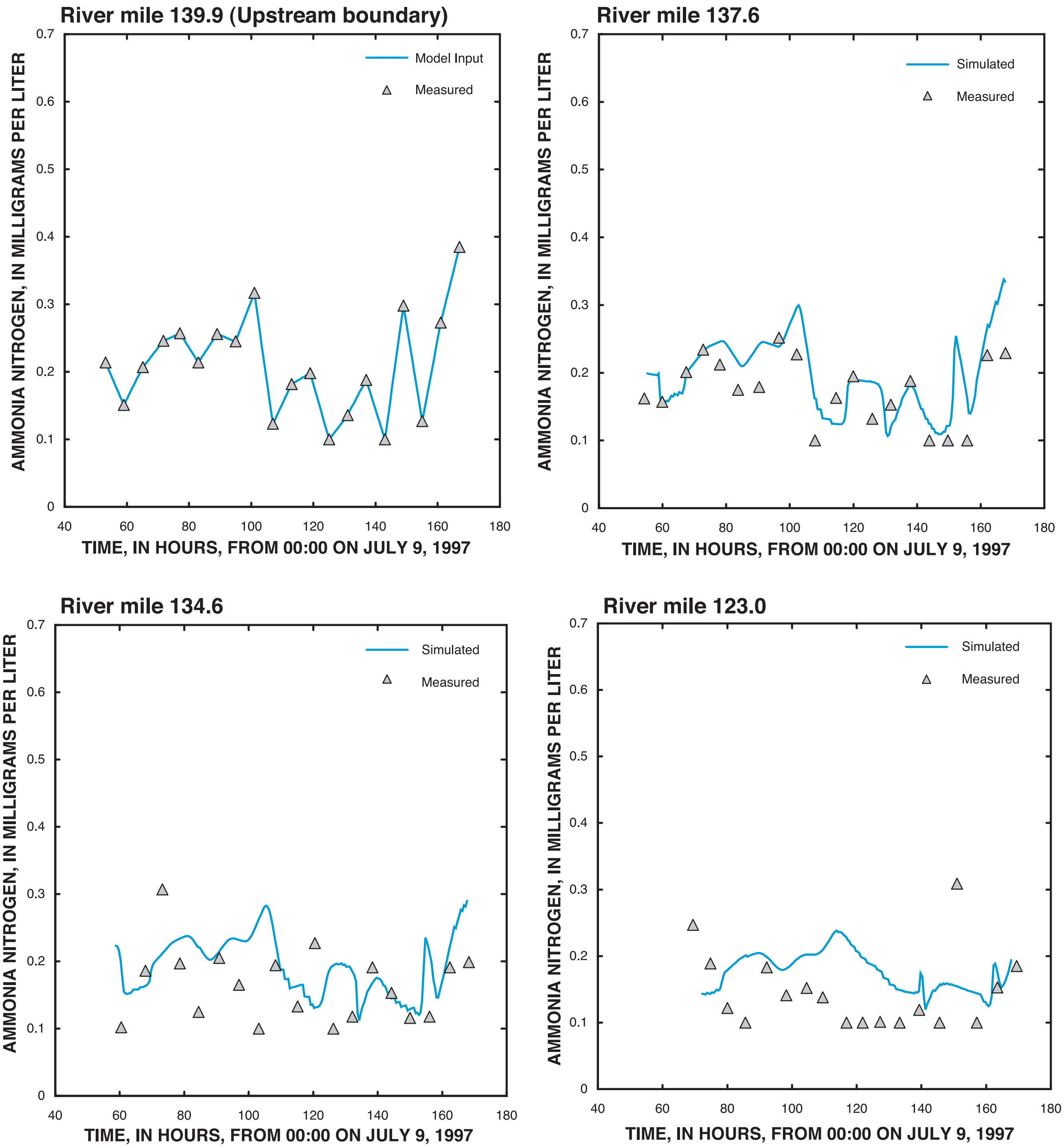


Figure 31. Measured and simulated ammonia nitrogen concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

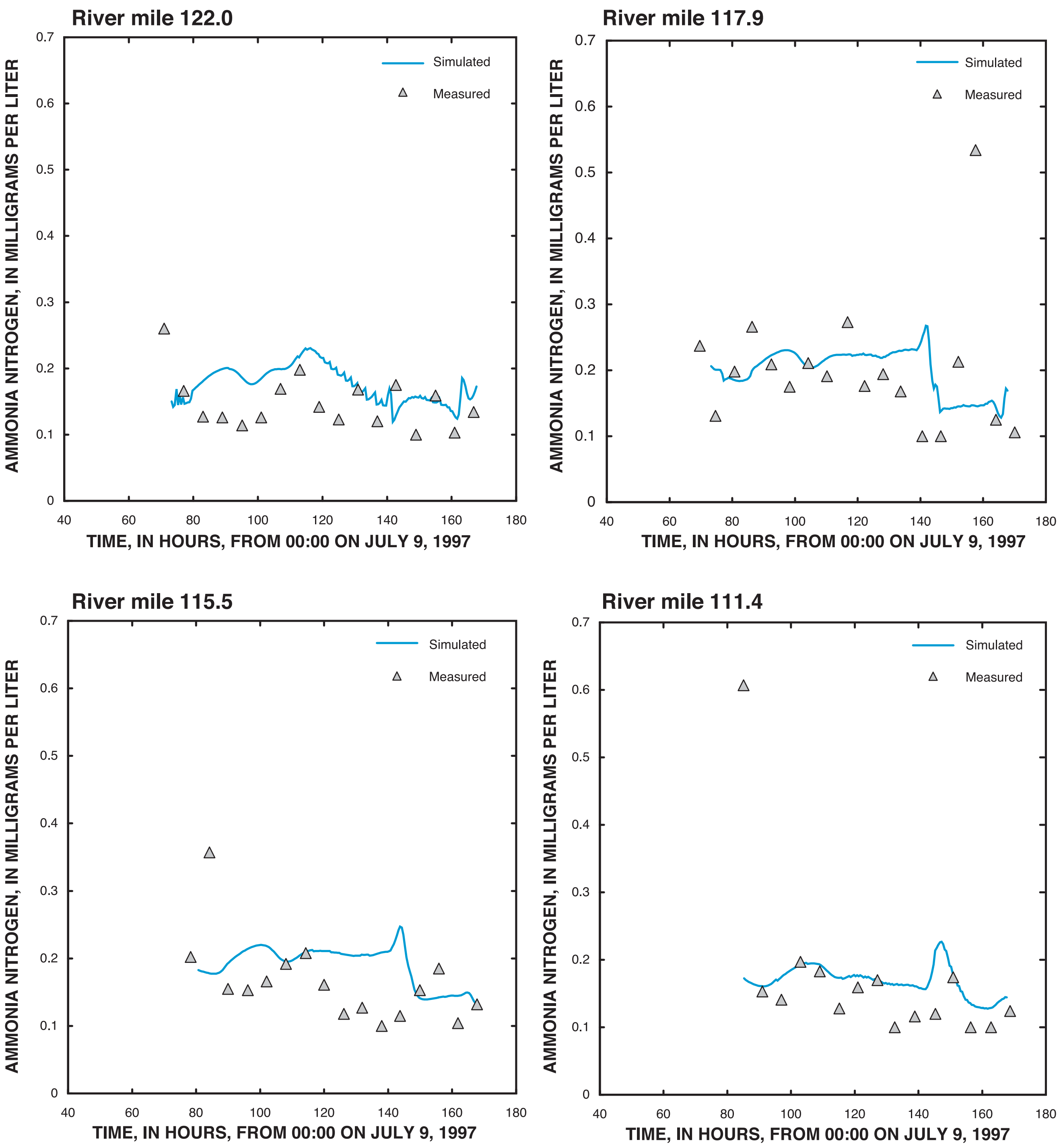


Figure 31 (Continued). Measured and simulated ammonia nitrogen concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

the measured data were underpredicted (table 23; fig. 32). The largest mean error occurred at RM 123.0. From the plots of simulated and measured data, it can be seen that the simulated nitrate nitrogen increases substantially between RM 134.6 and RM 123.0. The overall concentrations of the measured data at RM 123.0 are larger than those at RM 134.6, but do not show the same significant increase as the simulated data. In the model, the boundary data for Sugar Creek enter at RM 129.6. The median nitrate nitrogen concentration for the 1996 Sugar Creek samples was 2.75 mg/L, and the data ranged from 0.15 to 9.16 mg/L. Because of the rain event in 1996, flow during the calibration ranged from 128 to 1,650 ft^3/s. Consequently, given the high nitrate nitrogen concentrations and significant flow from Sugar Creek, the simulated values at RM 123.0 do not seem unreasonable. For the 1997 validation period, the mean errors ranged from -0.15 to 0.0 mg/L; six of the seven sites had negative mean errors, indicating that on average, the measured data were underpredicted (tables 24; fig. 33). In general, simulated values for the 1997 validation period were better than those for the 1996 period. For both periods, there were several measured values that seemed excessively higher than the general trend of the data and, therefore, may be questionable.

Algal biomass: For this study, chlorophyll *a* was used as an algal biomass indicator. According to Standard Methods for the Examination of Water and Wastewater, it can be assumed that chlorophyll *a* constitutes, on average, 1.5 percent of the dry weight of organic matter of algae. Therefore, the algal biomass can be estimated by multiplying the chlorophyll *a* concentration by a factor of 67, which is 1 divided by 0.015 (Greenberg and others, 1992). It should be noted that in this report the chlorophyll *a* concentrations are presented in micrograms per liter but the algal biomass concentrations are presented in milligrams per liter.

Simulated values of algal biomass for 1996 generally were in good agreement with measured values (fig. 34). The mean errors ranged from -0.01 to 0.26 mg/L. All but one of the sites (RM 134.6) had positive mean errors, indicating that on average, the model over simulated the algal biomass. The changing shape of the simulation graph is a result of dispersion and decay of the different constituents and of additional inputs from point sources and tributaries. As shown in figure 34, the general shape of the boundary data remained throughout the study reach, indicating the appreciable influence of the upstream boundary data. As previously mentioned, with the exception of two samples, the chlorophyll *a* concentrations for the 1997 samples were less than the detection limit of 0.075 µg/L. The simulations also showed only minor algal biomass changes through the study reach (fig. 35). The mean errors for the 1997 algal biomass simulations ranged from -0.007 to 0.003 mg/L (table 24).

Organic phosphorus: Simulated time series of organic phosphorus concentrations generally were in good agreement with measured values (figs. 36, 37), particularly for the 1996 sampling period. The mean errors ranged from -0.02 to 0.02 mg/L in the 1996 simulations (table 23). The mean errors for the 1997 period ranged from -0.07 to 0.06, indicating a slightly greater variation between simulated and measured data (table 24). A few high concentrations of phosphorus seem unusual when compared to upstream values and to point-source and tributary loadings. As in some of the previous data discussions, the high measured values relative to simulated values may be the result of analytical error or may indicate that daily mean concentrations from point-source dischargers do not adequately reflect the temporal variations in loadings to the river.

Orthophosphate: As with the organic phosphorus, the simulated time series of orthophosphate concentrations generally were in good agreement with measured values (figs. 38, 39), particularly for the 1996 sampling period. The mean errors ranged from -0.05 to 0.09 mg/L in the 1996 simulations (table 23). Five of the seven sites had negative mean errors, which indicates that on average, the orthophosphate concentrations were underpredicted. In 1996, the high measured values relative to simulated values at RM 115.5 and RM 111.4 may be the result of analytical error or may indicate that daily mean concentrations from point-source dischargers do not adequately reflect the temporal variations in loadings to the river. The mean errors for the 1997 period ranged from 0.00 to 0.09, indicating that on average, the orthophosphate concentrations were overpredicted (table 24). For both 1996 and 1997, there were substantial increases in the simulated orthophosphate concentrations at RM 123.0. In both cases, the higher simulated concentrations probably reflect the influence of Sugar Creek, which had orthophosphate concentrations significantly higher than the river concentrations, and the effects of a substantial rainfall event during the 1996 sampling period.

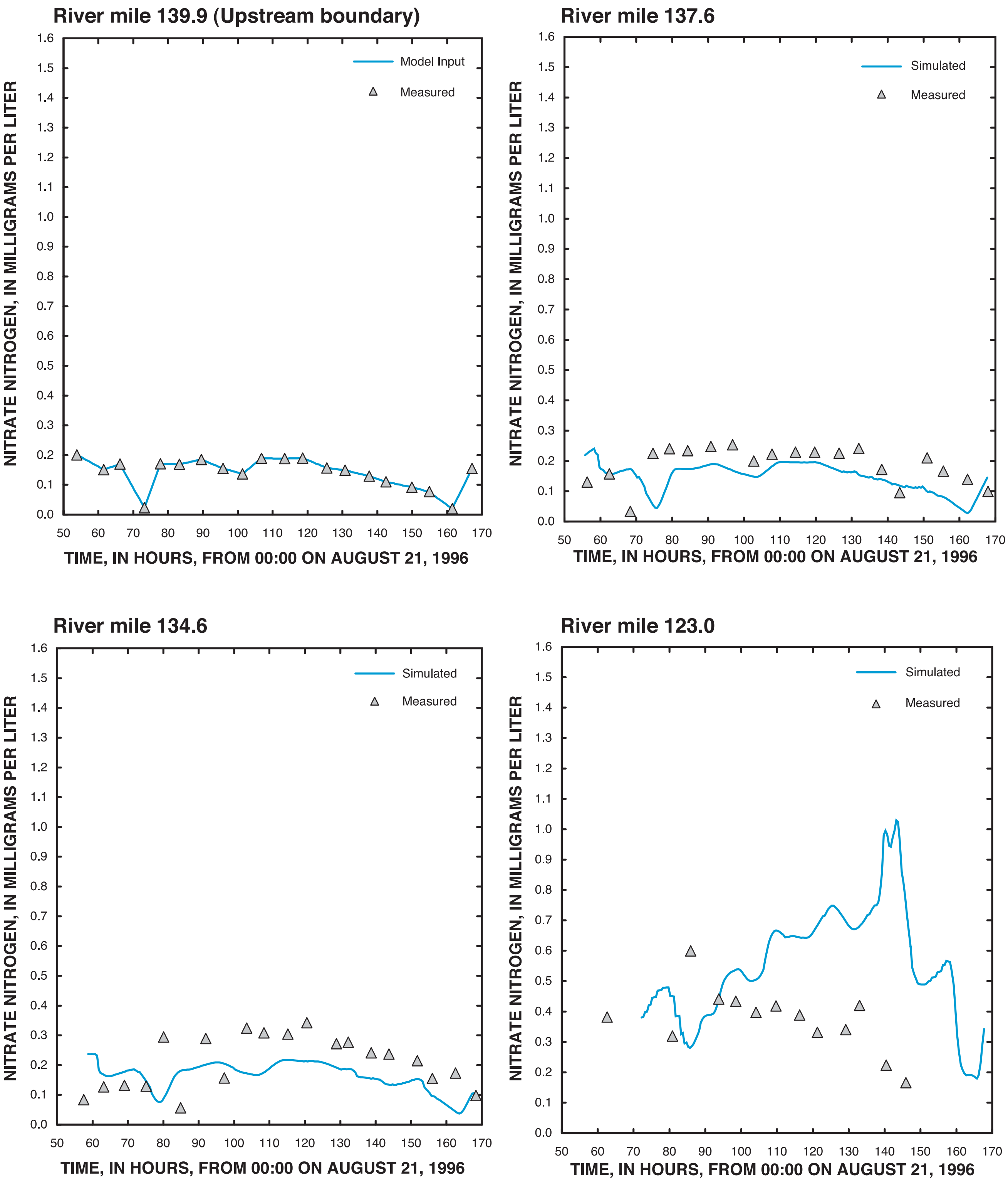


Figure 32. Measured and simulated nitrate nitrogen concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

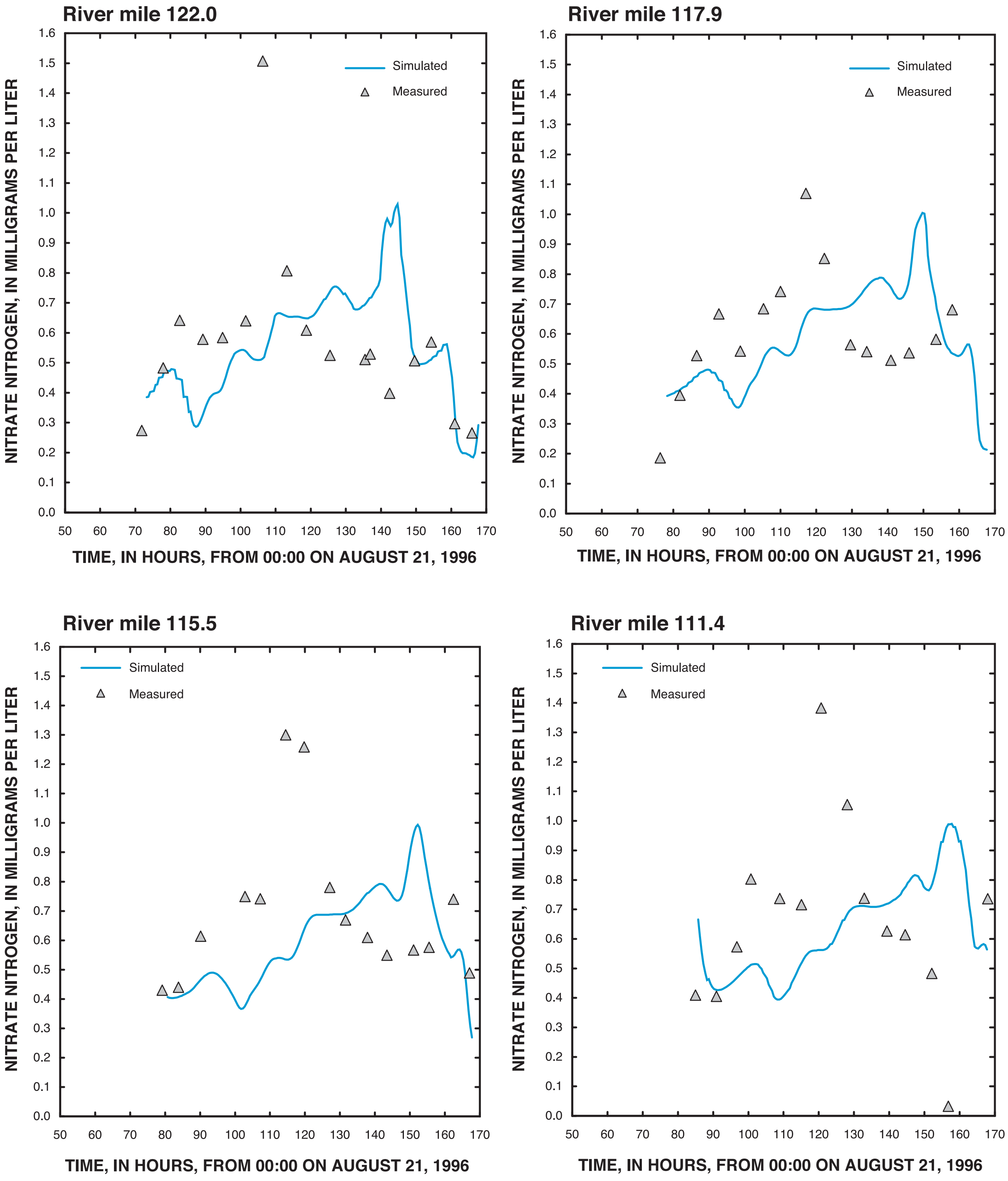


Figure 32 (Continued). Measured and simulated nitrate nitrogen concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

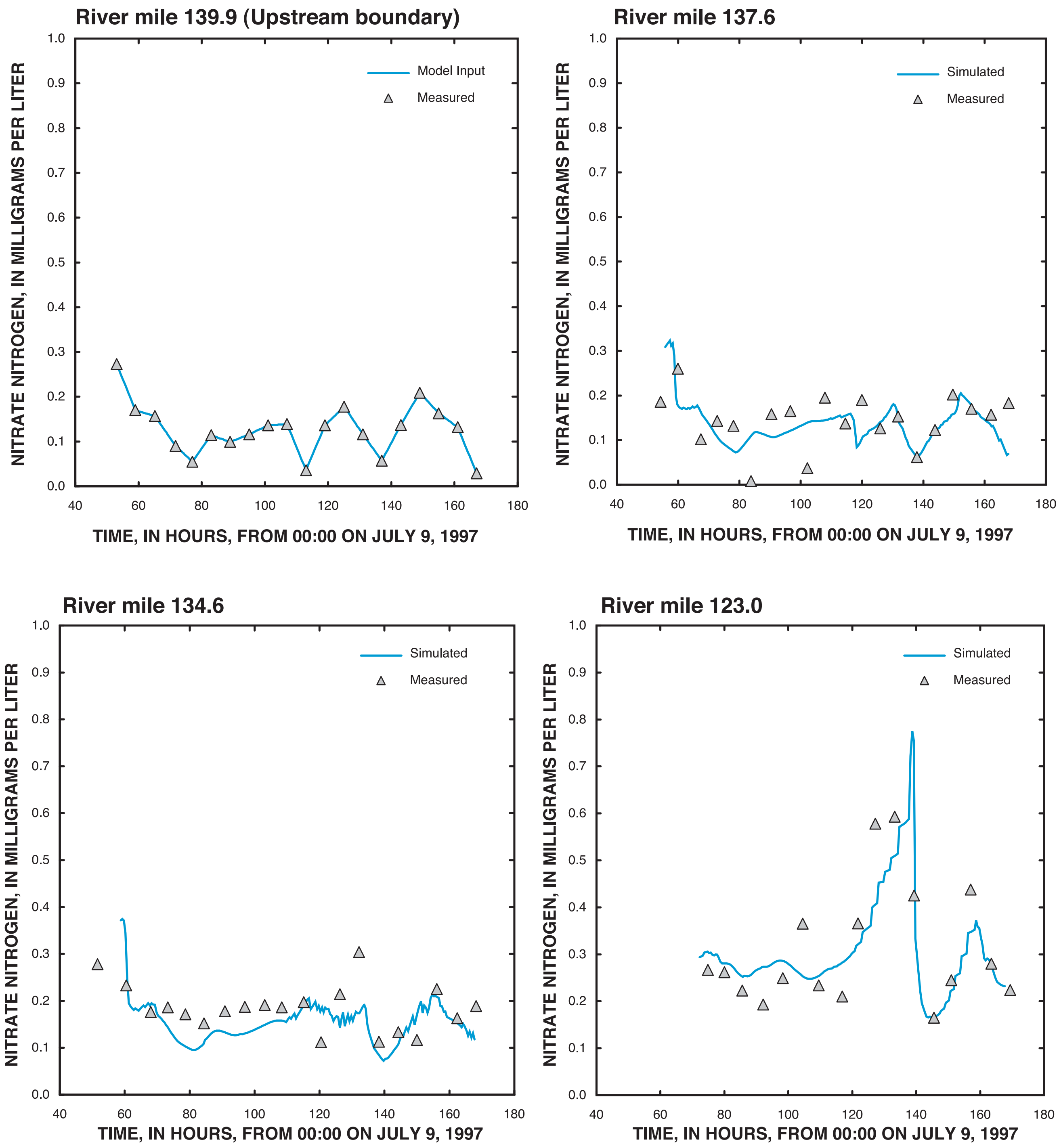


Figure 33. Measured and simulated nitrate nitrogen concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

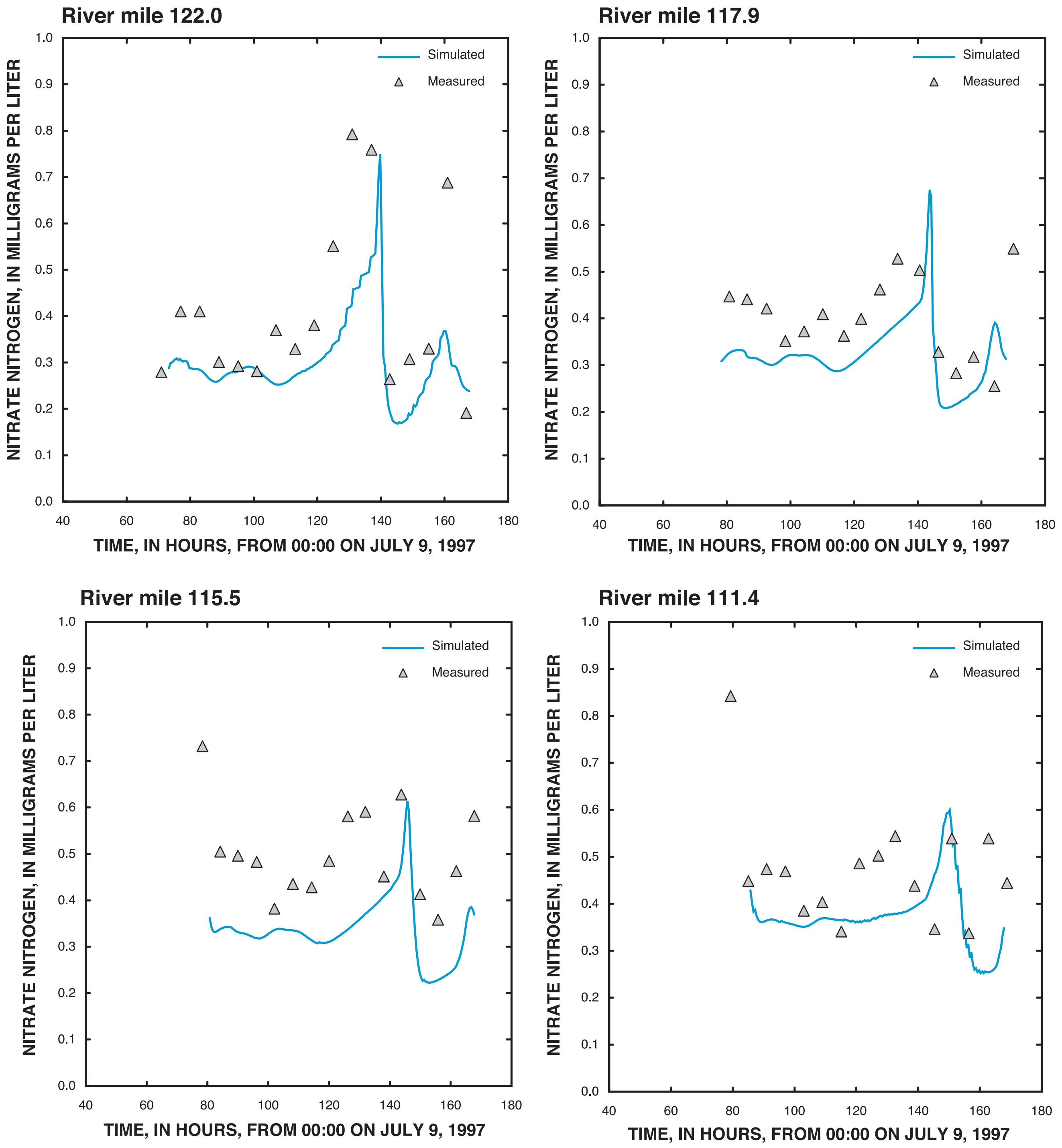


Figure 33 (Continued). Measured and simulated nitrate nitrogen concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

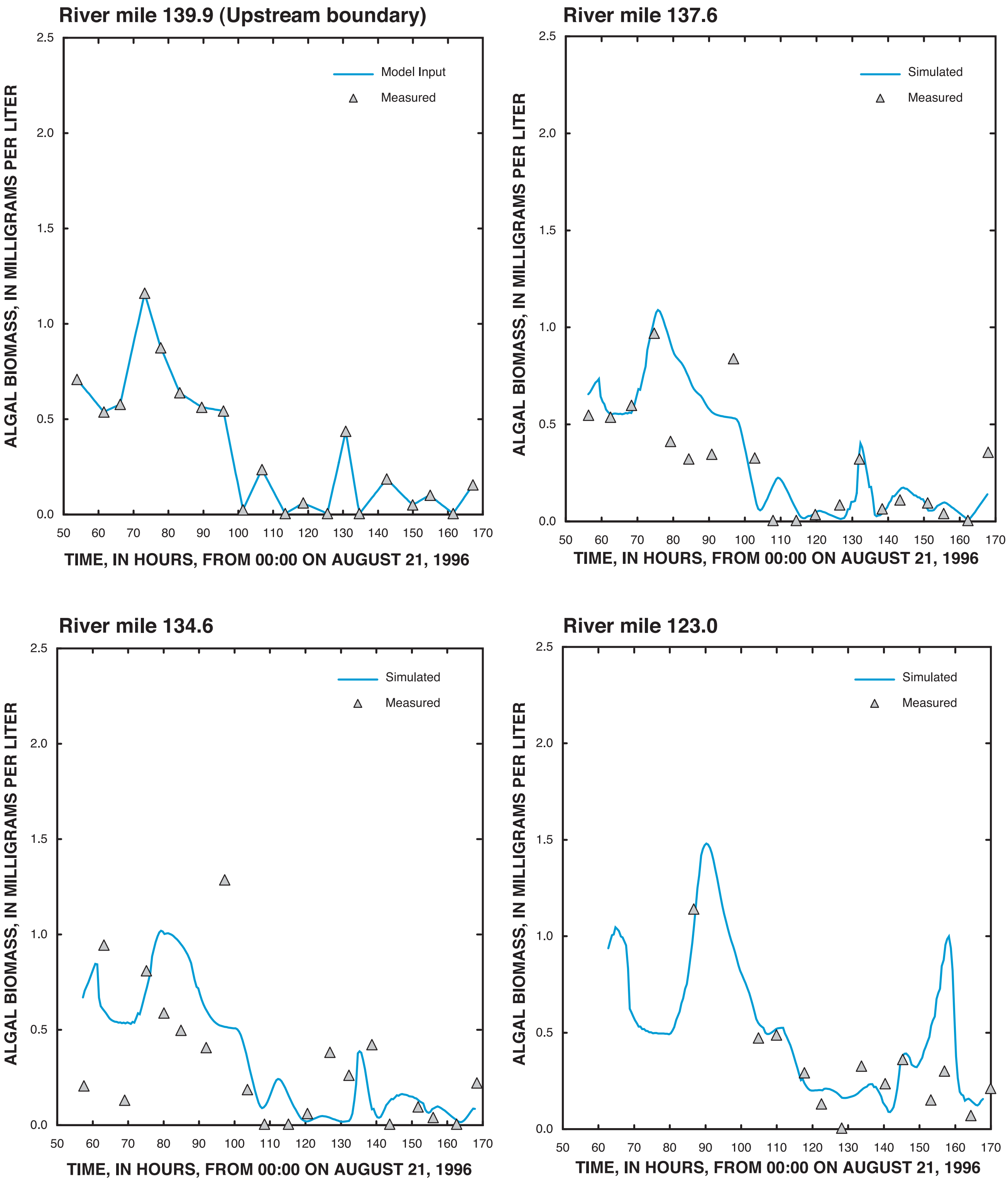


Figure 34. Measured and simulated algal biomass concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

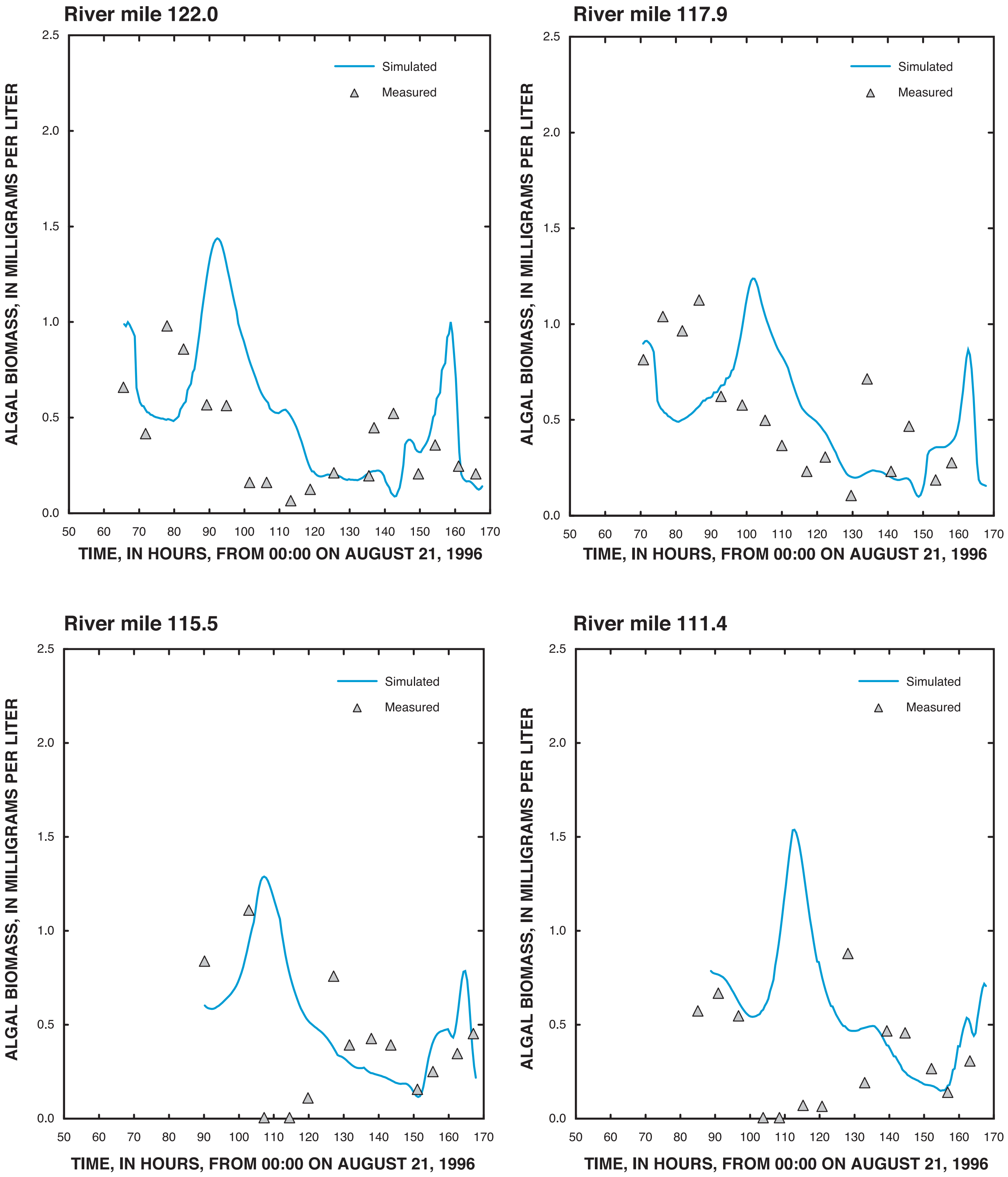


Figure 34 (Continued). Measured and simulated algal biomass concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

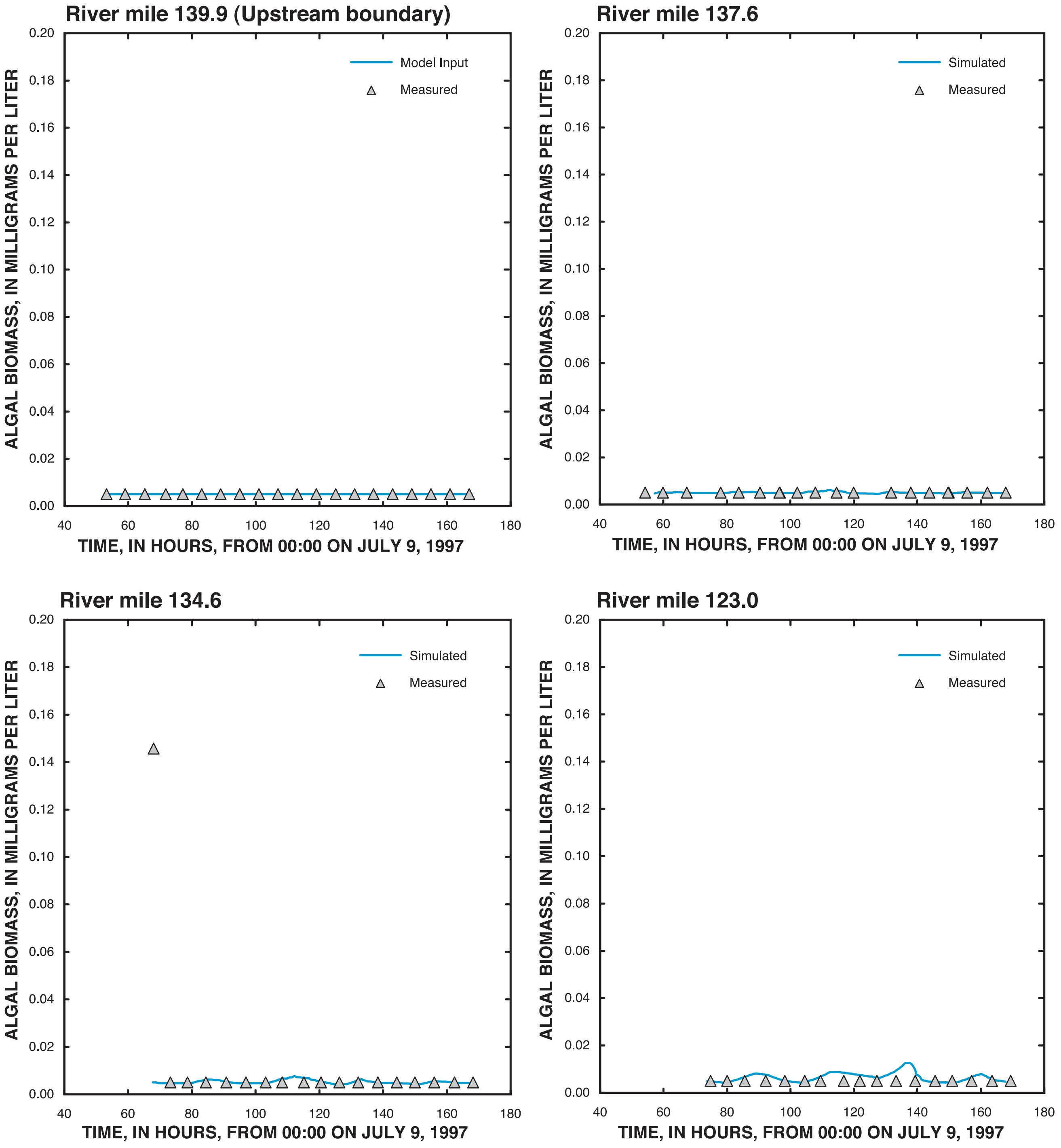


Figure 35. Measured and simulated algal biomass concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., July 11–15, 1997.

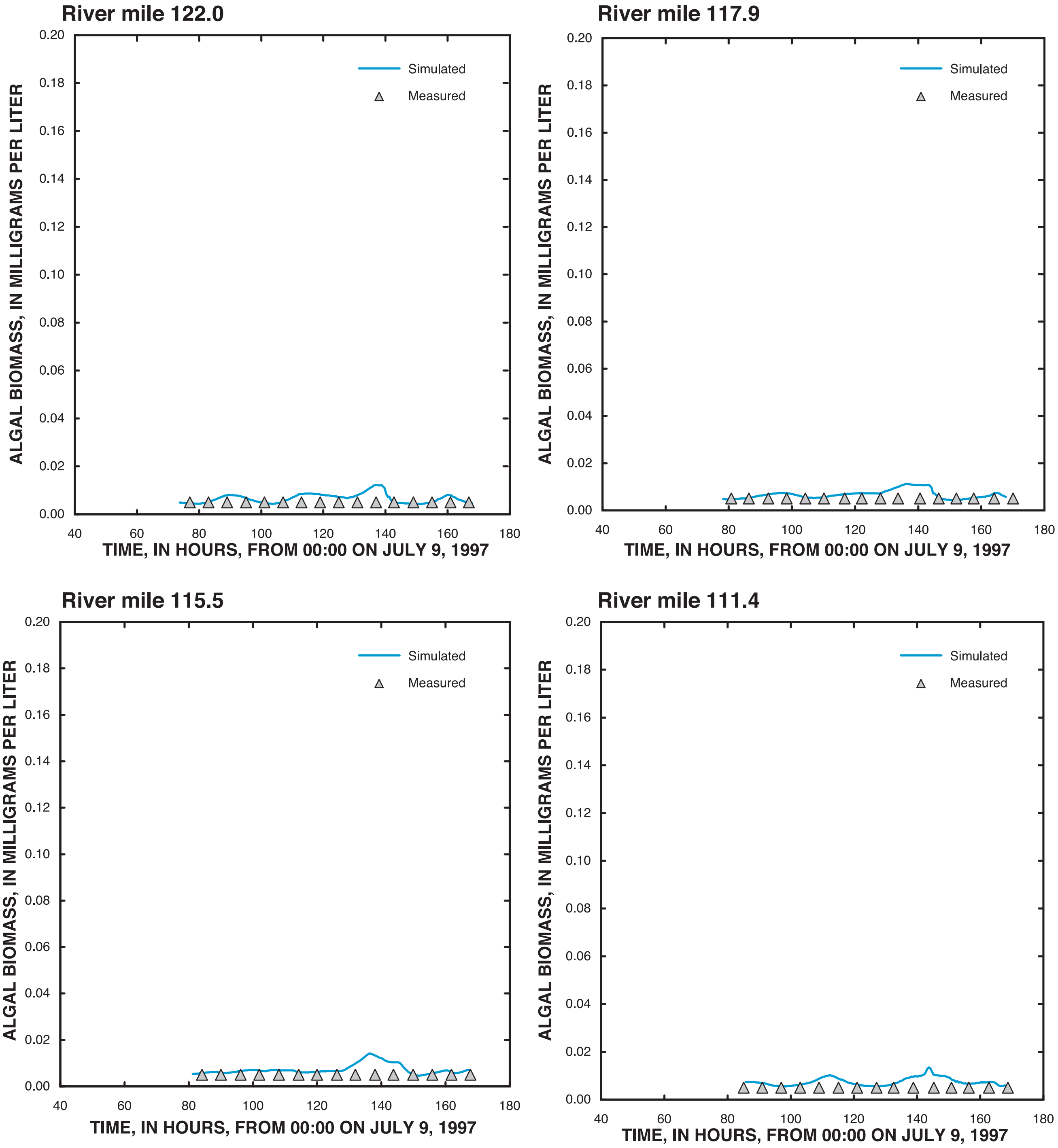


Figure 35 (Continued). Measured and simulated algal biomass concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., July 11–15, 1997.

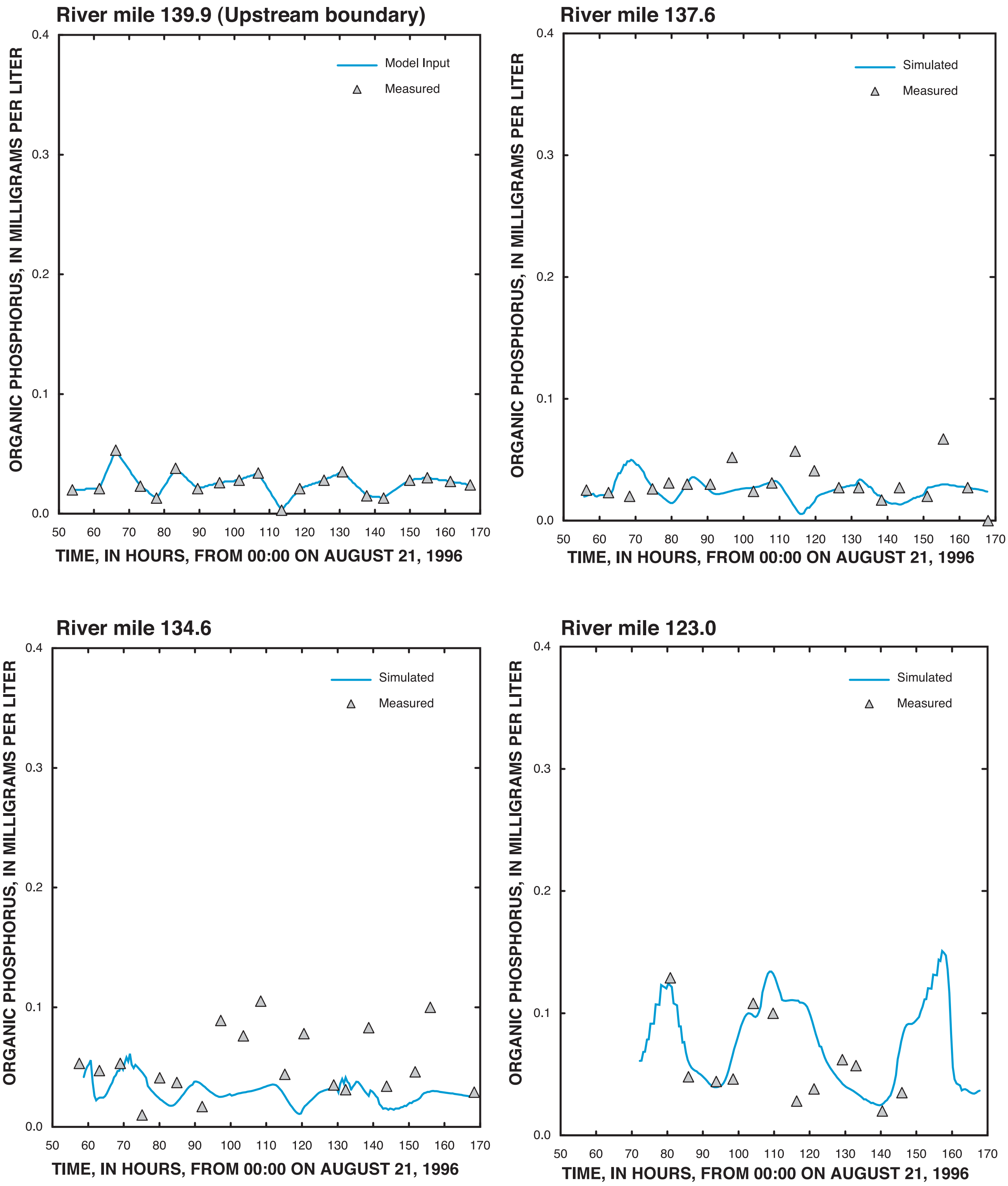


Figure 36. Measured and simulated organic phosphorus concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

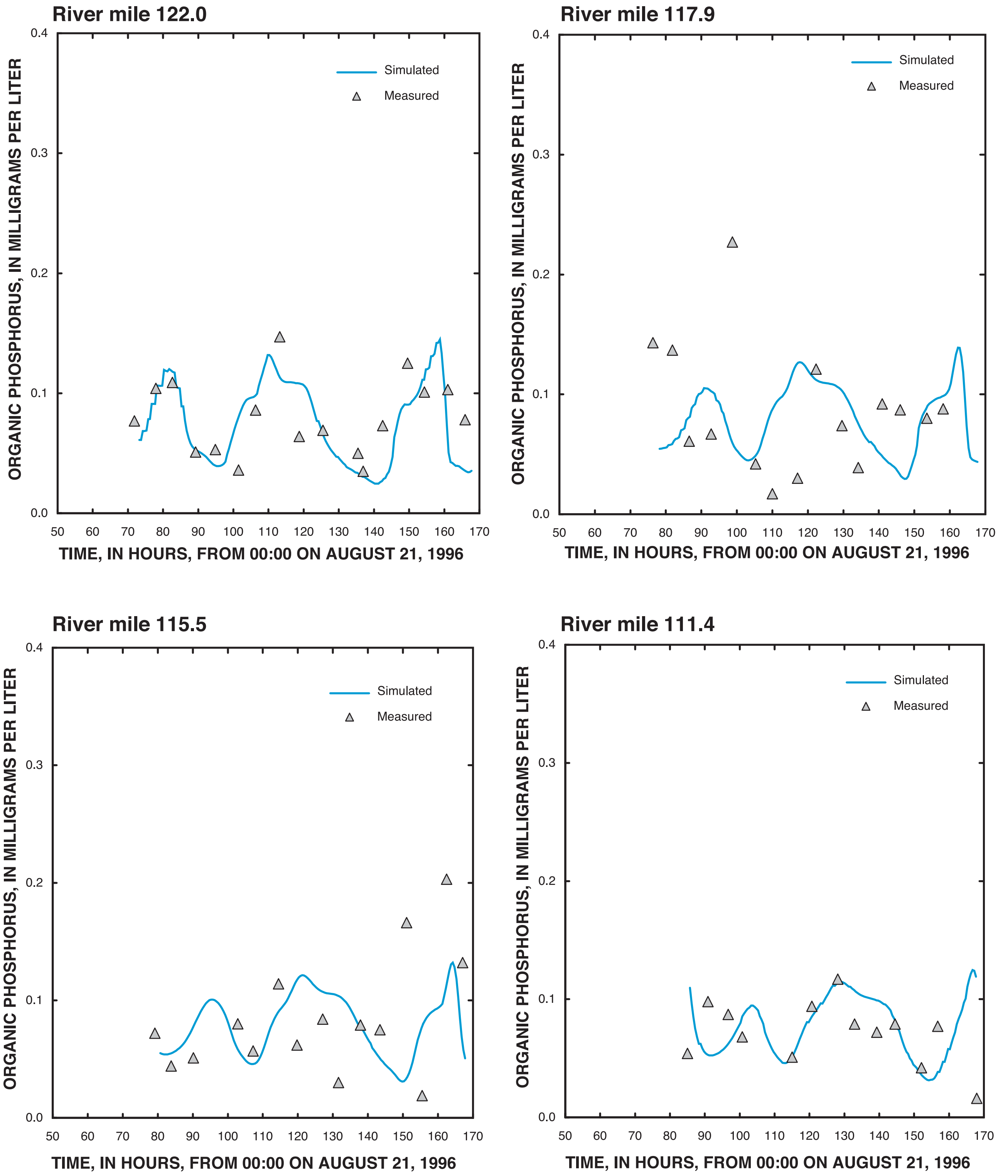
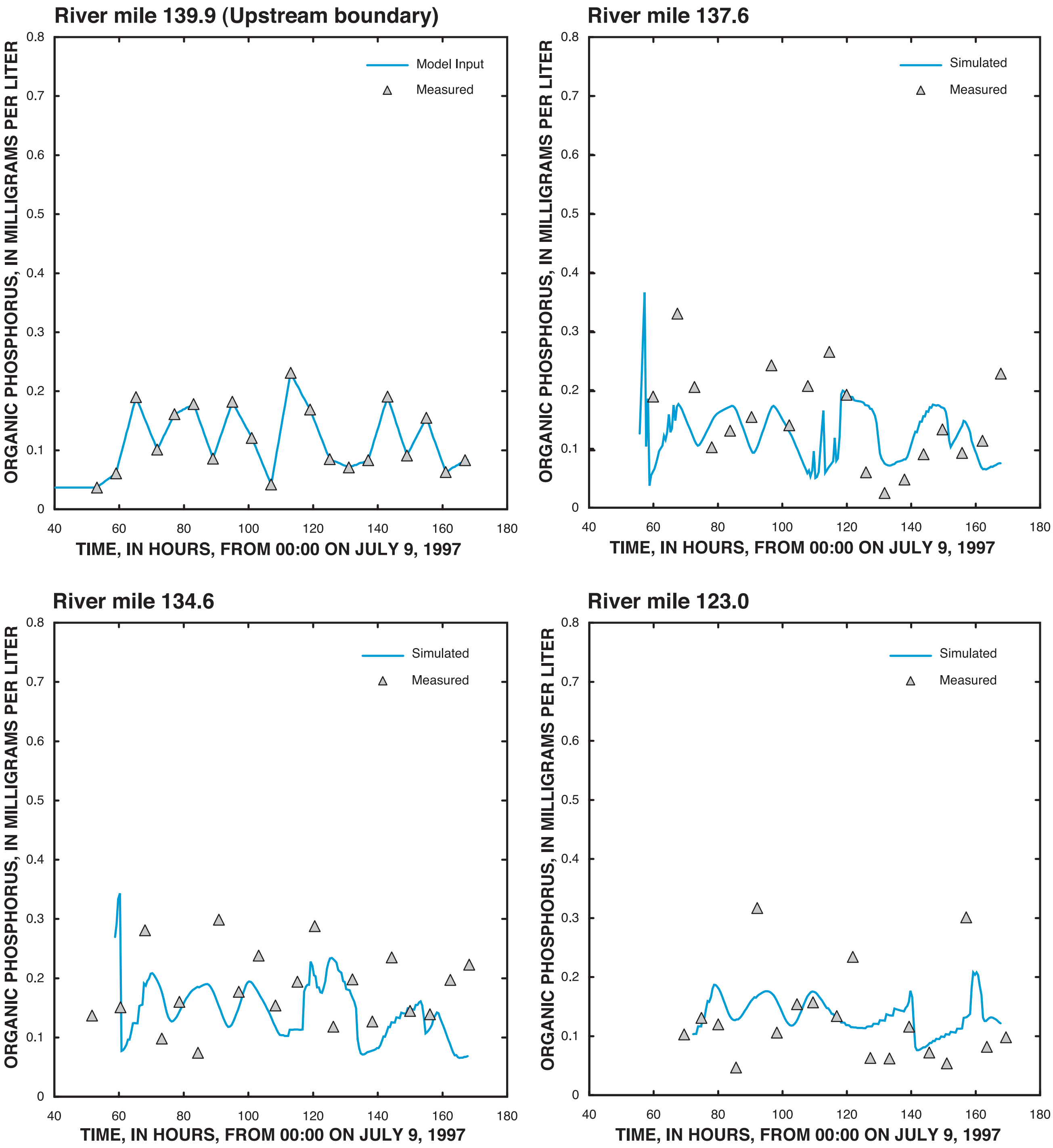


Figure 36 (Continued). Measured and simulated organic phosphorus concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.


River mile 139.9 (Upstream boundary)

River mile 137.6

River mile 134.6

River mile 123.0

Figure 37. Measured and simulated organic phosphorus concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

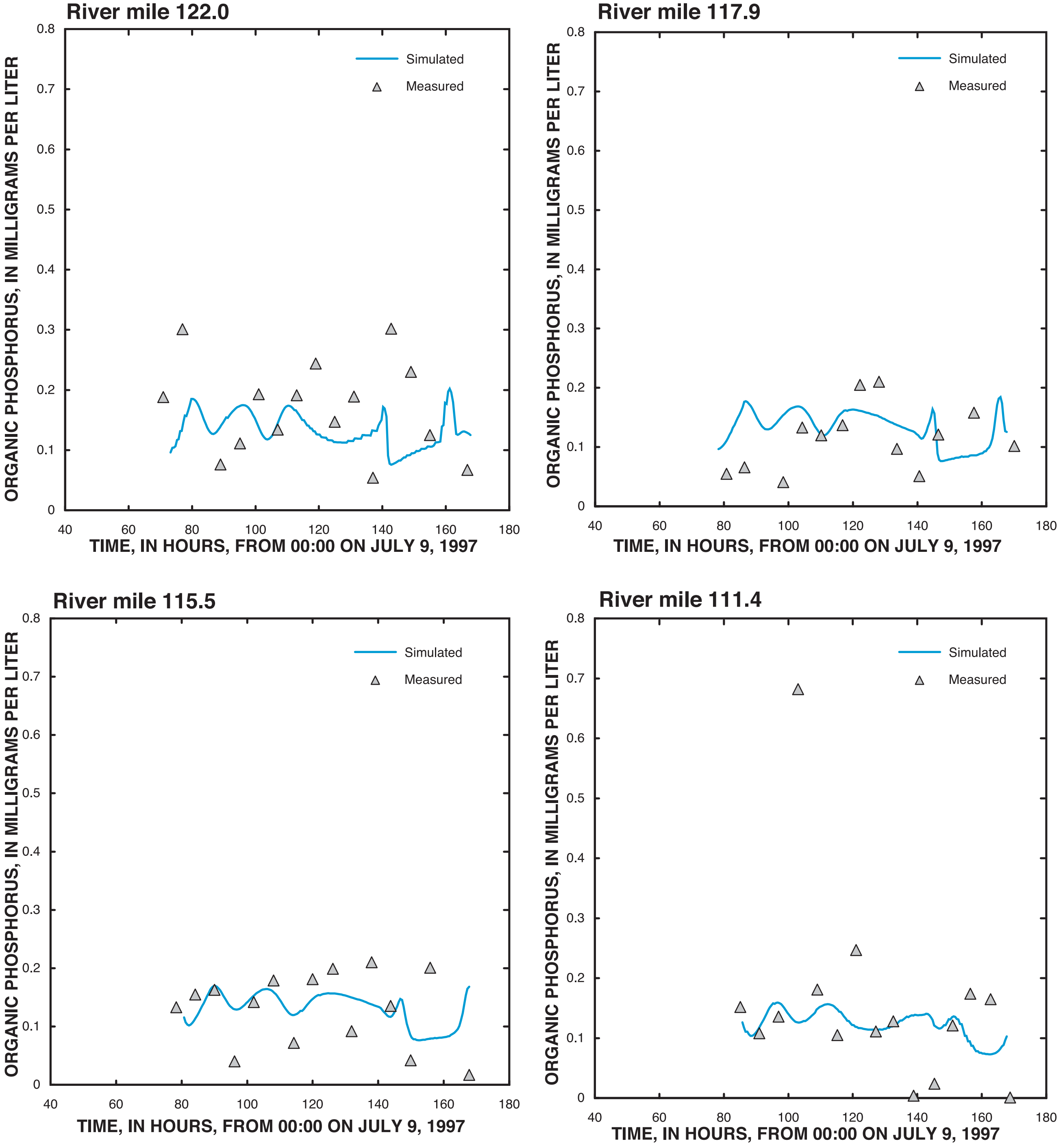
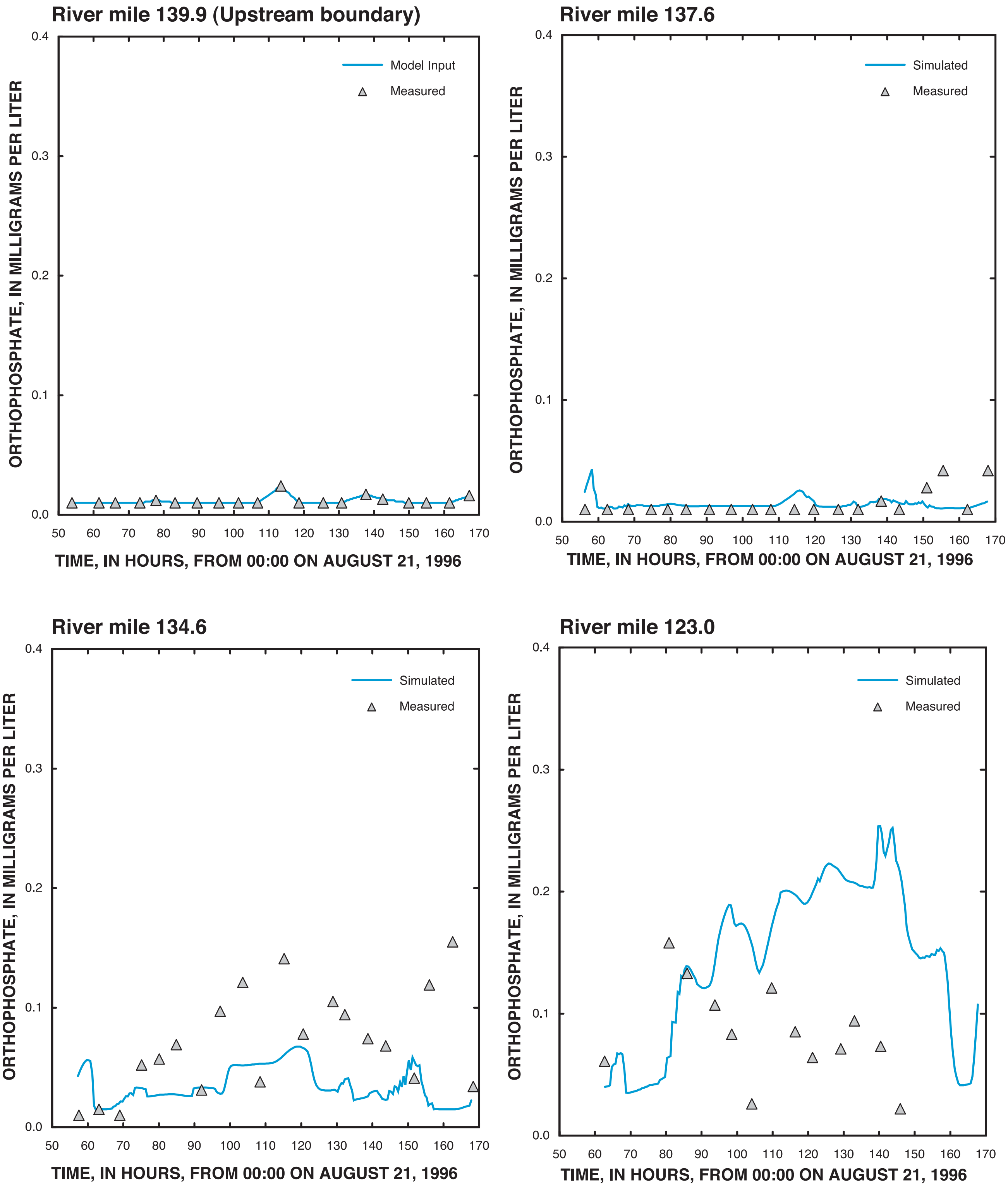


Figure 37 (Continued). Measured and simulated organic phosphorus concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.


River mile 139.9 (Upstream boundary)

River mile 137.6

River mile 134.6

River mile 123.0

Figure 38. Measured and simulated orthophosphate concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

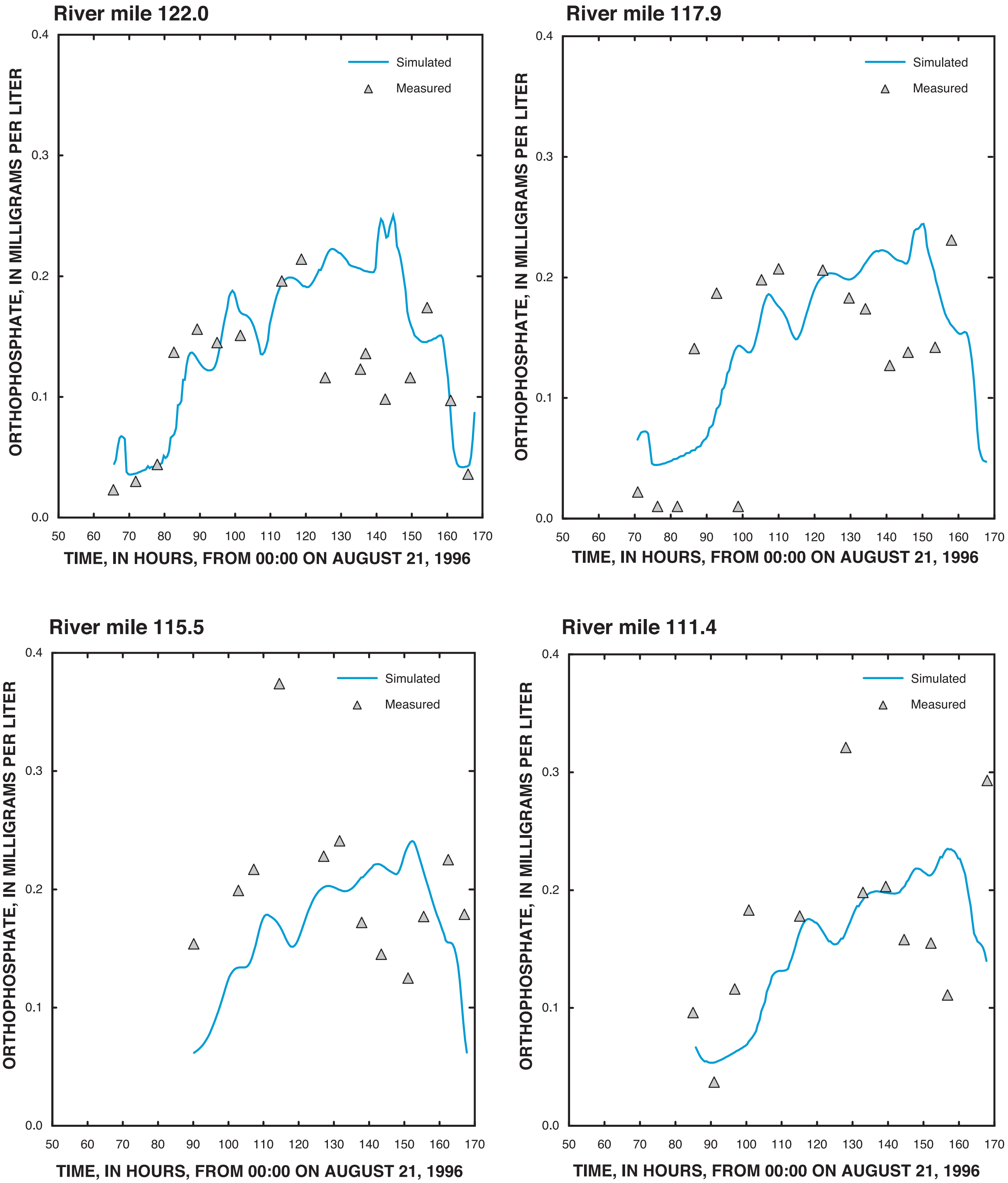


Figure 38 (Continued). Measured and simulated orthophosphate concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

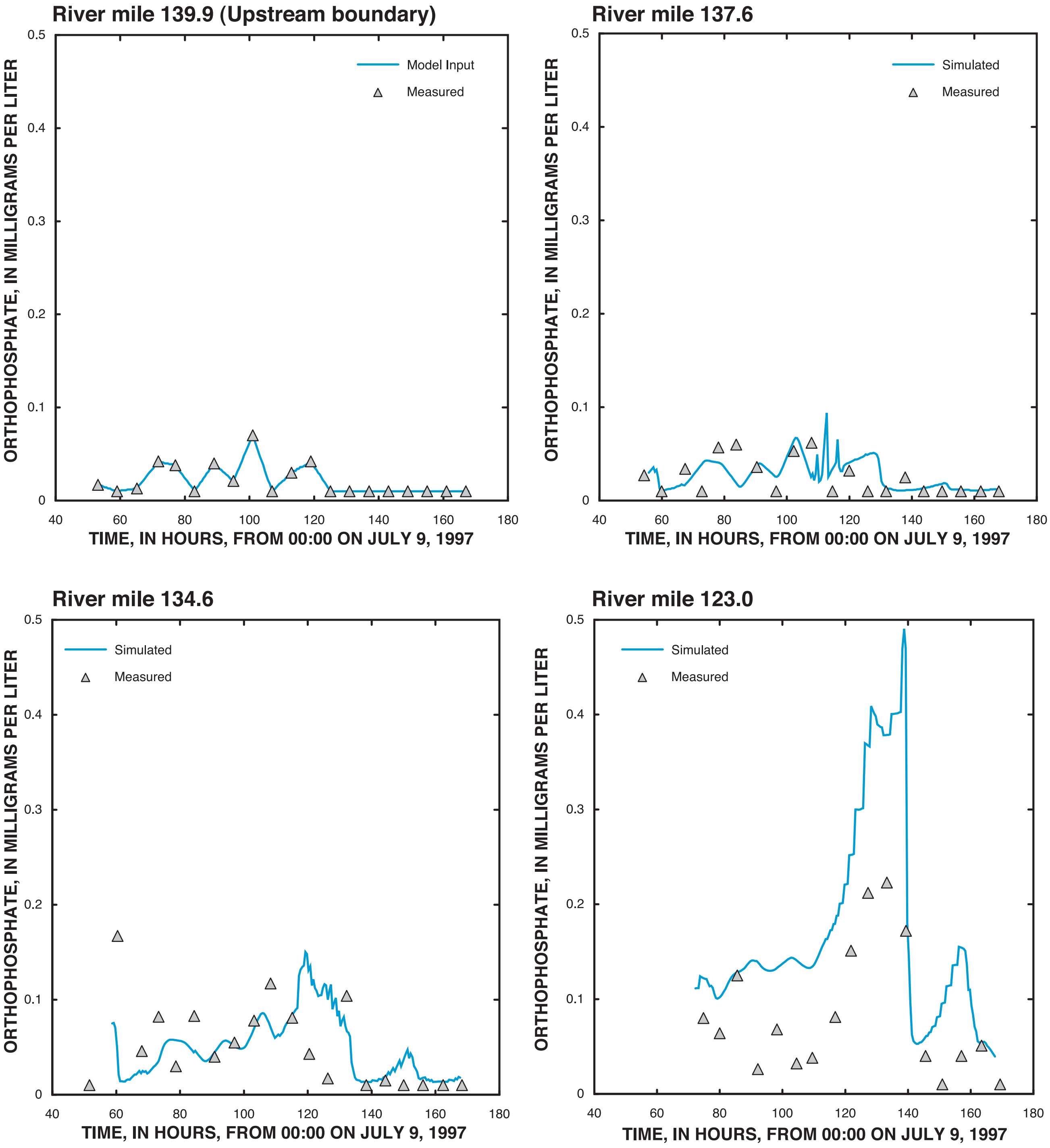


Figure 39. Measured and simulated orthophosphate concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

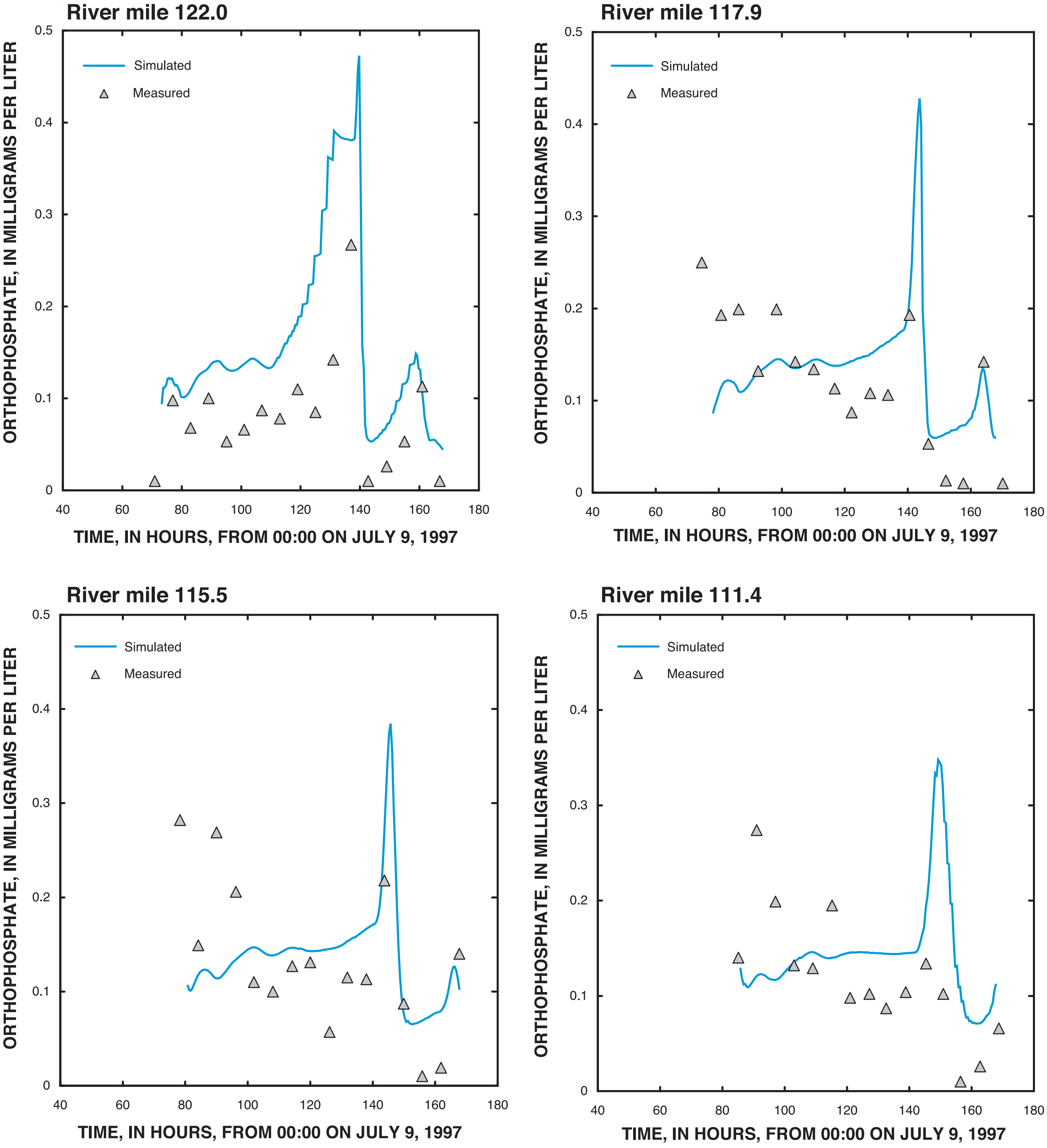
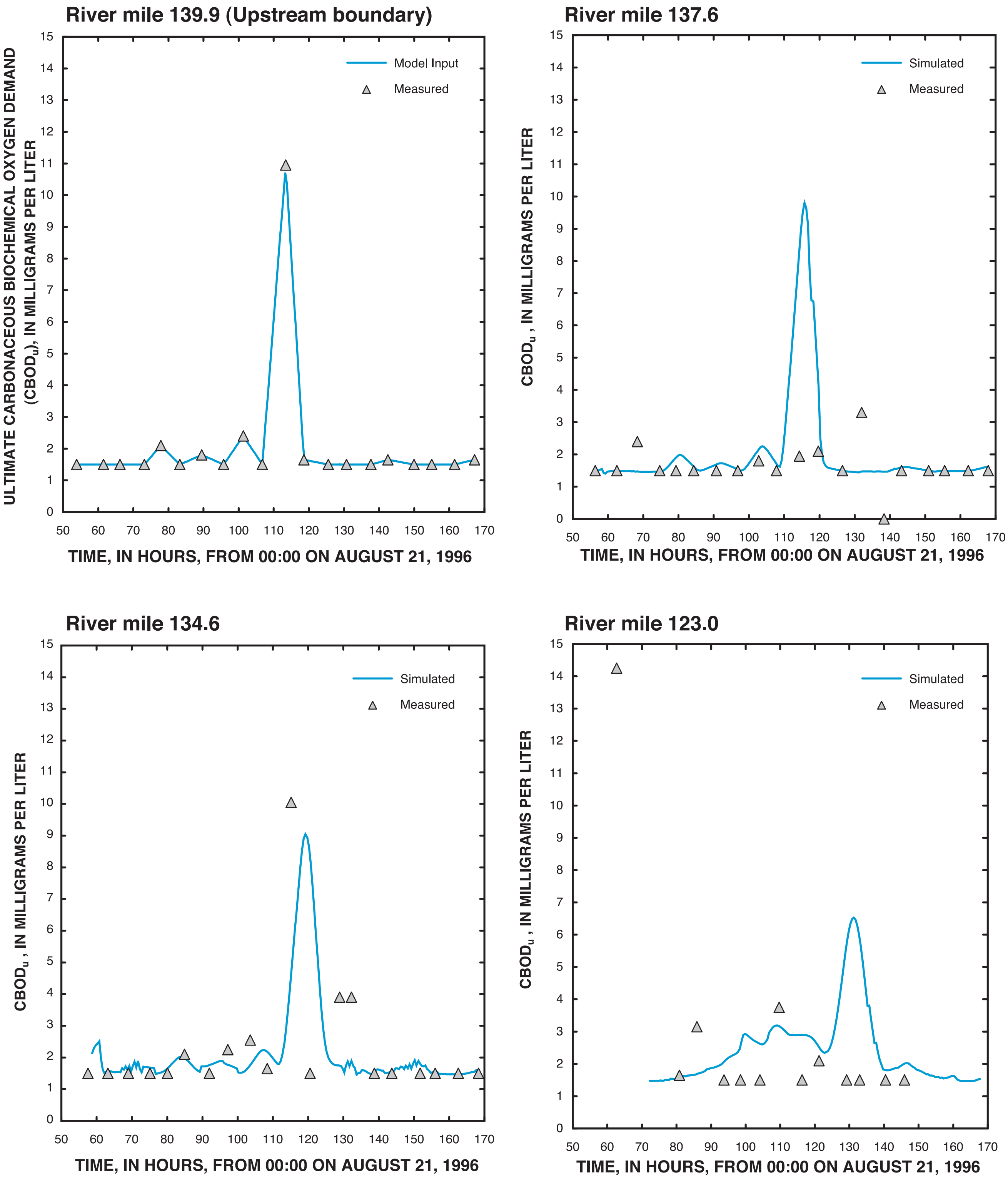


Figure 39 (Continued). Measured and simulated orthophosphate concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

Ultimate carbonaceous biochemical oxygen demand: For the 1996 calibration period, the mean errors for the ultimate carbonaceous biochemical oxygen demand simulations ranged from -0.13 to 1.62 mg/L (table 23). Six of seven sites had positive mean errors, indicating that on average, the simulations overestimated the measured data (table 23; fig. 40). From the graph of the upstream boundary $CBOD_u$ at RM 139.9, the influence of the initial concentrations can be seen. The maximum measured $CBOD_u$ value at the upstream boundary was 10.95 mg/L (fig. 40). As shown in the graphs for the other river miles, the spike is easily definable at every location. For the 1997 validation period, the mean errors ranged from -2.60 to -0.59 mg/L, indicating that on average, the simulations underpredicted the measured data at each site (table 24). As in some of the previous data discussions, the 1997 high measured values relative to simulated values may be the result of analytical error or may indicate that daily mean concentrations from point-source dischargers do not adequately reflect the temporal variations in loadings to the river (fig. 41).

Dissolved oxygen: For the 1996 simulations, the mean errors for simulated DO concentrations ranged from -0.20 to 0.73 mg/L. Five of seven sites had positive mean errors, indicating that on average, the simulations overestimated the measured data (table 23; fig. 42). For the 1997 simulations, the mean errors for simulated DO concentrations ranged from -0.56 to -0.03 mg/L, indicating that on average, the simulations underestimated the measured data (table 24; fig. 43). As shown in tables 23 and 24, the mean measured DO concentrations for the 1996 and 1997 study periods initially increase in the downstream direction and then begin to decrease to a minimum at RM 117.9. The mean simulated DO concentrations follow this same pattern with minimum DO concentration again occurring at RM 117.9 after the initial increase from RM 137.6. River mile 117.9 is at the head of Landsford Rapids where water is pooled to depths of 8 to 12 ft before flowing over a section control and through the rapids. Five wastewater discharges to the study reach and the confluence of Sugar Creek with the Catawba River are located upstream from RM 117.9. The steady-flow water-quality model developed by Davis and Floyd, Inc., (1984) also simulated a minimum DO concentration at RM 117.9.


River mile 139.9 (Upstream boundary)

River mile 137.6

River mile 134.6

River mile 123.0

Figure 40. Measured and simulated ultimate carbonaceous biochemical oxygen demand concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

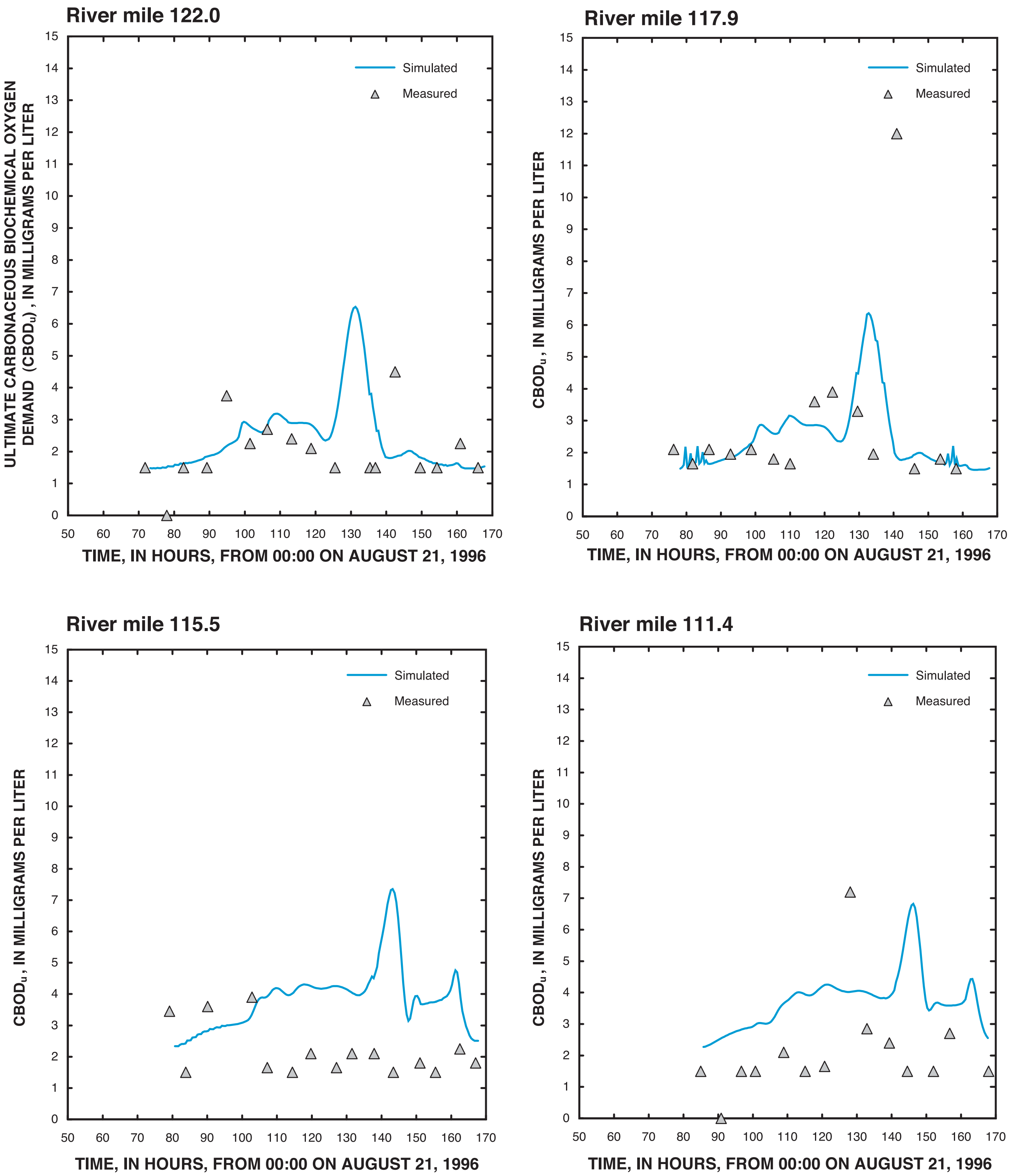


Figure 40 (Continued). Measured and simulated ultimate carbonaceous biochemical oxygen demand concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

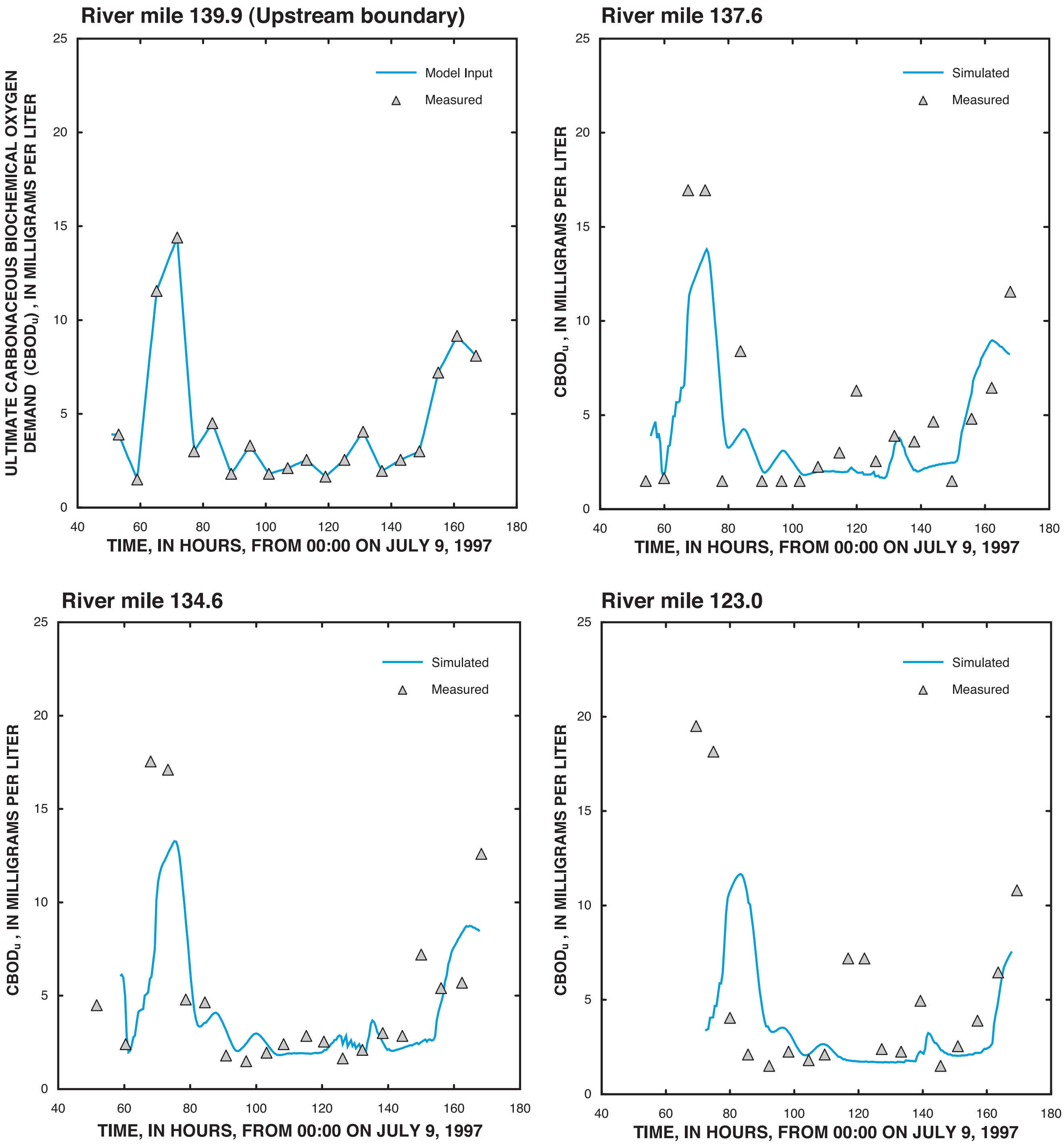
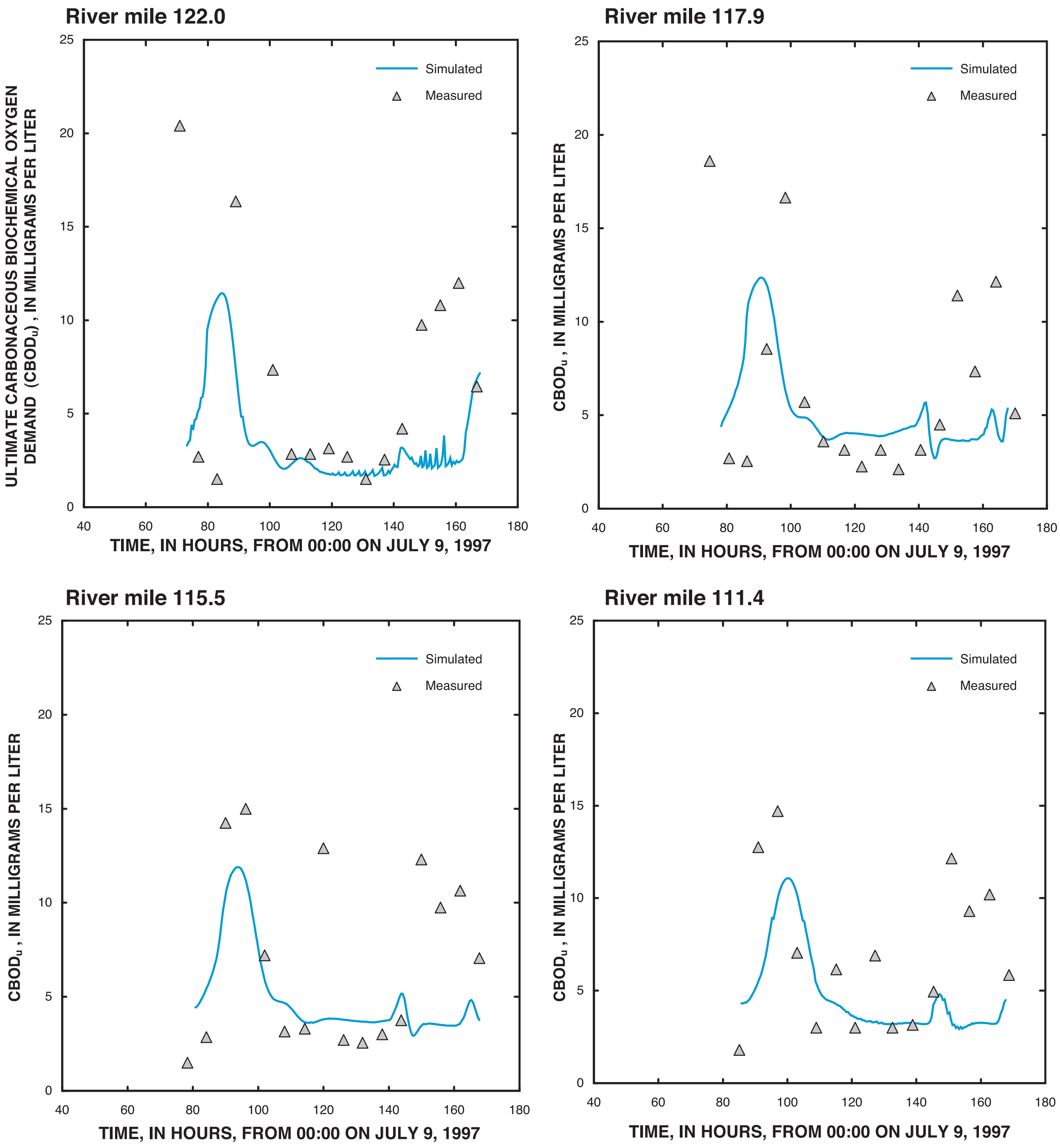


Figure 41. Measured and simulated ultimate carbonaceous biochemical oxygen demand concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.



River mile 122.0

River mile 117.9

River mile 115.5

River mile 111.4

Figure 41 (Continued). Measured and simulated ultimate carbonaceous biochemical oxygen demand concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

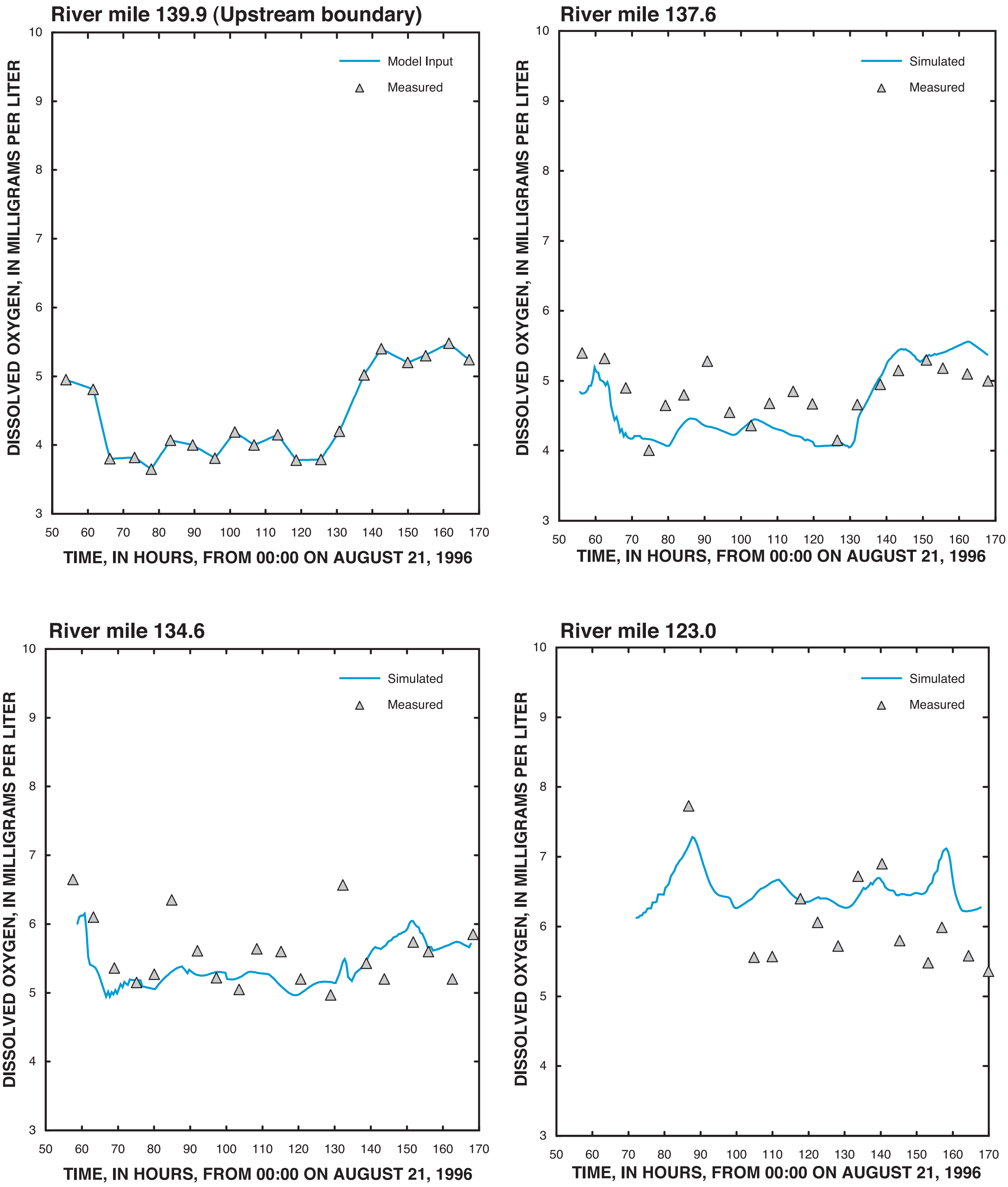


Figure 42. Measured and simulated dissolved-oxygen concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

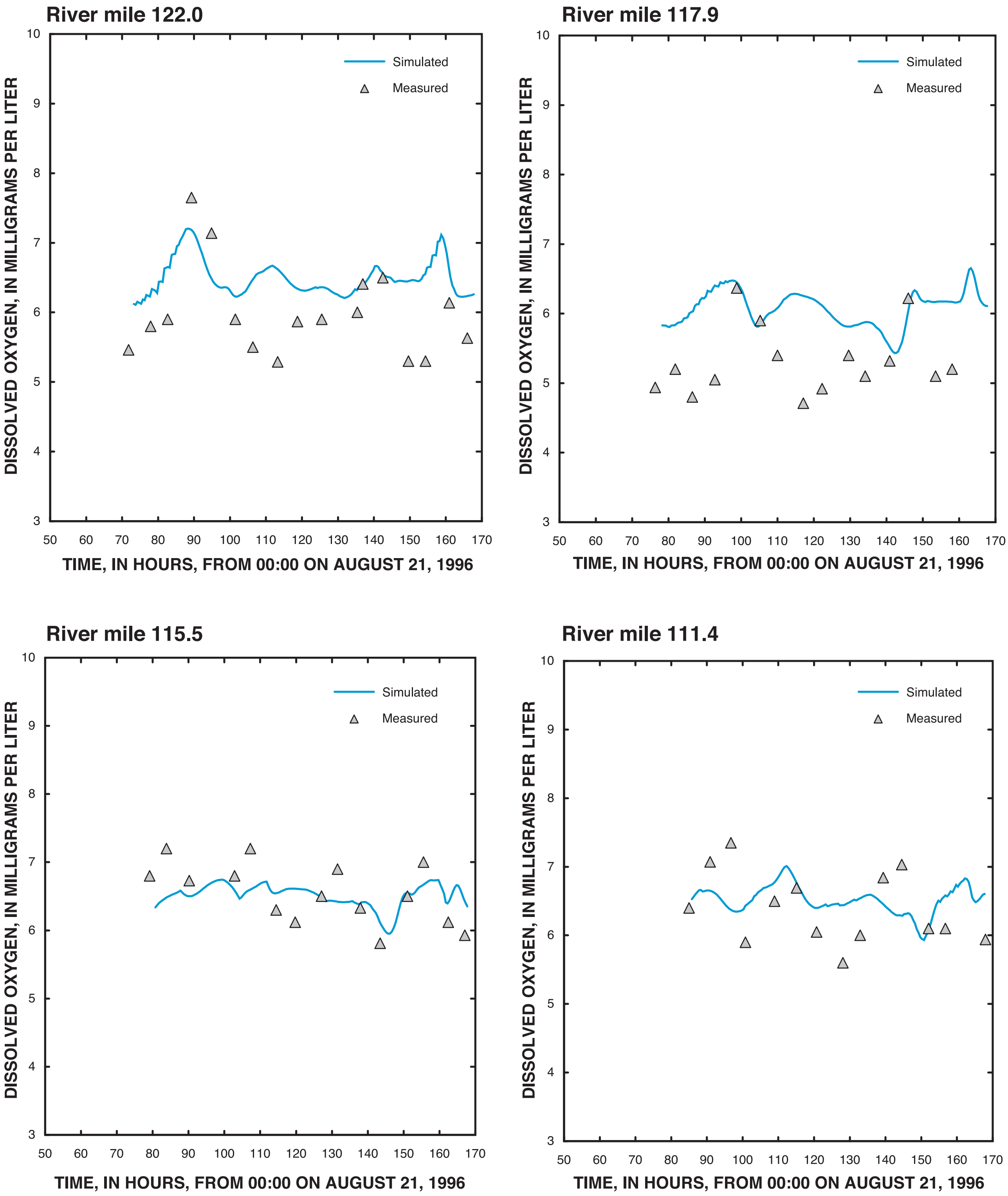


Figure 42 (Continued). Measured and simulated dissolved-oxygen concentrations used in the Branched Lagrangian Transport Model calibration for eight locations on the Catawba River, S.C., August 23–27, 1996.

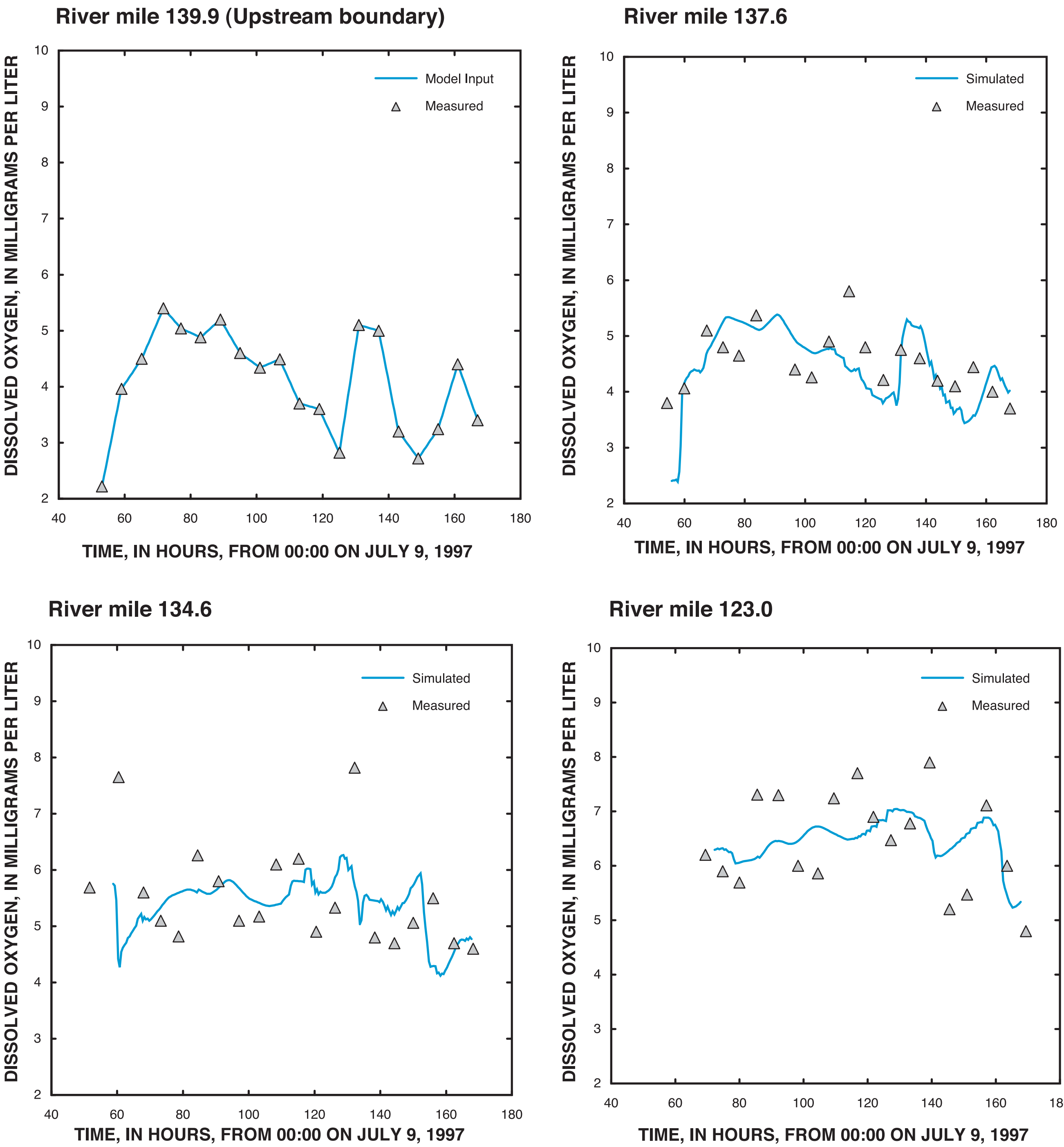


Figure 43. Measured and simulated dissolved-oxygen concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

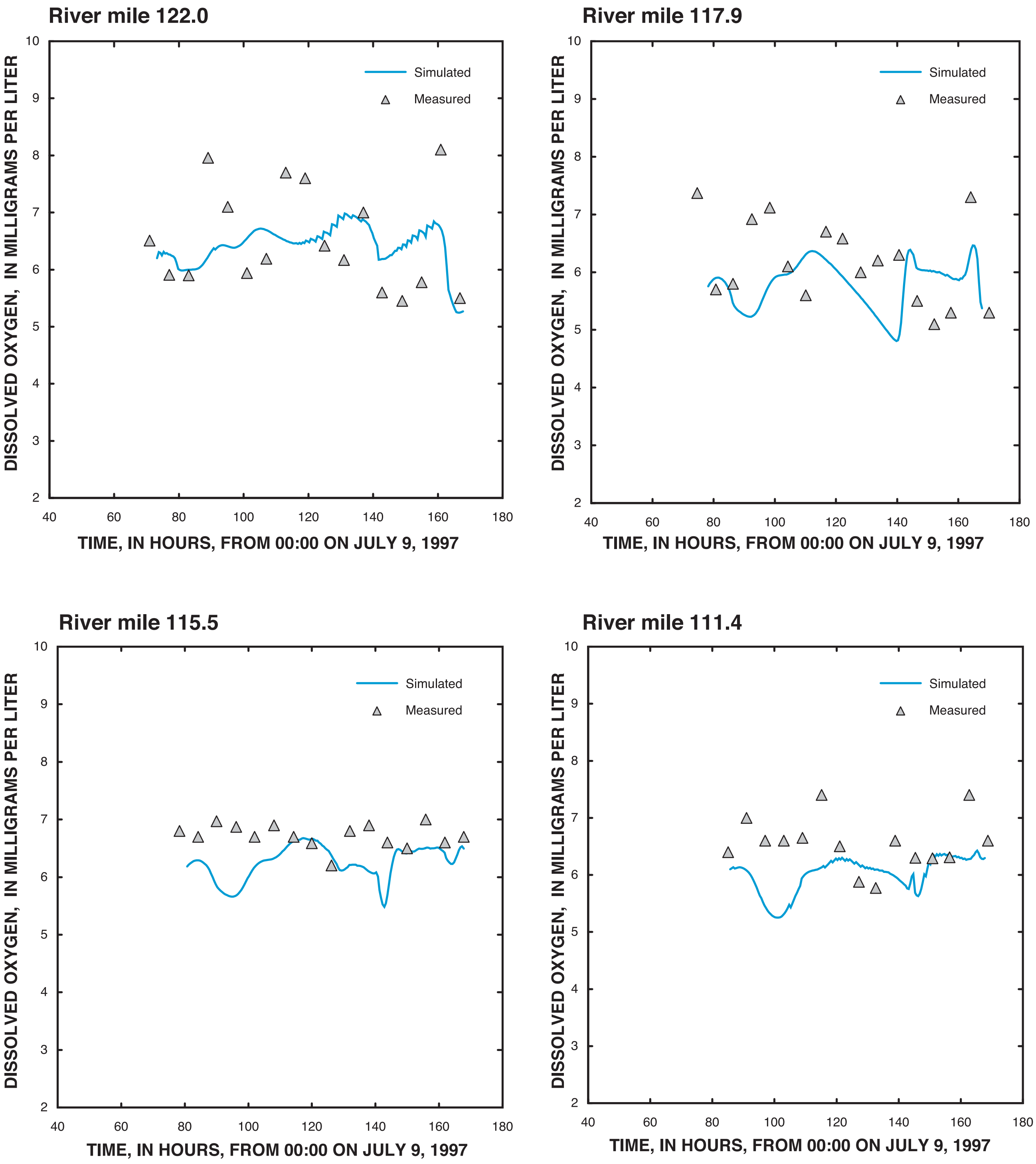


Figure 43 (Continued). Measured and simulated dissolved-oxygen concentrations used in the Branched Lagrangian Transport Model validation for eight locations on the Catawba River, S.C., July 11–15, 1997.

Sensitivity Analysis

A sensitivity analysis was performed to determine the sensitivity of simulated flows, travel times, and constituent concentrations to changes in model parameters and boundary conditions. The analysis was performed by changing one parameter (or set of boundary data) and comparing simulated results to results from the calibrated model. Comparisons were made by using simulations at RM 122.0 for August 23–27, 1996. River mile 122.0 is located downstream from the intervening drainage area from Lake Wylie Dam to Sugar Creek, as well as downstream from the mouths of Sugar and Twelvemile Creeks, but upstream from Waxhaw Creek (fig. 2). This analysis gives an indication of the stability of the model and provides insight into the parameters or data that need to be most carefully determined for model application.

The sensitivity of simulated streamflow to changes in channel cross-sectional area, flow-resistance coefficient, flow from Lake Wylie Dam, and tributary inflows in the BRANCH model was analyzed. The sensitivity of travel time and dispersion to changes in the storage term of the BRANCH model and the dispersion factor in the BLTM also were analyzed. The reasons these particular parameters were selected for the sensitivity analysis are as follows:

- Cross sections measured at discrete locations were used to represent the highly variable geometry of the study reach, so there was a need to evaluate the assumption that these cross sections adequately represented the model domain.
- Flow-resistance coefficients, the dispersion factor, and the storage term cannot be measured directly but are estimated. The sensitivity of simulated flows and transport rates to changes in these parameters was determined.
- Flows from Lake Wylie Dam were adjusted as previously described, and continuous tributary inflows were estimated. Consequently, there is some uncertainty about these boundary data, the effects of which were evaluated in the sensitivity analysis.

The area at all cross sections was increased by 25 percent. The cross-sectional area was then reduced by 25 percent. In the streamflow simulations at RM 122.0, the 25-percent increase in cross-sectional area had no effect on the peak flow after the values were rounded to three significant figures. A 25-percent decrease raised the peak flow by about 1.0 percent. When the streamflow was fairly steady, the effect on the simulated peak flows was minimal (fig. 44).

The flow-resistance coefficient was increased by 25 percent and reduced by 25 and 50 percent. The computations became numerically unstable when the flow-resistance coefficient was reduced by 50 percent. Decreasing the flow-resistance coefficient (smoother channel) by 25 percent increased the peak flow from 10,500 to 10,600 ft^3/s at RM 122.0 (fig. 44), and a 25-percent increase in the flow-resistance coefficient decreased the simulated peak flow to 10,400 ft^3/s. Changes in the flow-resistance coefficient also had a small effect on the timing of the peak flow (fig. 44). The 25-percent reduction in the flow-resistance coefficient resulted in the arrival of the simulated peak flow at RM 122.0 earlier by 0.5 hour than for the calibrated model, and the 25-percent increase in flow-resistance coefficient resulted in a 1.0 hour later arrival of the peak flow at RM 122.0 compared to the calibrated model.

Streamflow from Lake Wylie Dam was increased and decreased by 25 percent. The change in simulated streamflow was almost directly proportional to the increase or decrease in streamflow from Lake Wylie Dam (fig. 44) with the flows increasing and decreasing by approximately 22 percent. BRANCH model simulation results were not very sensitive to changes in tributary flows. The streamflow increase of 25 percent in the tributaries increased the peak streamflow by 3 percent, from 10,500 to 10,800 ft^3/s, at RM 122.0; whereas, a 25-percent decrease in the tributary flows reduced the peak flow to 10,200 ft^3/s, also a 3-percent change (fig. 44). During the steady-flow period on August 24, 1996, the flow increased by 2 percent, from about 2,520 to 2,570 ft^3/s, at RM 122.0 for the 25-percent increase in tributary flows and decreased slightly (1.6 percent) to 2,480 ft^3/s for the 25-percent decrease in flow (fig. 44).

Sensitivity of the transport component simulations to changes in the dispersion factor and storage value was evaluated. For the sensitivity analyses, a hypothetical conservative tracer was injected at RM 139.9, and the response was simulated at RM 122.0. The calibrated dispersion factor of 0.2 was reduced to 0.0, and then increased to 0.4. The change in the dispersion factor affected the peak concentration as well as the leading and trailing edges of the dye cloud. An increased dispersion factor resulted in the earlier arrival of the attenuated dye cloud relative to the calibrated condition (fig. 45A). The peak

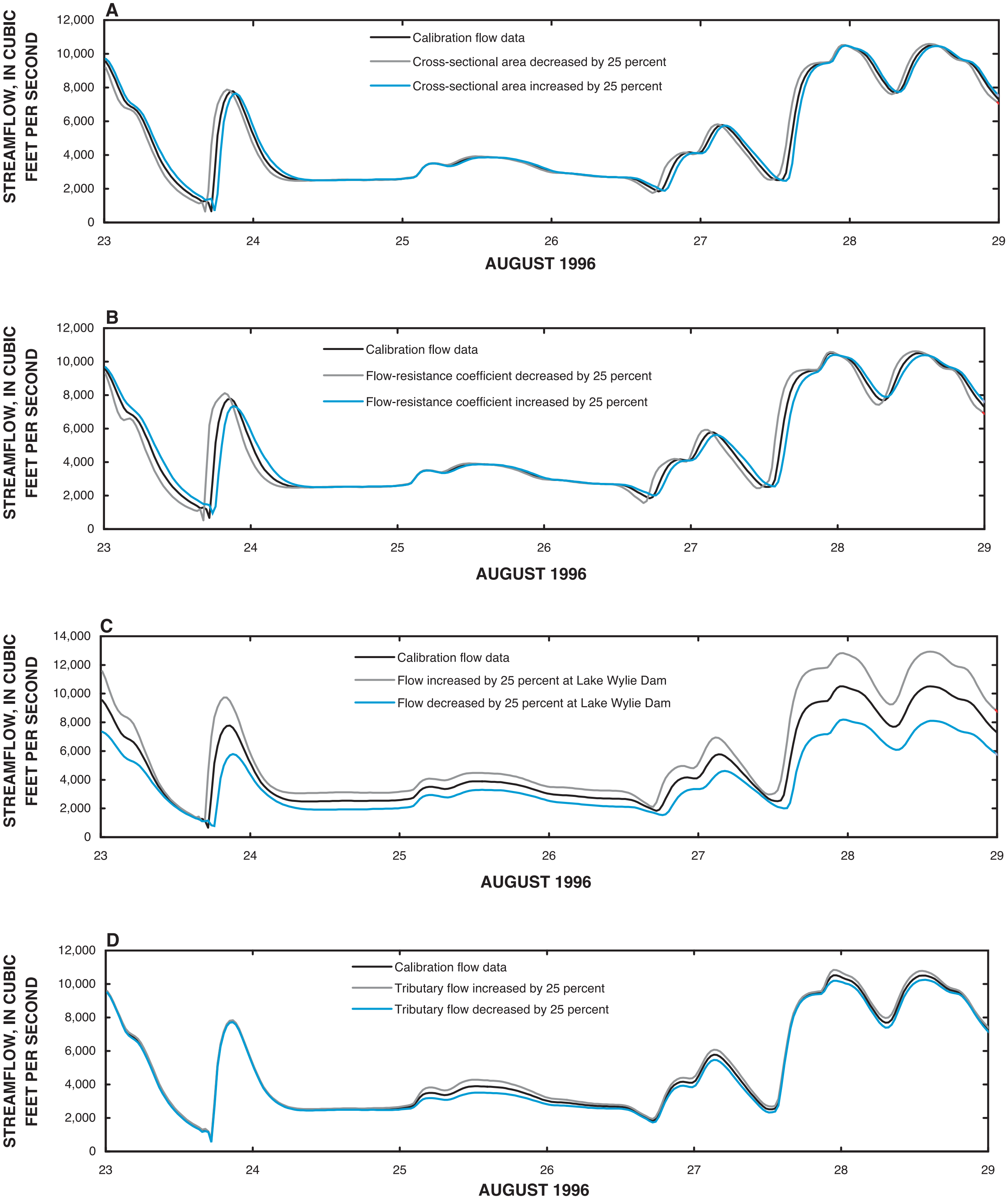


Figure 44. Sensitivity of simulated streamflow at river mile 122.0 on the Catawba River, S.C., to changes in (A) cross-sectional area, (B) flow-resistance coefficient, (C) inflows from Lake Wylie Dam, and (D) inflows from tributary streams, August 23–28, 1996.

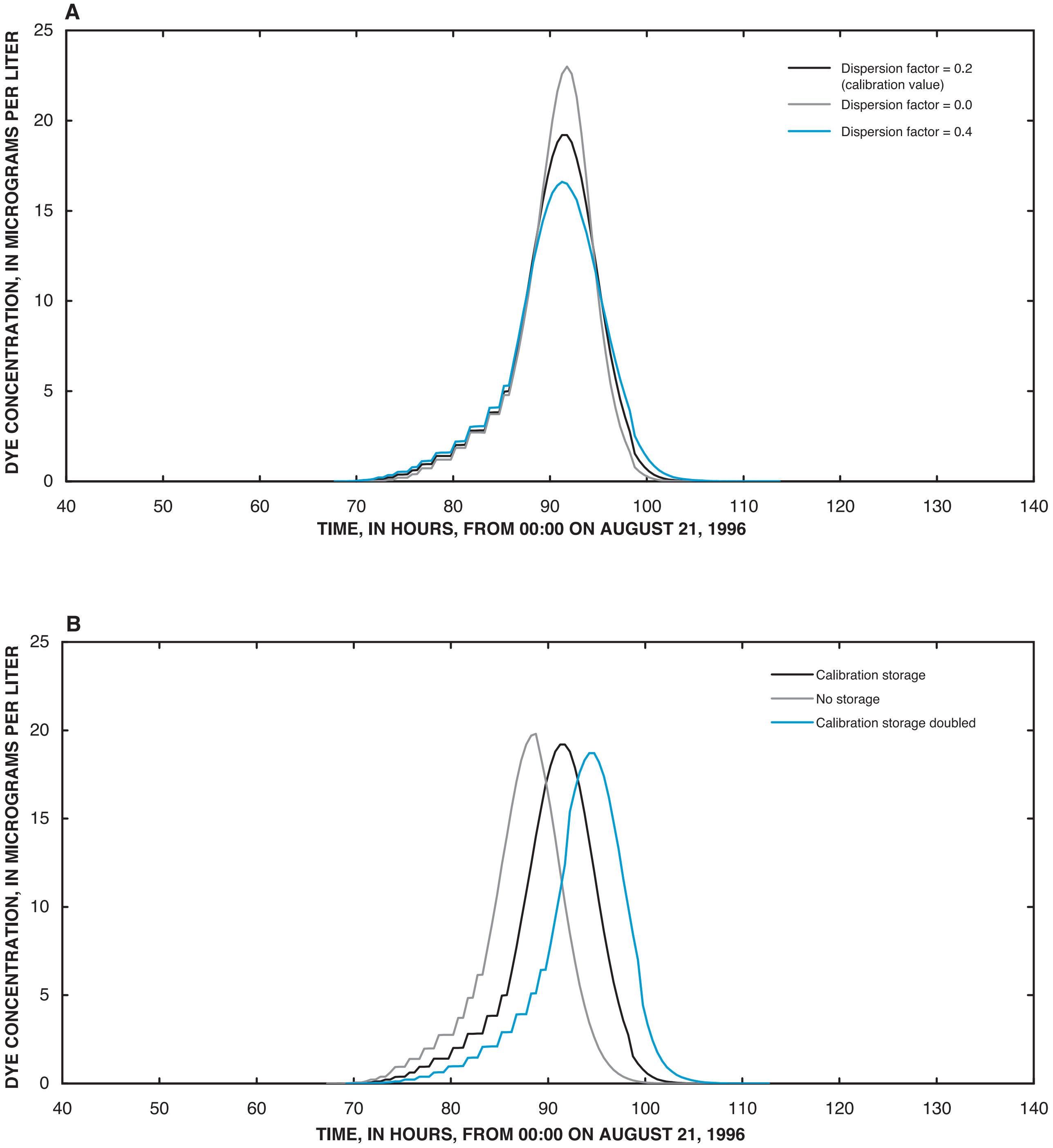


Figure 45. Sensitivity of the Branched Lagrangian Transport Model at river mile 122.0 on the Catawba River, S.C., to changes in (A) dispersion coefficient, and (B) storage values, August 23–27, 1996.

concentration for a dispersion factor of 0.0 was 20 percent greater than for the calibrated condition, with a dispersion factor of 0.2, and the peak concentration was 14 percent lower than the calibrated condition for a dispersion factor of 0.4. When compared to the calibration model dispersion factor of 0.2, the time of travel of the peak concentration did not change for the dispersion factor of 0.4, but did increase by 0.5 hour for the dispersion factor of 0.0.

As previously discussed, a storage area was added to each channel cross section in the BRANCH model in order to achieve calibration of the transport component. All storage values were below the minimum water level as described earlier in this text. In order to assess the sensitivity of the transport model to these storage values, the simulation using the calibrated model storage values was used to compare with a simulation that included zero storage and a second simulation with the calibrated model storage areas doubled. For the simulation with zero storage, the peak concentration increased slightly from 19.2 to 19.8 mg/L and arrived 2.5 hours earlier (fig. 45B). When the storage areas were doubled, the peak concentration decreased slightly from 19.2 to 18.7 mg/L and arrived 3.0 hours later. As the results indicate, the storage areas do not appreciably affect the peak concentrations, but do have a more pronounced effect on the time of travel.

The sensitivity of the water-quality simulations was analyzed by simulating the daily mean DO concentration at RM 122.0 on August 24, 1996, for various parameters and input values, comparing these values with the calibrated model value of 6.61 mg/L, and computing a normalized sensitivity index. Three groups of model inputs were evaluated—rate constants (table 22), meteorological conditions, and water-quality boundary conditions. For each simulation, one parameter (or set of boundary data) was increased approximately 35 percent relative to the calibrated model value. Time-dependent inputs (boundary conditions), such as meteorological input data and boundary constituent concentrations, were changed for each simulation timestep. In addition, the sensitivity of simulated DO concentrations at RM 122.0 to flow conditions was evaluated by using the validation model flows. Substituting the calibration model flows with the validation model flows increased the 24-hour mean upstream inflow for August 24, 1996, by about 79 percent (from 2,300 to 4,110 ft^3/s).

A normalized sensitivity index (S_{ij}) was calculated to indicate the percentage of change in the output variable (the simulated mean DO concentration at RM 122.0 for August 24, 1996) resulting from a 1-percent change in the rate constant or boundary condition (Brown and Barnwell, 1987). The normalized sensitivity index is defined as

$$S_{ij} = (DY_j / Y_j) / (DX_i / X_i) \quad , \qquad (14)$$

where,

S_{ij} is the normalized sensitivity index for output Y_j to input X_i,
DY_j is the change in the output variable,
Y_j is the original value of the output variable,
DX_i is the change in the input variable, and
X_i is the original value of the input variable.

A higher value (positive or negative) of S_{ij} indicates a greater sensitivity of the simulated value to changes in the input value.

Simulated DO concentrations at RM 122.0 were unaffected by changes in the following parameters: A1, ALGSET, ALPH1, ALPH2, ALPH6, B1, BET3, BET4, CKL, CKP, PN, SHAD1, SIG2, SIG3, SIG4, SIG5, SIG6, and SIG7 (table 25). In addition, changes in solar radiation, wind velocity, organic nitrogen boundary data, organic-phosphorus boundary data, and orthophosphate boundary data did not affect simulated DO concentrations at RM 122.0. For the rate coefficients, simulated DO concentrations at RM 122.0 were most sensitive to BET2, which is the biological oxidation rate of nitrite nitrogen to nitrate nitrogen (tables 22, 25). Of the chemical boundary conditions, DO simulations were most sensitive to water temperature and DO. Following the sensitivity to DO boundary data, simulations were somewhat sensitive to change in the upstream streamflow boundary data. For the meteorological boundary conditions, DO simulations were sensitive only to the equilibrium temperature.

As previously discussed, the atmospheric reaeration equation chosen for the Catawba River water-quality model was the Langbien and Durum (1967) equation. The model was run with three other potential reaeration equations, and the mean simulated DO for August 24, 1996, at RM 122.0 was compared with the base value of 6.61 mg/L (table 25). Using the O'Connor and Dobbins (1958) equation resulted in a mean DO concentration of 7.17 mg/L, an 8-percent

Table 25. Sensitivity indices for the Branched Lagrangian Transport Model inputs for river mile 122.0 on the Catawba River, S.C., August 24, 1996

[mg/L, milligram per liter; —, no data]

Parameter or input value	Original value	Test value	Mean simulated dissolved oxygen (mg/L)	Sensitivity index
Base value[a]	—	—	6.61	—
Rate coefficients (table 22)				
A1	1.000	1.350	6.61	0.00
ALGSET	0.200	0.270	6.61	.00
ALPH1	.070	.094	6.61	.00
ALPH2	.010	.014	6.61	.00
ALPH3	1.800	2.430	6.71	.04
ALPH4	1.600	2.160	6.58	-.01
ALPH5	3.450	4.660	6.56	-.02
ALPH6	1.140	1.539	6.60	.00
B1	1.130	1.530	6.61	.00
BET1	.400	.540	6.57	-.02
BET2	2.000	2.700	6.19	-.18
BET3	.100	.135	6.61	.00
BET4	.020	.027	6.61	.00
CK1	.180	.243	6.57	-.02
CK4	75.0	101.2	6.49	-.05
CKL	.020	.027	6.60	.00
CKN	.260	.351	6.58	-.01
CKP	.040	.054	6.60	.00
GRO	2.500	3.375	6.74	.06
PN	.250	.338	6.61	.00
RSPRT	.150	.202	6.57	-.02
SHAD0	.700	.945	6.58	-.01
SHAD1	.002	.003	6.60	.00
SIG2	2.000	2.700	6.61	.00
SIG3	2.000	2.700	6.61	.00
SIG4	.100	.135	6.61	.00
SIG5	.100	.135	6.61	.00
SIG6	1.000	1.350	6.61	.00
SIG7	.100	.135	6.61	.00
Meteorological data				
Equilibrium temperature[b]	1.000	1.350	6.39	-0.10
Solar radiation[b]	1.000	1.350	6.62	.00
Wind velocity[b]	1.000	1.350	6.61	.00
Constituent concentration data				
Algal biomass[b]	1.000	1.350	6.67	0.03
Ammonia[b]	1.000	1.350	6.57	-.02
$CBOD_u$[b]	1.000	1.350	6.56	-.02
Dissolved oxygen[b]	1.000	1.350	6.98	.16
Nitrate[b]	1.000	1.350	6.63	.01
Organic nitrogen[b]	1.000	1.350	6.61	.00
Organic phosphorus[b]	1.000	1.350	6.61	.00
Orthophosphate[b]	1.000	1.350	6.62	.00
Water temperature[b]	1.000	1.350	5.93	-0.29
Upstream inflow				
Streamflow[b]	1.000	1.790	5.93	-0.13

[a] Simulated mean dissolved-oxygen concentrations for August 24, 1996.

[b] Time-dependent model input; original and test values are multiplicative factors.

increase. Using the Churchill and others (1962) equation resulted in a mean DO concentration of 6.53 mg/L, a 1-percent decrease. Using the Owens and others (1964) equation resulted in a mean DO concentration of 6.86 mg/L, a 4-percent increase.

WATER-QUALITY MODEL APPLICATIONS

A water-quality model can be used by water-resource managers to evaluate the effect of wastewater loads on DO concentrations, especially in determining the amount of wastewater that a water body is able to assimilate. The assimilative capacity of a stream is its capacity to carry a particular pollutant without violating an instream water-quality standard. The capacity of a stream to assimilate oxygen-consuming substances is a function of many factors, including streamflow, water temperature, reaeration, benthic oxygen demand, and channel geometry. In terms of water-resource management, this capacity is expressed as pounds per day of ultimate oxygen demand (UOD) that can be assimilated without violating the State water-quality standard for DO.

Wastewater effluent contains many oxygen-consuming constituents, primarily ammonia and biodegradable organic substances. The UOD is the total, theoretical demand for oxygen from carbonaceous and nitrogenous sources. The SCDHEC defines the UOD by the following equation (Conrads, 1998):

$$UOD = (BOD_5 \times F_{ratio} \times NH_3\text{-}N \times 4.57) \times \text{Flow} \times 8.34, \qquad (15)$$

where,

UOD is the ultimate oxygen demand, in pounds per day;

BOD_5 is the 5-day carbonaceous biochemical oxygen demand, in milligrams per liter;

F_{ratio} is the conversion factor from BOD_5 to ultimate carbonaceous biochemical oxygen demand;

$NH_3\text{-}N$ is the ammonia concentration, in milligrams of nitrogen per liter;

4.57 is the stoichiometric ratio of the milligram of oxygen consumed per milligram of ammonia nitrogen oxidized;

Flow is wastewater-effluent flow, in million gallons per day; and

8.34 is the conversion factor to pounds per day.

The procedure for determining the assimilative capacity of an upland, unregulated stream is well established. The procedure involves a statistically computed steady-state, low-flow value, often referred to as the critical flow, that is used in conjunction with a critical water temperature in a simulation model. The results are interpreted according to a State water-quality standard.

There are, however, important issues to consider pertaining to the application of these procedures to the Catawba River. One issue is how to determine a critical flow on a regulated stream. Statistically computed low-flow values for unregulated streams represent flow conditions responding to the climatic conditions. Statistically computed low-flow values for regulated streams represent flow conditions responding to the management of the flow-control structure. Change in the operation of the flow-control structure changes the statistically computed flow values.

Another issue to address when determining the assimilative capacity is the question of what is the natural condition of waters that are downstream from manmade impoundments. Water-quality conditions in the upper reaches of a river, such as the Catawba River, are dependent on the water-quality conditions of the hypolimnetic releases from the Lake Wylie Dam. These conditions represent a shift from water-quality conditions based on riverine processes to conditions based on lacustrian processes. Resolving these issues is beyond the scope of this report. However, defensible determination of the assimilative capacity will require that these issues be addressed.

The BLTM water-quality model of the Catawba River was used to simulate three water-quality scenarios to evaluate their effects on the simulated longitudinal DO concentrations of the river and on phosphorus loadings to Fishing Creek Reservoir and, ultimately, to gain a better understanding of the river system. The results from these scenarios are intended to demonstrate the utility of the model in making water-resource management decisions and are not intended as a regulatory application of the model. The August 23–27, 1996, calibration data set was used for the scenario simulations. Simulated model output was analyzed at 21 locations selected to adequately define the longitudinal DO profile. Three types of scenarios were simulated. The first scenario simulated the effect of varying point-source loads. The second scenario evaluated the effect of an additional point-source load to the Catawba River. The third scenario evaluated the phosphorus loading to Fishing Creek Reservoir for various loading scenarios.

Although the effects of algae are often difficult to represent in dynamic modeling (Butcher, 1998), the algal component was included in the calibration and validation of the Catawba River model in an effort to accurately calibrate to the synoptically sampled data. The chlorophyll *a* samples collected in 1996 and 1997 indicate the variability in the sampled data and the difficulty in quantifying the algal population in the Catawba River. Butcher (1998) suggests that dynamic models be calibrated and validated by including the algal component, and ran with and without the algal component for analysis of various water-resource scenarios. For the purposes of this report, the scenarios were simulated only with the algal component.

Variation in Point-Source Loading

For this report, various point-source loading conditions were compared with a condition in which there were no point-source discharges into the system (a no-load condition). The effects of the point-source loadings were evaluated by comparing the differences in the DO concentrations for each simulation. The scenarios indicate how the model can be used to compare relative differences between various point-source loading conditions rather than to predict the absolute DO concentration of the system for a selected point-source loading, hydrological, and meteorological condition. The modeled absolute value may be in error, but relative differences in the simulated results are likely to be more accurate.

Simulations were compared for three different loading conditions from the effluent discharges—no effluent discharges (no-load condition), effluent levels measured during the calibration period (current condition), and effluent loads set to their fully permitted conditions (fully loaded condition). Because of the hydrograph control release (HCR) permit for one discharger, the fully loaded condition represents flow conditions during the calibration period and not the maximum loading conditions that occur during maximum flows from Lake Wylie. Permit limits by the SCDHEC for point-source discharges are for flow, and BOD_5, ammonia nitrogen, and DO concentrations. Constituents that are not listed in SCDHEC permits were assigned a value. Nitrate nitrogen values that were used were one-half of the ammonia nitrogen value. Orthophosphate and organic-phosphorus concentrations for the wastewater dischargers were set to the computed mean concentrations from sampling conducted during August 23–27, 1996. Organic nitrogen and nitrite nitrogen concentrations were set to zero. The water temperature of 27 °C was used to closely agree with water temperatures of the receiving streams, and an algal concentration of 0.0 mg/L was used. The SCDHEC NPDES permit levels for the six major dischargers are listed in table 3.

The different loading scenarios were analyzed using the simulated streamflow from the calibration period. For the current (calibration) conditions, the mean UOD for the dischargers was about 32,700 pounds per day, or 27 percent of the permitted level (about 122,000 pounds per day; table 26). When compared to the no-load simulation, the largest effect

Table 26. Comparison of the ultimate oxygen demand from wastewater effluent discharges on the Catawba River, S.C., for August 24, 1996, with the National Pollutant Discharge Elimination System permit levels

[UOD, ultimate oxygen demand; lbs/d, pounds per day; NPDES, National Pollutant Discharge Elimination System; BOD_5, 5-day biochemical oxygen demand; mg/L, milligram per liter; NH_3, ammonia nitrogen]

Wastewater effluent discharger (fig. 2)	Calibration period UOD (lbs/d)	NPDES permit UOD (lbs/d)	Percent of permitted UOD discharged to river during calibration period
A	248	2,522	9.8
B	122	1,710	7.1
C	2.8	73.5	3.9
D	2,520	15,500	16.3
E	26,300	91,900[a]	28.6
F	3,500	10,177	34.4
Total	32,693	121,883	26.8

[a] This discharger has a hydrograph control release permit; therefore, the UOD was computed using the mean streamflow for August 24, 1996, at station 02147020 and assuming a BOD_5 concentration of 60 mg/L and an NH_3 concentration of 5 mg/L.

of the calibration simulation was at RM 111.4, where the 24-hour minimum DO concentration decreased 0.48 mg/L, and the 24-hour mean DO concentration decreased 0.33 mg/L. The lowest DO concentrations in the reach occurred at RM 117.9, just upstream from the Landsford Rapids. As compared to the no-load condition, the 24-hour minimum and mean DO concentrations for the current conditions decreased 0.18 and 0.24 mg/L, respectively (table 27; fig. 46). For the fully loaded conditions, when compared to the no-load condition, the effect at site RM 117.9 decreased the 24-hour mean DO by 0.69 mg/L and the 24-hour minimum decreased 0.74 mg/L. Once again, the largest decrease occurred at RM 111.4, where the 24-hour minimum DO concentration decreased 1.04 mg/L, and the 24-hour mean DO concentration decreased 0.88 mg/L (table 27).

Table 27. Simulated 24-hour mean and minimum dissolved-oxygen concentrations for three point-source loading conditions at 21 locations on the Catawba River, S.C.

[DO, dissolved oxygen; mg/L, milligram per liter]

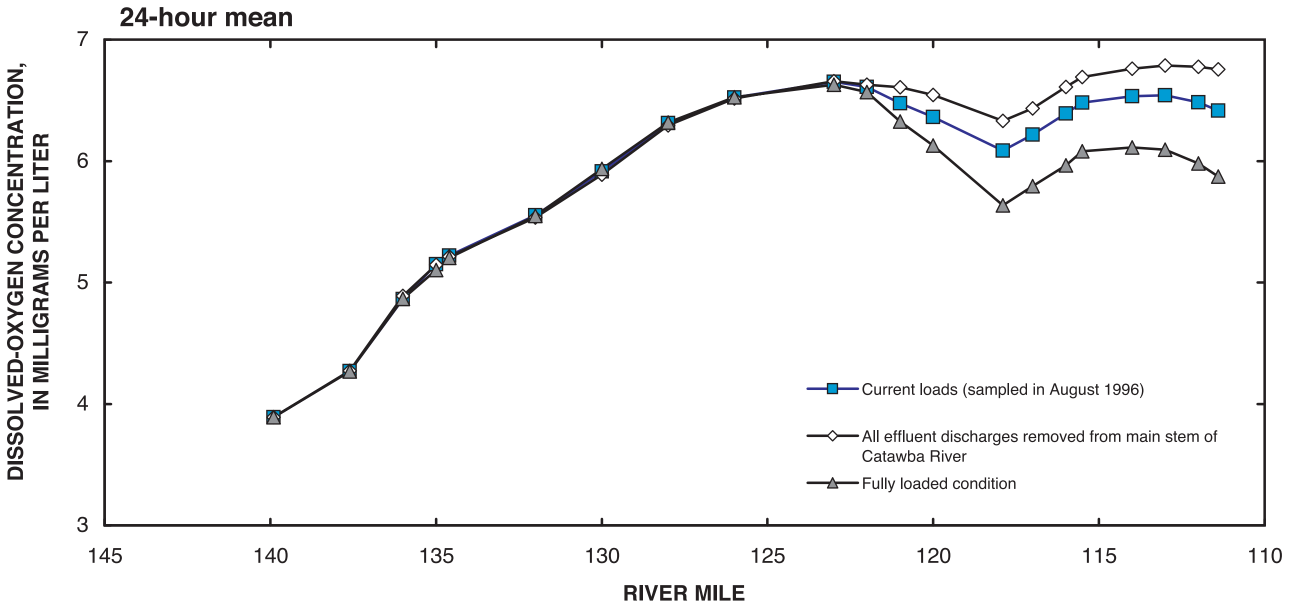
Location, in river miles (fig. 2)	No-load conditions	Calibration period conditions		Fully permitted conditions	
	DO, in mg/L	DO, in mg/L	Change from no-load condition	DO, in mg/L	Change from no-load condition
August 24, 1996, 24-hour mean DO concentrations					
139.9[a]	3.89	3.89	0.00	3.89	0.00
137.6[a]	4.27	4.27	.00	4.27	.00
136.0	4.89	4.86	-.03	4.86	-.03
135.0	5.15	5.15	.00	5.10	-.05
134.6[a]	5.21	5.22	.01	5.20	-.01
132.0	5.54	5.55	.01	5.55	.01
130.0	5.89	5.91	.02	5.93	.04
128.0	6.29	6.31	.02	6.32	.03
126.0	6.51	6.52	.01	6.53	.02
123.0[a]	6.66	6.65	-.01	6.63	-.03
122.0[a]	6.63	6.61	-.02	6.57	-.06
121.0	6.61	6.48	-.13	6.33	-.28
120.0	6.54	6.36	-.18	6.13	-.41
117.9[a]	6.33	6.09	-.24	5.64	-.69
117.0	6.43	6.22	-.21	5.79	-.64
116.0	6.61	6.39	-.22	5.96	-.65
115.5[a]	6.69	6.48	-.21	6.08	-.61
114.0	6.76	6.53	-.23	6.11	-.65
113.0	6.79	6.54	-.25	6.09	-.70
112.0	6.77	6.48	-.29	5.98	-.79
111.4[a]	6.75	6.42	-.33	5.87	-.88

Table 27. Simulated 24-hour mean and minimum dissolved-oxygen concentrations for three point-source loading conditions at 21 locations on the Catawba River, S.C.—Continued

[DO, dissolved oxygen; mg/L, milligram per liter]

Location, in river miles (fig. 2)	No-load conditions	Calibration period conditions		Fully permitted conditions	
	DO, in mg/L	DO, in mg/L	Change from no-load condition	DO, in mg/L	Change from no-load condition
August 24, 1996, 24-hour minimum DO concentrations					
139.9	3.65	3.65	0.00	3.65	0.00
137.6[a]	4.07	4.07	.00	4.07	.00
136.0	4.70	4.70	.00	4.70	.00
135.0	4.96	4.96	.00	4.93	-.03
134.6[a]	5.05	5.05	.00	5.02	-.03
132.0	5.31	5.29	-.02	5.25	-.06
130.0	5.45	5.43	-.02	5.43	-.02
128.0	5.81	5.80	-.01	5.78	-.03
126.0	6.03	6.02	-.01	6.00	-.03
123.0[a]	6.14	6.13	-.01	6.10	-.04
122.0[a]	6.11	6.09	-.02	6.05	-.06
121.0	6.11	6.02	-.09	5.81	-.30
120.0	6.09	5.96	-.13	5.66	-.43
117.9[a]	5.99	5.81	-.18	5.25	-.74
117.0	6.02	5.84	-.18	5.27	-.75
116.0	6.36	6.19	-.17	5.67	-.69
115.5[a]	6.47	6.31	-.16	5.81	-.66
114.0	6.59	6.29	-.30	5.81	-.78
113.0	6.66	6.31	-.35	5.73	-.93
112.0	6.68	6.26	-.42	5.65	-1.03
111.4[a]	6.64	6.16	-.48	5.60	-1.04

[a] Water-quality sampling location.



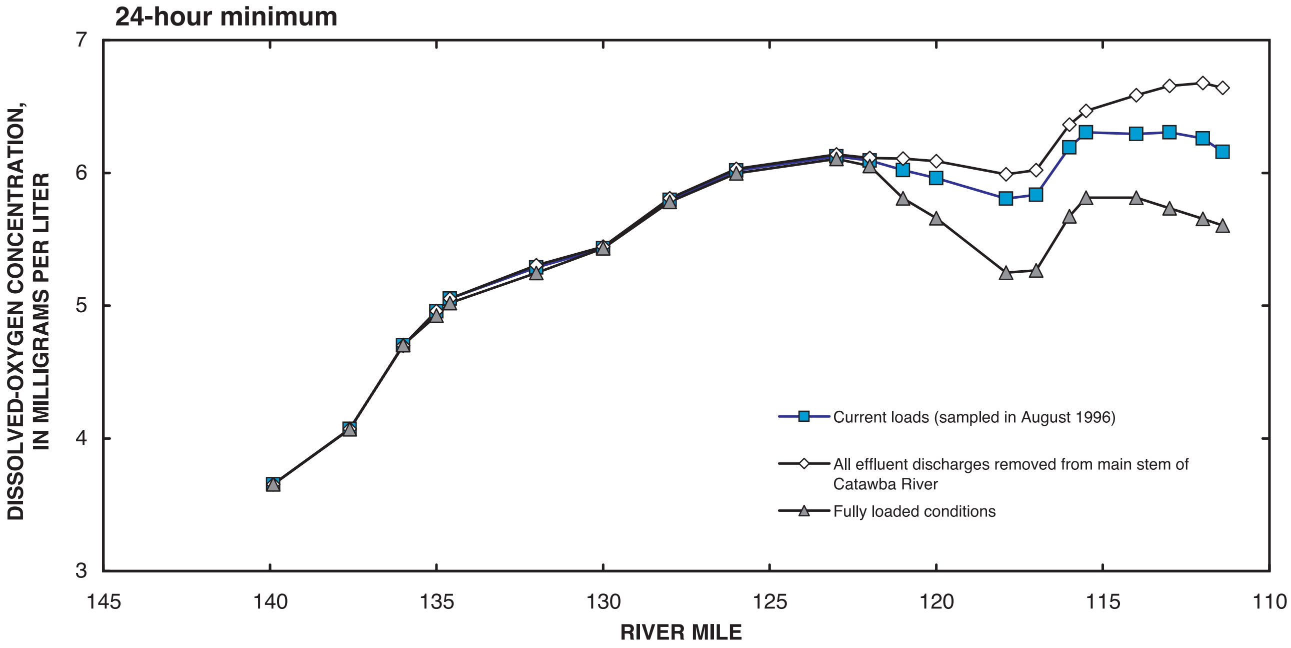


Figure 46. Longitudinal profiles of simulated 24-hour mean and minimum dissolved-oxygen concentrations for the no-load condition and two point-source loading conditions at 21 locations on the Catawba River, S.C., August 24, 1996.

Addition of Point-Source Load

To evaluate the effects of adding a point-source load to the Catawba River, two scenarios were simulated in which an additional point-source discharge was added at two alternate locations. For the first scenario, a hypothetical pipe was added at S.C. Highway 5 (RM 122.3). For the second scenario, a hypothetical pipe was added at a location near Culp Island (RM 119.2). The new point source at each of these locations was assumed to discharge 25 Mgal/d with the following characteristics: concentrations of BOD_5, 10.0 mg/L; ammonia nitrogen, 1.0 mg/L; nitrate nitrogen, 0.50 mg/L; and DO, 5.0 mg/L; and water temperature at 28.8 °C. In the calibration model, $CBOD_u$ was simulated instead of BOD_5. For this exercise, BOD_5 was converted to $CBOD_u$ by using an f-ratio of 3.0. The calibration period was used for the baseline condition. The hypothetical discharge pipe contributed an additional load of 7,210 pounds of UOD, or an increase of 22 percent from the simulated load during the calibration period.

The additional effluent load at S.C. Highway 5 (RM 122.3) or near Culp Island (RM 119.2) had little effect on the mean or minimum DO concentrations in the river (fig. 47; table 28). Differences between current conditions and the additional effluent load were on the order of 0.01 mg/L.

Phosphorus Loading

To evaluate the phosphorus loading of the Catawba River into Fishing Creek Reservoir, four conditions were simulated for the period August 25–27, 1996. Loads were computed as a total load for the 3-day period and reported in pounds (mean pounds per day for August 25–27, 1996, multiplied by 3). The conditions represent the effect of increasing and decreasing point-source phosphorus on the main stem of the Catawba River and the effect of removing the point and nonpoint sources from Sugar Creek. The first two conditions simulated are the fully permitted loading condition and the actual loading conditions as measured during the calibration period. The third loading condition simulated the effect of removing all of the point-source phosphorus loads from the main stem of the Catawba River. For the fourth condition, loads from Sugar Creek (both point- and nonpoint-source loads) were removed along with the point sources on the main stem of the Catawba River.

The organic phosphorus and orthophosphate loads for the calibration simulation for August 25–27, 1996, at RM 111.4 were 5,020 and 11,000 pounds, respectively. For the fully permitted condition, the organic phosphorus load increased by 19 percent from the calibration load and the orthophosphate load increased by 52 percent (fig. 48). Removing the phosphorus load completely from the point-source loads on the main stem of the Catawba River reduced the organic phosphorus and orthophosphate loads by 27 and 43 percent, respectively, from the fully permitted load. Reducing the point and nonpoint-source loading from Sugar Creek resulted in a further reduction in the organic phosphorus and orthophosphate load by 78 and 85 percent, respectively, from the fully permitted load.

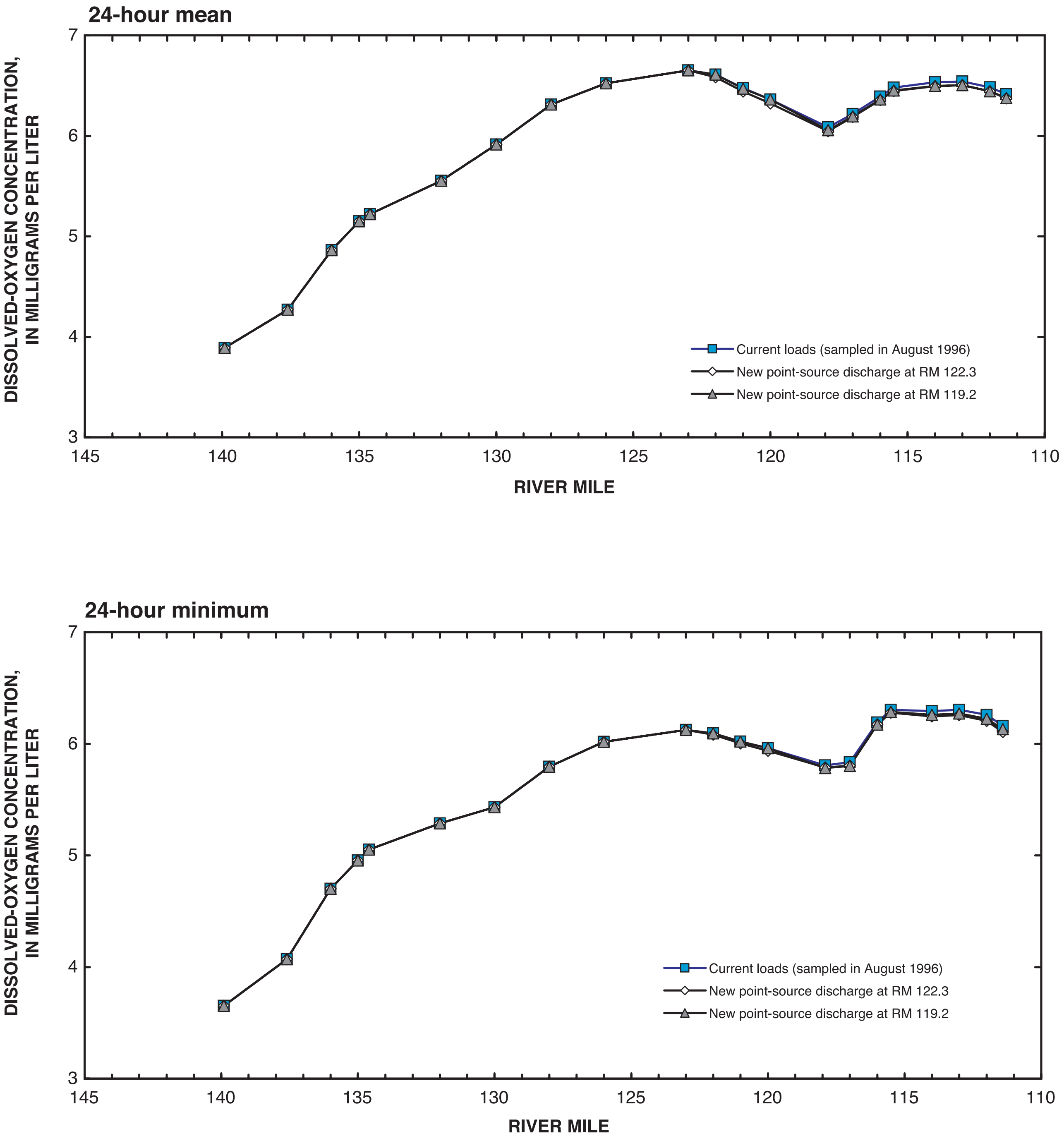


Figure 47. Longitudinal profiles for 24-hour (A) mean and (B) minimum dissolved-oxygen concentrations for August 24, 1996, from the addition of an effluent load near river miles 122.3 and RM 119.2.

Table 28. Simulated 24-hour mean and minimum dissolved-oxygen concentrations for three point-source loading conditions at 21 locations on the Catawba River, S.C.

[RM, river mile; DO, dissolved oxygen; mg/L, milligram per liter]

River mile (fig. 2)	Calibration period	Additional effluent at S.C. Highway 5 (RM 122.3)		Additional effluent discharge near Culp Island (RM 119.2)	
	DO (mg/L)	DO (mg/L)	Change from calibration period (mg/L)	DO (mg/L)	Change from calibration period (mg/L)
August 24, 1996, 24-hour mean DO concentrations					
139.9[a]	3.89	3.89	0.00	3.89	0.00
137.6[a]	4.27	4.27	.00	4.27	.00
136.0	4.86	4.86	.00	4.86	.00
135.0	5.15	5.15	.00	5.15	.00
134.6[a]	5.22	5.22	.00	5.22	.00
132.0	5.55	5.55	.00	5.55	.00
130.0	5.91	5.91	.00	5.91	.00
128.0	6.31	6.31	.00	6.31	.00
126.0	6.52	6.52	.00	6.52	.00
123.0[a]	6.65	6.65	.00	6.65	.00
122.0[a]	6.61	6.59	-.02	6.61	.00
121.0	6.48	6.44	-.04	6.47	-.01
120.0	6.36	6.32	-.04	6.36	.00
117.9[a]	6.09	6.05	-.04	6.06	-.03
117.0	6.22	6.19	-.03	6.19	-.02
116.0	6.39	6.36	-.03	6.36	-.03
115.5[a]	6.48	6.45	-.03	6.45	-.03
114.0	6.53	6.50	-.03	6.50	-.03
113.0	6.54	6.50	-.04	6.51	-.03
112.0	6.48	6.44	-.04	6.44	-.04
111.4[a]	6.42	6.37	-.05	6.37	-.05

Table 28. Simulated 24-hour mean and minimum dissolved-oxygen concentrations for three point-source loading conditions at 21 locations on the Catawba River, S.C.—Continued

[RM, river mile; DO, dissolved oxygen; mg/L, milligram per liter]

River mile (fig. 2)	Calibration period	Additional effluent at S.C. Highway 5 (RM 122.3)		Additional effluent discharge near Culp Island (RM 119.2)	
	DO (mg/L)	DO (mg/L)	Change from calibration period (mg/L)	DO (mg/L)	Change from calibration period (mg/L)
August 24, 1996, 24-hour minimum DO concentrations					
139.9[a]	3.65	3.65	0.00	3.65	0.00
137.6[a]	4.07	4.07	.00	4.07	.00
36.0	4.70	4.70	.00	4.70	.00
35.0	4.96	4.96	.00	4.96	.00
134.6[a]	5.05	5.05	.00	5.05	.00
132.0	5.29	5.29	.00	5.29	.00
130.0	5.43	5.43	.00	5.43	.00
128.0	5.80	5.80	.00	5.80	.00
126.0	6.02	6.02	.00	6.02	.00
123.0[a]	6.13	6.13	.00	6.13	.00
122.0[a]	6.09	6.08	-.01	6.09	.00
121.0	6.02	6.00	-.02	6.02	.00
120.0	5.96	5.94	-.02	5.96	.00
117.9[a]	5.81	5.79	-.02	5.78	-.03
117.0	5.84	5.81	-.03	5.80	-.04
116.0	6.19	6.17	-.02	6.17	-.02
115.5[a]	6.31	6.28	-.03	6.29	-.02
114.0	6.29	6.25	-.04	6.26	-.03
113.0	6.31	6.26	-.05	6.27	-.04
112.0	6.26	6.21	-.05	6.22	-.04
111.4[a]	6.16	6.11	-.05	6.13	-.03

[a] Water-quality sampling location.

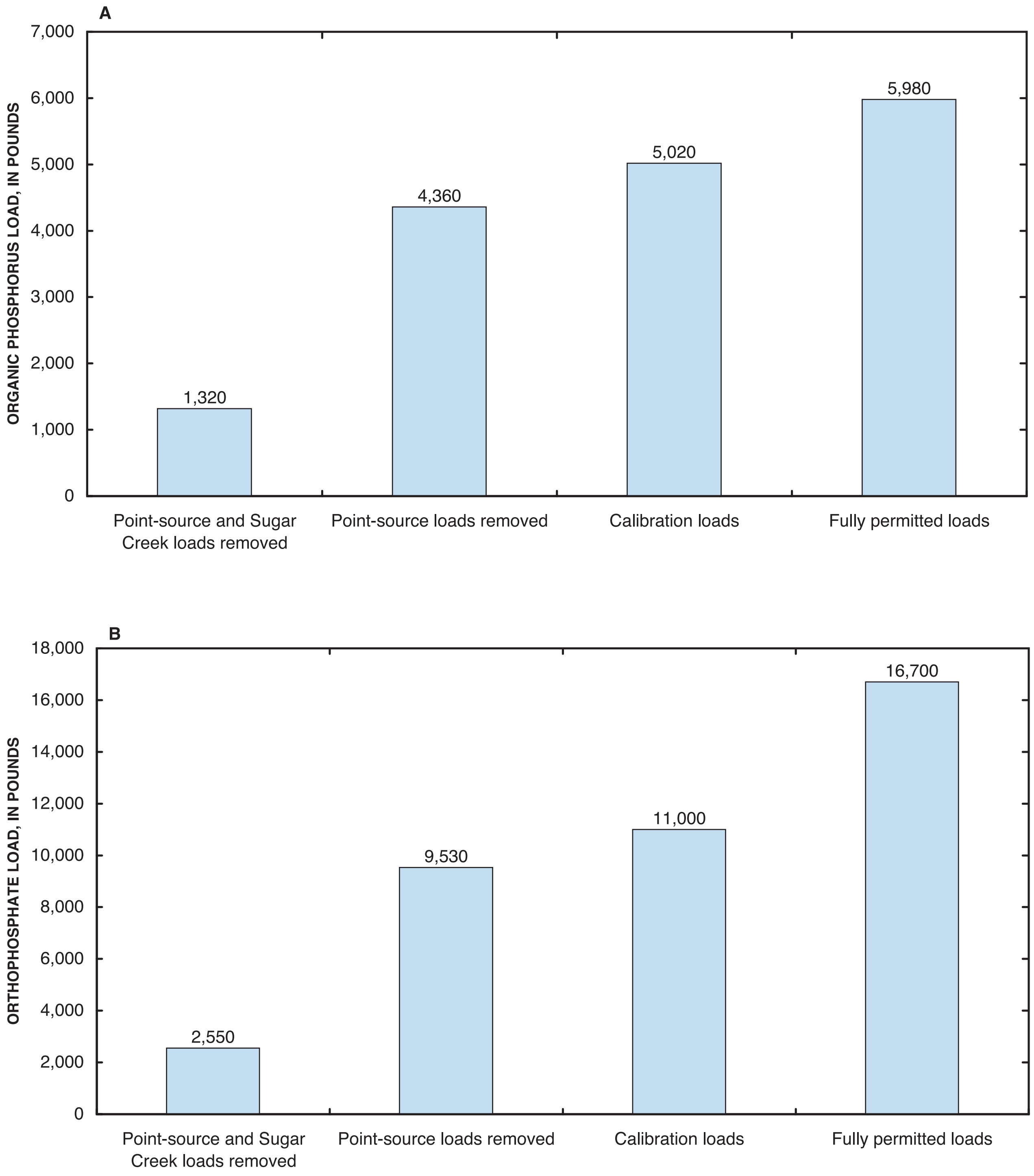


Figure 48. Loadings of (A) organic phosphorus and (B) orthophosphate at river mile 111.4 (S.C. Highway 9) for four loading conditions on the Catawba River, S.C., August 25–27, 1996.

SUMMARY

The U.S. Geological Survey (USGS) one-dimensional, dynamic-flow BRANCH model and the Branched Lagrangian Transport Model (BLTM) were calibrated and validated for the Catawba River downstream from Lake Wylie Dam (RM 141.0) to the headwaters of Fishing Creek Reservoir (RM 110.3) in South Carolina. The limits of the BLTM extend from river mile 139.9 to river mile 111.4 (S.C. Highway 9). Data used to calibrate and validate the flow model included time-series water levels at five locations and streamflow at two locations along the Catawba River, streamflow from the Lake Wylie Dam, estimated tributary flow, channel geometry, and flow-resistance coefficients. The transport model was calibrated and validated with time-of-travel and dispersion data collected along the Catawba River. Data used in calibrating and validating the water-quality model included concentrations of nutrients, chlorophyll *a*, and biochemical oxygen demand at 10 locations along the Catawba River and at 3 tributaries; continuous water temperature and dissolved-oxygen concentrations at 5 locations along the Catawba River; sediment- and substrate-oxygen demand data; reaeration data; wastewater-effluent flow and concentration data; and meteorological data. Water-quality data also were collected for selected tributaries to the Catawba River.

The streamflow, transport, and water-quality models were calibrated by adjusting the model parameters until simulated hydraulic and water-quality values were within the range of measured data. After calibration, the flow, transport, and water-quality models were validated to ensure that the models would accurately simulate conditions different from those existing during the calibration period. A sensitivity analysis was made on the streamflow, transport, and water-quality models.

The BLTM water-quality model of the Catawba River was used to simulate three point-source and nonpoint-source loading conditions to evaluate the effects on the system. Scenarios included setting point-source discharges at various loading levels to evaluate the effect on dissolved-oxygen (DO) concentrations, adding an additional point-source load to the river and evaluating the effect on DO concentration, and reducing point- and nonpoint-source phosphorus inputs to the river and evaluating the phosphorus load to Fishing Creek Reservoir.

Different point-source loading conditions to the system were evaluated. The current loading condition, as measured during the August 1996 calibration period, decreased the 24-hour mean and minimum DO concentration by a maximum of 0.33 and 0.48 mg/L, respectively. Fully permitted loading conditions decreased the 24-hour mean and minimum DO concentration by a maximum of 0.88 and 1.04 mg/L, respectively. An additional point-source load to the system of 7,210 pounds of UOD at Culp Island or at S.C. Highway 5 had no significant effect on the daily mean and daily minimum DO.

To evaluate the phosphorus loading into Fishing Creek Reservoir, four loading conditions of phosphorus into the Catawba River were simulated. The four conditions included fully permitted and actual loading conditions, removal of all point sources from the Catawba River, and removal of all point and nonpoint sources from Sugar Creek. Reducing the point-source inputs on the Catawba River and the point and nonpoint sources in Sugar Creek reduced the organic phosphorus and orthophosphate loading to Fishing Creek Reservoir by 78 and 85 percent, respectively.

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