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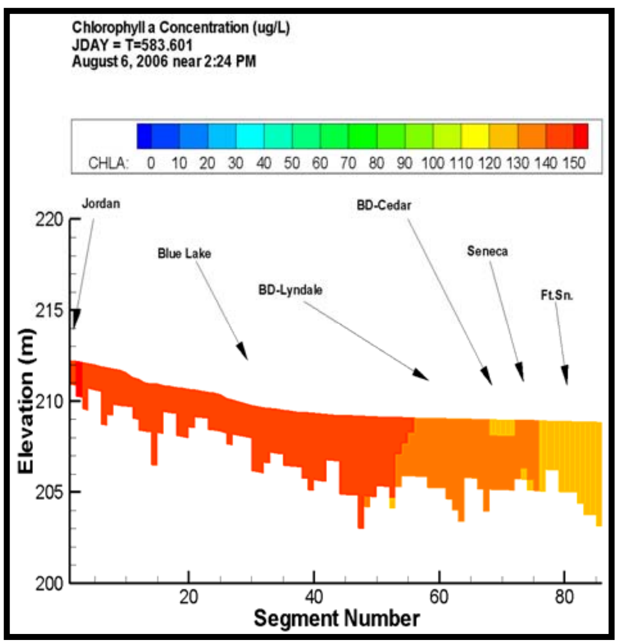
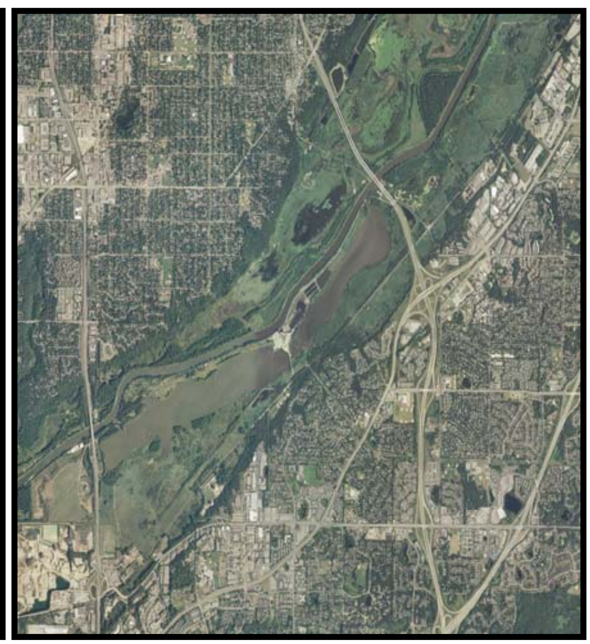
Modeling the Hydrodynamics and Water Quality of the Lower Minnesota River Using CE-QUAL-W2

A Report on the Development, Calibration, Verification, and Application of the Model

David L. Smith, Tammy L. Threadgill, and Catherine E. Larson

May 2012





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David L. Smith and Tammy L. Threadgill

Environmental Laboratory
U.S. Army Engineer Research and Development Center
3909 Halls Ferry Rd
Vicksburg, MS 39180-6199

Catherine E. Larson

Metropolitan Council Environmental Services
390 Robert Street North
St. Paul, MN 55101-1805

Final report

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St. Paul, MN
U.S. Army Engineer District, St. Paul

Abstract

The U.S. Army Engineer Research and Development Center (USACE ERDC) Environmental Lab (EL) and Metropolitan Council Environmental Services (MCES) developed an advanced water quality model of the Lower Minnesota River (Jordan, Minnesota, to the mouth) using the CE-QUAL-W2 modeling framework. This portion of the river is a highly impaired system with a very rich set of monitored data. Model development consisted of calibration and validation of seven water years: 1988 (low flow) and 2001-2006. Data from 2006 were first used to calibrate the model, and the same parameter values were applied to all other years for validation. The 2006 parameter set worked well for all years except 1988. The model was then recalibrated using data from 1988 and verified by applying the revised parameter set to the other six years. The model output agrees to an acceptable level with observed data for every water year simulated. The Lower Minnesota River Model (LMRM) provides a tool for load allocation studies and facility or watershed planning, in addition to providing a bridge to other water quality modeling efforts in the area.

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Preface

This report was written to detail development, calibration, validation, and application of the Lower Minnesota River Model (LMRM) Project. The LMRM serves two important purposes for stakeholders and regulators:

1. LMRM is a tool for load allocation studies and facility or watershed planning.
2. LMRM is a bridge to other water quality models in the area.

Dr. David Smith and Tammy Threadgill, both of the Water Quality and Contaminant Modeling Branch (WQCMB), Environmental Processes and Engineering Division (EPED), of the Environmental Laboratory (EL), U.S. Army Engineer Research and Development Center (ERDC), Vicksburg, Mississippi, conducted this study with assistance from Catherine Larson and Karen Jensen, Metropolitan Council Environmental Services (MCES) St. Paul, Minnesota. Dr. Smith, Threadgill and Larson participated in preparing this report. Dr. Smith served as the principal investigator and study point of contact. This study was jointly funded by Metropolitan Council Environmental Services and the U.S. Army Corps of Engineers, St. Paul District.

This work was conducted under the general supervision of Dr. Quan Dong, Chief, WQCMB; and Warren Lorentz, Chief, EPED. Dr. Beth Fleming was Director of EL. COL Kevin J. Wilson was Commander of ERDC. Dr. Jeffery P. Holland was Director.

Unit Conversion Factors

Multiply	By	To Obtain
cubic feet	0.02831685	cubic meters
degrees Fahrenheit	(F-32)/1.8	degrees Celsius
feet	0.3048	meters
square miles	2.589998 E+06	square meters
langley per day	0.48	Watts per square meter

Acronyms and Units

ACHLA	Ratio of algal biomass to chlorophyll a, mg algae/µg chla a
ALG1	Algal group #1 assigned to diatoms, mg/L dry wt
ALG2	Algal group #2 assigned to blue-green algae, mg/L dry wt
ALG3	Algal group #3 assigned to other algae (mostly green), mg/L dry wt
BOD	Biochemical oxygen demand, mg/L
BODC	Stoichiometric equivalent between CBOD decay and carbon
BODN	Stoichiometric equivalent between CBOD decay and nitrogen
BODP	Stoichiometric equivalent between CBOD decay and phosphorus
CBOD	Carbonaceous biochemical oxygen demand, 5-day (5) or ultimate (U)
CHLA	Chlorophyll *a* associated with live phytoplankton, µg/L
DO	Dissolved oxygen, mg/L
DOC	Dissolved organic carbon, mg C/L
DSI	Dissolved silica, mg/L
ERDC	U.S. Army Engineer Research and Development Center
GP	Black Dog Generating Plant
ISS	Inorganic suspended solids, mg/L
LDOM	Labile dissolved organic matter, mg/L dry wt (decomposes at a fast rate)

LMRM	Lower Minnesota River Model
LPOM	Labile particulate organic matter, mg/L dry wt (decomposes at a fast rate)
MAC	Metropolitan Airports Commission
MCES	Metropolitan Council Environmental Services
MPCA	Minnesota Pollution Control Agency
MRBDC	Minnesota River Basin Data Center
MRCC	Midwestern Regional Climate Center
MSP	Minneapolis-St. Paul International Airport
NH4	Ammonium nitrogen, mg N/L
NO3	Nitrate nitrogen, mg N/L
OM	Organic matter, mg/L dry wt
ORGN	Stoichiometric equivalent between organic matter and nitrogen
ORGP	Stoichiometric equivalent between organic matter and phosphorus
PO4	Orthophosphate phosphorus, mg PO_4 as P/L
POMS	Particulate organic matter settling rate, 1/day
RDOM	Refractory dissolved organic matter, mg/L dry wt (decomposes at a slow rate)
RM	River mile as measured from mouth
RPOM	Refractory particulate organic matter, mg/L dry wt (decomposes at a slow rate)
SSS	Suspended solids settling rate, 1/day

TDS	Total dissolved solids, mg/L
TKN	Total Kjeldahl nitrogen, mg N/L
TP	Total phosphorus, mg P/L
UMSP	University of Minnesota, St. Paul campus
USACE	U.S. Army Corps of Engineers
USGS	US Geological Survey
W2	CE-QUAL-W2 model
WY	Water year (October 1 through September 30)

1 Introduction

This report details the development, calibration, validation, and application of a hydrodynamic water quality model for the Lower Minnesota River from Jordan, Minnesota, to its confluence with the Mississippi River in St. Paul, Minnesota. The Lower Minnesota River Model (LMRM) will assist with estimating impacts of point and nonpoint source management actions aimed at improving water quality. The model may also provide a bridge to other modeling efforts, such as the Minnesota River Basin Model, developed for the Minnesota Pollution Control Agency (MPCA) by Tetra Tech (2008); and the Upper Mississippi River-Lake Pepin Model, developed for the MPCA by LimnoTech (2009).

Background and objectives

The goal of this project is to provide a calibrated and validated water quality model for approximately the lower 40 miles of the Minnesota River. This is a reach that extends from just below Jordan, Minnesota, down to the confluence with the Mississippi River in St. Paul, Minnesota. This reach of the river has been listed as impaired due to low levels of dissolved oxygen and high levels of turbidity, bacteria, mercury, and PCBs (MPCA 2008). Figure 1 provides an overview of the study area.

Over the past two decades, several studies and assessment reports have documented impairments of the water quality of the lower Minnesota River. In 1985, the MPCA conducted a wasteload allocation study (MPCA 1985). The study concluded that, in order to meet dissolved oxygen standards in the river, greater-than-secondary treatment would be needed at the two wastewater facilities, along with a 40% reduction in loads of oxygen-demanding material from nonpoint sources. Later, the MPCA linked high phosphorus concentrations to the oxygen impairment via the stimulation of excessive algal growth (MPCA 2004). As the algae respire and decay, they contribute to high oxygen demand.

Water-quality concerns over the entire Minnesota River Basin fall into three major categories: excessive sediment, nutrient enrichment, and environmental health risks (Minnesota River Basin Data Center (MRBDC) 2007). In turn, the Minnesota River contributes the highest sediment and

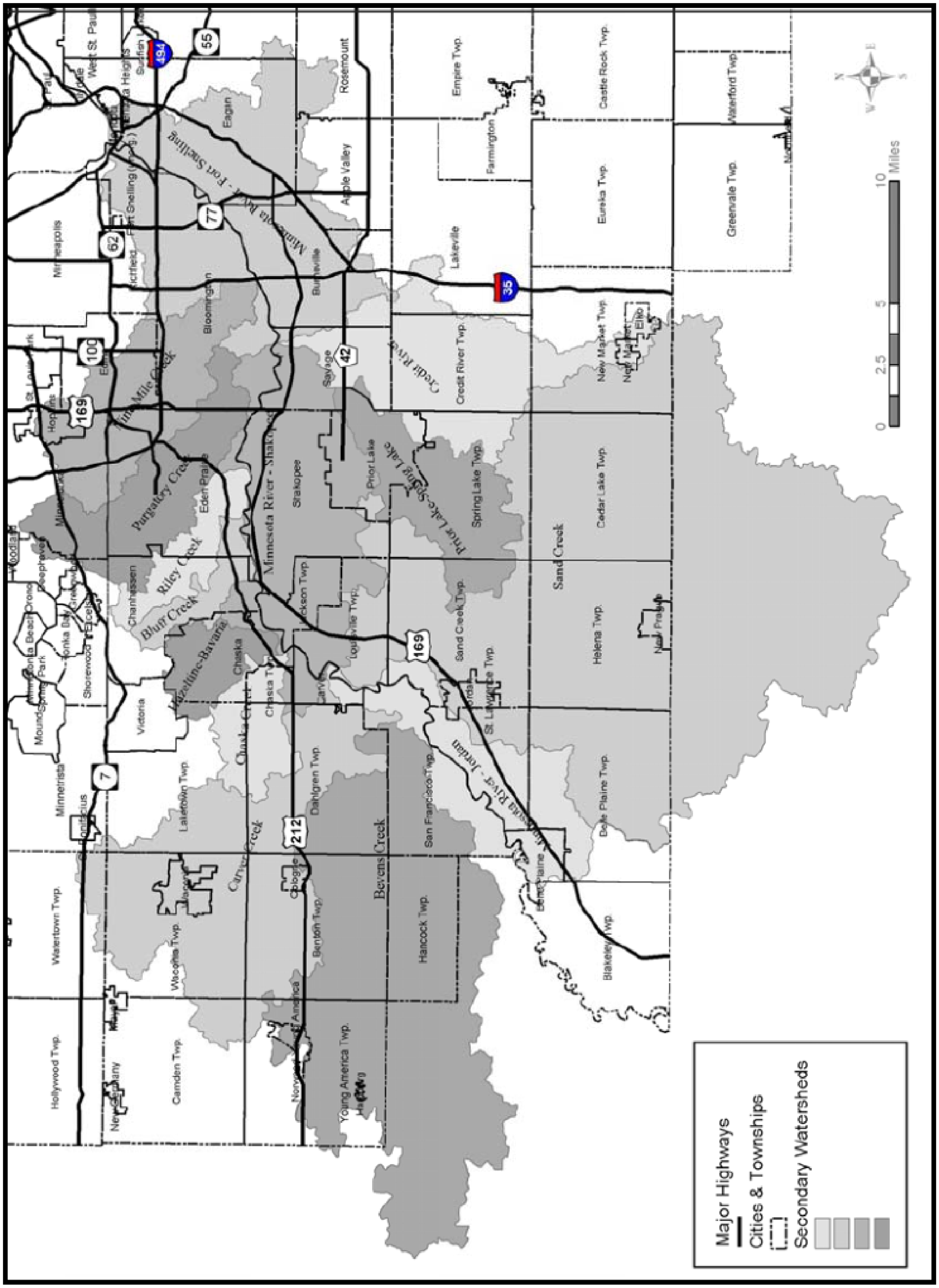


Figure 1. Lower Minnesota River Model Study Area (Matthew McGuire, MCES).

nutrient loads to the Mississippi River upstream of Lake Pepin, a natural impoundment in Navigation Pool 4 (St. Paul Metropolitan Council 2002, 2004). A number of other studies provide further evidence of poor water quality in the lower Minnesota River (Larson 2004).

In 1999 the MPCA and MCES began meeting to share plans and discuss needs for water-quality modeling in the Metro Area. The joint workgroup identified the need to update the wasteload allocation study of the lower Minnesota River and ranked it a high priority. Further discussions resulted in a project proposal for the Lower Minnesota River Model (Larson 2004). In 2003 the Metropolitan Council started coordinating a six-year project to develop the model. An interagency group formed to sponsor the project and guide the technical aspects. In the first year they selected a model framework (CE-QUAL-W2) and designed a three-year monitoring program to support it (Larson 2006). The monitoring program was implemented during water years (WY) 2004-2006. In 2005 the Metropolitan Council entered a cost-sharing agreement with the U.S. Army Engineer Research and Development Center (ERDC) to develop a hydrodynamic and water-quality model of the lower Minnesota River using the CE-QUAL-W2 framework.

The proposal outlined the model features, capabilities, and selection criteria needed to meet the project objectives and priorities (Larson 2004). The top priority was developing a tool for setting effluent limitations for expanded wastewater treatment facilities and other point sources. Second was determining pollutant loads from the headwaters and tributaries and reductions needed to meet water-quality standards. Modeling and monitoring would focus on the following variables, in order of priority: dissolved oxygen, ammonia, nutrients, and sediment.

Several objectives were defined for the modeling project:

1. Develop, calibrate, and validate a model for the three extensively monitored water years: 2004, 2005, and 2006.
2. Run the model for further validation, and possibly recalibration, for four earlier years: 2001, 2002, 2003, and 1988. The four years were chosen to provide a range of conditions from drought (1988) to flood (2001).
3. Provide MCES with a complete, calibrated, and validated model for use in load allocation studies and facility or watershed planning.
4. Provide MCES with a post-processor for viewing LMRM output and technical support during model delivery.

Site description

The Minnesota River watershed covers approximately 16,900 square miles and encompasses about 20% of the total area of Minnesota. It drains the southwestern and south central part of the state. Due to its relatively flat topography and rich soils, the Minnesota River basin is well suited for agriculture. In 1997, over 70% of the watershed was classified as cultivated cropland. Though land use is primarily agriculture in the western watersheds, it becomes increasingly developed toward the confluence of the Mississippi River. The model domain encompasses the lower 40 miles of the Minnesota River, which lie within the seven-county Twin Cities Metropolitan Area (Metro Area).

Roughly a dozen named tributaries enter the Metro-Area reach of the Minnesota River. The state's third and fourth largest wastewater treatment plants, Blue Lake and Seneca, respectively, also discharge to this reach. The lower 40 miles receive permitted discharges from several other facilities, notably stormwater discharges from the Minneapolis/St. Paul International Airport and cooling-water discharges from the Black Dog Generating Plant, a power generating plant owned and operated by Xcel Energy. The lower 15 miles of the river are maintained as a navigation channel for commercial barge traffic. The backwater pool behind Lock and Dam No. 2 on the Mississippi River also affects the hydrology of the lower Minnesota River (MRBDC 1999). Figure 2 is a detailed map of the project study area, including all major tributaries, wastewater treatment plants, power plant, and airport outfalls.

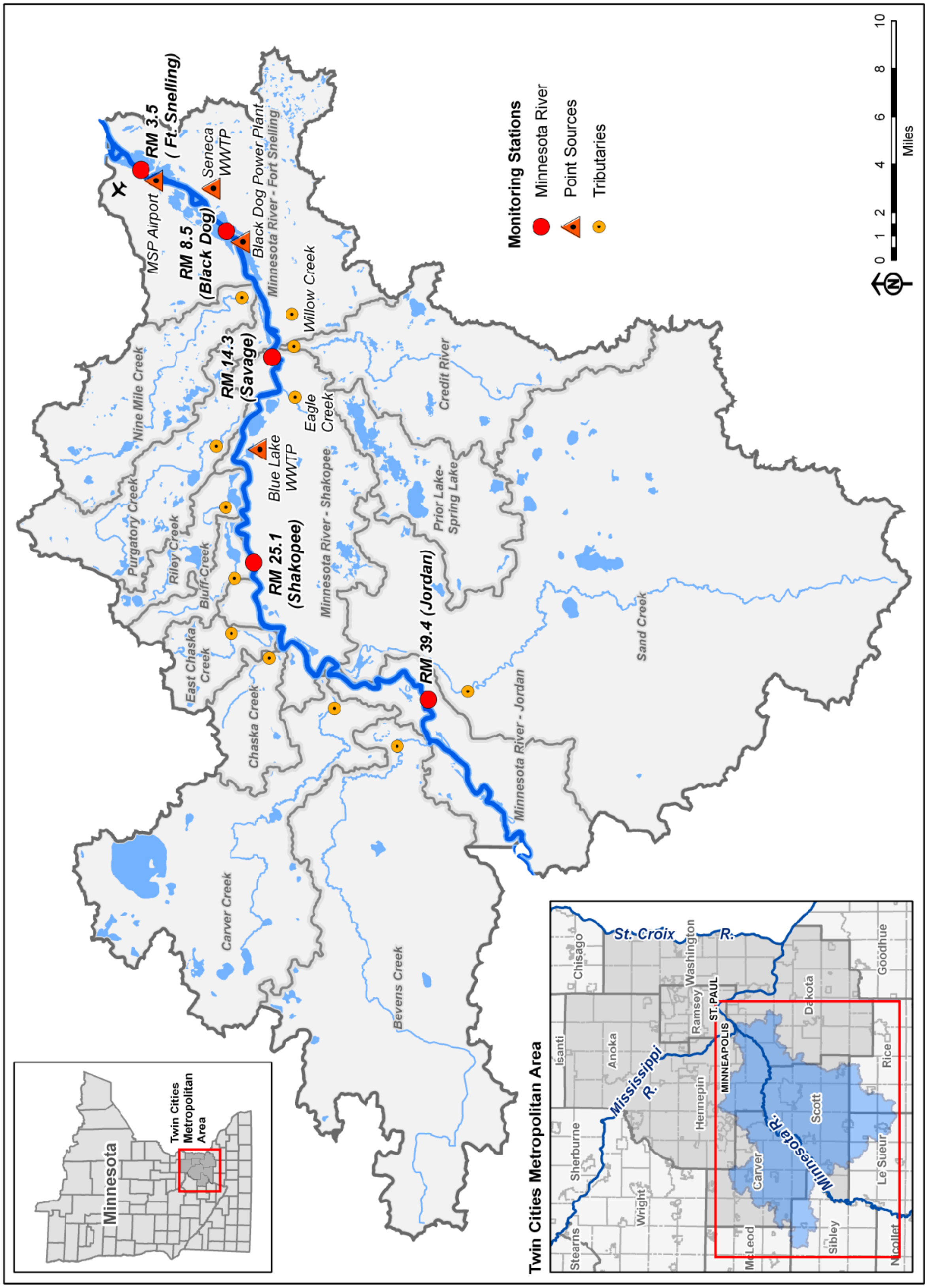


Figure 2. Detailed Minnesota River Study Area (Craig Skone, MCES).

2 Model Selection and Development Approach

CE-QUAL-W2 (W2) is the code selected to develop the LMRM. W2 is a two-dimensional longitudinal-vertical hydrodynamics and water quality model. It is capable of modeling basic eutrophication processes and is best suited for long narrow waterbodies that do not exhibit substantial lateral variation. W2 has been applied to hundreds of studies on various types of waterbodies (rivers, reservoirs, lakes, and estuaries) all over the world. For a list of the model applications, see the CE-QUAL-W2 website: http://www.ce.pdx.edu/w2/.

CE-QUAL-W2 description

The numerical modeling code known as CE-QUAL-W2, version 3.6 (Cole and Wells 2008), was configured for application to the lower Minnesota River. W2 uses a finite difference solution of the laterally averaged equations of fluid motion (Cole and Wells 2008). It allows for application to very complex water systems because it accommodates multiple branches and multiple waterbody types. W2 allows the user to set up variable grid spacing (longitudinally and vertically), time variable boundary conditions, multiple inflows and outflows, and time variable concentrations for each water quality constituent being modeled.

W2 is capable of modeling water elevation, flow, water temperature, and 28 water quality constituents such as total dissolved solids (TDS), inorganic suspended solids (ISS), ammonium (NH4), biochemical oxygen demand (BOD), nitrate (NO3), phytoplankton, dissolved oxygen (DO), and organic matter (OM). The constituents modeled in this study can be found in Table 1. In addition to modeling several state variables, W2 can also calculate over 60 derived variables such as total phosphorus (TP), chlorophyll *a* (CHLA), dissolved organic carbon (DOC), and total Kjeldahl nitrogen (TKN).

Hydrodynamics are updated at every time-step in the model; kinetics are updated based on a user-defined parameter in the control file, constituent update frequency (CUF) (Cole and Wells 2008). For the LMRM model, kinetics are updated every 10 time-steps. The time-step chosen allows for the model to adequately predict temporal and diurnal variations.

Table 1. CE-QUAL-W2 constituents used in the LMRM project.

Water Temperature	Total Dissolved Solids (TDS)	Dissolved Oxygen (DO)	Orthophosphate (PO4)
Ammonium (NH4)	Nitrate (NO3)	Dissolved Silica (DSI)	Inorganic Suspended Solids (ISS)
Labile Dissolved Organic Matter (LDOM)	Refractory Dissolved Organic Matter (RDOM)	Labile Particulate Organic Matter (LPOM)	Refractory Particulate Organic Matter (RPOM)
Carbonaceous Biochemical Oxygen Demand (CBODU1-6)	Diatoms (ALG1)	Blue-Green Algae (ALG2)	Other Algae (ALG3)

Project approach

CE-QUAL-W2 is well suited for application to the lower Minnesota River because of the following:

1. W2 is appropriate for modeling long, narrow waterbodies with spatially varying depths.
2. W2 is capable of modeling all constituents of concern in the river, including dissolved oxygen, ammonium, orthophosphate, phytoplankton, non-living organic matter, and suspended solids.
3. W2 has been applied to hundreds of water systems and is well-known, understood, and widely accepted.
4. W2 is capable of providing a wide variety of model output for comparison to observed data.
5. W2 is able to simulate various responses due to changes in loads and rates.

Seven monitoring stations were used to evaluate model performance during calibration. Locations with monitoring data are: River Mile (RM) 39.4, RM 25.1, RM 14.3, RM 13.0, RM 11.7, RM 8.5, and RM 3.5. RM 39.4 represents the inflow boundary condition at Jordan, and RM 3.5, or Fort Snelling, contains the most complete calibration data set. RM 3.5 was used as the primary calibration site because it is near the Minnesota River mouth, is below all point sources, and is in a reach with the most significant water quality problems.

Calibration strategy

Despite an outstanding data set that spanned the study reach and covered seven years, it proved difficult to implement a calibration and validation approach where some years or some sampling stations are used for calibration and others are used for validation. Two factors contributed to this difficulty. First, a consistent and substantial longitudinal decrease in model performance was evident. There were five water quality sampling stations. The first at Jordan was used to establish the time-varying boundary conditions. Thus, four other stations were available between Jordan and Fort Snelling at RM 3.5 that could have been paired for model calibration and validation. However, model performance between stations was not comparable because of the longitudinal decrease in model performance from Jordan to Fort Snelling. Any comparison between two stations in a given year would have reflected this dominant model performance trend.

Second, deciding which years among the seven were suitable for calibration and which were suitable for validation was arbitrary due to the hydrologic and water quality variability. In effect, no two years were comparable especially after a detailed inspection of flow and water quality data.

For these reasons, calibration was approached in a new and different way. W2 was first applied and calibrated to water year 2006. The same model parameters were used for the remaining six years. This yielded reasonable results in most cases with the notable exceptions of 1988 and summer low flow periods in general. To improve the calibration in 1988, a number of changes were made (listed below), but the most important were to add non-living organic matter, adjust the algal parameters, and adjust particle settling rates. These combined changes improved model performance for NH4 and DO in 1988 and other summer low flow periods. These changes were then applied to 2001 through 2006 and resulted in reasonable model performance. In effect, coefficients that reproduced water quality trends for 2006 did not perform well for 1988. However, coefficients that improved 1988 also reproduced measured water quality trends for all modeled years. The result was one set of coefficients that provide reasonable model performance over a wide range of water years. Moreover, 2001 through 2006 were modeled continuously as one complete model. Continuous model runs eliminate the arbitrary split between calibration and validation and suggested that one set of coefficients was suitable for all years (see Dr. Lung's comments in Appendix B for continous model output).

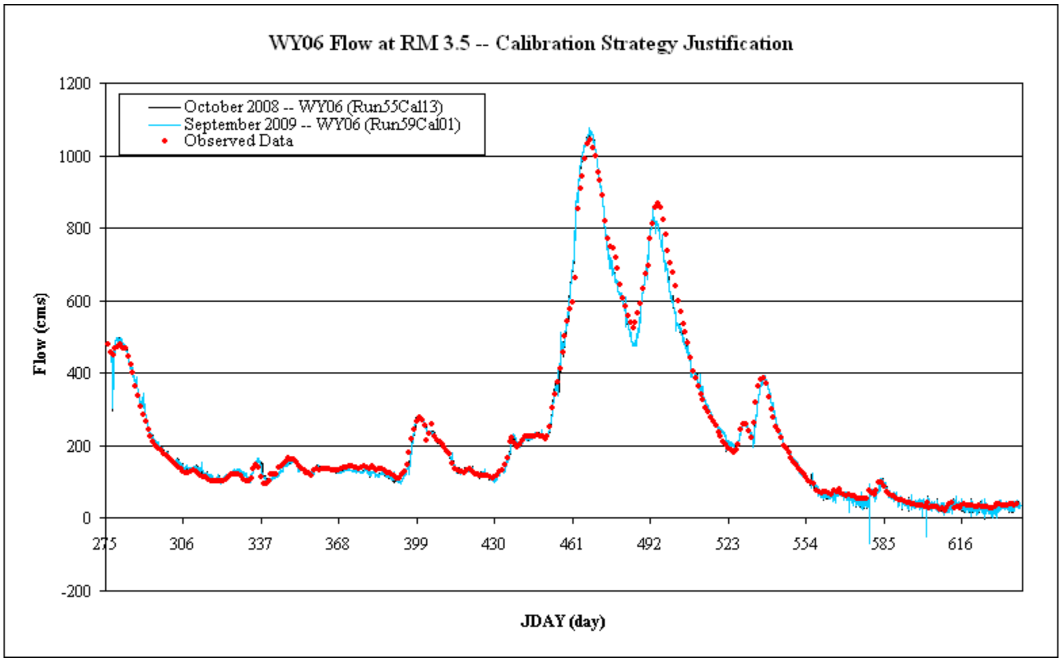
Figures 3-7 highlight the model output and measured data used during the calibration. The black line in the figures represents the initial calibration (labeled "October 2008"): calibrate WY 2006 and apply that parameter set backwards to all other water years. The same calibration parameters that worked well for water quality in water years 2004, 2005, and 2006 did not work sufficiently enough for the earlier years, especially 1988. ERDC then decided to recalibrate the model for WY 1988 and apply that parameter set forward to water years 2001-2006. The blue line represents this final calibration (labeled “September 2009”). Notice the improvements made to the water quality constituents, especially NH4 and DO in 1988 (Figures 5 and 6). The changes made in 2009 also improved the calibration for water years 2004-2006.

Changes made between the initial and final calibrations that led to this improvement were as follows:

1. Six BOD groups were initially defined in the model. However, after further review and calibration modifications, once the organic matter compartments (LDOM, RDOM, LPOM, RPOM) were turned ‘on’ in the model, only three unique BOD groups needed to be modeled—one for Blue Lake, one for Seneca, and one for the airport. For the other three BOD groups (RM 39.4, RM 3.5, and the tributaries), organic matter was substituted for BOD. Instead of modifying the input files to remove the extra BOD groups, the corresponding input values were set to 0.0 mg/L.
2. Three algal groups were modeled consistently across all three years—diatoms, bluegreens, and others. For the years when no data were available, monthly average splits based on all available measured data were applied to the total biomass measured.
3. Organic matter (labile and refractory dissolved organic matter, LDOM and RDOM, and labile and refractory particulate organic matter, LPOM and RPOM) was calculated based on measured dissolved organic carbon and volatile suspended solids. In the initial calibration, these organic matter groups were modeled; however, all of them were input as 0.0 mg/L, and the initial concentration of RDOM was set to 8.0 mg/L in the CE-QUAL-W2 control file. (See Appendix A for information on how the four groups were defined.)
4. Light extinction coefficients were set to correspond to Dr. R.O. Megard’s (2007) research (see Appendix C).
5. The suspended solids settling rate (SSS) was decreased from 1.0 to 0.15 m/day.
6. The ratio of algal biomass to chlorophyll-a (ACHLA) was reduced from 0.135 to 0.0675 mg algae/μg chla.
7. The algal growth rate for ALG1 (diatoms) was decreased from 2.3/day to 1.9/day and the rate for ALG3 (mostly green algae) was decreased from

2.5/day to 2.3/day. The algal temperature coefficients were also modified. In general, the temperature coefficients were increased.

8. The particulate organic matter settling rate (POMS) was increased from 0.10 to 0.80 m/day.
9. The stoichiometric equivalent between organic matter and nitrogen (ORGN) was decreased from 0.08 to 0.05.
10. For airport BOD (BOD4), the stoichiometric equivalents were changed to BODP = BODN = 0.0 mg/L and BODC = 0.387. These equivalents were determined based on the deicing material.



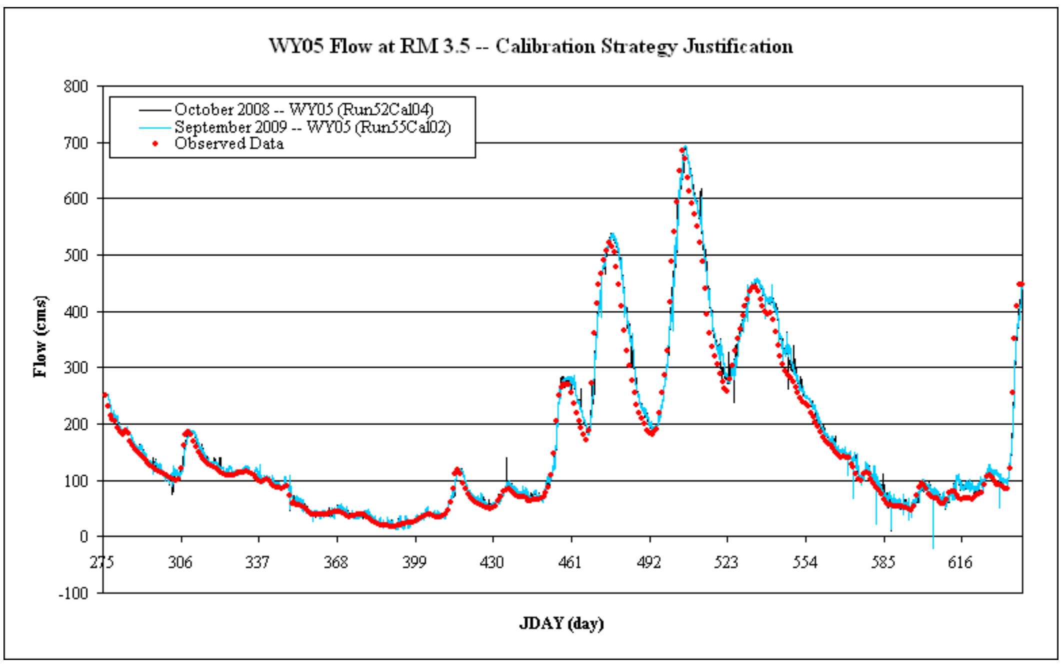
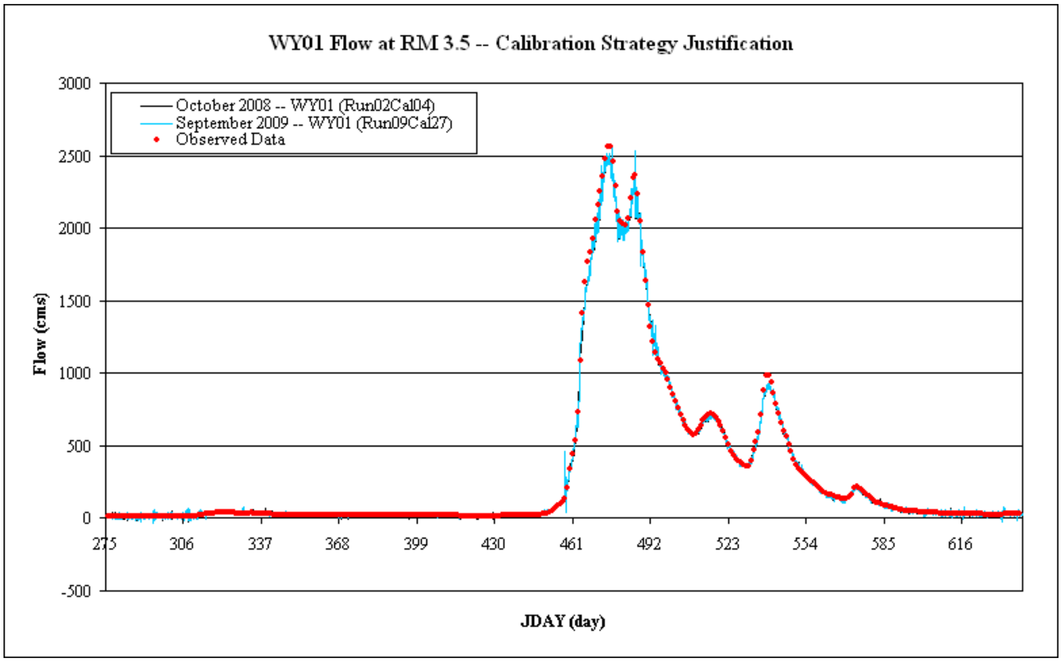


Figure 3. Calibration justification – Flow at RM 3.5 (continued).



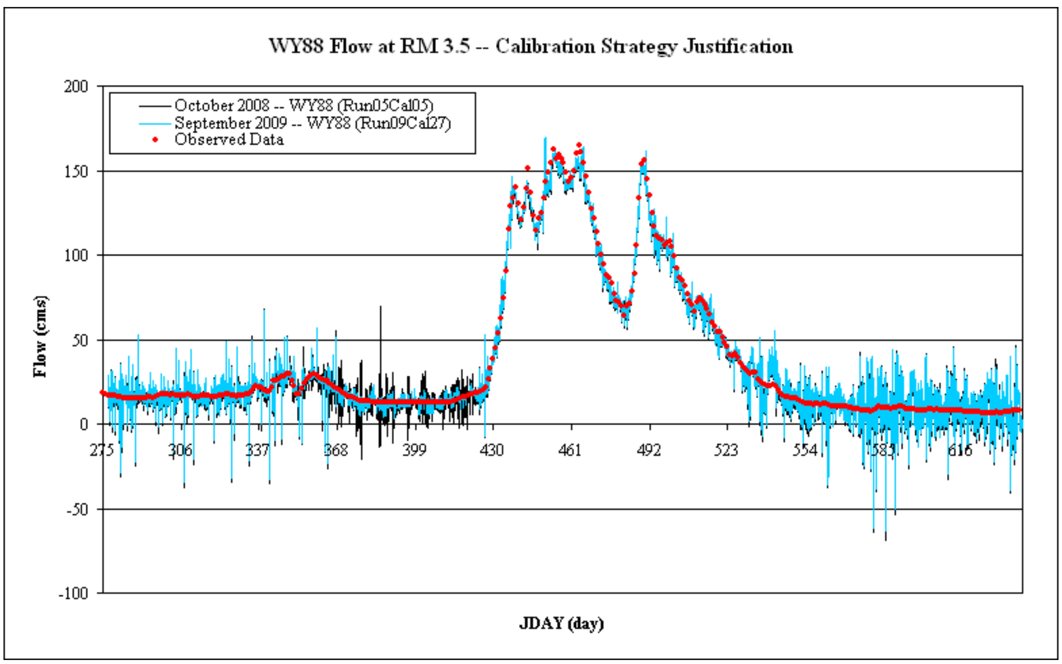
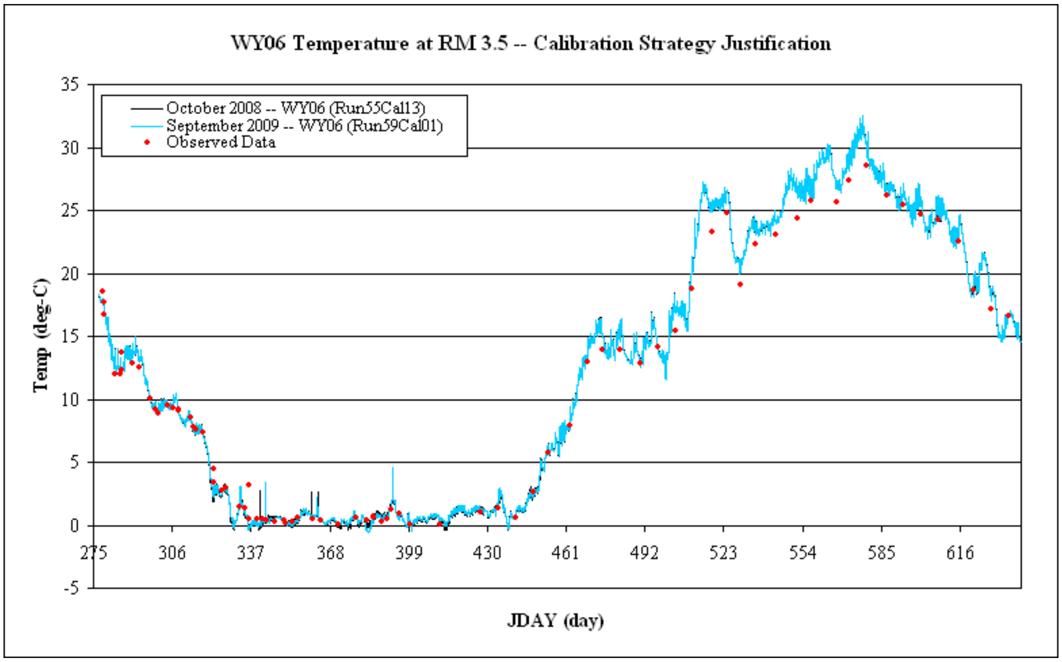


Figure 3. (concluded).



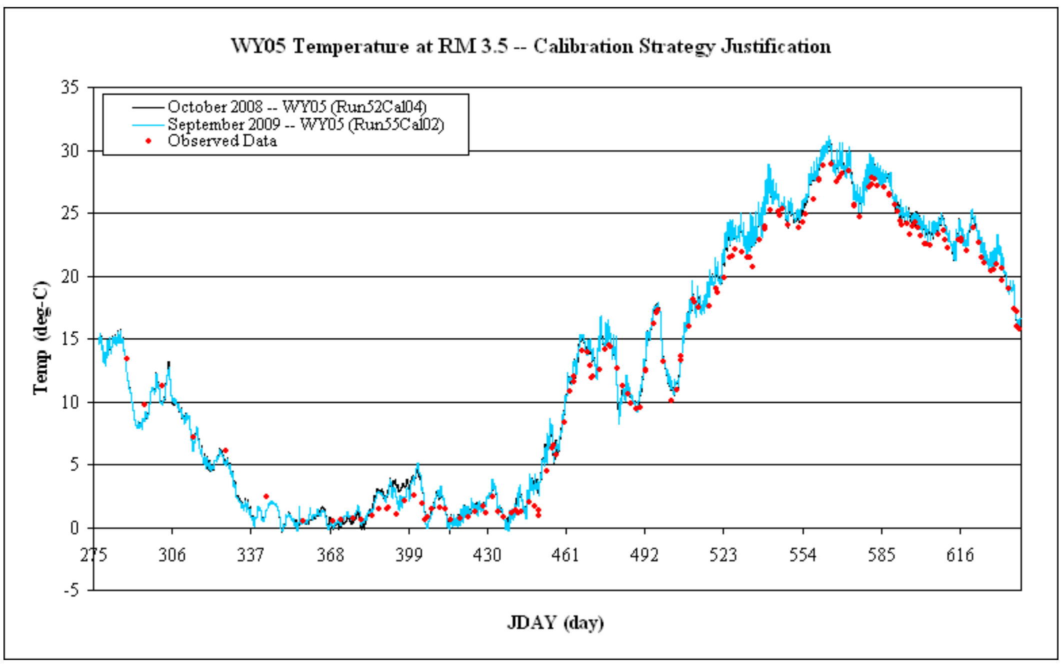
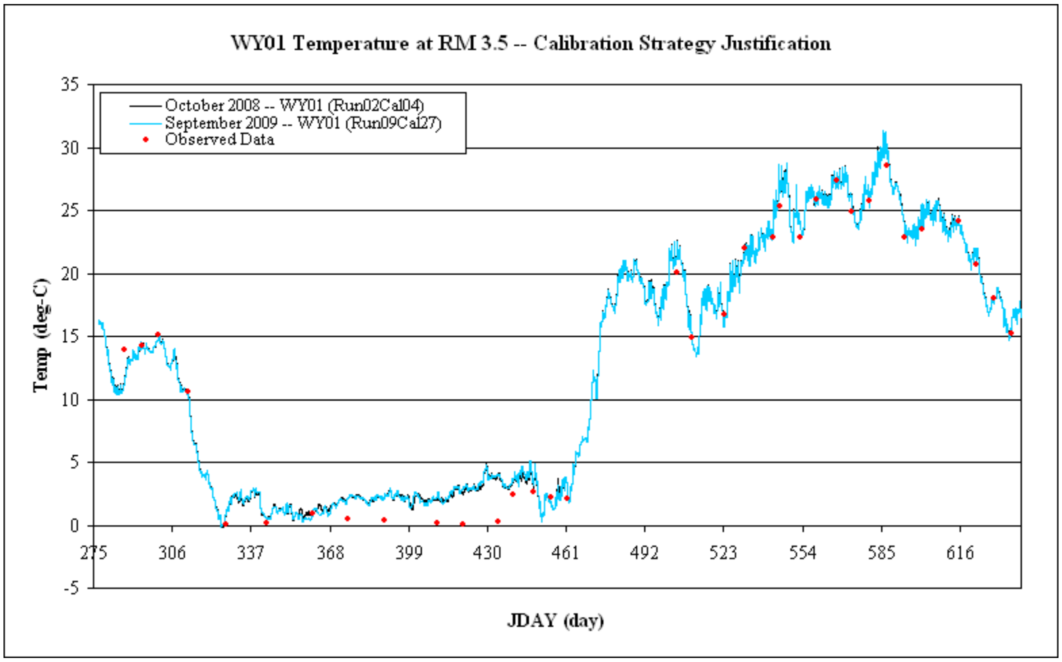
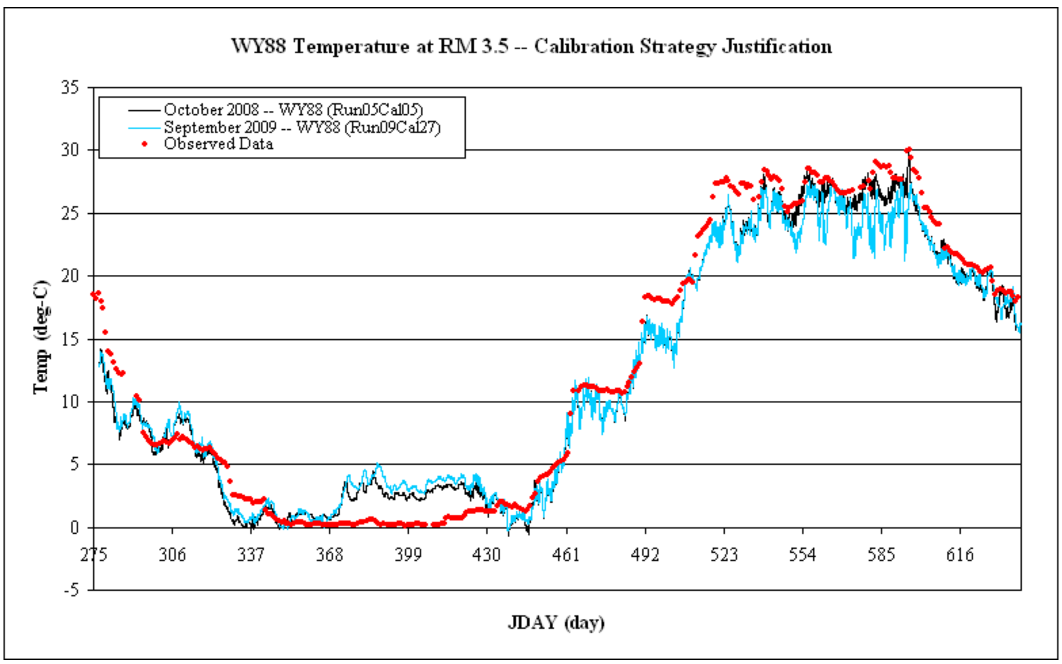


Figure 4. Calibration justification - Temperature at RM 3.5 (continued).





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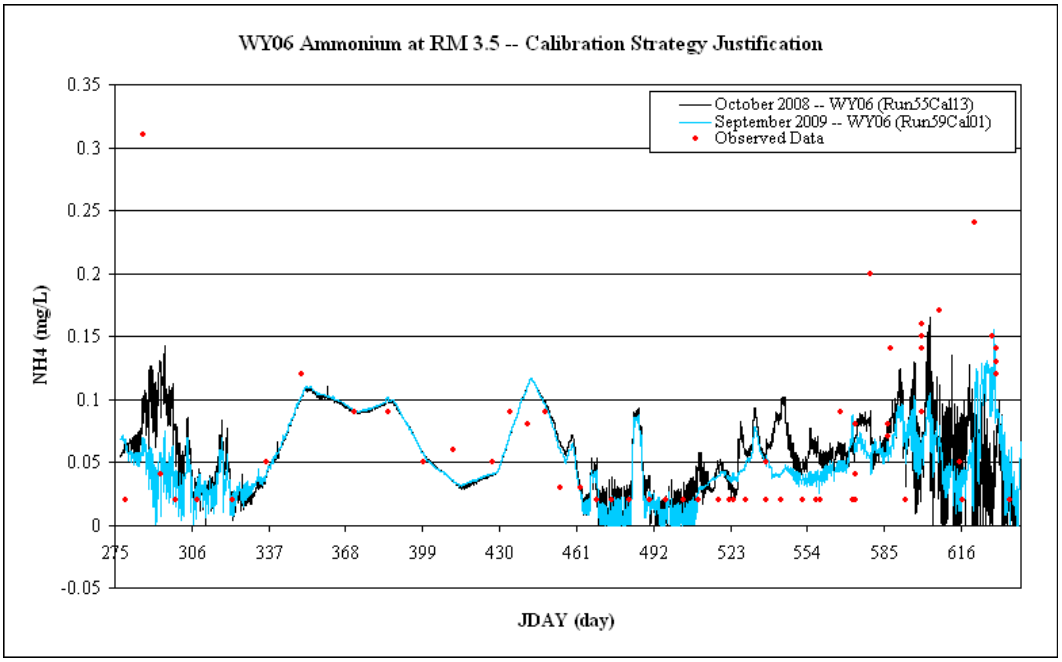


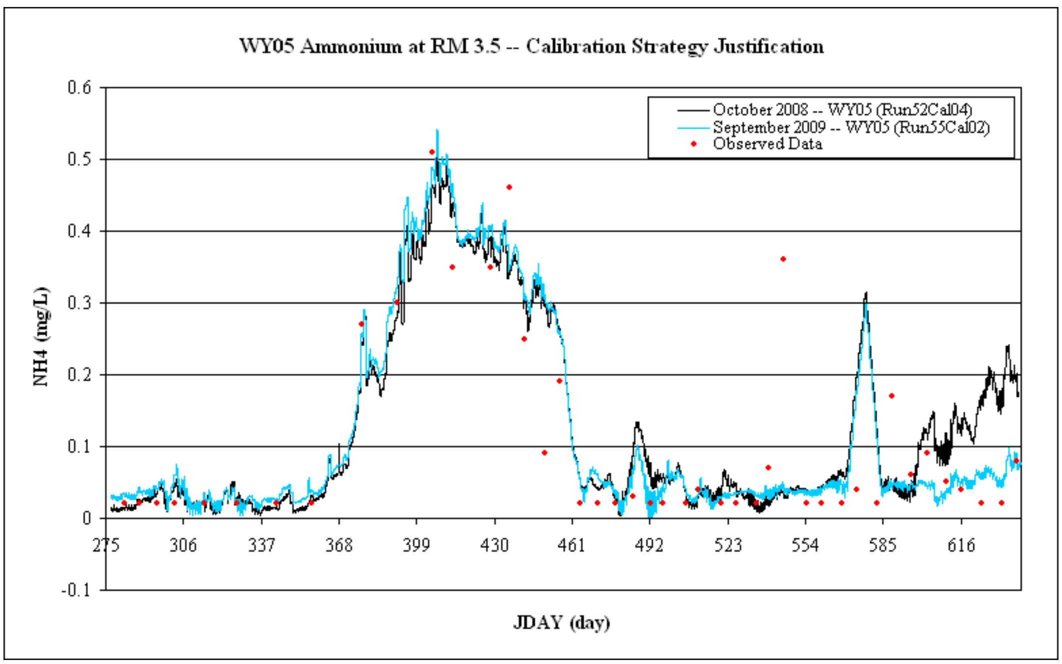


Figure 5. Calibration justification - Ammonium at RM 3.5 (continued).

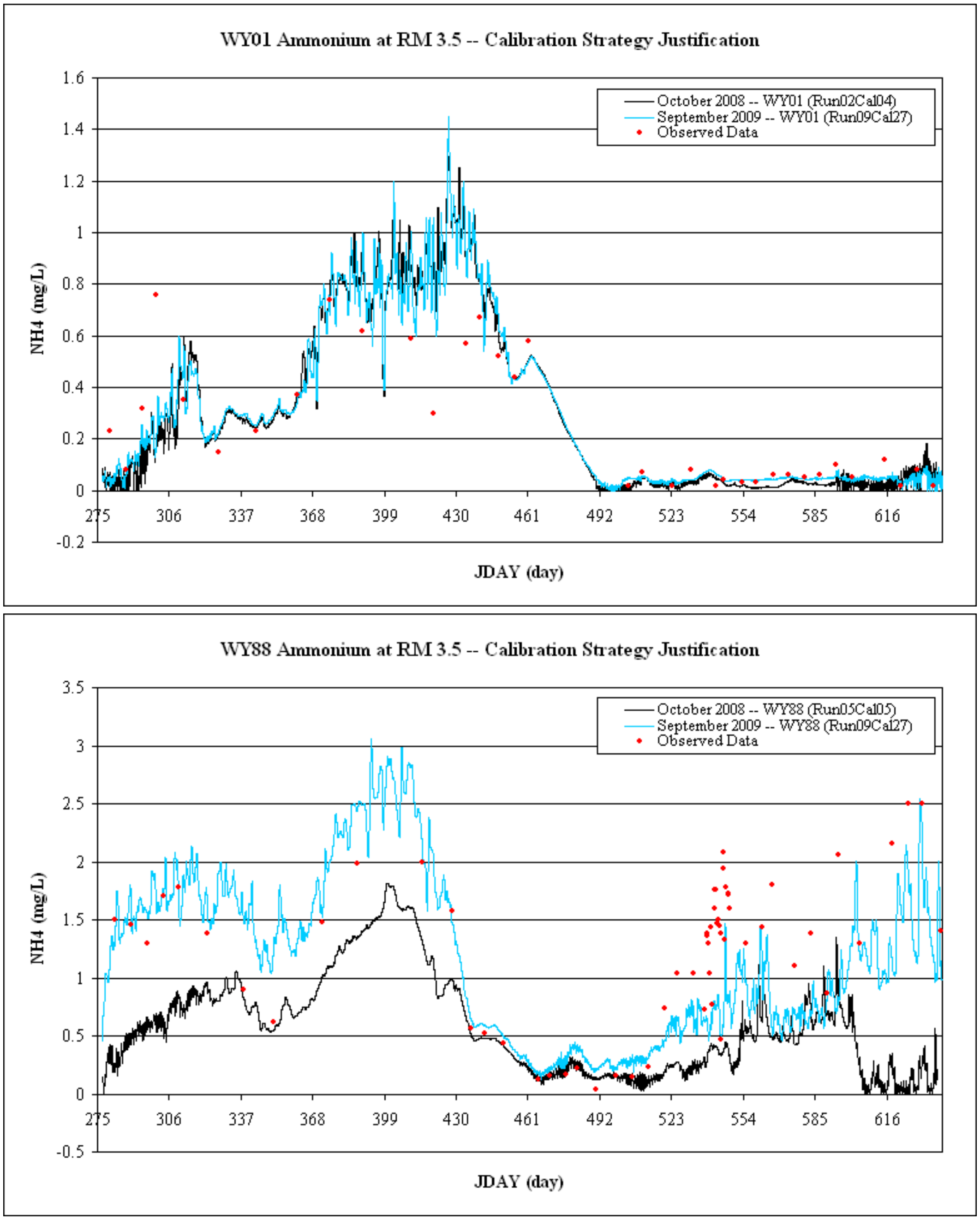
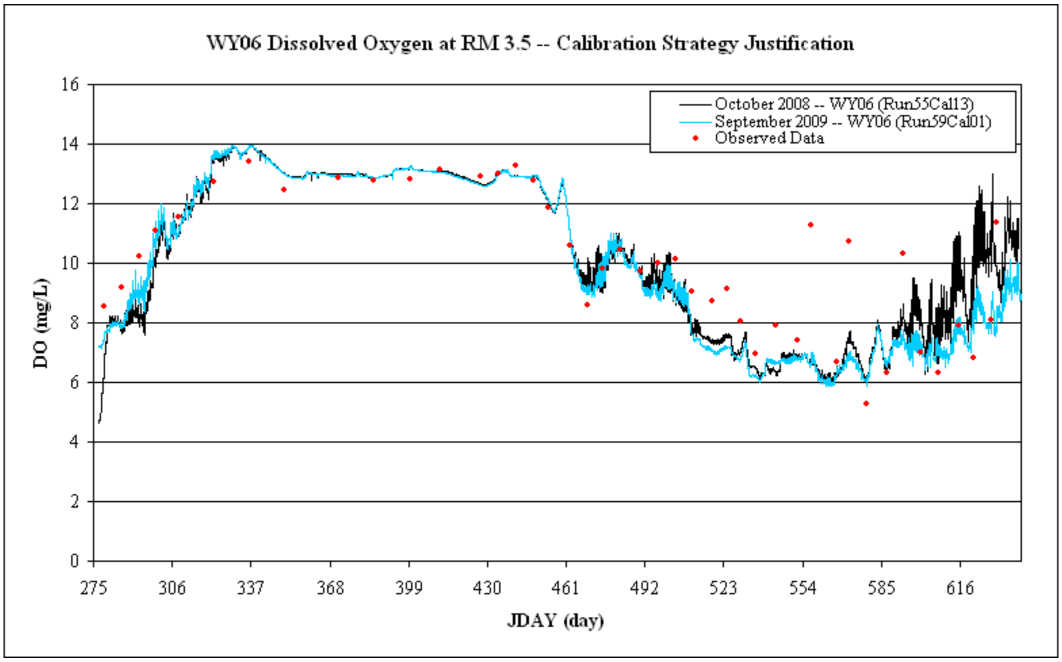


Figure 5. (concluded).



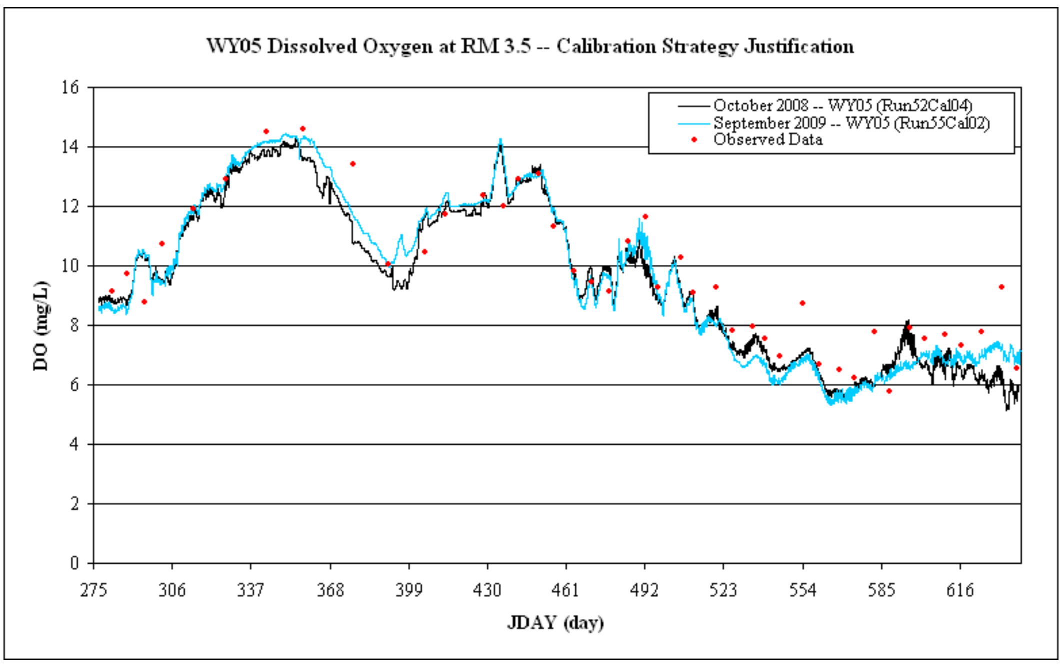
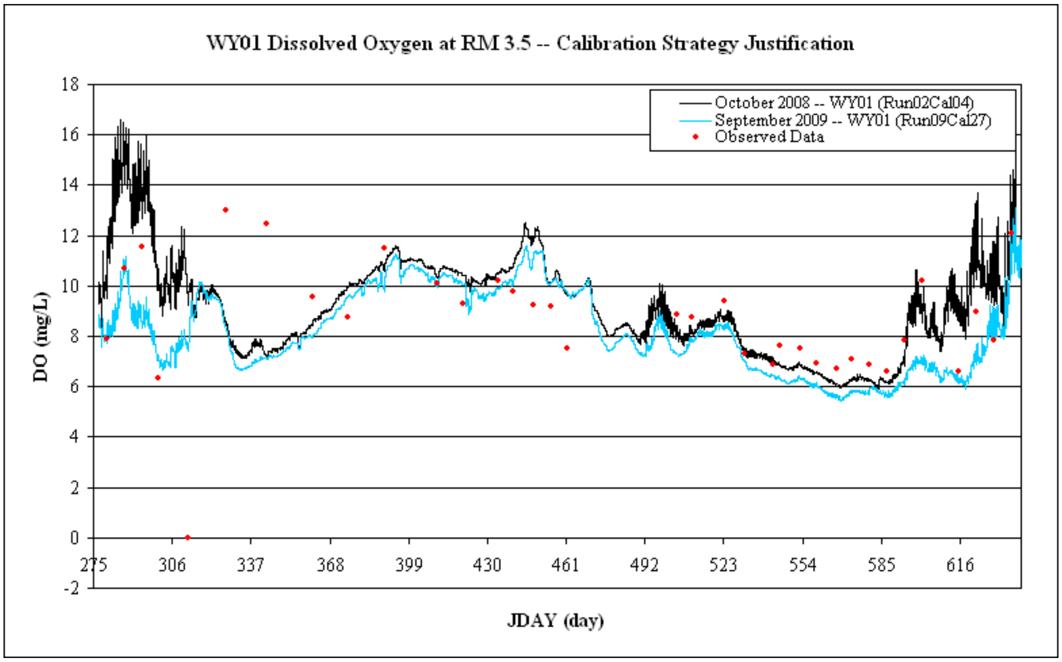


Figure 6. Calibration justification - Dissolved oxygen at RM 3.5 (continued).



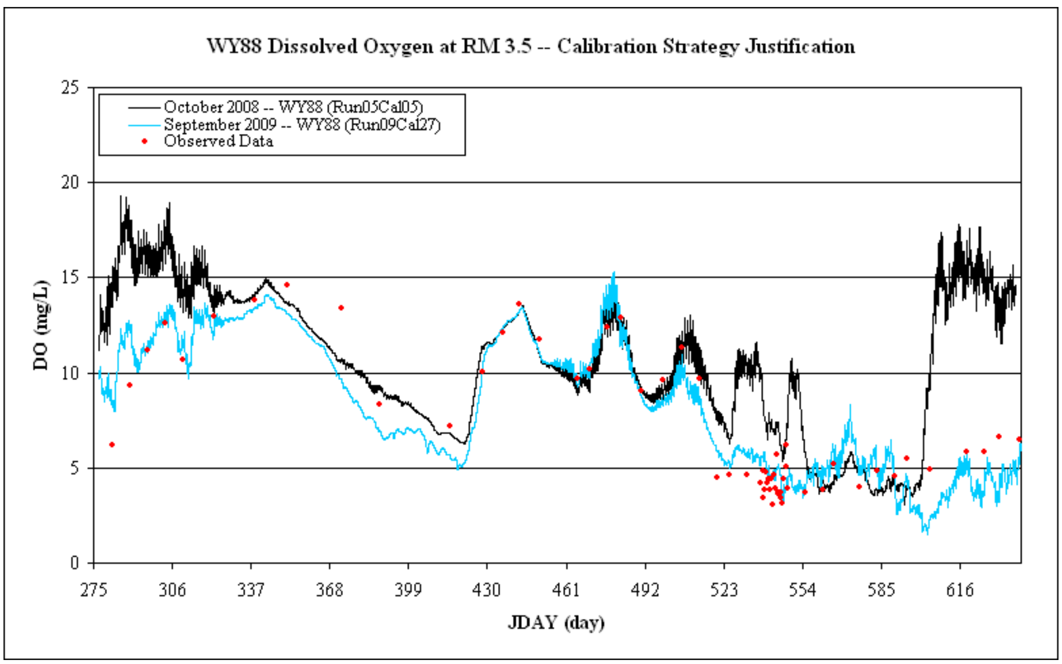
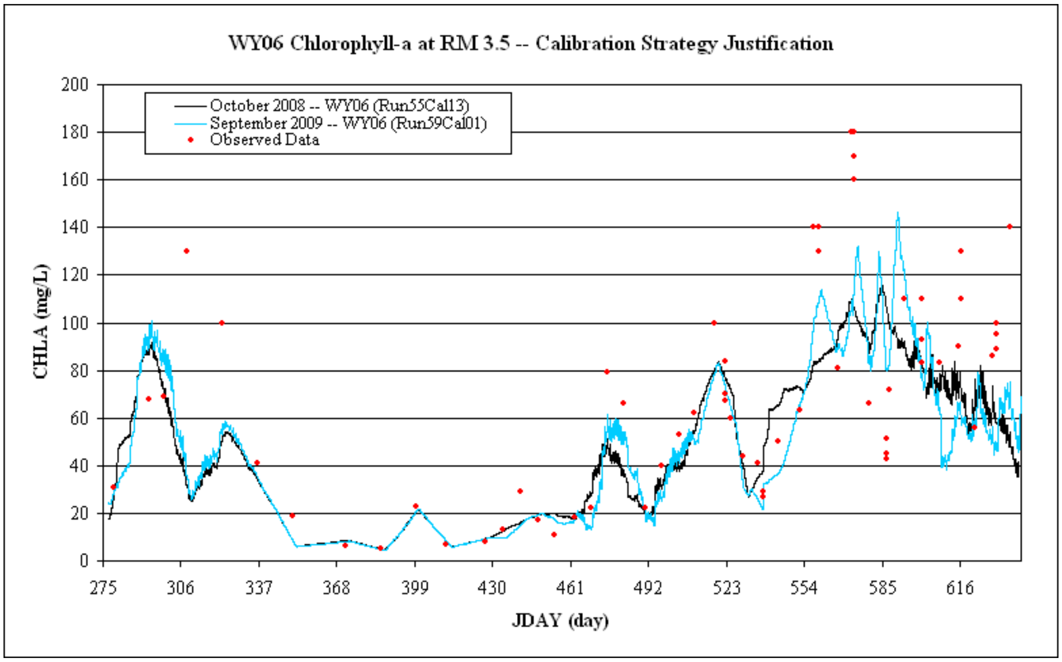


Figure 6. (concluded).



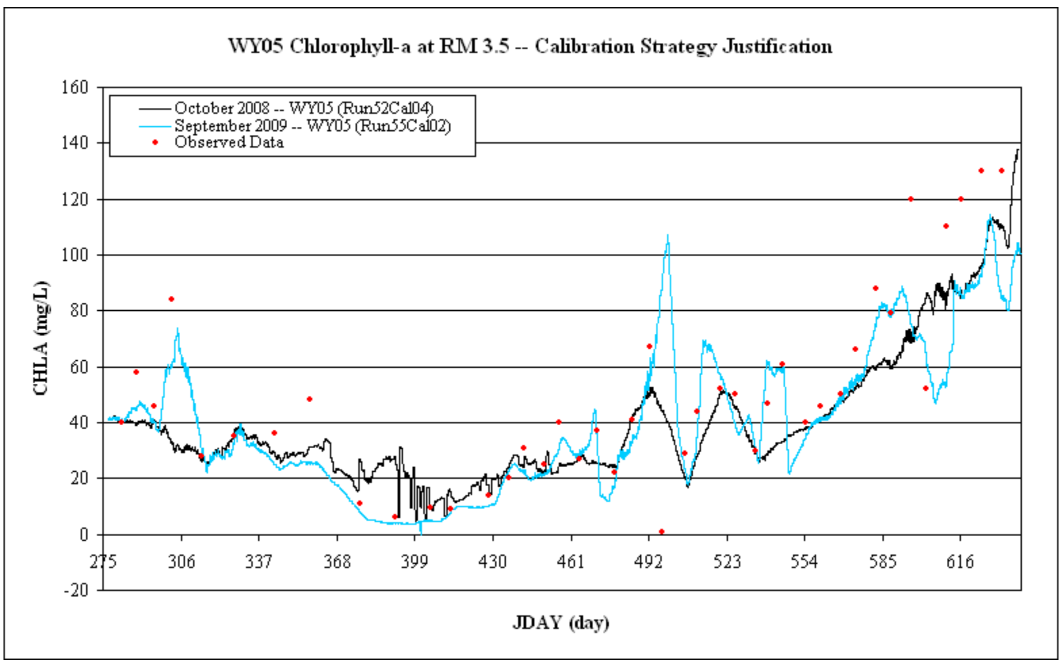


Figure 7. Calibration justification - Chlorophyll-a at RM 3.5 (continued).

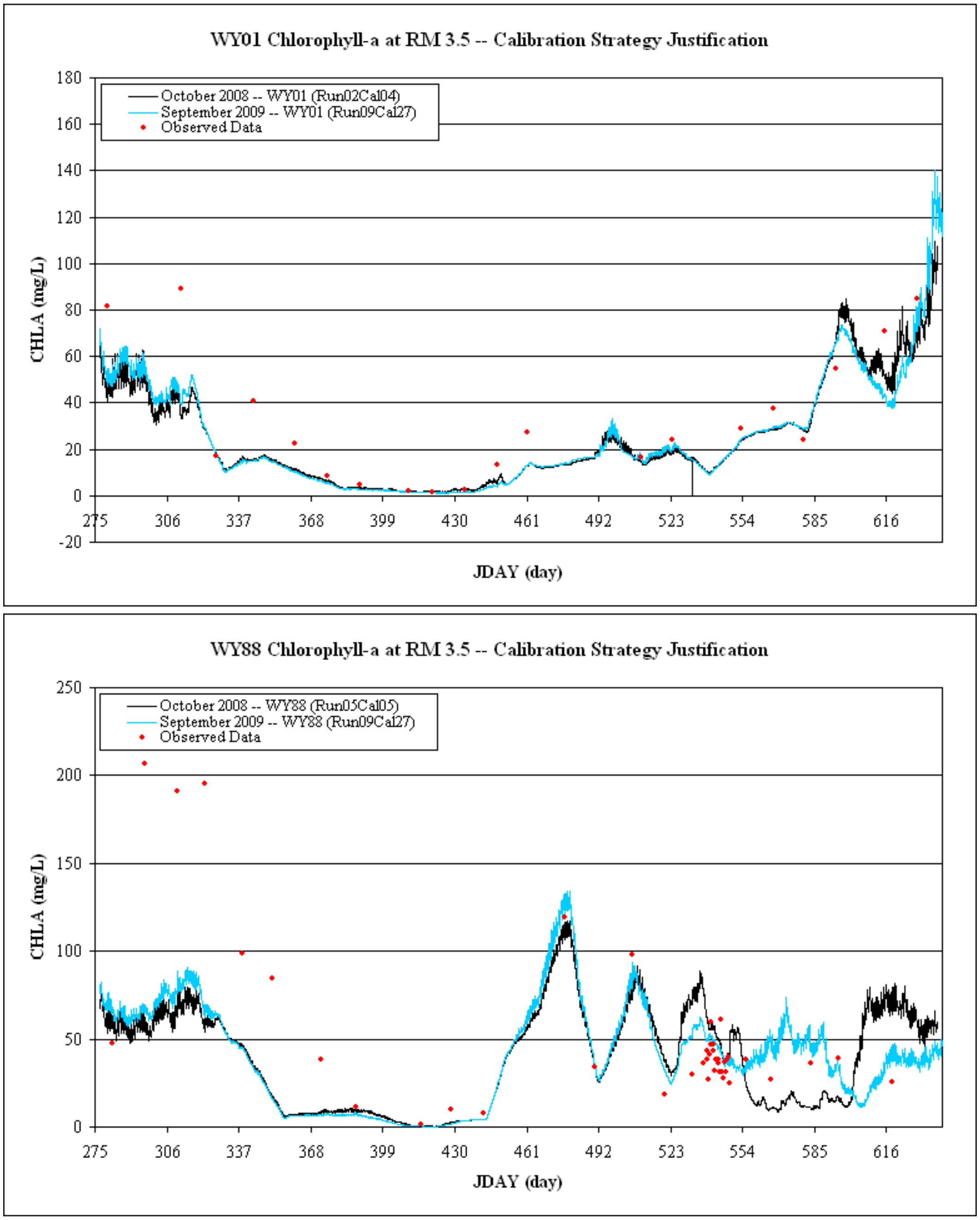


Figure 7. (concluded).

3 Data Analysis and Model Preparation

This chapter reviews the available data and how they were used to define the final calibration input files. W2 has several data requirements that must be met before simulations can begin:

1. Bathymetry of the river.
2. Flow, temperature, and water quality characteristics for boundaries, major tributaries, and point sources.
3. Stage data.
4. Meteorological conditions: air temperature, dew point temperature, wind speed, wind direction, cloud cover, and short wave solar radiation.

Model geometry

Bathymetry data

The bathymetry file for the LMRM was originally developed from a former bathymetry file used for a HEC-RAS model developed for the lower Minnesota River by the USACE, St. Paul District. The HEC-RAS model's grid consisted of cross-section data for RM 0.0 to RM 36.3. The data used for RM 0-15 consisted of 47 USACE cross sections from the late 1990s to 2000. For RM 14.5-35.92, 41 USGS cross sections obtained in 2000 were used. The grid was also very refined around structures; however, due to the lateral averaging of the W2 model, the grid was coarsened to fit within the W2 recommendations for a good grid.

Model grid development

The Minnesota River was split into two branches with Branch 1 extending from Jordan to Savage, MN, and Branch 2 extending from Savage to the mouth near St. Louis. The river was modeled with 90 longitudinal segments, varying in length from 134.0-2321.4 m, and 111 vertical segments, varying in height from 0.2-0.6 m. Each branch has a different slope. Table 2 describes of the branches in the river; the segment numbers also include the inactive (or "null") segments that start and end each branch. Figure 8 shows the longitudinal segments used in the model, along with the branch configuration.

Table 2. River characteristics.

Description	Branch	Segment Start	Segment End	# Segments	Slope
Jordan to Savage	1	1	52	52	0.00007
Savage to Fort Snelling	2	53	90	38	0.00002

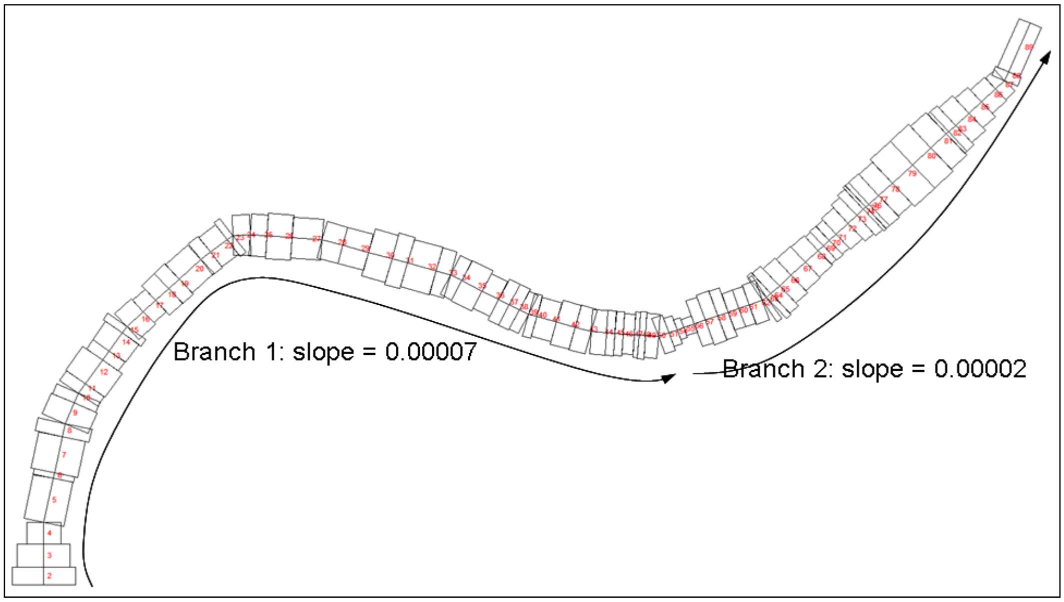


Figure 8. Longitudinal segments with branch configuration.

Tributary, point source, and withdrawal locations

Table 3 presents an abbreviated list of segment numbers in the LMRM bathymetry along with a brief description of the site located at the segment. For example, Blue Lake WWTP is located at segment 30 in the LMRM bathymetry.

Flow and elevations

Model boundaries

At the upstream boundary, located near Jordan (RM 39.4), mean daily flow was available from the United States Geological Survey (USGS) for every water year modeled. All available elevations were recorded or adjusted to datum NGVD 1929. Since the model is driven by flow, time-varying elevations were not used at the upstream boundary. At the downstream boundary, located at the mouth (RM 0.0), hourly elevations from the Mississippi River (RM 840.4) were available for most of 1988 and for 2001-2006. For 1988, where RM 840.4 elevations were unknown, data from RM 833.7 were used instead.

Table 3. Model segments of important locations.

Segment	Distance downstream (m)	Cumulative distance (m)	Cumulative distance (miles)	River Mile	Location
1	0.000	0.000	0.000	36.305	Upstream Boundary
2	754.250	754.250	0.468	35.836	Sand Creek, Jordan; Calibration Site
4	953.040	2710.010	1.683	34.622	Carver Creek
8	575.540	7111.860	4.416	31.888	Chaska Creek
11	506.080	9065.820	5.630	30.675	East Chaska Creek EC3 Outlet
12	1256.250	10322.070	6.410	29.895	East Chaska Creek EC1 Outlet
13	491.050	10813.120	6.715	29.590	1988 Chaska WWTP
23	311.540	18131.310	11.260	25.045	Calibration Site
27	1278.810	21936.440	13.623	22.682	Bluff Creek
28	907.380	22843.820	14.186	22.119	Riley Creek
30	1030.860	25014.940	15.534	20.771	Blue Lake WWTP
32	1268.030	26974.990	16.751	19.553	Purgatory Creek
41	885.510	32587.480	20.237	16.068	Eagle Creek
44	641.780	34760.280	21.586	14.719	1988 Savage WWTP
46	344.110	35473.100	22.029	14.276	Calibration Site
49	274.480	36431.010	22.624	13.681	Credit River
51	471.610	37453.930	23.259	13.046	Savage Gage (WSL)
52	0.000	37453.930	23.259	13.046	Branch 1 Downstream Boundary
53	0.000	37453.930	23.259	13.046	Branch 2 Upstream Boundary
55	286.950	38114.110	23.669	12.636	Nine Mile Creek
58	558.230	39663.220	24.631	11.674	Calibration Site
60	474.900	40606.770	25.217	11.088	Willow Creek
61	469.640	41076.410	25.508	10.796	Black Dog Lyndale Outfall
67	827.390	44140.390	27.411	8.894	Black Dog Withdrawal
68	757.370	44897.760	27.882	8.423	Calibration Site
71	400.780	45937.780	28.527	7.777	Black Dog Cedar Outfall
76	188.210	47848.630	29.714	6.591	Seneca WWTP
81	886.840	52340.270	32.503	3.801	Airport Outfall 040
82	134.600	52474.870	32.587	3.718	Airport Outfall 020
83	401.450	52876.320	32.836	3.469	Fort Snelling; Calibration Site
84	781.280	53657.600	33.321	2.983	Airport Outfall 030
90	0.000	58461.810	36.305	0.000	Downstream Boundary

The elevation and flow data available at RM 3.5 and RM 13.0 were used solely for model-to-data comparison. On January 22, 2004, the USGS deployed a stream-flow gaging station for the Minnesota River at Fort Snelling State Park. Before this date, mean daily flows at this location were estimated by MCES by lagging flows at Jordan by one day and multiplying them by 1.05. The formula was based on a comparison of measured flows at the two sites during 2004-2006 ($R^2 = 0.99$). Travel time can vary from hours at high flows to days at low flows, so this formula may not work well at extreme flows.

Table 4 shows the data sources for flow and elevation for various locations: the upstream boundary (RM 39.4), the downstream boundary (Mississippi RM 840.4/833.7), and two calibration locations in the Minnesota River (RM 13.0 and RM 3.5). Flow and elevation data were obtained from MCES; none of these files were modified. Figure 9 is a plot of all flow data used as input for the model at the upstream boundary for all seven water years. The blue vertical lines simply represent a water year division.

Tributaries

More than 40 streams of various sizes discharge to the lower Minnesota River, but monitoring has been limited to the larger tributaries. During 2004-2006, stream monitoring was enhanced and expanded for the model and other purposes, so that inputs for 11 tributaries could be compiled (Figure 2 and Table 5). Fewer data were available for 2001-2003, so inputs were compiled for only the four largest tributaries: Sand Creek, Carver

Table 4. Data sources for flow and elevation at the model boundaries.

River	Mile	Location and ID	Source	Variable	Water Year
Minnesota	39.4	Jordan USGS #05330000	USGS	Flow, Daily	1988, 2001-2006
Minnesota	39.4	Jordan NWSID JDNM5	USGS	Elevation, Hourly	2001-2006
Minnesota	39.4	Jordan NWSID JDNM5	NWS	Elevation, Daily	1988
Minnesota	13.0	Savage NWSID SAVM5	USACE	Elevation, Hourly	2001-2006
Minnesota	13.0	Savage NWSID SAVM5	NWS	Elevation, Daily	1988
Minnesota	3.5	Fort Snelling USGS #05330920	USGS	Flow, Daily	2004 (partial), 2005-2006
Minnesota	3.5	Fort Snelling USGS #05330920	USGS	Elevation, 15-minute	2004 (partial), 2005-2006
Mississippi	840.4	St. Paul NWSID STPM5	USACE	Elevation, Hourly	1988 (partial),2001-2006
Mississippi	833.7	South St. Paul NWSID SSPM5	USACE	Elevation, Daily	1988

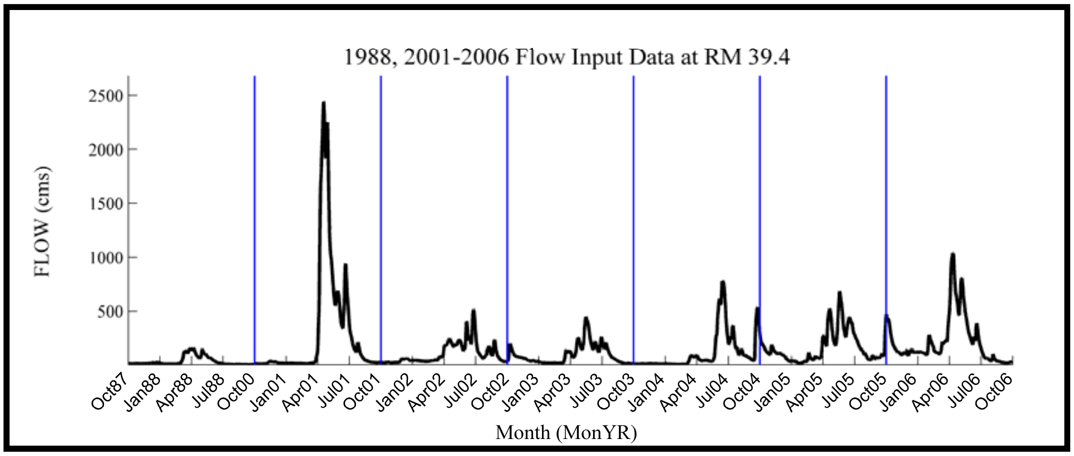


Figure 9. Flow input data for RM 39.4 for 1988, 2001-2006.

Table 5. Data sources and availability for tributary flows.

Tributary	River Mile	Source	Variable	Water Year
Sand Creek	35.5	MCES	Flow, Daily	2001-2006
Carver Creek	34.1	MCES	Flow, Daily	2001-2006[1]
Chaska Creek	31.6	Carver County	Flow, Daily	2004-2006
E. Chaska Creek, upstream	30.3	Carver County	Flow, Daily	2004-2006, partial
E. Chaska Creek, downstream	30.0	Carver County	Flow, Daily	2004-2006, partial
Bluff Creek	22.5	MCES	Flow, Daily	2004-2006
Riley Creek	22.3	MCES	Flow, Daily	2004, 2005 (partial)
Purgatory Creek	19.6	Barr Engineering	Flow, Daily	2004-2006
Eagle Creek	15.8	MCES	Flow, Daily	2004-2006
Credit River	13.7	MCES	Flow, Daily	2001-2006[1]
Nine Mile Creek	12.5	MCES	Flow, Daily	2001-2006
Willow Creek	11.0	MCES	Flow, Daily	2004-2006

[1] Gaps were filled as described in the text.

Creek, Credit River, and Nine Mile Creek. The MCES stream monitoring program began in 1989, so very few data were available for 1988. Conditions were extremely dry that year, so tributaries likely contributed light flows and loads to the lower Minnesota River. For these reasons, no tributaries were defined in the 1988 model.

The 11 tributaries that were monitored in 2004-2006 have a combined watershed area of approximately 1250 km^2 or roughly two-thirds of the total watershed area of the lower Minnesota River (1860 km^2). The four major tributaries represent a watershed area of approximately 1050 km^2 or nearly 60% of the total. No attempt was made to estimate flows or loads from unmonitored areas. James (2007) compiled annual loading budgets for the lower Minnesota River, and the 11 tributaries together contributed less than 10% of sediment and nutrient loads to the river in 2004-2006.

Kloiber (2006) used landscape variables to estimate water yield and pollutant loads from watersheds in the Twin Cities Metropolitan Area. Using his estimates for 2001-2003, the 11 tributaries listed in Table 5 delivered the following percentages of total flow and load from all tributaries to the Minnesota River downstream of Jordan: flow, 66%; TSS, 94%; TP, 77%; NO3, 76%; and TKN, 72%. By defining inputs for the 11 tributaries in the models for 2004-2006, the majority of flows and loads from local watersheds were represented.

Flow records for the four major tributaries (Sand, Carver, Credit, and Nine Mile) were complete for water years 2001-2006 with these exceptions:

- Bridge construction on Carver Creek halted monitoring from 5/1/03 to 9/30/04. Flows for this period were estimated by MCES with a SWAT watershed model.
- Flows at Credit River were missing for October-December 2000 and January-December 2002. MCES estimated flows by using a linear regression to flows at Sand Creek.

Flow records for the minor tributaries varied in quantity during WY 2004-2006. Records were complete for Bluff, Eagle, and Willow Creeks. Flows for Purgatory Creek were provisional but fairly complete; a few short gaps were filled via linear interpolation. Large gaps during the winter at Chaska Creek were filled with estimated base flow from the preceding fall. Large gaps in the records for East Chaska Creek (October to mid-March each year) and Riley Creek (much of May 2005 through 2006) were left blank, which the model interprets as zero.

Point sources

A total of six point sources with ten discharge or intake locations were identified for use in the LMRM project. Table 6 presents all point sources (discharge or intake) modeled in the various water years, along with their location and source for flow data. Within the W2 model, the discharge sources are modeled as tributaries and the Black Dog Plant intake is modeled as a withdrawal. Due to insufficient data at the airport outfalls, these were not modeled in water year 1988. MCES, Xcel Energy, and Metropolitan Airports Commission (MAC) provided flow, temperature, and water quality data from their monitoring stations.

Table 6. Data sources and availability for point source flows.

Discharge or Intake	River Mile	Source	1988	2001-2004	2005-2006
Chaska Wastewater Treatment Plant	29.4	MCES	Flow, Daily	Closed	Closed
Blue Lake Wastewater Treatment Plant	20.5	MCES	Flow, Daily	Flow, Daily	Flow, Daily
Savage Wastewater Treatment Plant	14.8	MCES	Flow, Daily	Closed	Closed
Black Dog Plant Lyndale Outfall	10.7	Xcel Energy	Flow, Daily	Flow, Hourly	Flow, Daily
Black Dog Plant Intake	8.8	Xcel Energy	Flow, Daily	Flow, Daily	Flow, Daily
Black Dog Plant Cedar Outfall	7.6	Xcel Energy	Flow, Daily	Flow, Hourly	Flow, Daily
Seneca Wastewater Treatment Plant	6.5	MCES	Flow, Daily	Flow, Daily	Flow, Daily
MSP Airport Outfall 040 (now SD012)[1]	4.1	MAC	None	Flow, Daily	Flow, Daily
MSP Airport Outfall 020 (now SD010)	3.8	MAC	None	Flow, Daily	Flow, Daily
MSP Airport Outfall 030 (now SD006)	3.0	MAC	None	Flow, Daily	Flow, Daily

[1] This flow was rerouted to RM 3.8 in 2005 but the model does not reflect this change.

Temperature

Model boundaries

For WY 2004-2006, temperature at the upstream boundary was defined with MCES continuous temperature at Jordan when available. Gaps in the RM 39.4 record were filled with mean daily or hourly temperature from the Xcel Energy monitor at RM 11.5. During WY 1988 and WY 2001-2003, MCES collected only weekly grab measurements of temperature at RM 39.4, so mean hourly temperature from the Xcel Energy monitor at RM 11.5 were applied to the upstream boundary. The temperature input files for 2004 were created using mean hourly temperature at RM 39.4 (when available) or RM 11.5. The input files for 2005 were created using mean daily temperature at RM 11.5 for the first six months and mean hourly temperature at RM 39.4 for the last six months. For water year 2006, continuous temperature data at RM 39.4 were aggregated to 15-min data for the input files.

For the temperature input file at the downstream boundary, a small number of sample dates (no more than four data samples) were considered sufficient to define the input file for every water year. Temperature data at RM 25.1, RM 14.3, RM 8.5, and RM 3.5 were used as calibration data for the model. Table 7 presents the locations and sources for temperature data, and Figure 10 provides a time-series plot of temperature at RM 39.4 as defined in the models for 1988 and 2001-2006.

Table 7. Data sources and availability for river temperature.

Location	River Mile	Data Source	Variable, Resolution	Water Year
Minnesota River at Jordan	39.4	MCES	Temperature, Weekly Grab	1988, 2001-2006
Minnesota River at Jordan	39.4	MCES	Temperature, Continuous	2004-2006 (partial record)
Minnesota River at Shakopee	25.1	MCES	Temperature, Weekly Grab	1988, 2001-2006
Minnesota River at Savage	14.3	MCES	Temperature, Weekly Grab	1988, 2001-2006
Minnesota River at I35W Bridge	11.5	Xcel Energy	Temperature, Mean Daily & Mean Hourly	1988 (daily), 2001-2006 (daily & hourly)
Minnesota River at Black Dog	8.5	MCES	Temperature, Weekly Grab	1988, 2001-2006
Minnesota River at Fort Snelling	3.5	MCES	Temperature, Weekly Grab	1988, 2001-2006
Minnesota River at Fort Snelling	3.5	MCES	Temperature, Continuous	1988, 2001-2006

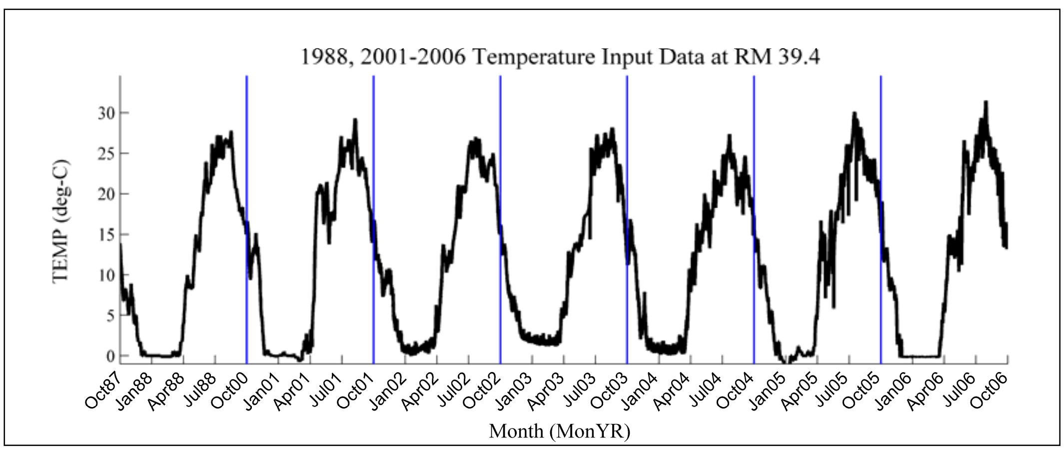


Figure 10. Temperature input data for RM 39.4 for 1988, 2001-2006.

Tributaries

For water year 1988, since flow was not defined for any tributaries, temperature was also not defined for any tributaries. For 2001-2003, temperature was only defined for the four major tributaries (Sand Creek, Carver Creek, Credit River, and Nine Mile Creek), but complete temperature data were only available for Nine Mile Creek. For the other three tributaries defined in 2001-2003, temperatures were defined by MCES based on regression relationships to temperatures at Nine Mile Creek. Temperature monitoring started in Sand Creek on March 19, 2003 and in Credit River on January 1, 2003. For 2004-2006, temperature inputs were defined for all 11 tributaries. Where data were not available, temperature from a nearby or similar tributary was used by MCES to estimate inputs. Riley Creek was not modeled in 2006 due to the lack of flow data. Table 8 presents all tributaries modeled in the various water years, along with their locations and sources for temperature data. In general, all continuous data reported for the tributaries were aggregated into mean hourly temperatures for input into the model.

Table 8. Data sources and availability for tributary temperature.

Location	River Mile	Source	Variable, Resolution	Water Year
Sand Creek	35.5	MCES	Temperature, Continuous	2003-04 (partial), 2005-06
Carver Creek	34.1	MCES	Temperature, Continuous	2005 (partial) and 2006 (full)
Chaska Creek	31.6	Carver County		None available
E. Chaska Creek	30.3, 30.0	Carver County	Temperature, Continuous	2005 and 2006 (partial)
Bluff Creek	22.5	MCES	Temperature, Continuous	2004-06 (partial), 2005 (full)
Riley Creek	22.3	MCES	Temperature, Continuous	2004
Purgatory Creek	19.6	Barr Engineering	Temperature, Continuous	2004 (partial), 2005-2006
Eagle Creek	15.8	MCES	Temperature, Continuous	2004-2006
Credit River	13.7	MCES	Temperature, Continuous	2004-2006
Nine Mile Creek	12.5	MCES	Temperature, Continuous	2001-2006
Willow Creek	11.0	MCES	Temperature, Continuous	2004-2006

Point sources

Temperature was monitored at least daily in 1988 and 2001-2006 at each of the major point sources. Mean hourly flow and temperature at the Black Dog GP were available for water years 2001-2004. CE-QUAL-W2 produced better results using a higher frequency of temperature and flow data. Table 9 presents all point sources modeled in the various water years, along with their locations and sources for temperature data.

Water quality

Several W2 state variables were defined for the model. Descriptions and brief definitions are included in Table 10. Detailed descriptions of how each state variable was handled for use in the input files will be discussed.

Table 9. Data sources and availability for point source temperature.

Location	River Mile	Data Source	Variable, Resolution	Water Year
Chaska WWTP	29.4	MCES	Temperature, Daily Grab	1988 (closed before 2001)
Blue Lake WWTP	20.5	MCES	Temperature, Daily Grab	1988, 2001-2006
Savage WWTP	14.8	MCES	Temperature, Daily Grab	1988 (closed before 2001)
Black Dog GP at Lyndale Outfall	10.7	Xcel Energy	Temperature, Mean Daily & Mean Hourly	All years (daily), 2001-2004 (daily & hourly)
Black Dog GP at Cedar Outfall	7.6	Xcel Energy	Temperature, Mean Daily & Mean Hourly	All years (daily), 2001-2004 (daily & hourly)
Seneca WWTP	6.5	MCES	Temperature, Daily Grab	1988, 2001-2006
MSP Airport Outfall SD012	4.1	MAC	Temperature, Daily Grab	2001-2006
MSP Airport Outfall SD010	3.8	MAC	Temperature, Daily Grab	2001-2006
MSP Airport Outfall SD006	3.0	MAC	Temperature, Daily Grab	2001-2006

Table 10. CE-QUAL-W2 state variables as defined in the LMRM.

ID	Description	Definition
TDS	Total dissolved solids	TDS or estimated from conductivity
ISS	Inorganic suspended solids	River & Tribs: Total SS – volatile SS (TSS-VSS) WWTPs & MSP: TSS * estimated ISS
PO4	Bioavailable phosphorus	River & Tribs: Soluble reactive P (SRP) WWTPs: SRP or estimated from total P
NH4	Ammonium nitrogen	Ammonium N
NO3	Nitrate-nitrite nitrogen	Nitrate N + nitrite N (NO3 + NO2)
DSI	Dissolved silica	Soluble reactive silica
LDOM	Labile dissolved organic matter	River & Tribs : 0.15 * (dissolved organic carbon / 0.45) WWTPs & MSP: Set to zero
RDOM	Refractory dissolved organic matter	River & Tribs: 0.85 * (dissolved organic carbon / 0.45) WWTPs & MSP: Set to zero
LPOM	Labile particulate organic matter	River & Tribs: 0.15 * (DOM + VSS – algal biomass) WWTPs & MSP: Set to zero
RPOM	Refractory particulate organic matter	River & Tribs: 0.85 * (DOM + VSS – algal biomass) WWTPs & MSP: Set to zero
CBOD1-CBOD6	Carbonaceous biochemical oxygen demand	River & Tribs: Set to zero. Replace with OM groups. WWTPs & MSP: CBOD5 * CBODU:CBOD5
ALG1	Diatom, biomass	Chlorophyll-a (ug/L) * 0.0675 * % diatoms
ALG2	Blue-green algae, biomass	Chlorophyll-a (ug/L) * 0.0675 * % blue-greens
ALG3	Other algae, biomass	Chlorophyll-a (ug/L) * 0.0675 * % other algae
DO	Dissolved oxygen	River: DO measured in field or lab WWTPs: DO measured in effluent Tributaries: Estimated from temperature

Sampling frequencies for the variables in Table 10 varied from daily to monthly depending on the site, variable, season, and year. Very often, however, the constituents were not sampled at the same frequency, resulting in many data gaps. Though CE-QUAL-W2 is capable of interpolating data, the model cannot interpolate the missing value of one constituent when other constituents were sampled. W2 would assume that the missing value is zero; in most cases, this is an incorrect assumption. In these cases, ERDC used a Microsoft Excel Add-In developed by DigDB (http://digdb.com) to quickly and efficiently linearly interpolate any data gaps in the water quality data samples. For the tributaries, MCES provided monthly average concentrations estimated with the FLUX program (Walker 1996). Gaps in the tributary records were filled with estimates from nearby tributaries with similar land use.

Where data were unavailable to define state variables, MCES estimated the inputs using the best data available and professional judgment. For example, dissolved silica was not monitored in the earlier years, so mean monthly concentrations from 2004-2006 were applied to the upstream boundary for 1988 and 2001-2003.

Total dissolved solids (TDS)

TDS was routinely monitored at most stations. However, when TDS was unknown, it was estimated from conductivity. All input files (model boundaries, tributaries, and point sources) were developed in the same manner for TDS. TDS was not monitored at the airport outfalls; for these files, TDS was set to 0.0 mg/L. Figure 11 is a plot of the total dissolved solids used in the LMRM for the upstream model boundary at RM 39.4 near Jordan. The blue vertical lines are included to highlight the division between individual water years.

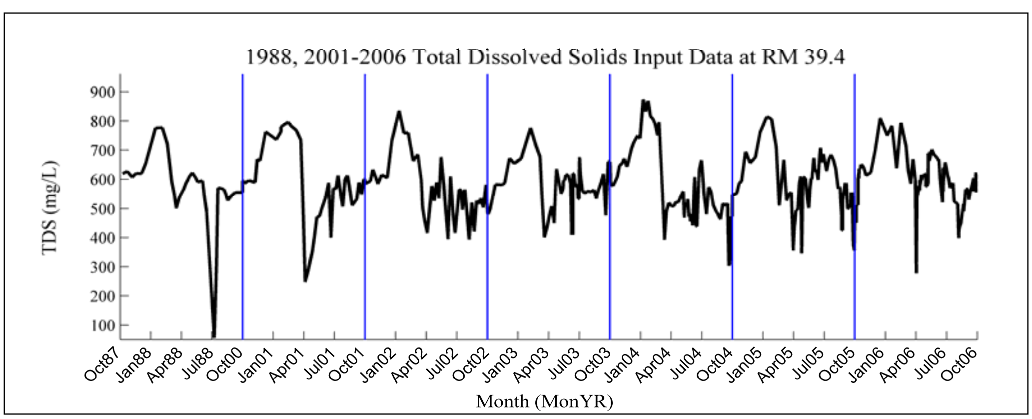


Figure 11. TDS input data for RM 39.4 for 1988, 2001-2006.

Inorganic suspended solids (ISS)

ISS was not directly monitored; instead, for the model boundaries, tributaries, and the Black Dog outfalls, it was estimated as total suspended solids less the volatile suspended solids. For the wastewater treatment plants and airport outfalls, ISS was estimated by multiplying the measured TSS by the fraction ISS/TSS determined from available paired samples (Blue Lake, 0.34; Seneca, 0.19; and airport, 0.43). Figure 12 is a plot of inorganic suspended solids input data used in the LMRM at RM 39.4.

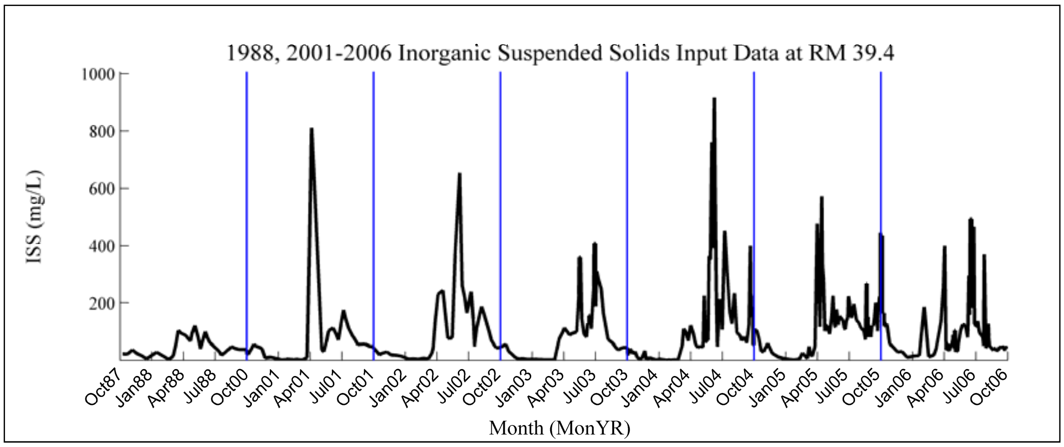


Figure 12. ISS input data for RM 39.4 for 1988, 2001-2006.

Bioavailable phosphorus (PO_4)

Bioavailable phosphorus or orthophosphate (PO4) was measured as soluble reactive phosphorus in the laboratory, and it was available for the model boundary conditions and the tributary input files. For the wastewater treatment plants, when PO4 was not available, it was estimated from total phosphorus based on measured PO4/TP fractions from WY 2004-2006 (Blue Lake, 0.90; Seneca, 0.81). The fraction was based on a linear regression that tends to overpredict PO4 at low TP concentrations, and the mean PO4/TP ratio was approximately 0.60 during this period. PO4 was infrequently monitored at the airport outfalls and typically low; for these files, PO4 was input as 0.0 mg/L. Figure 13 is a plot of orthophosphate input data used in the LMRM for RM 39.4.

Ammonium nitrogen (NH_4)

MCES routinely monitored NH4 at all monitoring stations. When the samples measured were below the detection limit, the values were set to 0.02 mg/L. Figure 14 is a plot of ammonium nitrogen input data used in the LMRM at RM 39.4.

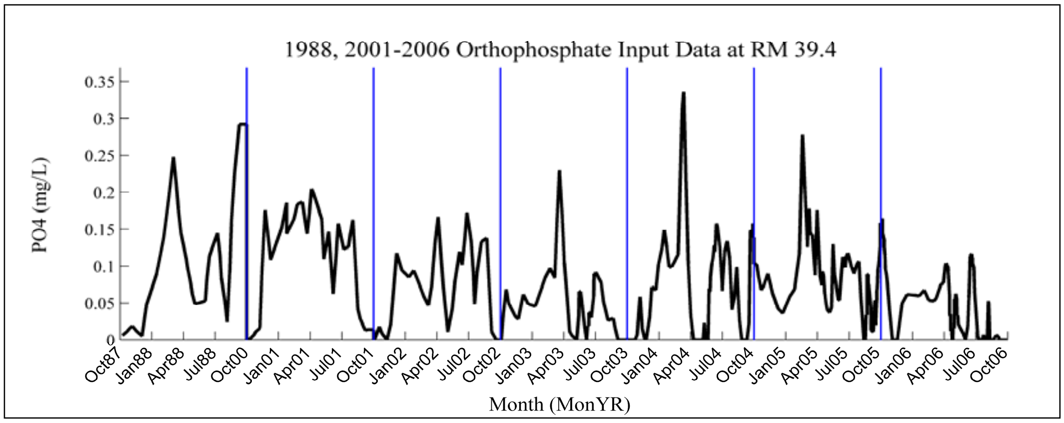


Figure 13. PO4 input data for RM 39.4 for 1988, 2001-2006.

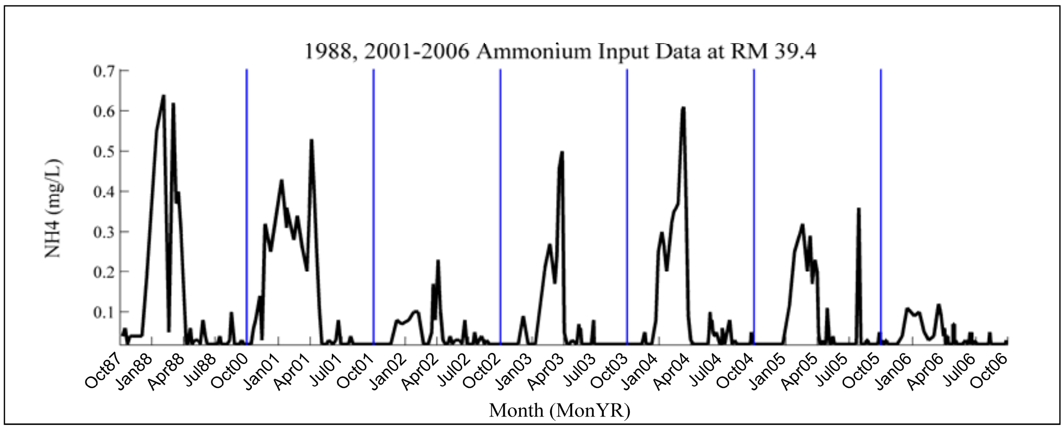


Figure 14. NH4 input data for RM 39.4 for 1988, 2001-2006.

Nitrate-nitrite nitrogen (NO_3)

NO3 and NO2 were both monitored at most stations. For the airport outfalls, NO3 was input as 0.0 mg/L because it was not monitored. Figure 15 is a plot of nitrate-nitrite nitrogen input data used in the LMRM at the upstream boundary for RM 39.4.

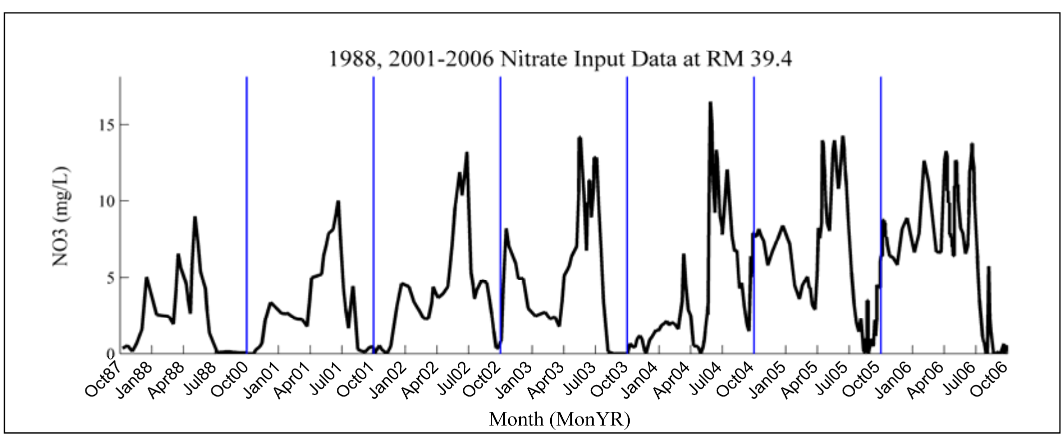


Figure 15. NO3-NO2 input data for RM 39.4 for 1988, 2001-2006.

Dissolved silica (DSI)

DSI was monitored as soluble reactive silica at most stations except the airport outfalls. For the airport input files, DSI was input as 0.0 mg/L. Figure 16 is a plot of dissolved silica input data used in the LMRM for RM 39.4. For water years 1988 and 2001-2003, where DSI data were not monitored, the monthly mean concentrations from 2004-2006 were used.

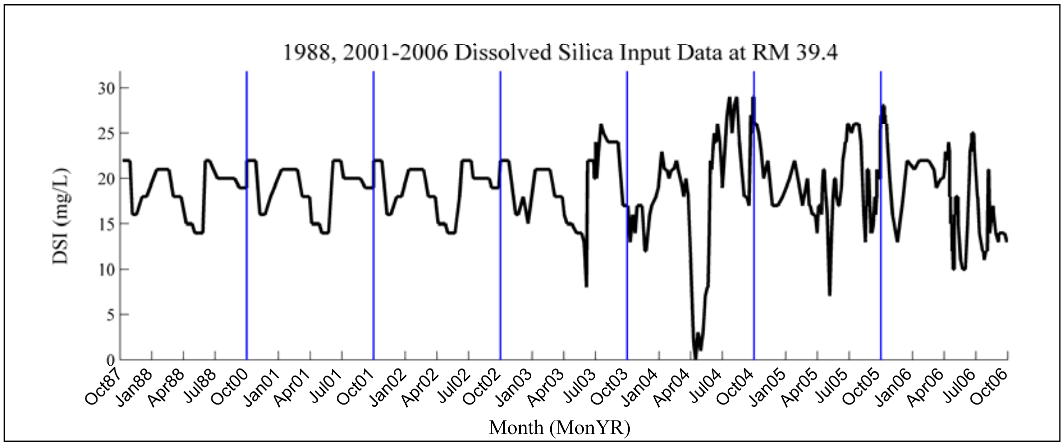


Figure 16. DSI input data for RM 39.4 for 1988, 2001-2006.

Organic matter (OM) and carbonaceous biochemical oxygen demand (CBOD)

Non-living organic matter can be defined in CE-QUAL-W2 either as organic matter (OM) expressed as biomass in terms of labile/refractory dissolved/particulate organic matter (LDOM, RDOM, LPOM, RPOM) or carbonaceous biochemical oxygen demand (CBOD). That is, non-living organic matter is input exclusively as OM or exclusively as CBOD for each station. Both are stoichiometrically associated with organic carbon, nitrogen, and phosphorus in the model. CBOD was monitored at all of the monitoring stations; however, organic matter was not. Dissolved organic carbon and volatile suspended solids, which can be used to estimate organic matter, were sampled or could be estimated at most of the MCES monitoring sites. Organic matter estimates were needed to specify four model constituents (LDOM, RDOM, LPOM, RPOM) at the river and tributary sites. CBOD was used to define organic matter at the wastewater treatment plants and airport stormwater outfalls in order to track these sources for regulatory purposes. Table 11 provides the reader with a listing of the BOD groups that were modeled in the LMRM.

Table 11. BOD groups defined in the LMRM.

BOD group	Site	Final Inputs
BOD1	Upstream Boundary (Jordan)	Set to 0.0. Used OM and ALG.
BOD2	Blue Lake WWTP (Chaska & Savage in 1988)	Used CBOD. Set OM and ALG to 0.0.
BOD3	Seneca WWTP	Used CBOD. Set OM and ALG to 0.0.
BOD4	Airport Stormwater Outfalls	Used CBOD. Set OM and ALG to 0.0.
BOD5	Tributaries and Black Dog GP Outfalls	Set to 0.0. Used OM and ALG.
BOD6	Downstream Boundary	Set to 0.0. Used OM and ALG.

In the initial calibration, CBOD inputs were defined for the two river boundaries and tributaries (BOD groups 1, 5, and 6) in addition to the WWTPs and airport (BOD groups 2, 3, and 4). However, CE-QUAL-W2 allows simulation of living (algae) and nonliving organic matter. Algae are modeled separately and become part of the organic matter budget when they die and excrete. Also, care must be taken not to 'double-count' any organic matter. Based on this information, CBOD was set to 0.0 mg/L at the river boundaries, tributaries, and Black Dog outfalls in the final calibration, and nonliving organic matter inputs were added. Also, since algae were modeled separately, the algal contribution was subtracted from the total organic matter. An approach similar to Lung (1993) was used to estimate the OM groups. Detailed information on how these inputs were developed can be found in Appendix A.

Dissolved organic carbon was not monitored in 1988 and 2001-2003. For these years, everywhere that volatile suspended solids were measured, DOC was assumed to be 6.0 mg/L. Once that assumption was made, organic matter estimates were made exactly as described in Appendix A. From Figures 17 and 18, the impact of assuming a constant DOC for 1988 and 2001-2003 can be seen.

Figures 17-20 are plots of labile dissolved organic matter, refractory dissolved organic matter, labile particulate organic matter, and refractory particulate organic matter input data, respectively, used at the upstream boundary in the LMRM model.

Algae: Diatoms (ALG_1), bluegreens (ALG_2), and others (ALG_3)

Three algal groups are modeled in the LMRM: diatoms (ALG1), bluegreens (ALG2), and others (ALG3). Total algal biomass estimated from pheophytin-corrected chlorophyll *a* was available for most monitoring

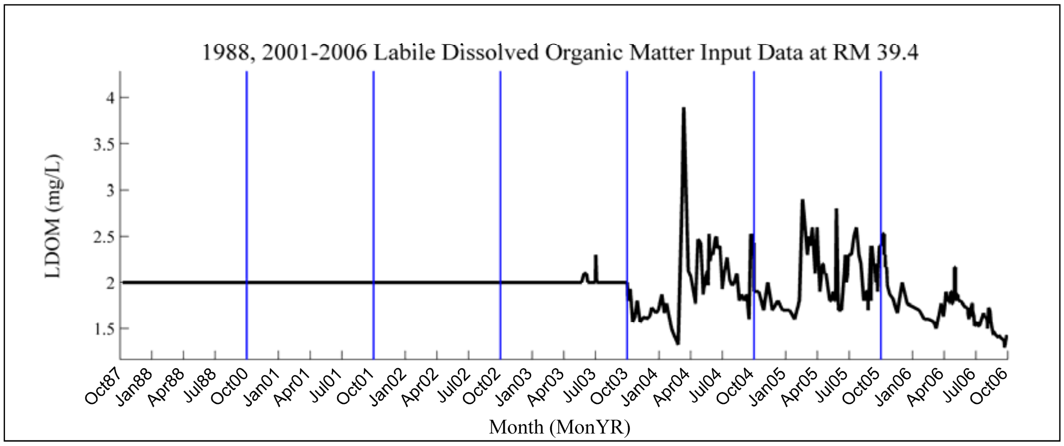


Figure 17. LDOM input data for RM 39.4 for 1988, 2001-2006.

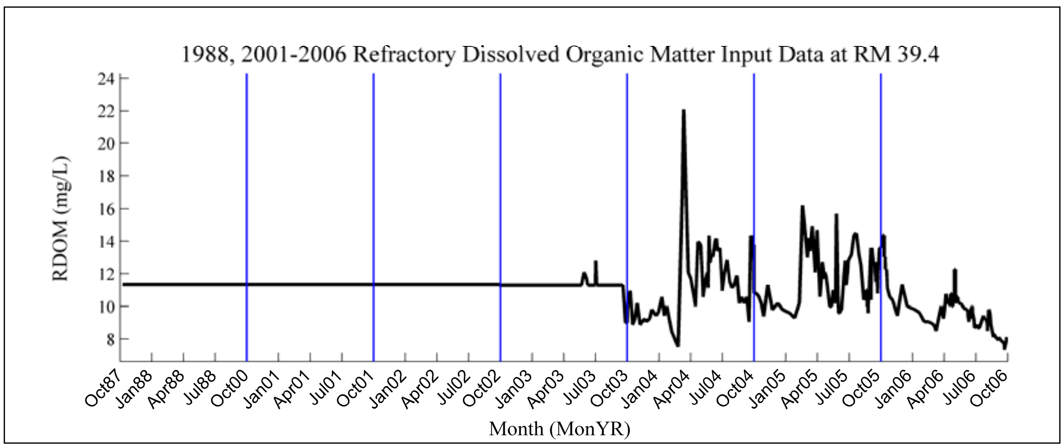


Figure 18. RDOM input data for RM 39.4 for 1988, 2001-2006.

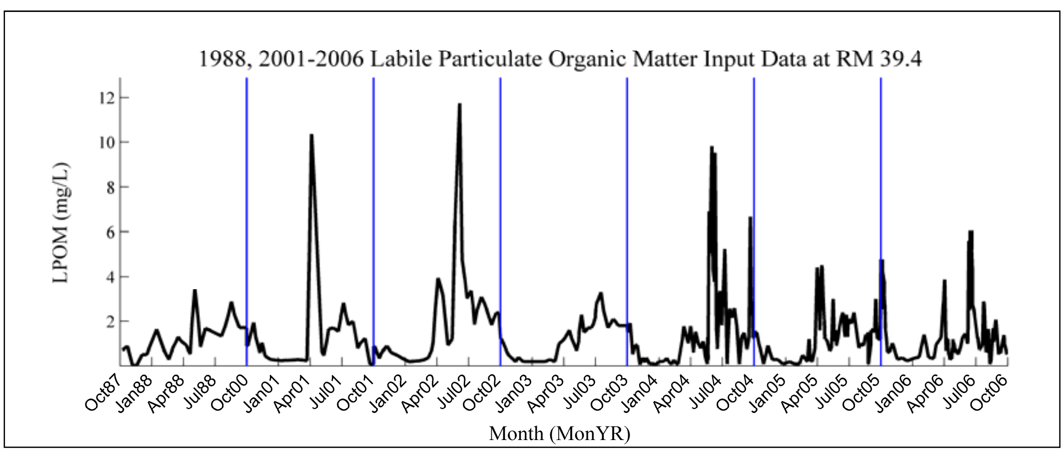


Figure 19. LPOM input data for RM 39.4 for 1988, 2001-2006.

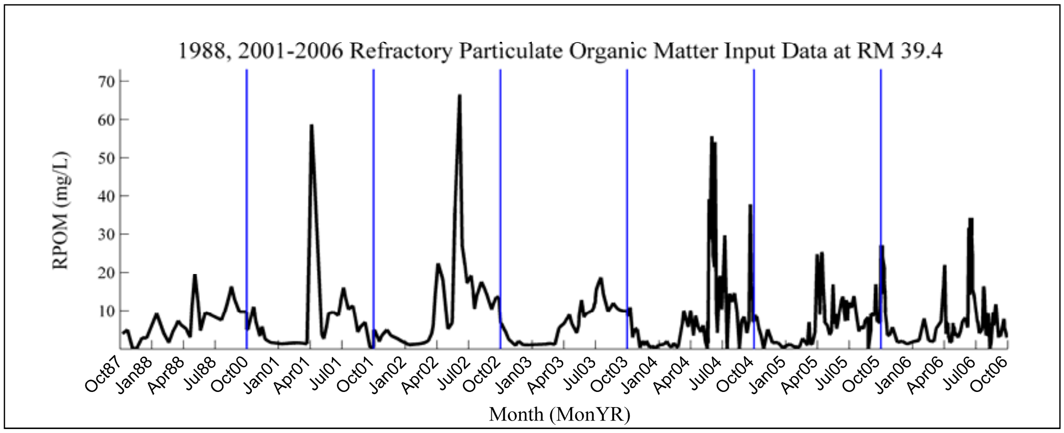


Figure 20. RPOM input data for RM 39.4 for 1988, 2001-2006.

stations and all water years; however, actual phytoplankton biomass and identifications were only available at RM 3.5 for WY 2004-2006 and at RM 39.4 for 2005 and 2006. A few samples were also available during 1988 and 1996. Since phytoplankton data were not consistently available for 1988 and 2001-2003, MCES calculated monthly average algal splits from all available data at RM 3.5 and provided these algal splits to apply to the total biomass. Table 12 presents these splits in terms of percentages. These splits were applied to the 15th day of every month; where monitored data were available, the data were used. The data gaps left in the input files were then linearly interpolated using DigDB. Figures 21-23 show the input plots for the LMRM at RM 39.4, the upstream boundary.

Table 12. MCES suggested algal splits for historical years.

Algae	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Diatoms	88.8%	62.9%	60.8%	97.8%	91.4%	84.6%	75.3%	86.7%	77.8%	65.4%	97.6%	99.4%
Blue-Greens	1.2%	6.5%	3.0%	0.2%	3.8%	6.2%	17.7%	5.8%	14.7%	26.6%	1.2%	0.2%
Other	9.9%	30.6%	36.2%	2.0%	4.8%	9.3%	7.0%	7.5%	7.5%	7.9%	1.3%	0.4%

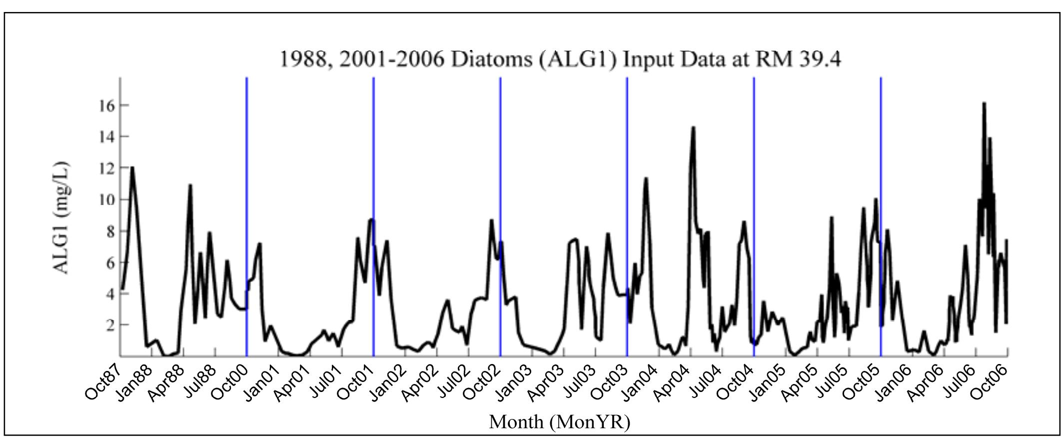


Figure 21. ALG1 (diatoms) input data for RM 39.4 for 1988, 2001-2006.

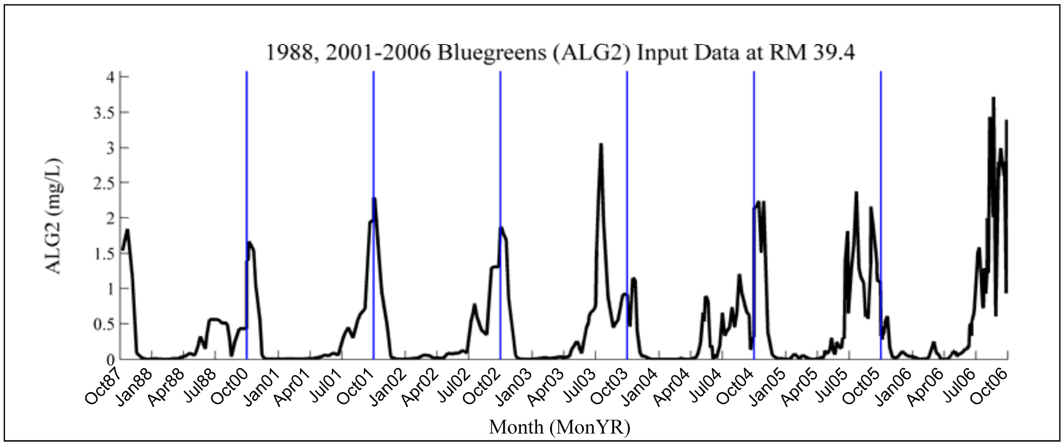


Figure 22. ALG2 (bluegreens) input data for RM 39.4 for 1988, 2001-2006.

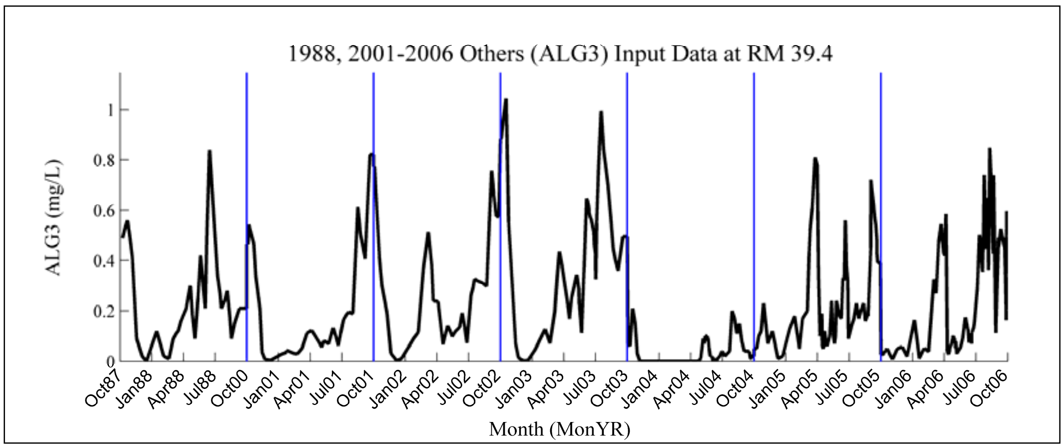


Figure 23. ALG3 (others) input data for RM 39.4 for 1988, 2001-2006.

Dissolved oxygen (DO)

MCES monitored DO on a weekly or continuous basis at all river monitoring stations and daily at the treatment plants. While some continuous DO measurements were available at RM 39.4, weekly measurements were used to define the DO inputs at the upstream boundary so they were at the same frequency as other constituents. DO was not routinely monitored at the tributaries. For the tributaries, DO was estimated from temperature using equations provided by MCES. Dissolved oxygen was not monitored at the airport outfalls during 2005-2006; for these years DO was input as an average value, 3.0 mg/L, obtained from measured data in 2001-2004. Figure 24 is a plot of dissolved oxygen input data used in the LMRM at RM 39.4.

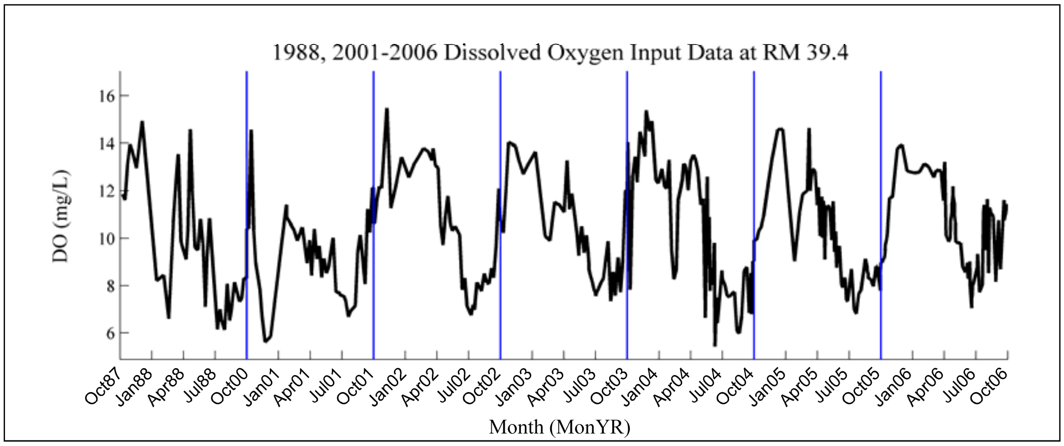


Figure 24. DO input data for RM 39.4 for 1988, 2001-2006.

Problem with defining water quality at the Black Dog outfalls

Water quality samples were only collected on 15 dates during the late summers of 2005 and 2006 at the Black Dog Generating Plant outfalls: Black Dog Lyndale (RM 10.7) and Black Dog Cedar (RM 7.6). This is significant because the Black Dog discharges often equal a substantial portion of the river flow. This lack of measured data required that ERDC make a decision on how to best handle the water quality concentrations at the Black Dog outfalls.

To test the significance of water quality inputs at Black Dog, two different runs were set up using the 2006 model. The first model run assumed that no data were collected, so no inputs except flow and temperature could be defined. That is, the water quality input files for both of the Black Dog outfalls had zeroes for the first day and the last day of the model simulation. This run indicated what to expect in the years when no measured data were available: 1988 and 2001-2004. The results for DO for this model run are shown in Figure 25. Notice that the model underpredicts DO throughout the entire water year. This is directly attributed to the fact that no water quality data were input for the outfalls; however, the facility was still withdrawing and discharging flow in the model. This demonstrated that it was important to account for the mass of DO and other constituents routed through Black Dog Lake.

The second run assumed that the effect of Black Dog on water quality was negligible; that is, water quality downstream of the plant was similar to upstream water quality. To accomplish this, model output from the segment just upstream of the Black Dog Lyndale outfall (segment 60) was used with

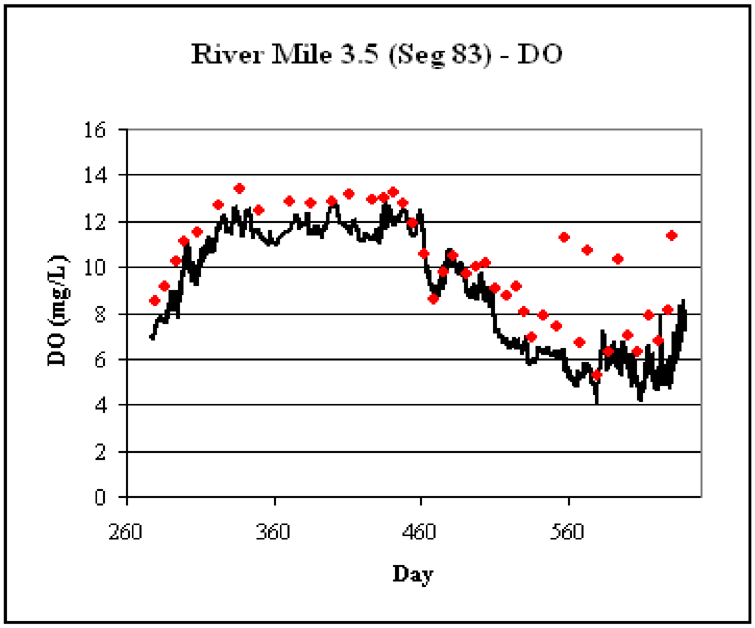


Figure 25. WY06 DO at Fort Snelling – No water quality inputs at Black Dog outfalls.

the predicted water quality values to define the input files for both of the outfalls. These files were termed “reflective input files.” Using this assumption, the model no longer underpredicted DO values throughout the entire water year, as shown in Figure 26. Model performance improved with reflective boundary conditions for Black Dog, but a major drawback is that two model runs are required: an initial run to generate results at segment 60 to create the reflective inputs and a final run that applies the reflective inputs. In the final calibration, reflective input files were used for the Black Dog outfalls except when data were available (i.e., portions of the summer in 2005 and 2006).

Meteorological data

MCES collected 15-minute meteorological data on the left bank of the Minnesota River at RM 3.5 from April 2005 through September 2006. These data were tested in the model; however, hourly data collected from the University of Minnesota at St. Paul produced better results. The data obtained did not include the cloud cover, so hourly cloud cover data were requested from the 14th Weather Squadron at the Minneapolis-St. Paul Airport. Table 13 lists the data sources used to obtain meteorological data for each water year, which include the National Weather Service at Minneapolis-St. Paul Airport (MSP), University of Minnesota at St. Paul (UMSP), and Midwestern Regional Climate Center estimates for MSP (MRCC).

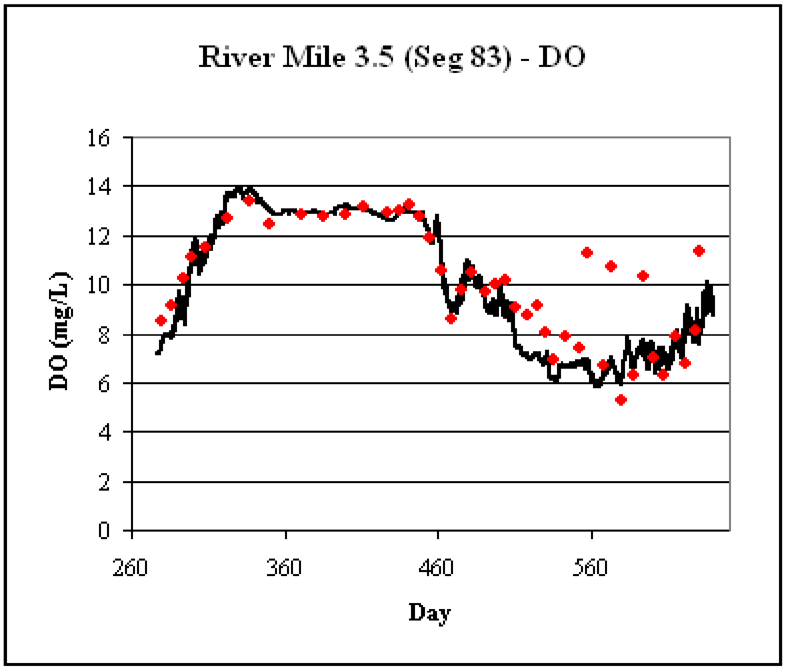


Figure 26. WY06 DO at Fort Snelling – Reflective input files for Black Dog outfalls.

Table 13. Data sources for hourly meteorological inputs.

Variable	1988	2001-2003	2004-2006
Air temperature	MSP	MSP	UMSP
Dew-point temperature	MSP	MSP	UMSP
Wind speed	MSP	MSP	UMSP
Wind direction	MSP	MSP	UMSP
Solar radiation	MRCC	UMSP	UMSP
Cloud cover	MSP	MSP	MSP

CE-QUAL-W2 control file

Each of the model year control files can be found in Appendix E. In order to keep this section concise, only a few important parameters will be discussed.

Transport scheme and heat exchange

The transport solution scheme used in the LMRM is the ULTIMATE scheme. This scheme is a higher order solution scheme that reduces numerical diffusion and eliminates the over- and undershoots that the QUICKEST scheme generates near regions of shear concentration gradients (Cole and Wells 2008).

In the W2 control file, the user must specify heat exchange parameters. The first parameter specified is the approach used for computing surface heat exchange, SLHTC. For the LMRM, the ERDC chose to use SLHTC = ET because the equilibrium temperature approach consistently produces better results for various systems according to Cole and Wells (2008). Since the meteorological data files contain short wave solar radiation, the model setting SROC was set to ON, which specifies that W2 needs to read an extra column from the meteorological input file. Although the ERDC was provided with hourly meteorological data, W2 was still allowed to interpolate the input data to correspond to the model time-step by setting the parameter METIC to ON. The wind speed measurement height was set to 10.0 m in the LMRM. All other heat exchange parameters were set to the suggested default values.

Due to the very cold temperatures in Minnesota and based on ice observations collected from field crews, the LMRM allows for ice calculations (ICEC = ON). For WY 2005, all ice cover parameters were set to the suggested default values in the latest version of the W2 V3.6 manual. For all other water years, the coefficient of water-ice heat exchange, HWICE, was set to 0.10, and the temperature above which ice formation cannot occur, ICET2, was set to 4.0 deg-C. WY 2005 was originally run identically to the other six water years, but the run time in W2 version 3.6 was 12 hr. Version 3.6 resolves several “glitches” to the ice routine, and since WY 2005 was a much colder year, ERDC decided to change the two variables as described above. Making the changes in WY 2005 did not impact the results in any way, but it significantly improved run time. Run time for WY 2005 was reduced from 12 hr to 2 hr. The control files for the remaining water years were left unchanged.

Sediment oxygen demand (SOD)

In the W2 control file, the user is allowed to specify spatially variable zero-order SOD based on segment location. W2 does not have a complete sediment diagenesis model in the current version of the model; however, future work includes adding it. Currently the user has two options for specifying the method to which sediment contributions to nutrients and DO are simulated: a zero-order method and a first-order method. The ERDC chose to model SOD as a zero-order process. The zero-order process does not depend on the sediment concentrations; it uses the specified SOD (see Table 14) and temperature-dependent anoxic release rates. The minimum oxygen value, O2LIM (specified in the control file), determines

Table 14. Mean SOD from HydrO2 assessment.

River Mile	18-24 July 2006		September 2006	
	Mean SOD (gm O2/m2/day)	Water Temp (deg-C)	Mean SOD (gm O2/m2/day)	Water Temp (deg-C)
RM 1.0	1.49	27.0		
RM 6.5	1.29	25.8		
RM 11.0	0.26	25.0	1.72	20.1
RM 15.0	1.65	25.8	2.76	22.5
RM 21.4	4.00	25.8	1.52	22.9
RM 39.4	0.21	28.7		

when nutrient releases occur. If the oxygen concentration is above the minimum value, then nutrient releases will not occur (Cole and Wells 2008). Nutrient release is specified as a fraction of SOD rates: 0.010 for NH4 and 0.001 for PO4.

HydrO$_2$ (2006) performed an oxygen dynamics assessment for the lower Minnesota River in July and September of 2006. Their findings are summarized in Table 14. In order to account for temporal variability, the control file also requires temperature rate multipliers. Figure 27 presents SOD values as used in the LMRM for all water years. ERDC chose to represent the higher SOD values in the river to get the greatest impact on DO. These values were applied to the entire reach; they were not interpolated.

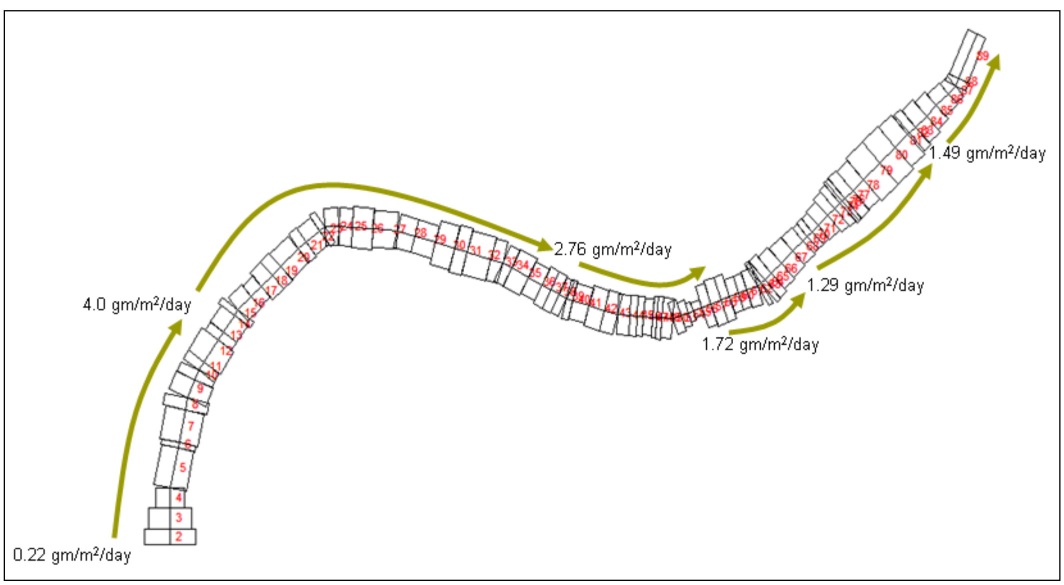


Figure 27. SOD values used in the LMRM.

Light extinction coefficients

Based on an analysis of data from the Lower Minnesota River by Dr. R. Megard (2007; Appendix C), University of Minnesota, several extinction coefficients were determined. Megard's findings are summarized in Figure 28. In the plot, VSS represents volatile suspended solids, NVSS represents non-volatile suspended solids, and DOC represents dissolved organic carbon.

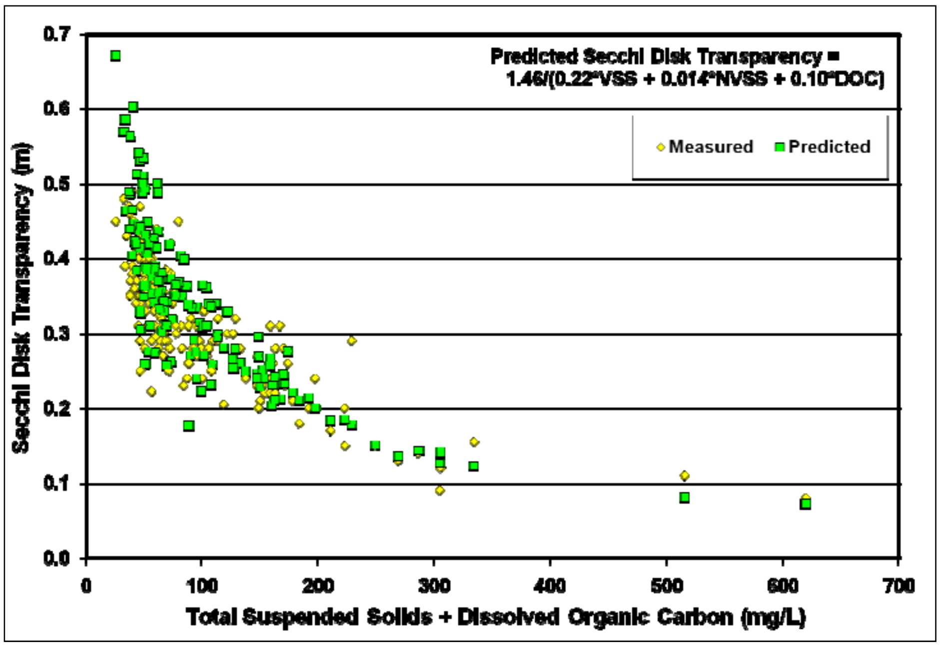


Figure 28. Transparency vs. TSS+DOC (mg/L) (Megard 2007).

The LMRM has EXSS = 0.14 m^{-1}, EXOM = EXA1 = EXA2 = EXA3 = 0.22 m^{-1}, and EXH20 = 0.10*(median DOC concentration) or ~ 0.581 m^{-1}. EXSS is the extinction due to inorganic suspended solids, EXOM is the extinction due to organic suspended solids, EXH20 is the extinction for pure water, and EXA1, EXA2, and EXA3 are the extinctions due to diatoms, bluegreens, and other algae (greens), respectively.

Algal parameters

LimnoTech (2007, 2008) presented their work to the MPCA for the Upper Mississippi River - Lake Pepin Water Quality Model. The algal parameters used in their model are presented in Tables 15 and 16. Notice that Table 15 has values from the 2007 version of LimnoTech's report. These were the first values that ERDC used for calibration. Table 16 presents the coefficients that LimnoTech used in the final calibration of the Lake Pepin Model.

Table 15. RCA algal coefficients from LimnoTech Report (2007).

Algal Coefficients	Units	Diatoms	Blue-greens	Greens
Maximum Growth Rate	1/day	2.3	1.9	2.3
Optimal Growth Temperature	deg-C	15.0	25.0	22.0
Saturating Light Intensity	ly/day[1]	150	100	150
Half-Saturation constant for N	mg-N/L	0.05	0.005	0.005
Half-Saturation constant for P	mg-P/L	0.001	0.001	0.001
Half-Saturation constant for Si	mg-Si/L	0.002	0.002	0.002
Respiration Rate	1/day	0.14	0.2	0.14
Settling Rate	m/day	0.35	-0.1	0.3
C:Chl ratio	mg-C/mg-Chla	50	33	33

[1] 1 ly/day = 0.48 W/m^2

Table 16. RCA algal coefficients from LimnoTech Report (2008).

Algal Coefficients	Units	Diatoms	Blue-greens	Greens
Maximum Growth Rate	1/day	2.3	1.9	2.2
Optimal Growth Temperature	deg-C	15.0	30.0	25.0
Saturating Light Intensity	ly/day[1]	150	150	200
Half-Saturation constant for N	mg-N/L	0.02	0.01	0.02
Half-Saturation constant for P	mg-P/L	0.005	0.005	0.005
Half-Saturation constant for Si	mg-Si/L	0.02	NA	0.02
Respiration Rate	1/day	0.14	0.2	0.14
Settling Rate	m/day	0.3	-2	0.2
C:Chl ratio	mg-C/mg-Chla	50	33	33

[1] 1 ly/day = 0.48 W/m^2

Although many of the parameters used in the LMRM were based on LimnoTech's coefficients, some of the parameters were used as calibration parameters and were modified as necessary. In order to determine the best optimal temperatures for algal growth, values from Nielson (2005) and Cole and Wells (2008) were also considered. Table 17 presents the algal parameters used in the LMRM model for all water years. All model coefficients are listed in Appendix D.

Table 17. LMRM algal coefficients used for all water years.

Algal Coefficients	Units	Diatoms	Blue-greens	Greens
Maximum Growth Rate	1/day	1.9	1.9	2.3
Lower Temp for Algal Growth	deg-C	0.5	15.0	10.0
Lower Temp for Maximum Algal Growth	deg-C	10.0	20.0	15.0
Upper Temp for Maximum Algal Growth	deg-C	16.0	25.0	20.0
Upper Temp for Algal Growth	deg-C	20.0	40.0	25.0
Saturating Light Intensity	W/m^2	72.64	48.43	72.64
Half-Saturation constant for N	mg-N/L	0.05	0.005	0.005
Half-Saturation constant for P	mg-P/L	0.001	0.001	0.001
Half-Saturation constant for Si	mg-Si/L	0.002	0.002	0.002
Respiration Rate	1/day	0.14	0.20	0.14
Settling Rate	m/day	0.25	0.00	0.20
Algal biomass:Chl a ratio	mg algae/μg Chl a	0.0675	0.0675	0.0675

4 Model Calibration and Verification

Final calibration results are presented in this chapter. In all of the time series plots shown, a black solid line represents model output, a solid red circle represents measured data, and the blue vertical lines represent a division between water years. These plots present all model output and measured data for the seven water years modeled. Three statistics are also presented in the charts: mean error (ME), absolute mean error (AME), and root mean square error (RMSE). These statistics are calculated as shown in Equations 1-3 and represent seven-year average statistics. The model was output every 0.02 day; when making comparisons to the observed data, a tolerance of 0.02 day was used for the model output so that model output and measured data were compared spatially and temporally with minimal averaging.

$$ME = \frac{\sum_{1}^{n}(model - data)}{n} \tag{1}$$

$$AME = \frac{\sum_{1}^{n}abs(model - data)}{n} \tag{2}$$

$$RMSE = \sqrt{\frac{\sum_{1}^{n}(model - data)^2}{n}} \tag{3}$$

Cumulative distribution plots are also presented in this section. For these plots, the solid black line represents model output and the dashed red line represents observed data. Again, these plots represent a combination of all model and measured data over all seven water years. Individual year time series plots are shown in Appendix F, individual year cumulative distribution plots are presented in Appendix G, individual year scatter plots are presented in Appendix H, and Appendix I presents statistical information in tabular form.

A general rule of thumb for water quality calibration is that the absolute mean error should be within 10% of the range of monitored data.[1] Equation 4 is used to calculate the target values for AME. These target values were calculated for the seven years of data and will be presented in tabular form in the following sections. Units for these targets are consistent with the minimum and maximum values for each constituent. For example, for flow, the minimum, maximum, AME, and 10% target are presented in cubic meters per second.

$$\text{Target} = 0.10*((\text{maximum observed value}) - (\text{minimum observed value})) \quad (4)$$

Flow

Model output, along with observed data for all seven water years, is shown in Figures 29 and 30. The model output tends to predict flow well. The AME for all data pairs for all seven years at RM 3.5 is 10.51 cms, which is less than 0.5% of the measured range of flows for all seven years. Table 18 presents the 1% AME target that ERDC attempted to reach. Based on Figure 30, the slope of the trendline fitted through the data pairs is 0.97 and the R-squared value is 0.995. Overall, the model only overpredicts flow at RM 3.5 by 0.651 cms.

Temperature

Time series plots and statistical plots are presented in Figures 31 and 32. The model output tends to predict temperature well. The AME for all data pairs for all seven years at RM 3.5 is 1.34 deg-C, which is less than 5% of the measured range of temperatures for all seven years. Table 19 presents the 10% AME targets for each monitoring station. Based on Figure 32 (RM 3.5), the slope of the trendline fitted through the data pairs is 0.935 and the R-squared value is 0.970.

Table 18. 1% target for flow (cms) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	1% Target
RM 39.4	6.23	2441.18	3.29	24.350
RM 3.5	6.54	2563.23	10.51	25.564

[1] Personal Communication. 2008. Scott Wells, Professor and Chair, Department of Civil and Environmental Engineering, Portland State University, Portland, OR.

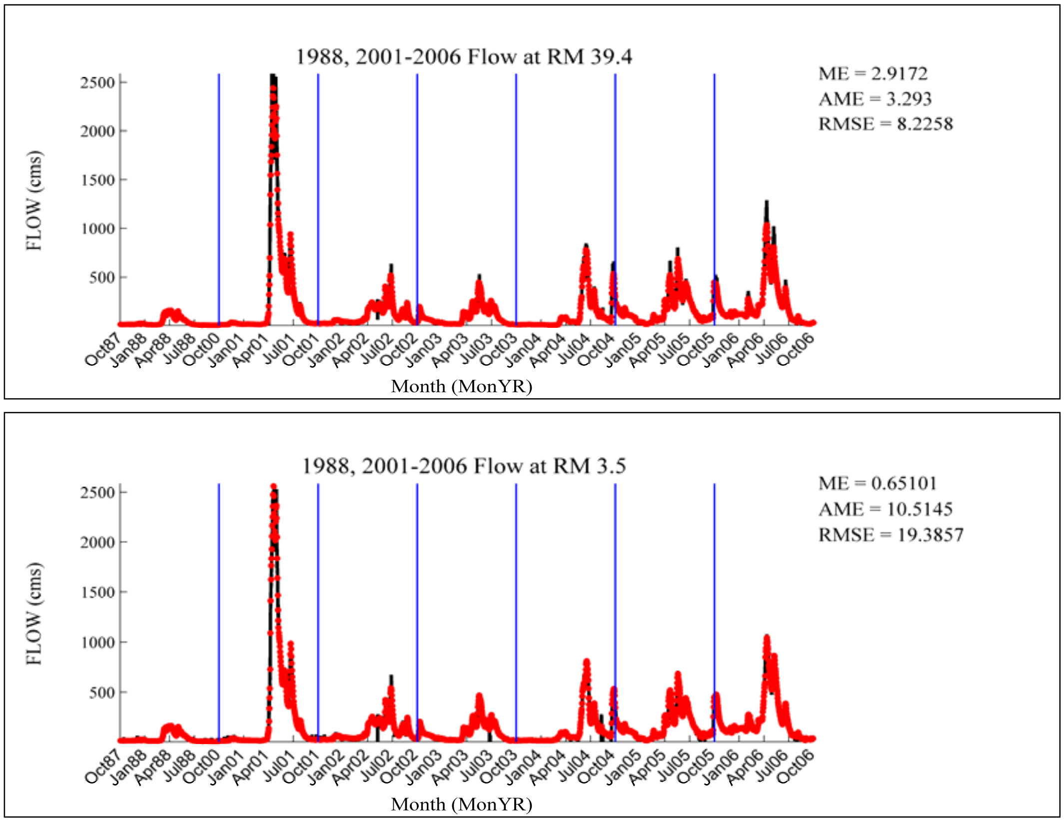


Figure 29. Flow at various calibration stations.

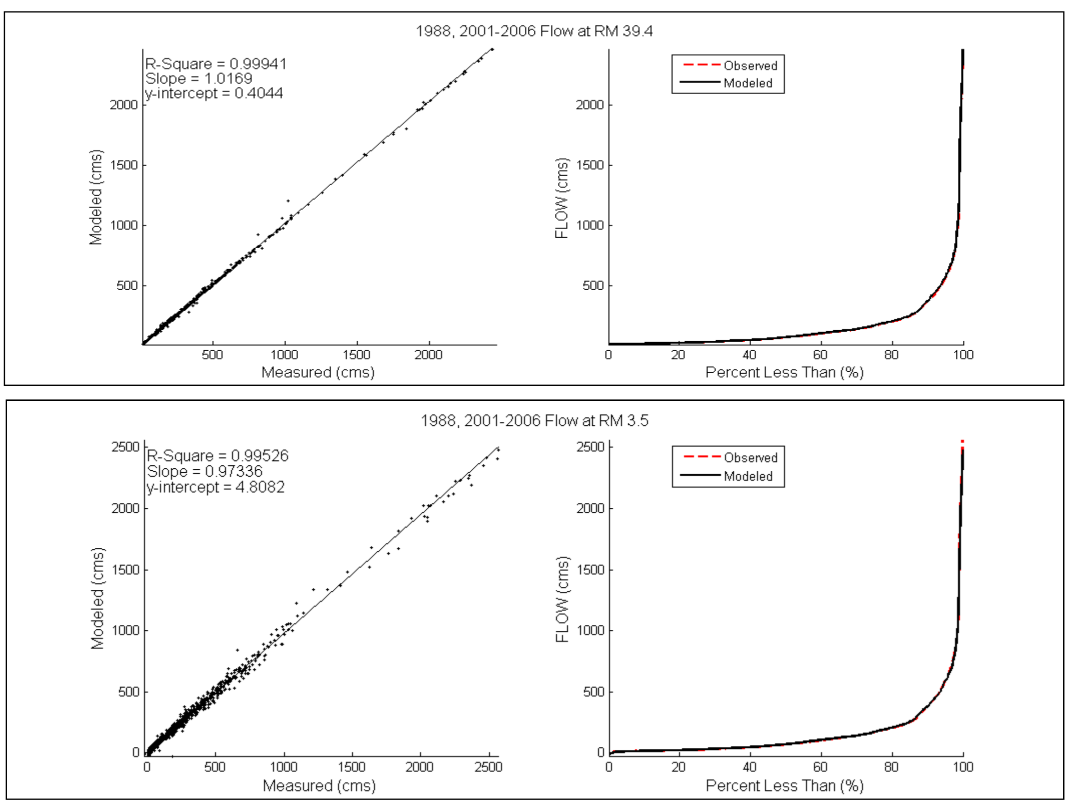


Figure 30. Flow linear and cumulative distribution plots at various calibration stations.

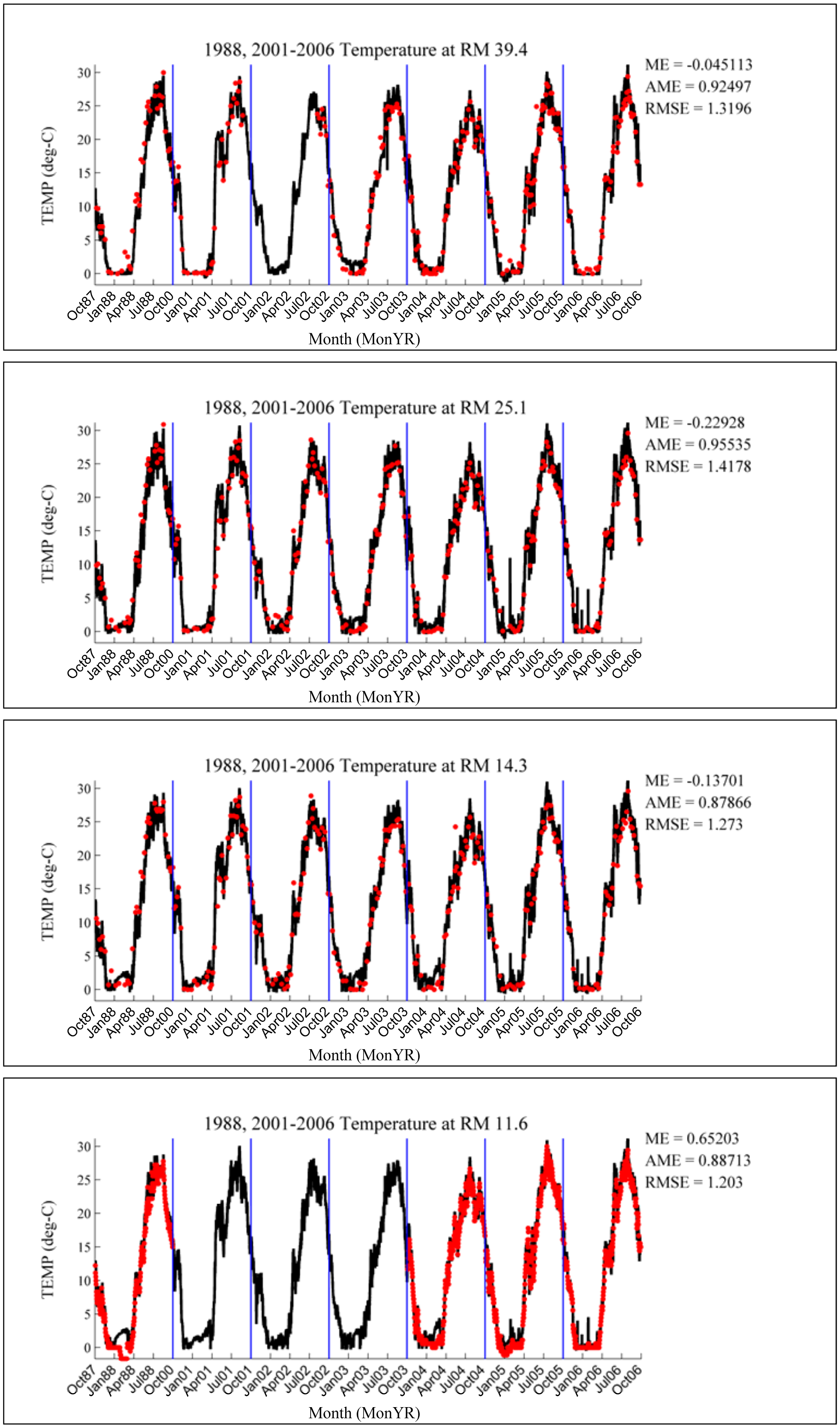
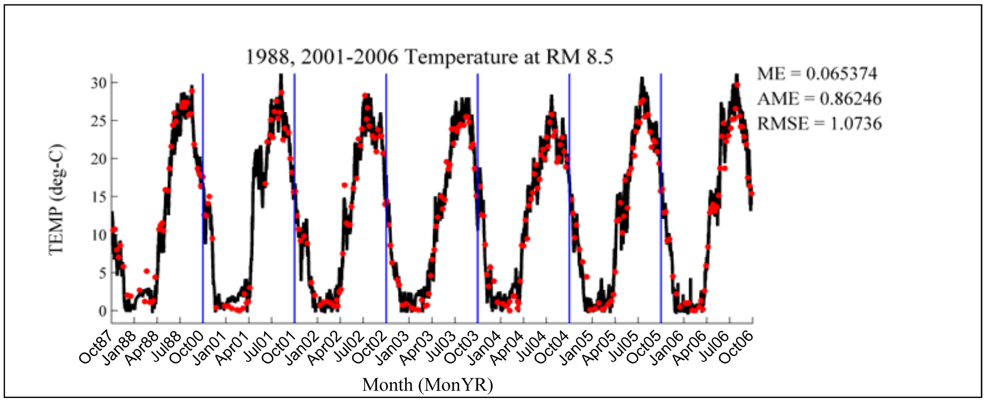


Figure 31. Temperature at various calibration stations (continued).



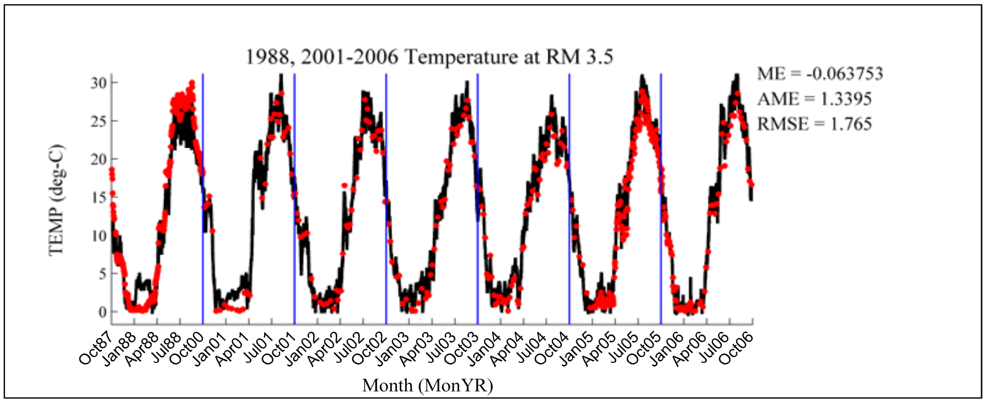
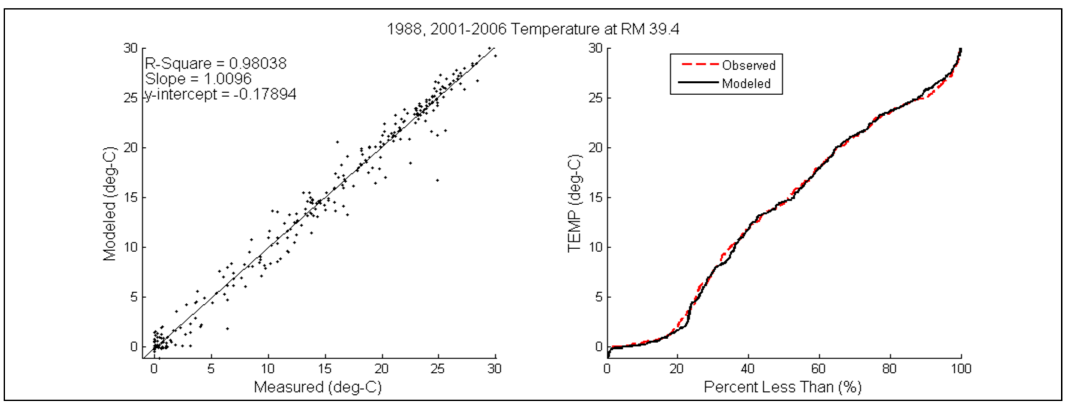


Figure 31. (concluded).



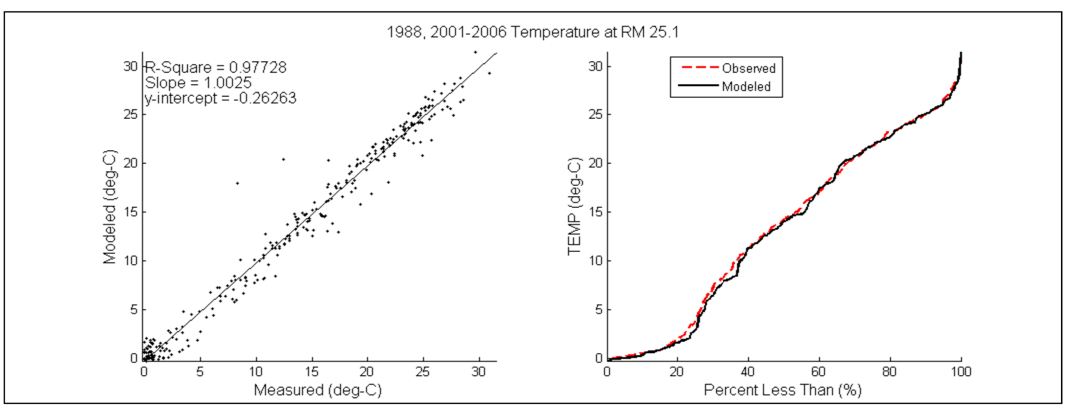


Figure 32. Temperature linear and cumulative distribution plots at various calibration stations.

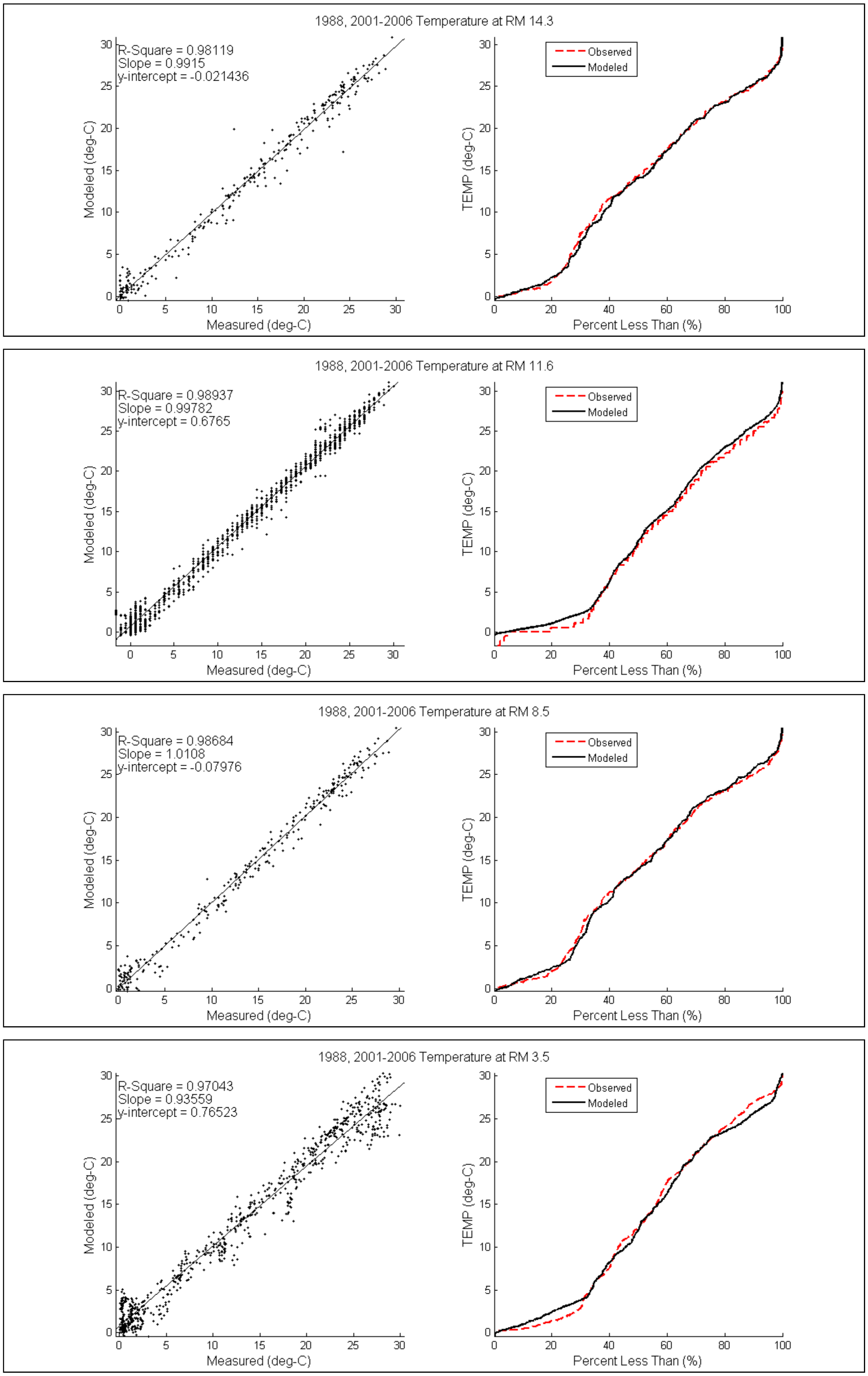


Figure 32. (concluded).

Table 19. 10% target for temperature (deg-C) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	0.00	30.00	0.92	3.00
RM 25.1	0.00	30.90	0.96	3.09
RM 14.3	0.00	29.60	0.88	2.96
RM 11.7	-1.67	30.00	0.89	3.17
RM 8.5	0.00	29.70	0.86	2.97
RM 3.5	0.04	30.00	1.34	3.00

Water surface elevation

Time series plots of water surface elevation are shown in Figure 33. The model tends to better predict water surface elevations as RM 3.5 is approached. At RM 3.5, the AME = 0.09 m, which is less than 2% of the range of seven years of measured water surface elevations (see Table 20), and the model only underpredicts water surface elevations by 0.029 m for about 80% of the data (see Figure 34). According to the statistical plots shown in Figure 34, the trendline through the paired data shows very good correlation because the slope is 0.995 and the R-squared value is 0.976.

Dissolved oxygen

As can be seen in Figures 35 and 36, the model output tends to predict DO concentrations fairly well, especially in the upper reach of the river; however, the seven-year mean error for DO indicates that the model slightly underpredicts DO. This is especially prevalent during the summer periods in most water years. Notice at the lower 60% of measured values, the model tends to underpredict the data by approximately 0.63 mg/L at RM 3.5. Although the model underpredicts DO levels, the model is well within the standard accepted level of tolerance for DO, 1.00 mg/L, and is well within the 10% AME target found in Table 21.

Table 20. 10% target for ELWS (m) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	211.28	220.38	0.32	0.91
RM 13.0	209.21	214.52	0.26	0.53
RM 3.5	209.17	215.55	0.09	0.64

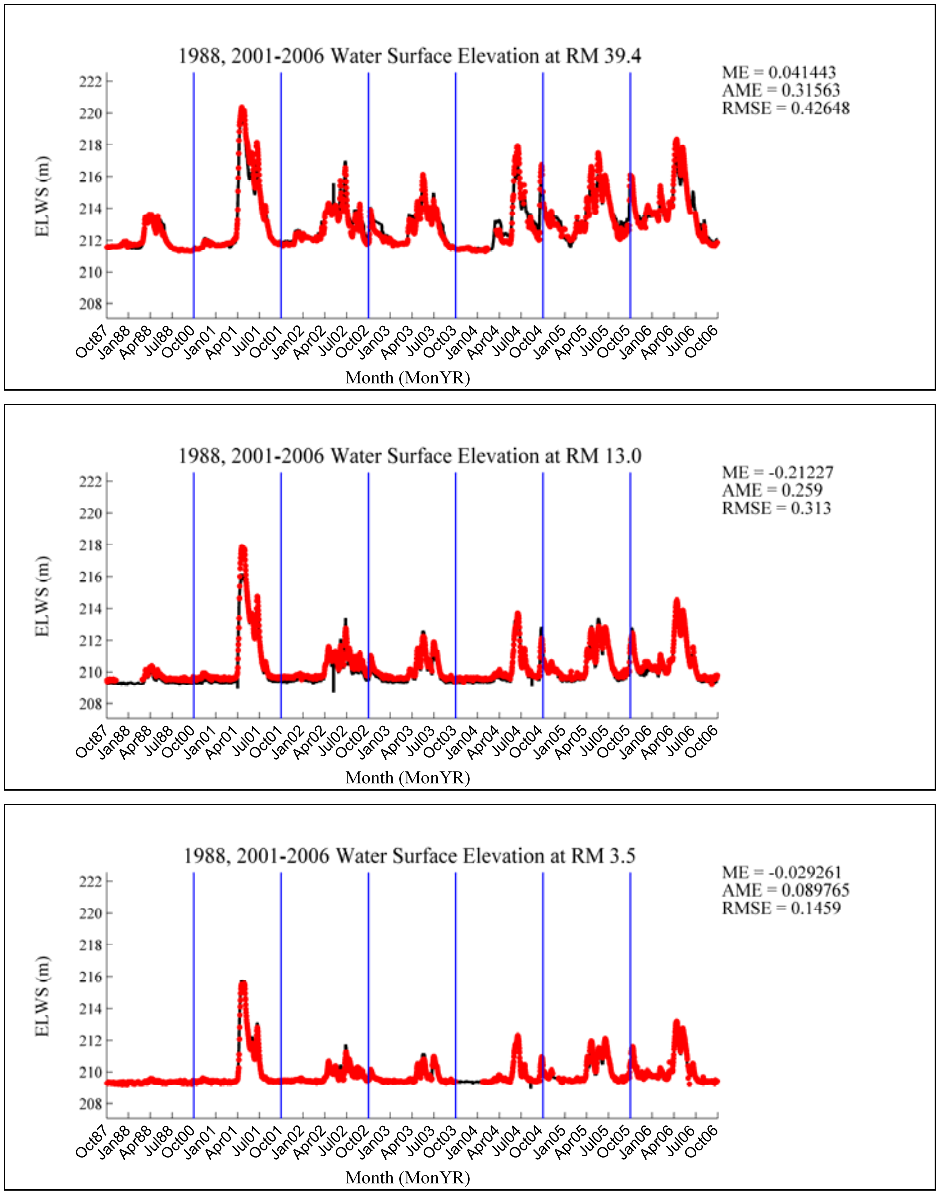


Figure 33. ELWS at various calibration stations.

Table 21. 10% target for DO (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	5.44	15.40	0.08	1.00
RM 25.1	5.11	17.06	0.66	1.20
RM 14.3	4.31	16.25	1.10	1.19
RM 8.5	3.54	16.14	1.01	1.26
RM 3.5	3.72	16.50	1.09	1.28

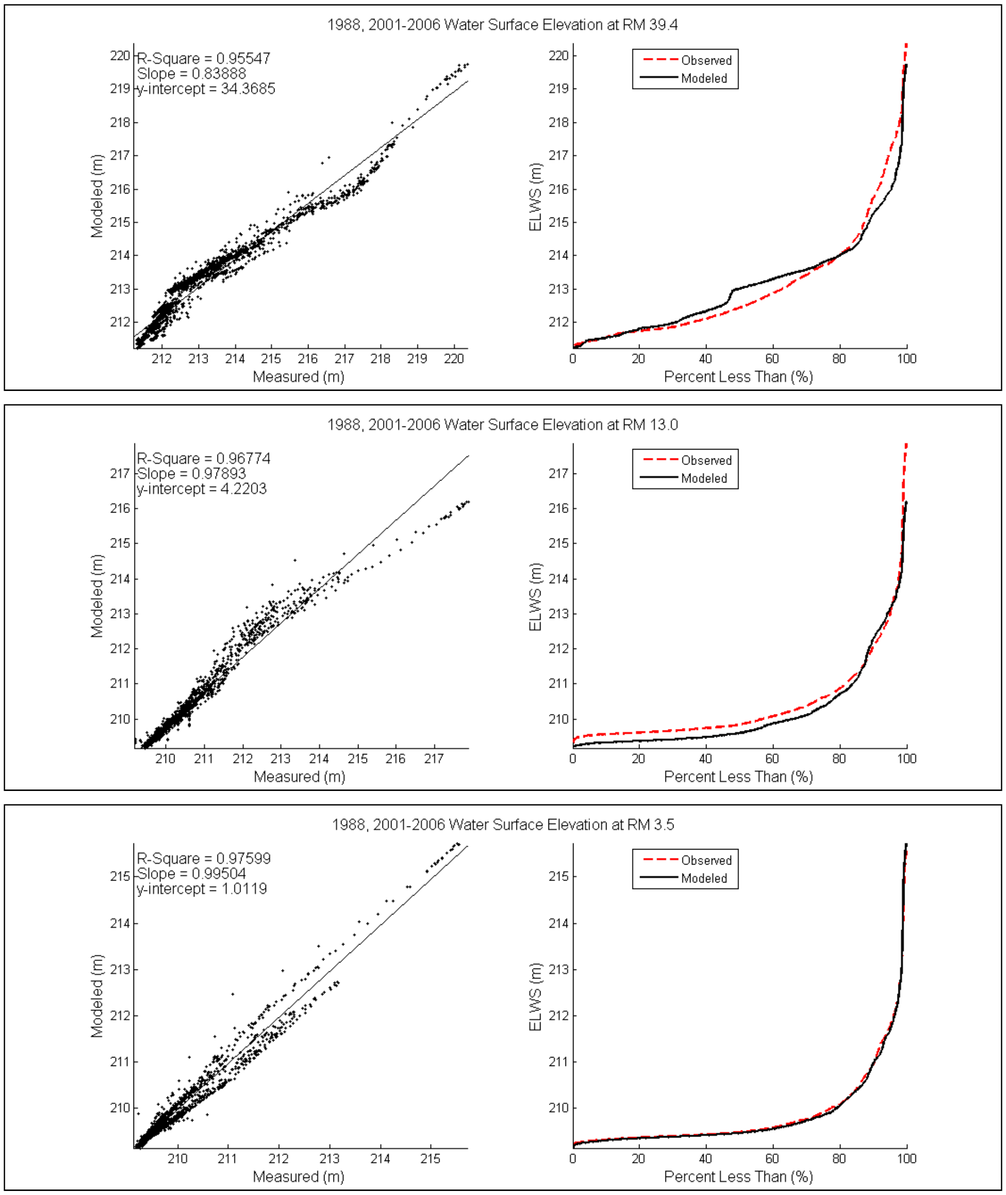


Figure 34. ELWS linear and cumulative distribution plots at various calibration stations.

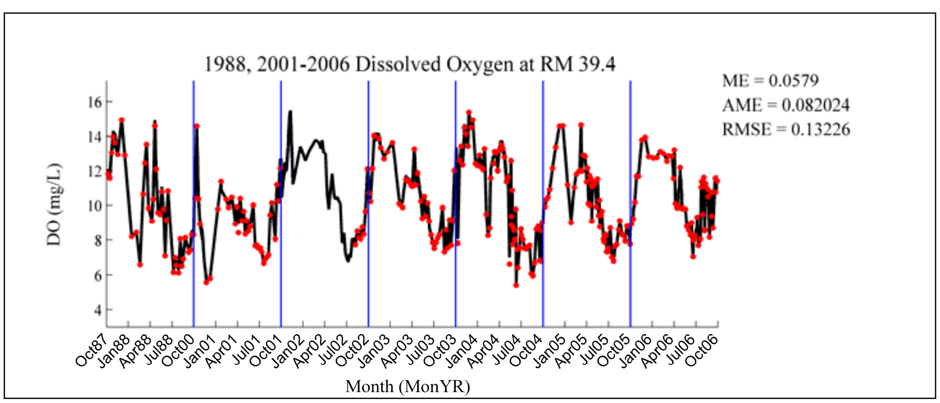


Figure 35. DO at various calibration stations (continued).

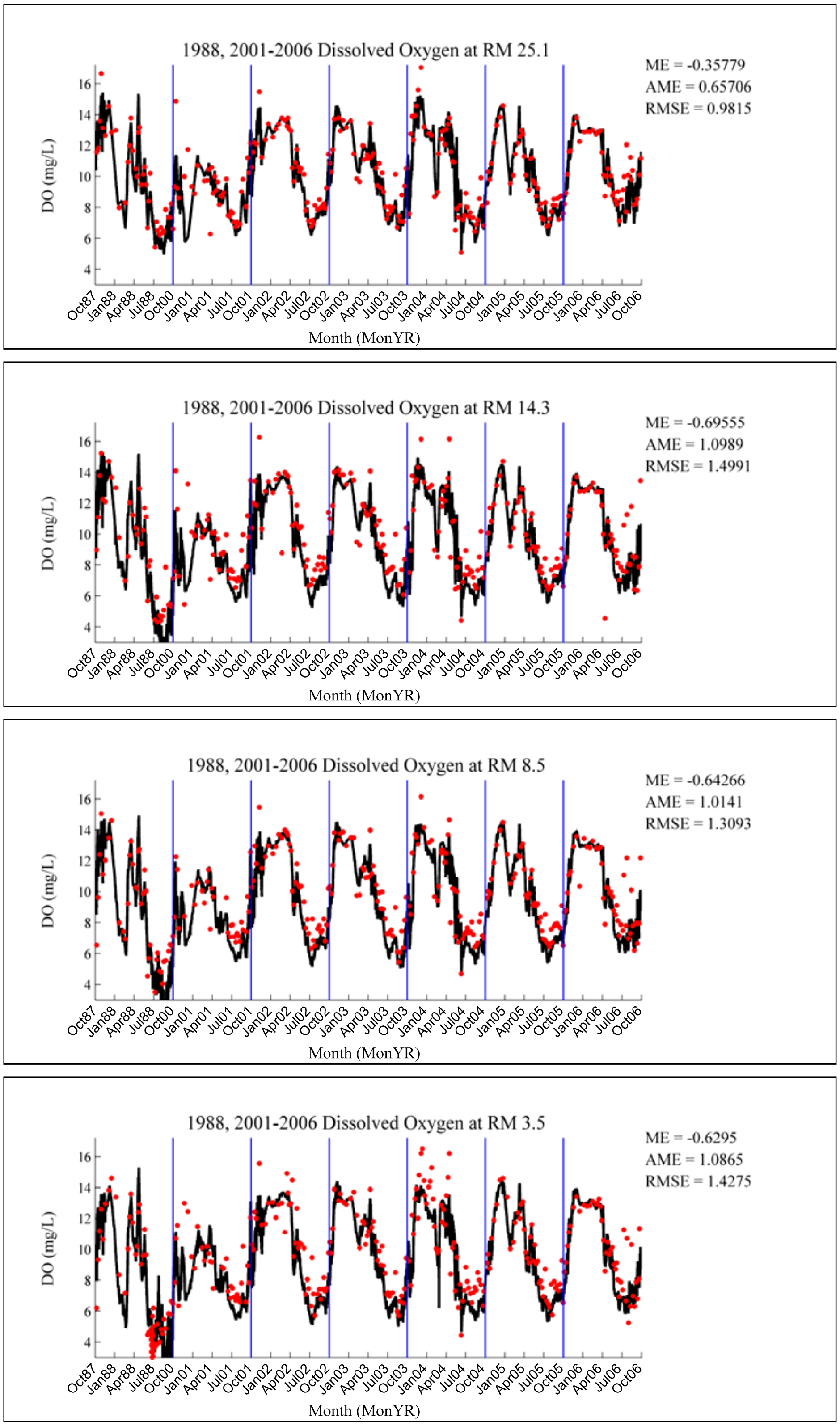


Figure 35. (concluded).

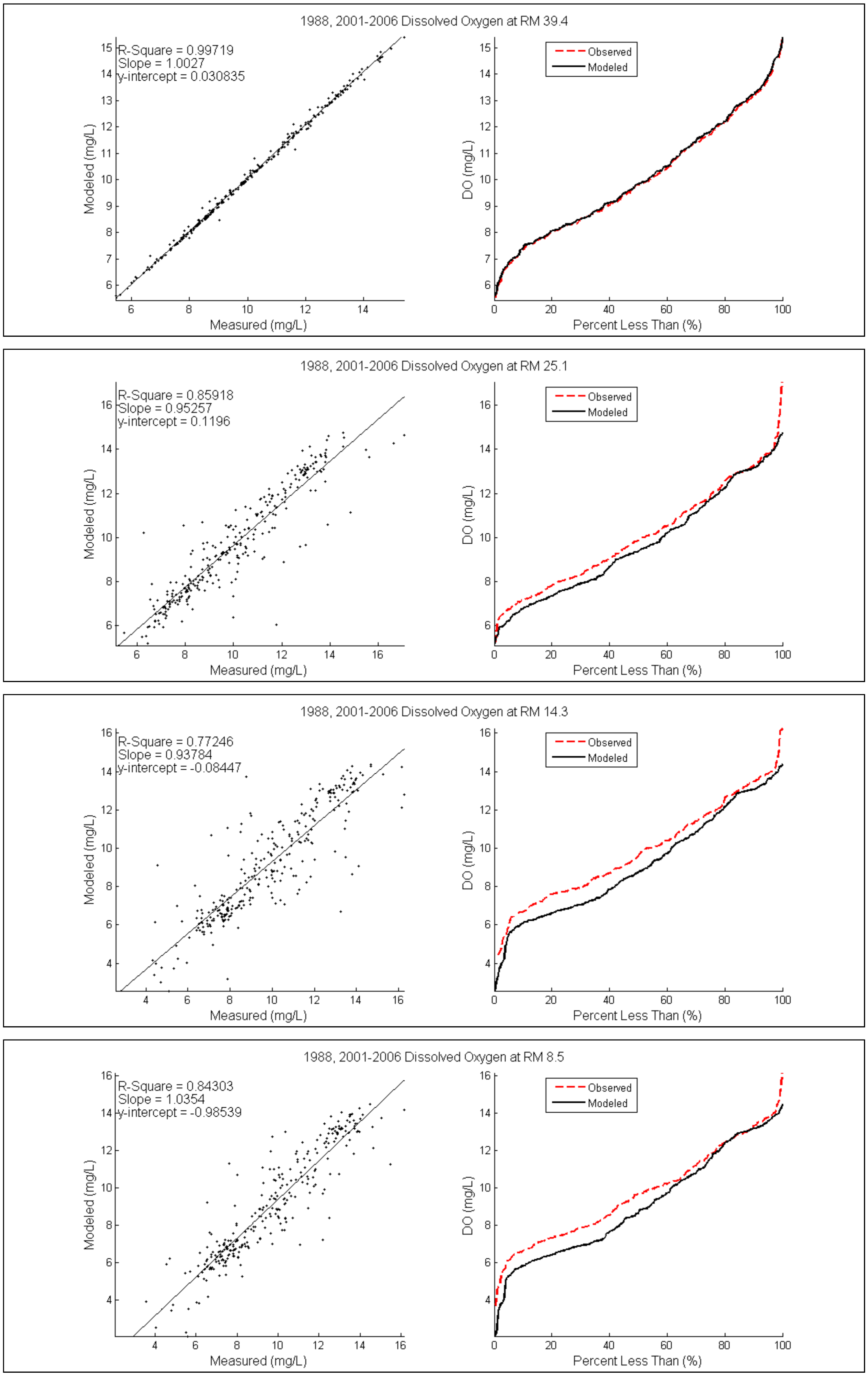


Figure 36. DO linear and cumulative distribution plots at various calibration stations (continued).

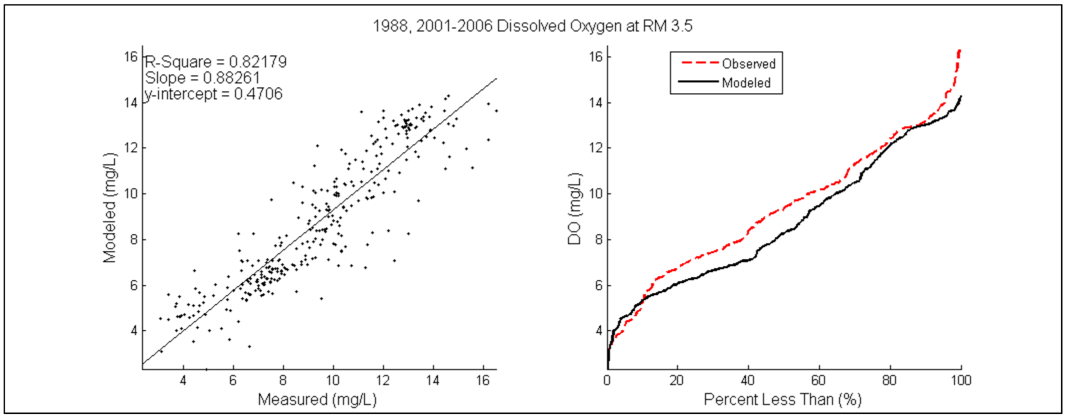
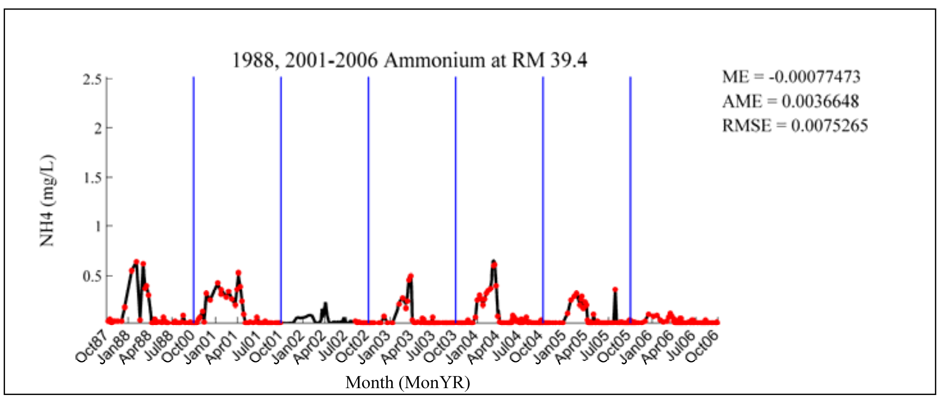


Figure 36. (concluded).

Ammonium nitrogen

As can be seen in Figures 37 and 38, the model performs well with ammonium nitrogen predictions. The AME increases as the river approaches the mouth; however, even at Fort Snelling (RM 3.5), the AME is 0.122 mg/L, which is much less than the 10% AME target found in Table 22 (0.25 mg/L). During the summer of 1988, the model underpredicts NH4 beginning downstream (see RM 14.3) from three wastewater treatment plants: Chaska WWTP, Blue Lake WWTP, and Savage WWTP.



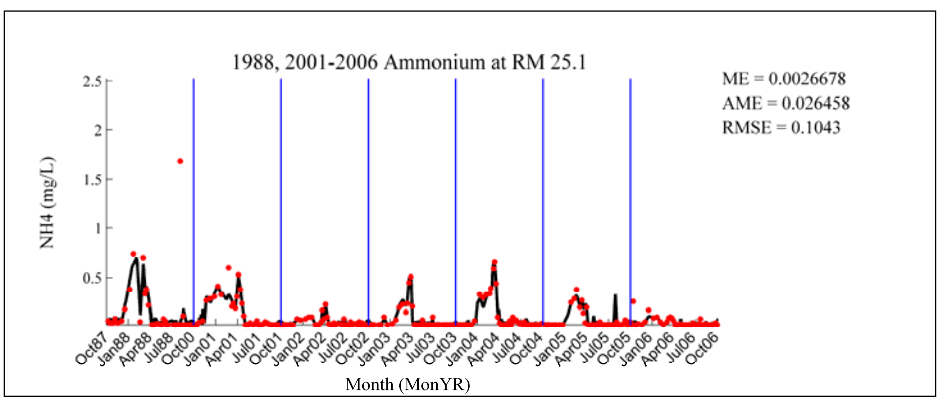


Figure 37. NH4 at various calibration stations (continued).

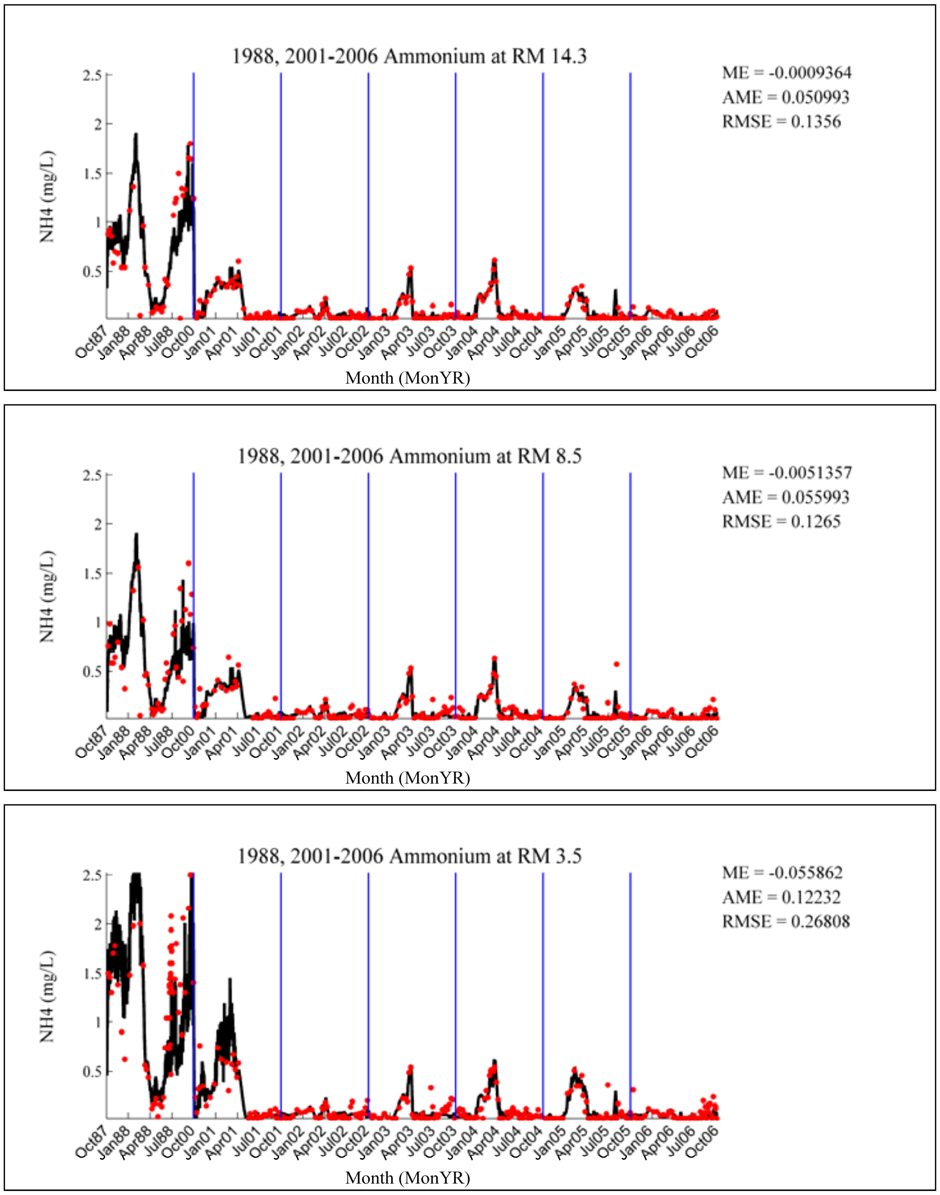


Figure 37. (concluded).

Table 22. 10% target for NH_4 (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	0.02	0.64	0.00	0.06
RM 25.1	0.02	1.68	0.03	0.17
RM 14.3	0.02	1.80	0.05	0.18
RM 8.5	0.02	1.60	0.06	0.16
RM 3.5	0.02	2.50	0.12	0.25

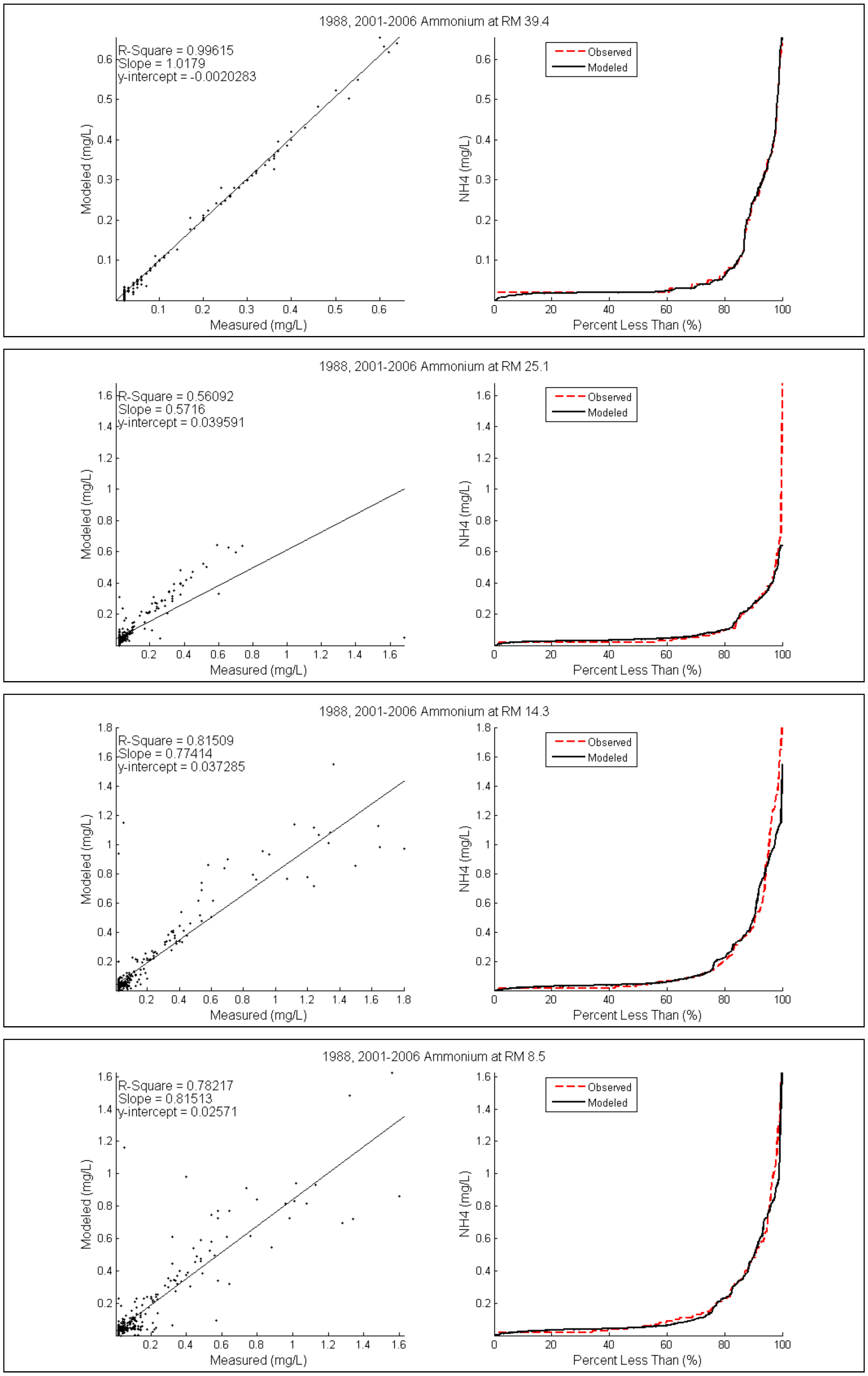


Figure 38. NH4 linear and cumulative distribution plots at various calibration stations (continued).

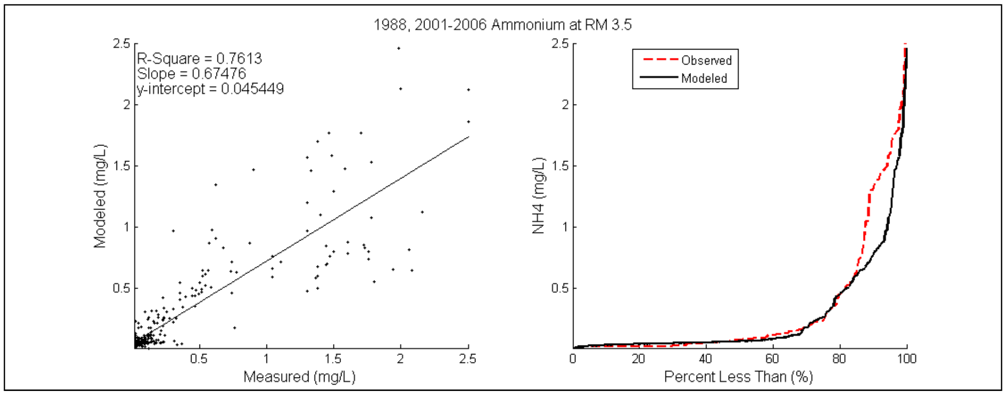


Figure 38. (concluded).

Algae and chlorophyll *a*

As can be seen in Figures 39 and 40, the model tends to do well with chlorophyll *a* predictions. The AME increases as the river approaches the mouth and culminates at Fort Snelling (RM 3.5) with an AME = 20.33 ug/L, which is still less than the 10% AME target found in Table 23 (23.90 ug/L). Notice from the cumulative distribution plots that at higher concentrations, the model tends to underpredict chlorophyll a. Phytoplankton biomass was only available at both RM 39.4 and RM 3.5 for water years 2005 and 2006; the time series and cumulative distribution plots for the three algal groups are found in Figures 41-46. The 10% AME targets for the individual algal groups are found in Table 24.

Table 23. 10% target for CHLA (ug/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	0.25	270.00	3.06	26.98
RM 25.1	3.50	270.00	13.97	26.65
RM 14.3	1.00	230.00	20.15	22.90
RM 8.5	0.79	277.20	20.13	27.64
RM 3.5	1.00	240.00	20.33	23.90

Table 24. 10% target for algae (biomass mg/L dry wt) for 2005-2006.

Algal Group	River Mile	Minimum	Maximum	AME	10% Target
Diatoms	RM 39.4	0.08	12.05	0.07	1.20
Diatoms	RM 3.5	0.04	13.45	1.07	1.34
Bluegreens	RM 39.4	0.00	3.15	0.01	0.32
Bluegreens	RM 3.5	0.00	1.96	0.18	0.20
Others	RM 39.4	0.00	1.78	0.01	0.18
Others	RM 3.5	0.01	1.25	0.09	0.12

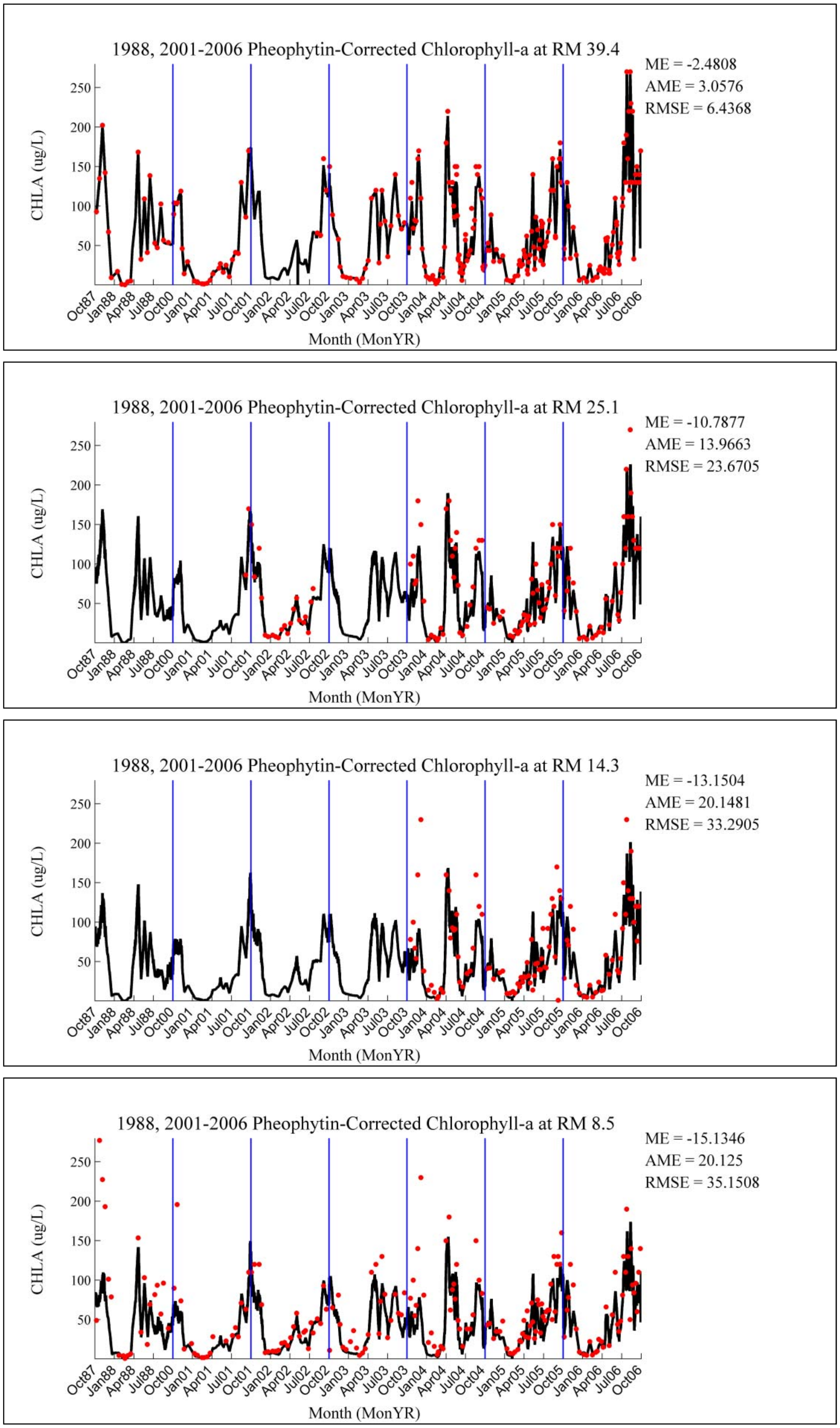


Figure 39. CHLA at various calibration stations (continued).

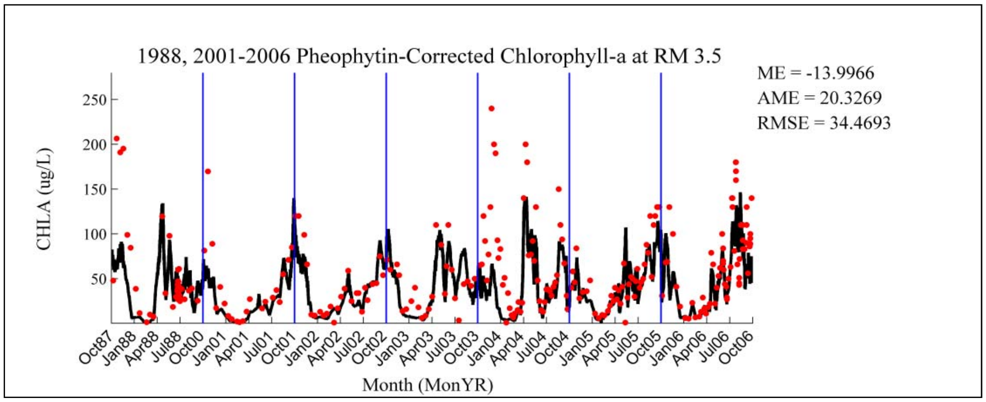


Figure 39. (concluded).

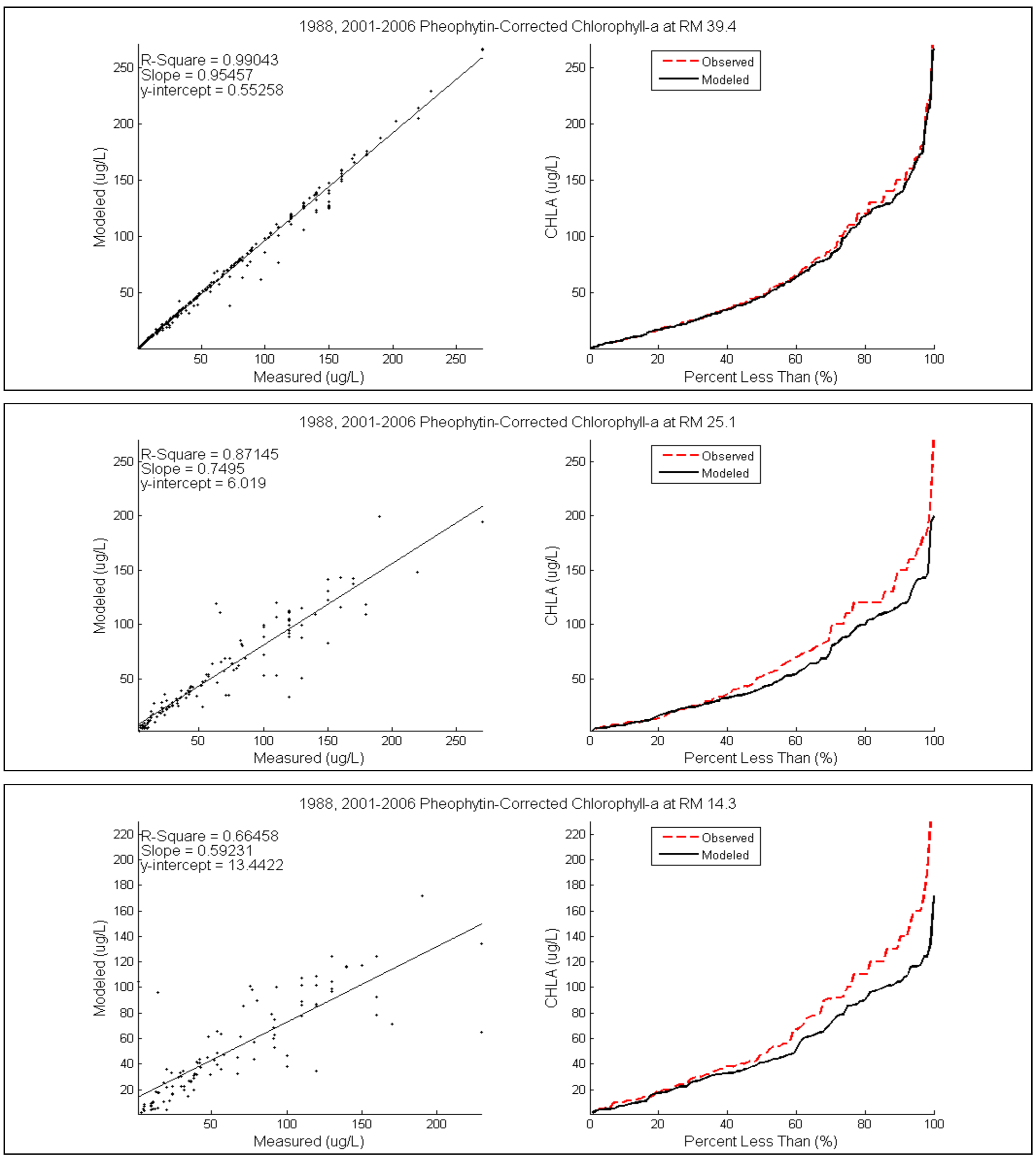


Figure 40. CHLA linear and cumulative distribution plots at various calibration stations (continued).

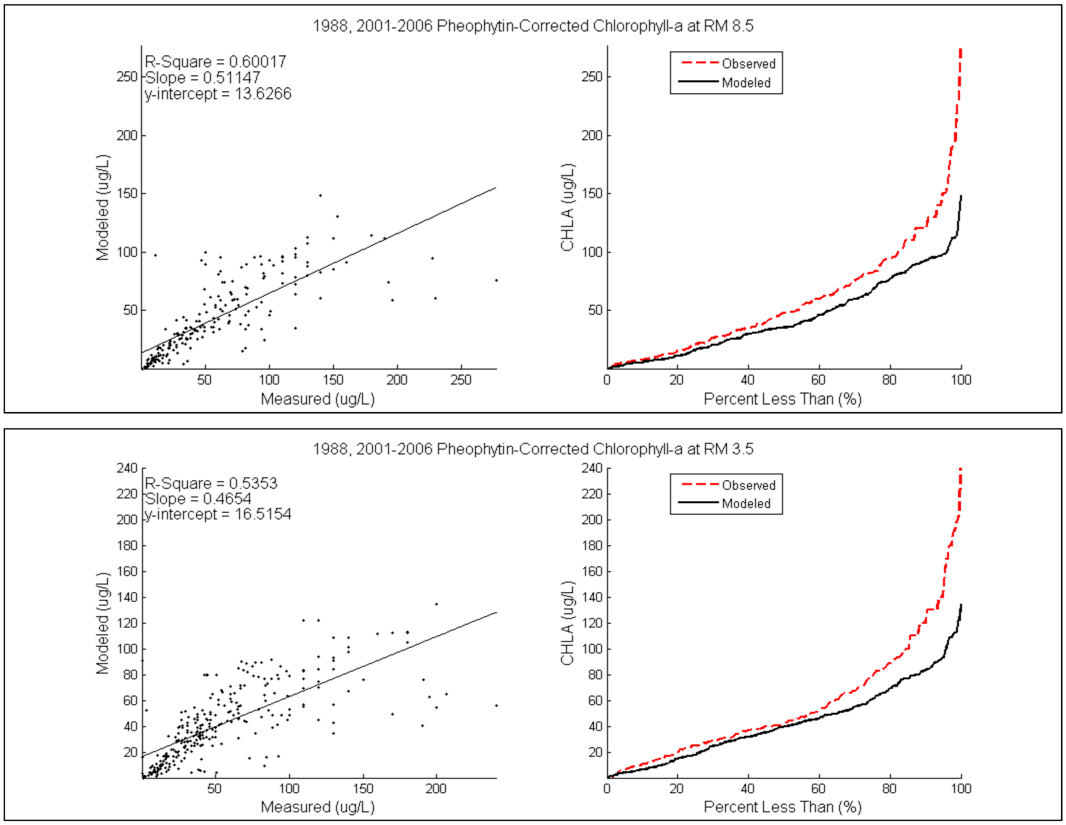


Figure 40. (concluded).

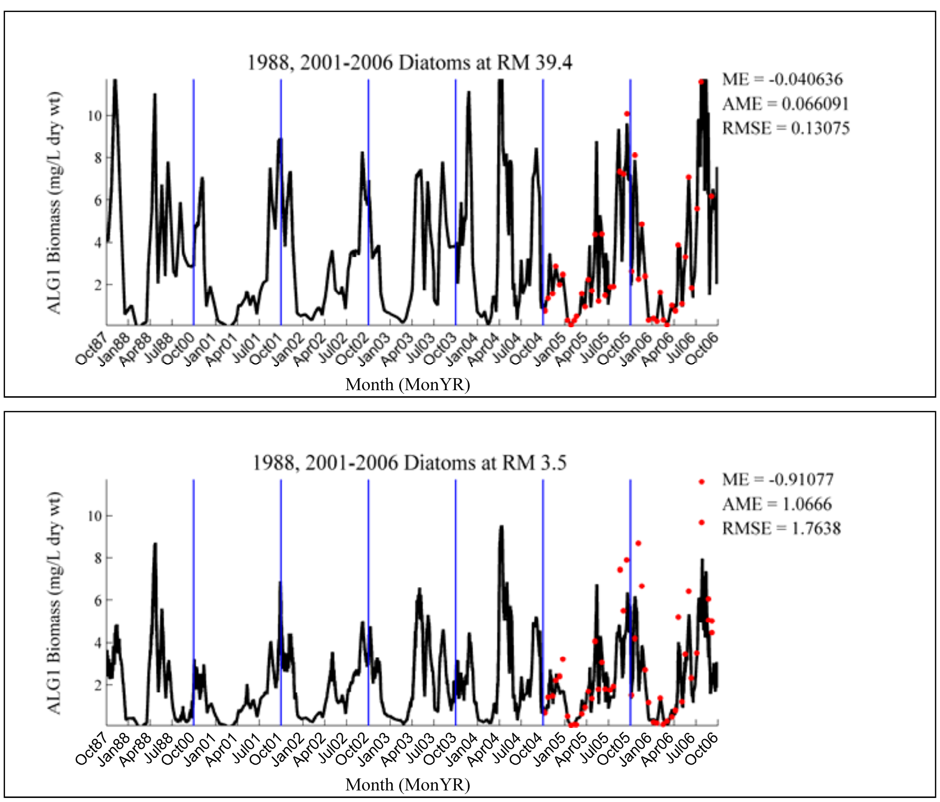


Figure 41. Diatoms (ALG1) time series plots at RM 39.4 and RM 3.5.

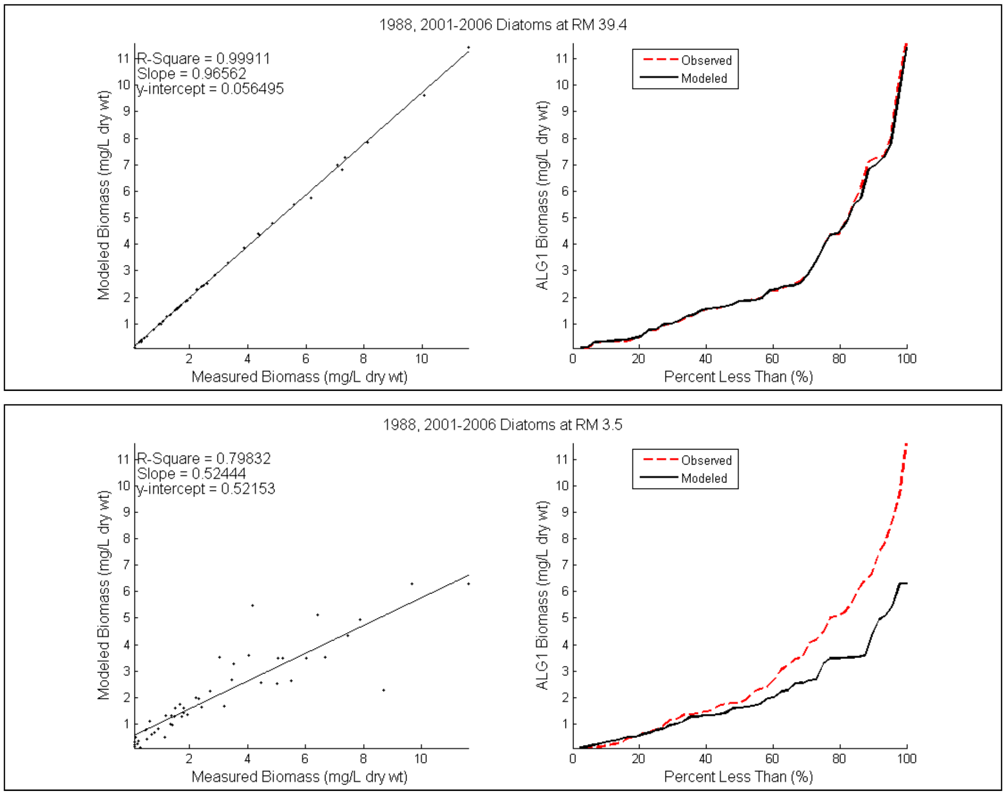


Figure 42. Diatoms (ALG1) linear and cumulative distribution plots at RM 39.4 and RM 3.5.

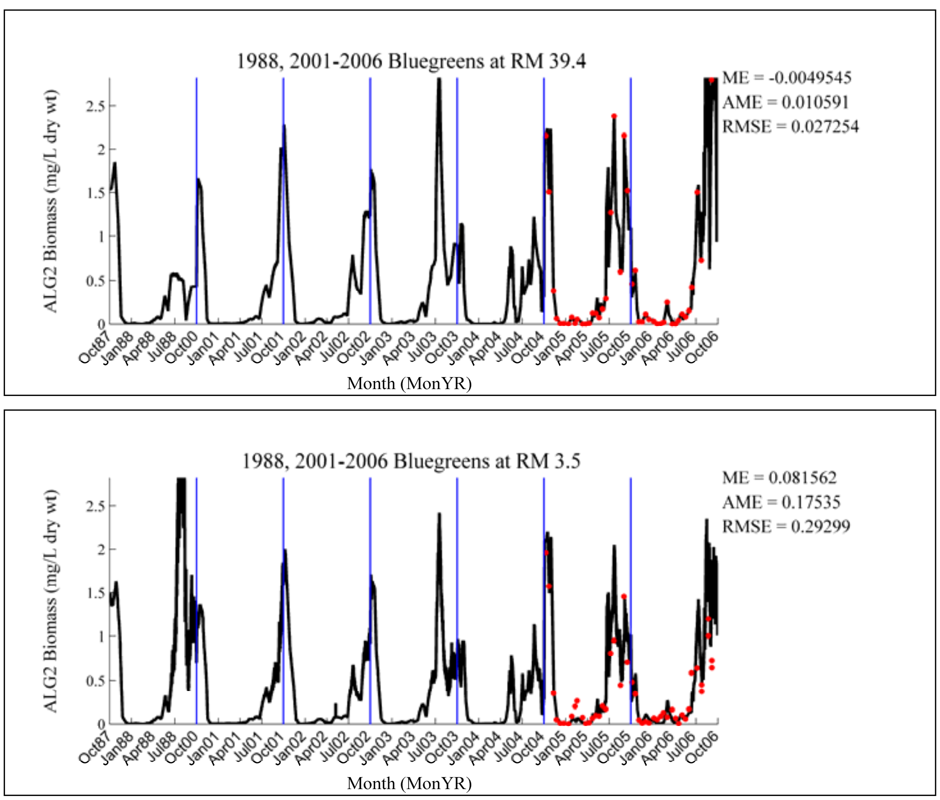


Figure 43. Bluegreens (ALG2) time series plots at RM 39.4 and RM 3.5.

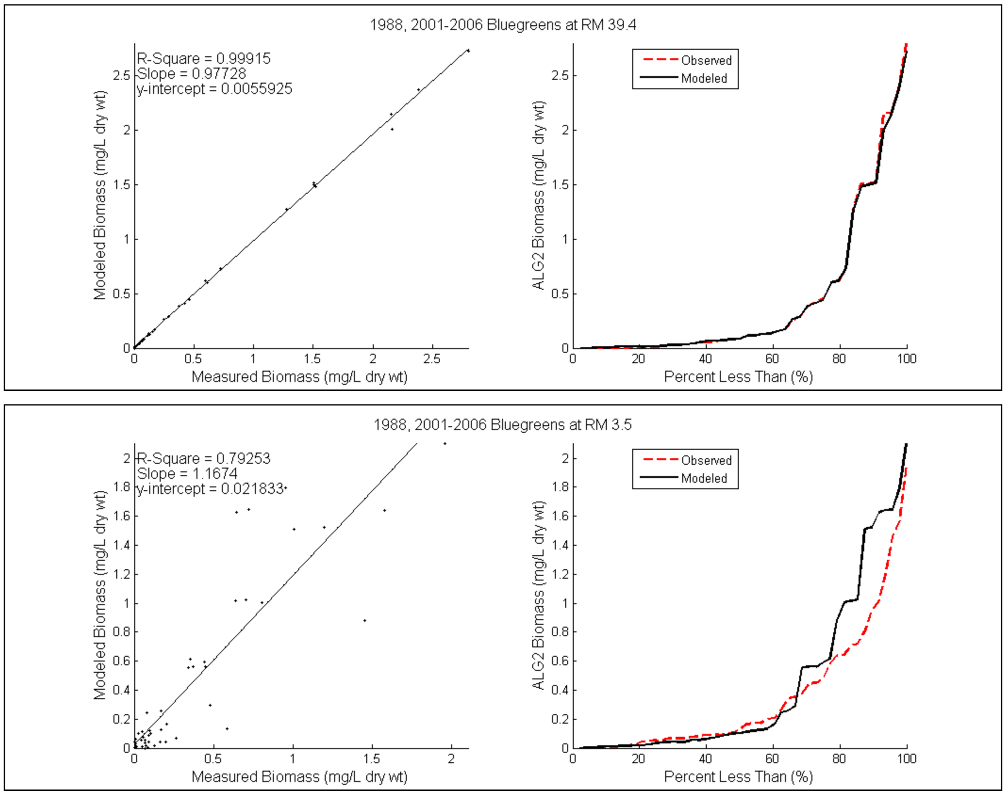


Figure 44. Bluegreens (ALG2) linear and cumulative distribution plots at RM 39.4 and RM 3.5.

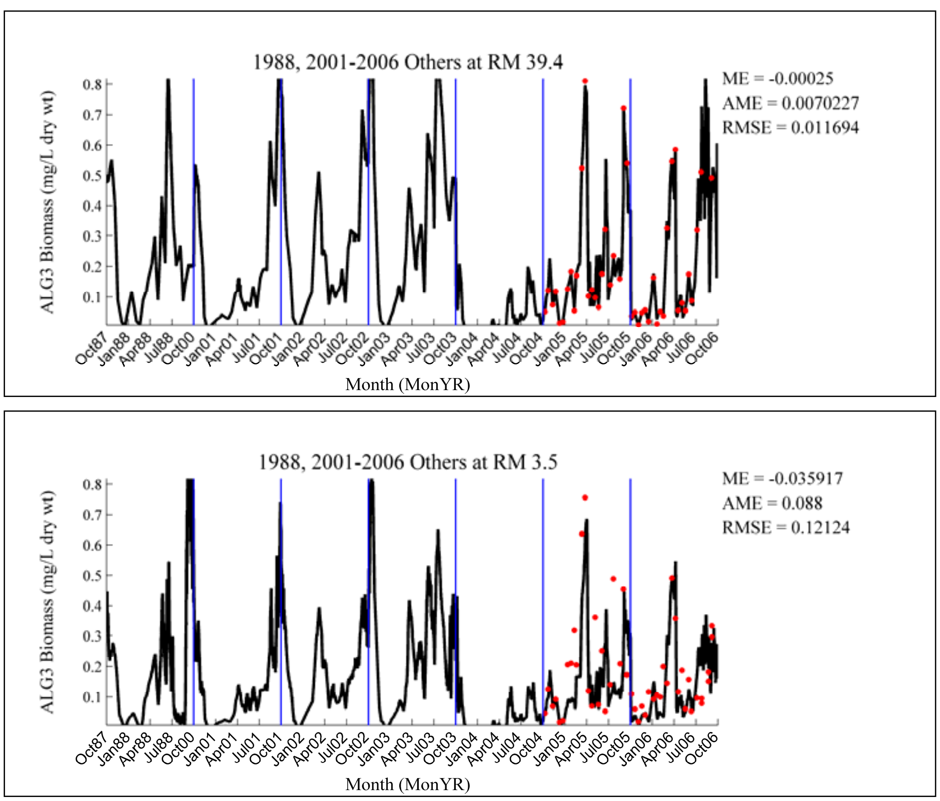


Figure 45. Others (ALG3) time series plots at RM 39.4 and RM 3.5.

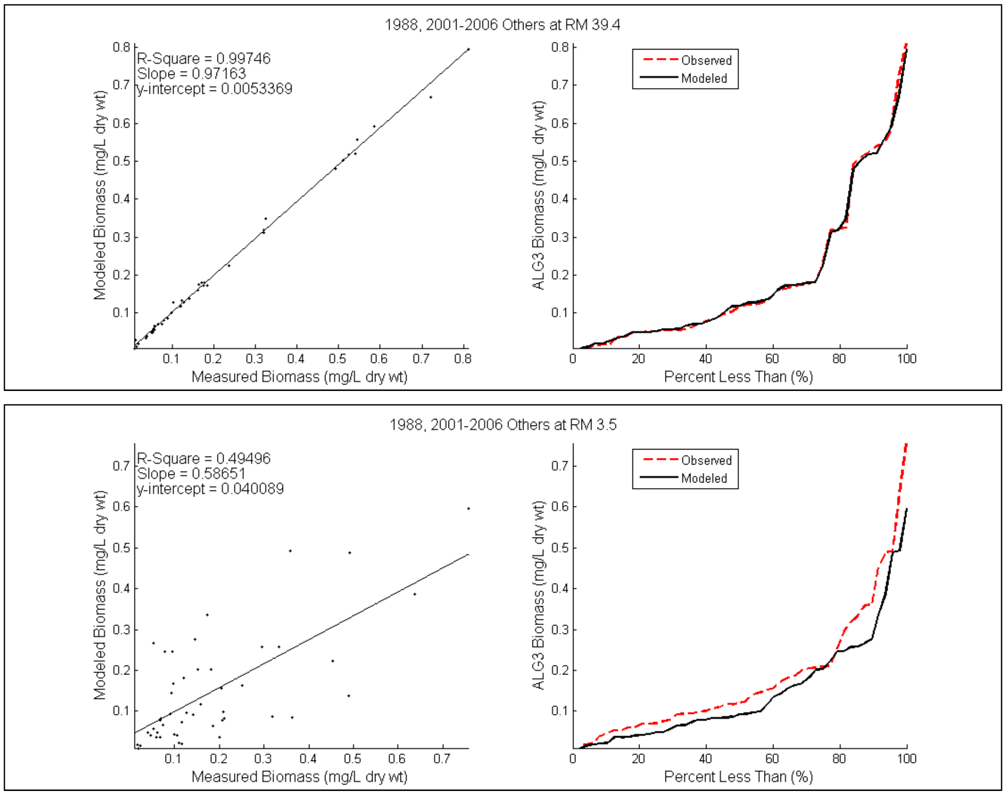


Figure 46. Others (ALG3) linear and cumulative distribution plots at RM 39.4 and RM 3.5.

Total suspended solids

TSS time series plots and linear and cumulative distribution plots are found in Figures 47 and 48, respectively. According to these figures, the model tends to do well with total suspended solids predictions. The AME = 38.31 mg/L at Fort Snelling (RM 3.5), which is well below the 10% AME target found in Table 25 (151.60 mg/L). On average, the model over-predicts TSS for the middle 40% of observed data.

Table 25. 10% target for TSS (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	2.00	943.00	9.54	94.10
RM 25.1	3.00	734.00	25.12	73.10
RM 14.3	2.00	600.00	34.12	59.80
RM 8.5	2.00	884.00	31.28	88.20
RM 3.5	4.00	1520.00	38.31	151.60

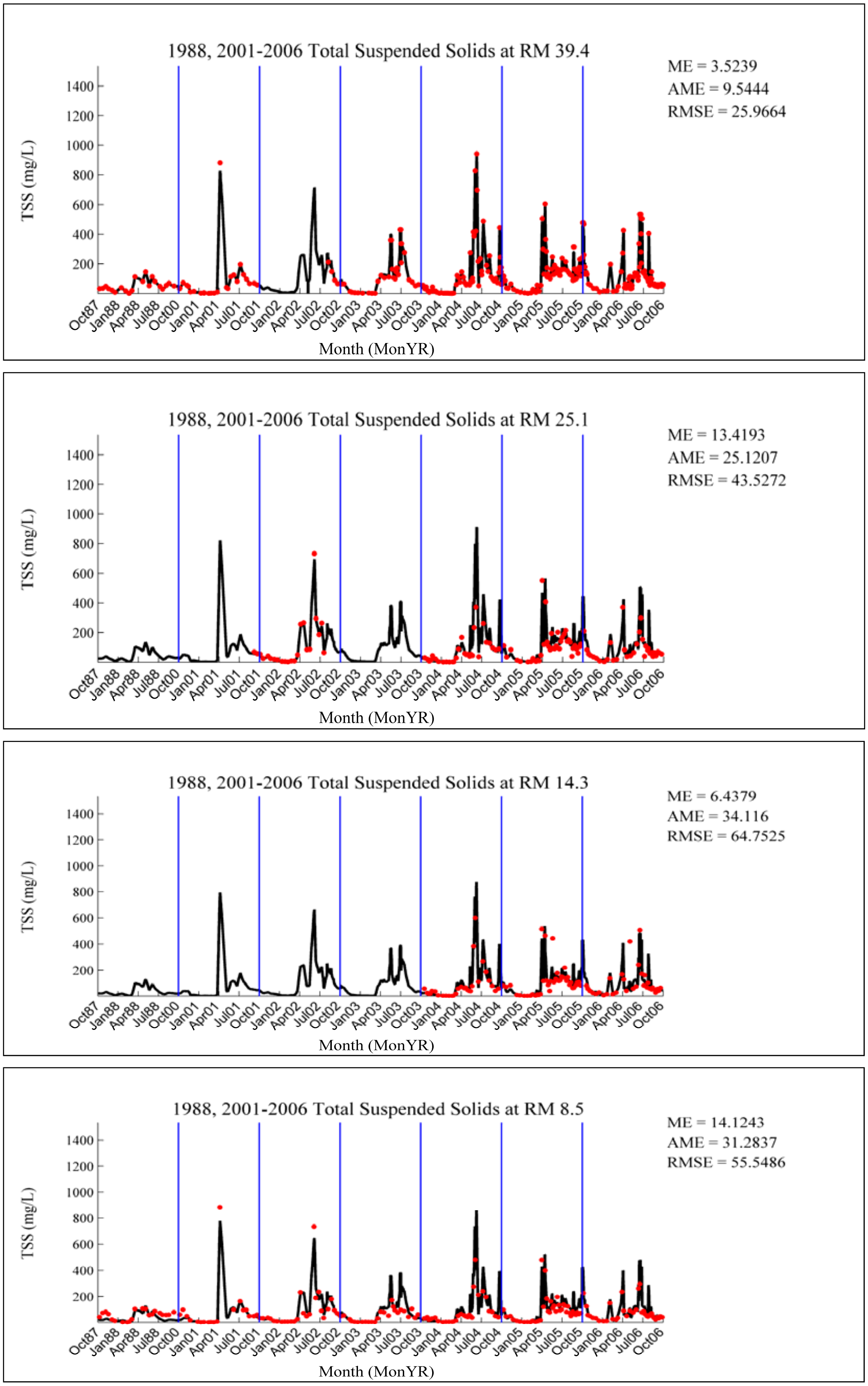


Figure 47. TSS at various calibration stations (continued).

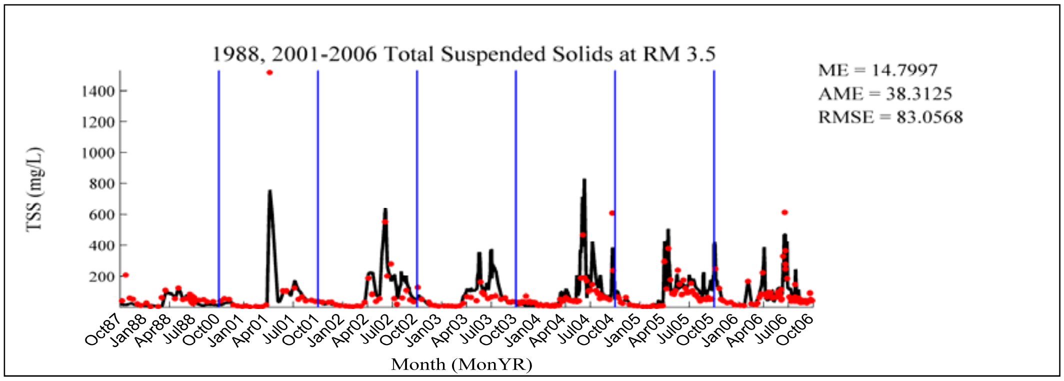


Figure 47. (concluded).

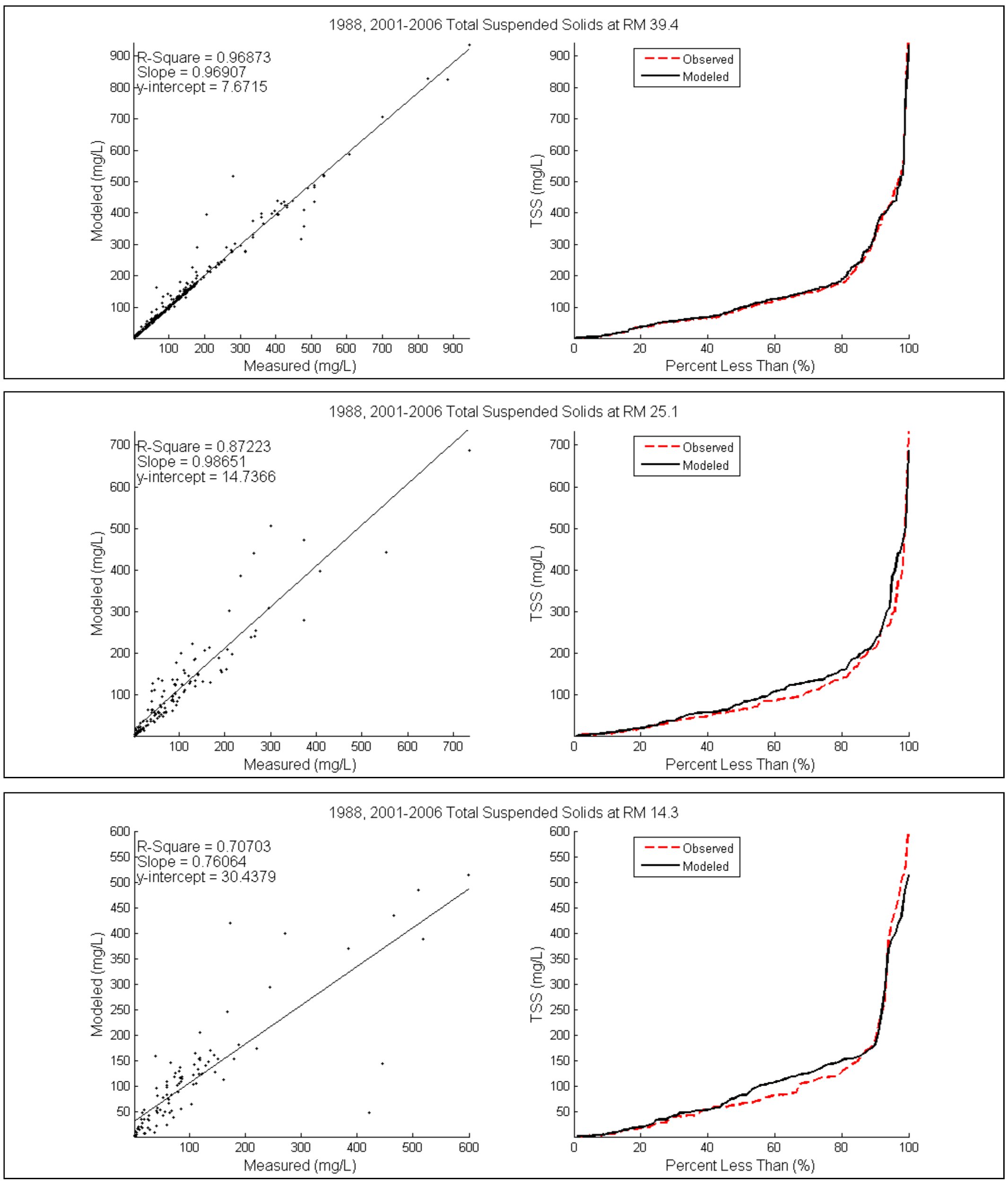


Figure 48. TSS linear and cumulative distribution plots at various calibration stations (continued).

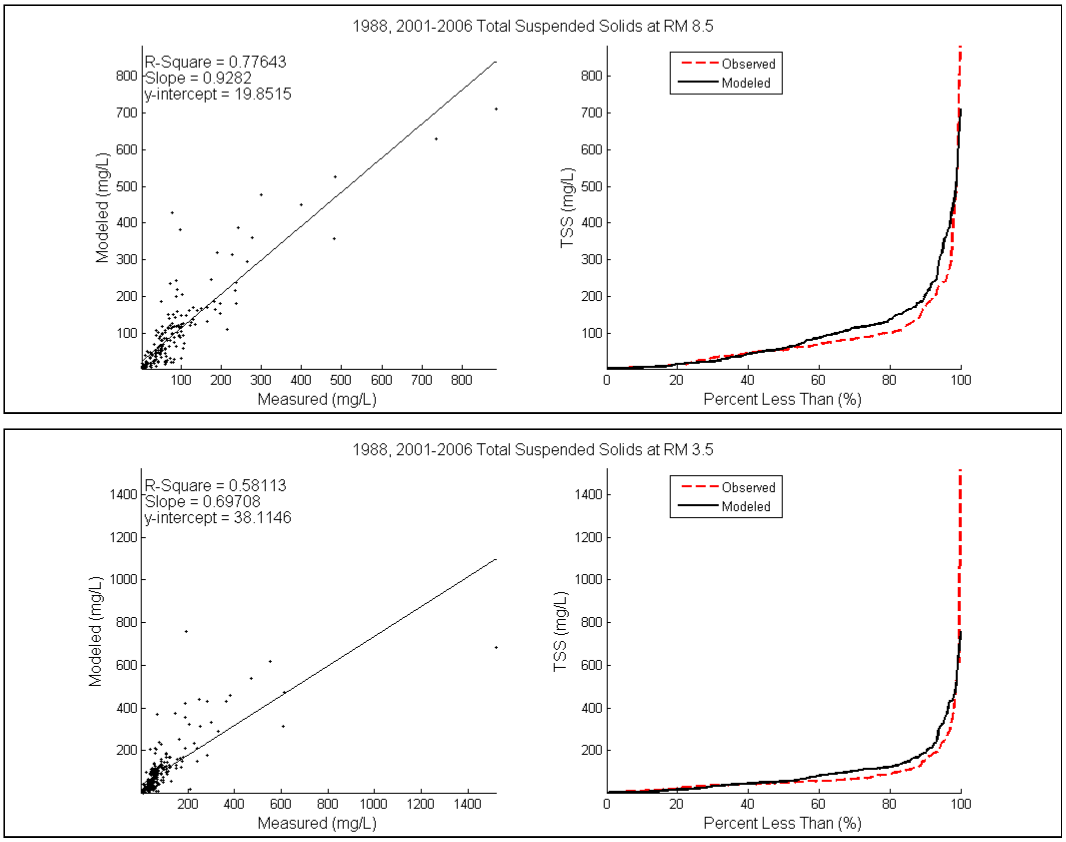


Figure 48. (concluded).

Dissolved organic carbon

Time series plots and linear and cumulative distribution plots for dissolved organic carbon are found in Figures 49 and 50, respectively. According to these figures, the model tends to do well with dissolved organic carbon predictions. The exception to that is at Fort Snelling, where the AME = 1.117 (mg/L) is above the 10% AME target found in Table 26. BOD data for the airport stormwater varied greatly, which led to uncertainty in the loading estimates and characteristics (i.e., U:5 ratios and decay rates). This, in turn, affected model results for the derived variables DOC and BOD at RM 3.5. These inputs would benefit from further work.

Table 26. 10% target for DOC (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	3.90	11.70	0.09	0.78
RM 25.1	4.00	9.60	0.35	0.56
RM 14.3	3.90	9.80	0.38	0.59
RM 8.5	3.80	10.20	0.44	0.64
RM 3.5	4.30	11.40	1.11	0.71

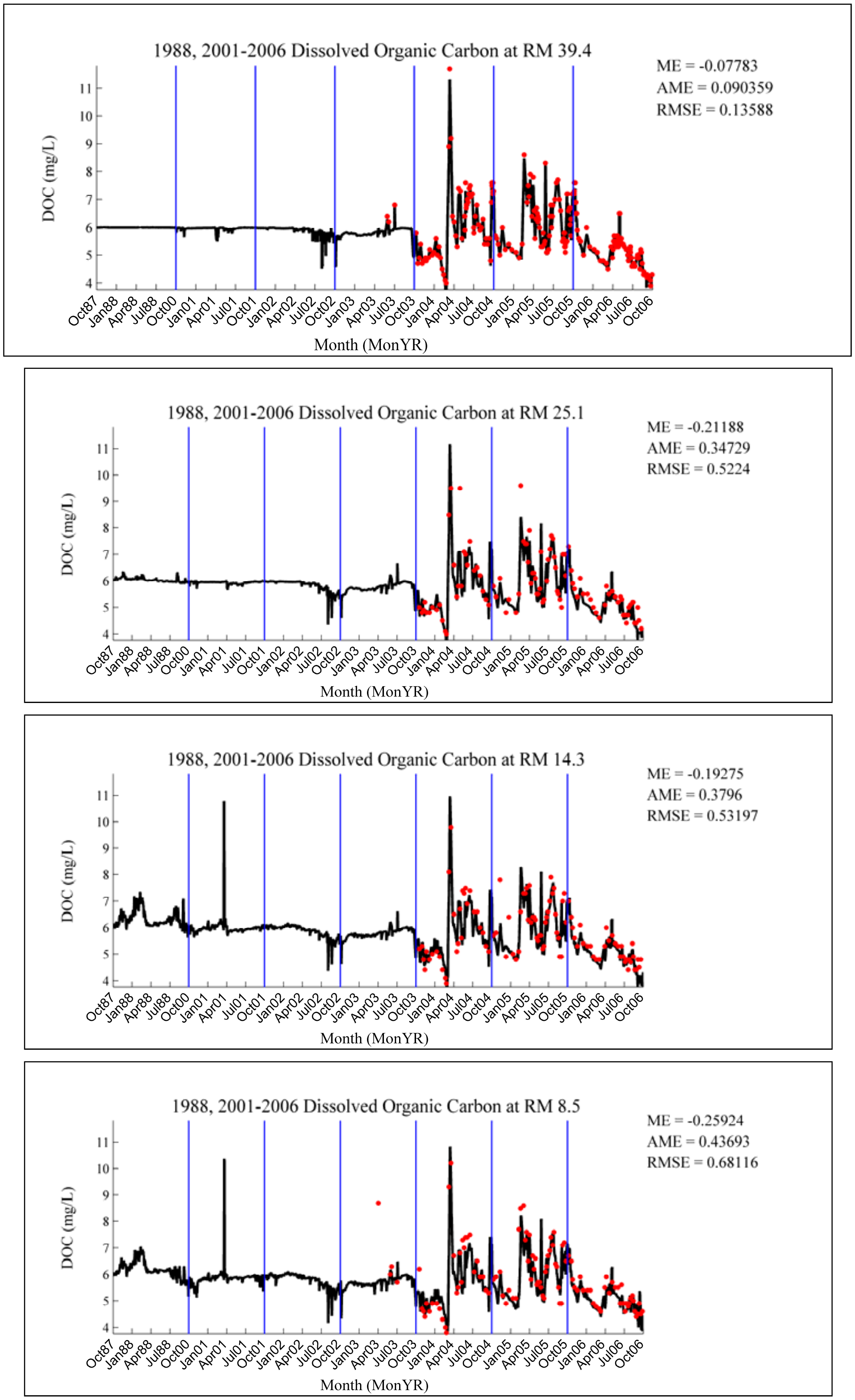


Figure 49. DOC at various calibration stations (continued).

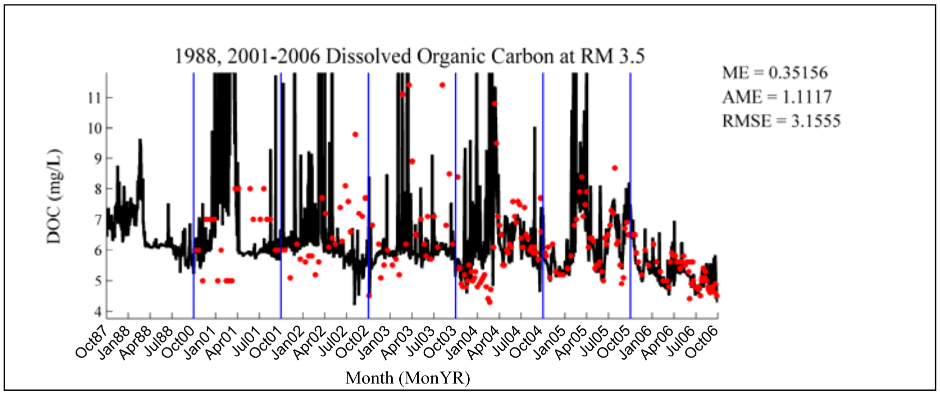


Figure 49. (concluded).

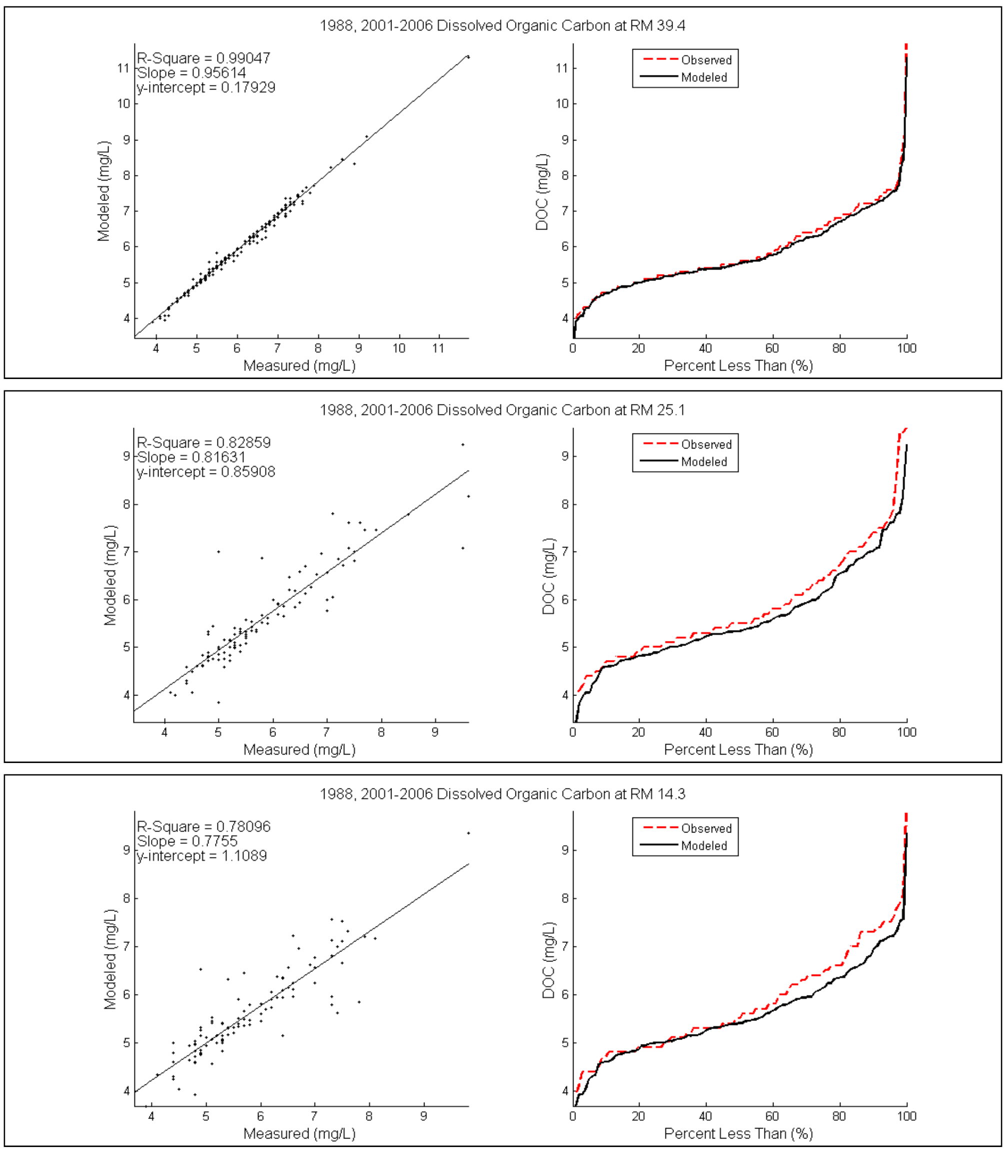


Figure 50. DOC linear and cumulative distribution plots at various calibration stations (continued).

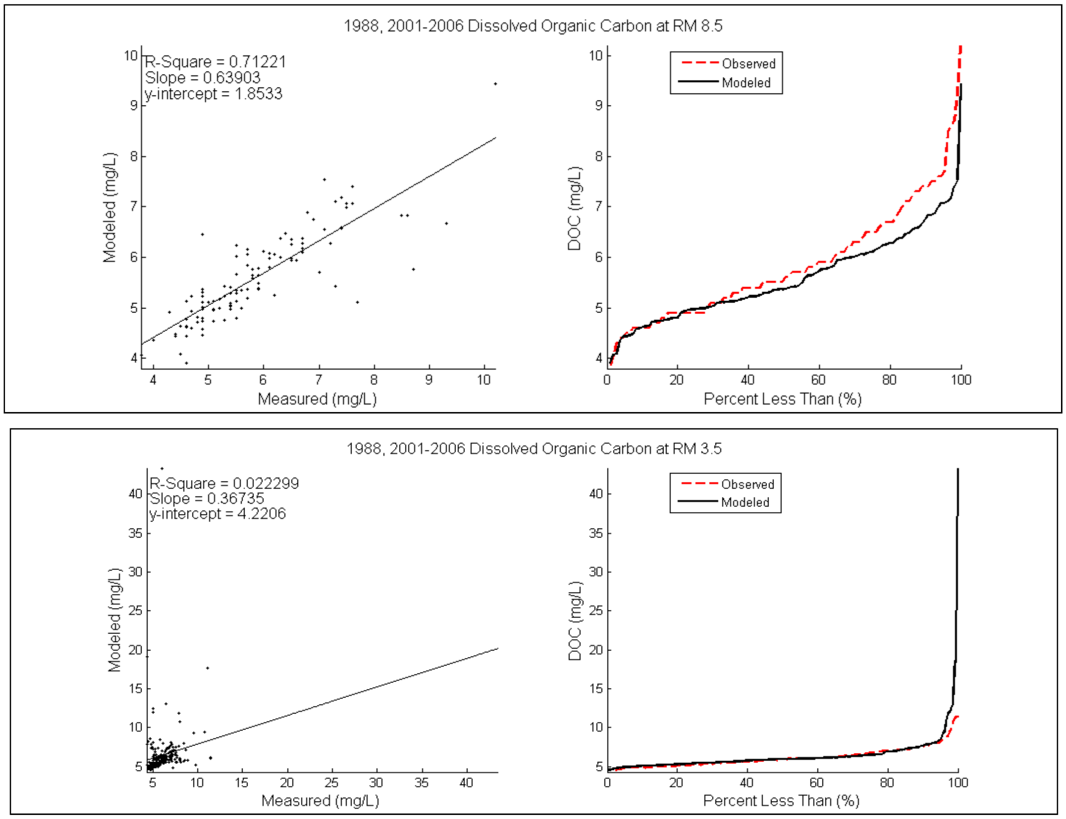


Figure 50. (concluded).

Dissolved silica

Figures 51 and 52 show time series plots and linear and cumulative distribution plots, respectively, for dissolved silica. According to these figures, the model tends to do well with dissolved silica predictions. At Fort Snelling, the AME = 1.96 mg/L, which is below the 10% AME target found in Table 27 (2.8 mg/L). At RM 3.5, the model overpredicts approximately the lower 80% of measured data by about 1.0 mg/L.

As indicated previously, DSI was not monitored at Jordan until WY 2004. By studying the time-series plot at RM 3.5 below, one can see how sampling improved model calibration. When monitoring data were not available, the model tended to miss the minimums and maximums of DSI. However, once DSI data were available beginning in WY 2004, model predictions fell more in line with the observed data at Fort Snelling (RM 3.5).

Table 27. 10% target for DSI (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	1.00	29.00	0.17	2.80
RM 25.1	1.00	28.00	0.56	2.70
RM 14.3	1.00	28.00	0.78	2.70
RM 8.5	1.00	29.00	1.20	2.80
RM 3.5	1.00	29.00	1.96	2.80

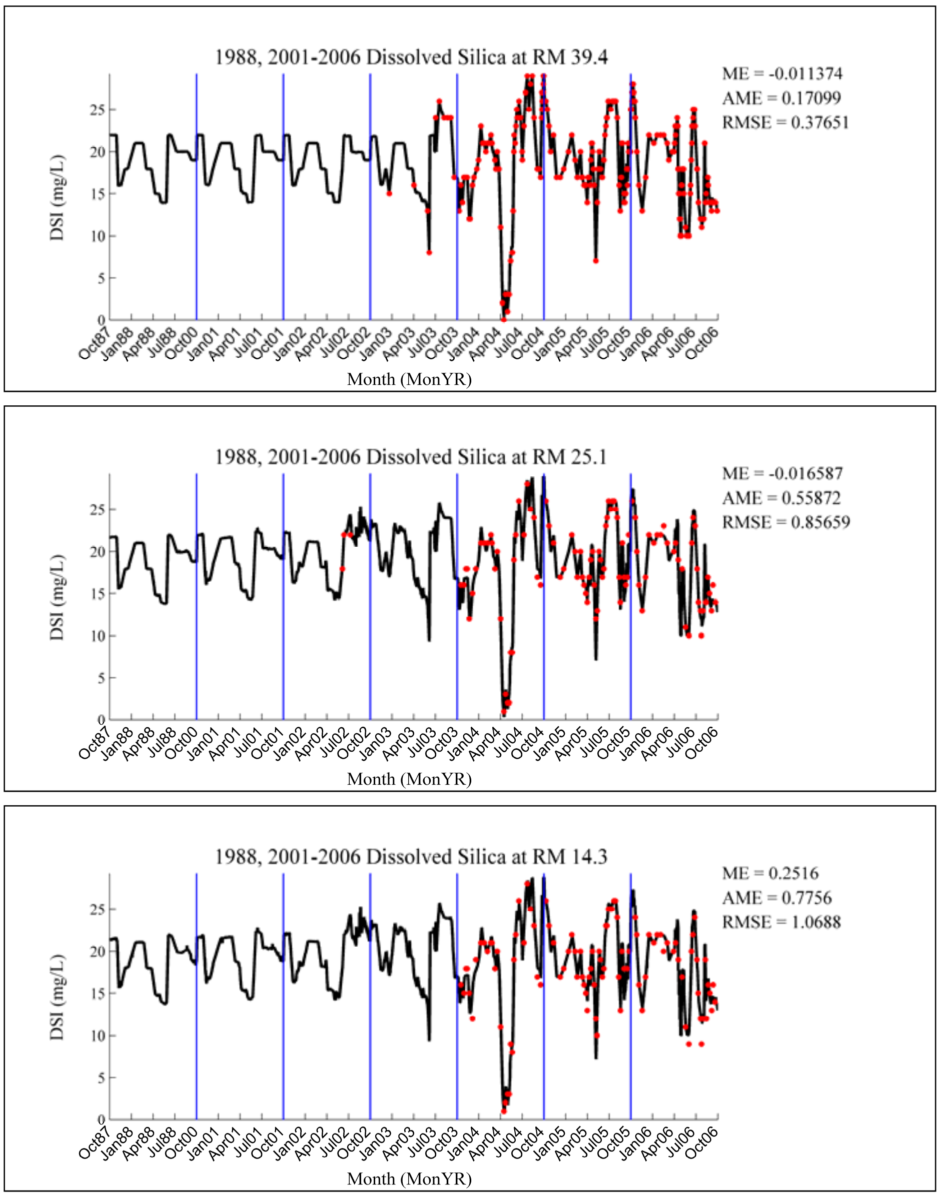


Figure 51. DSI at various calibration stations (continued).

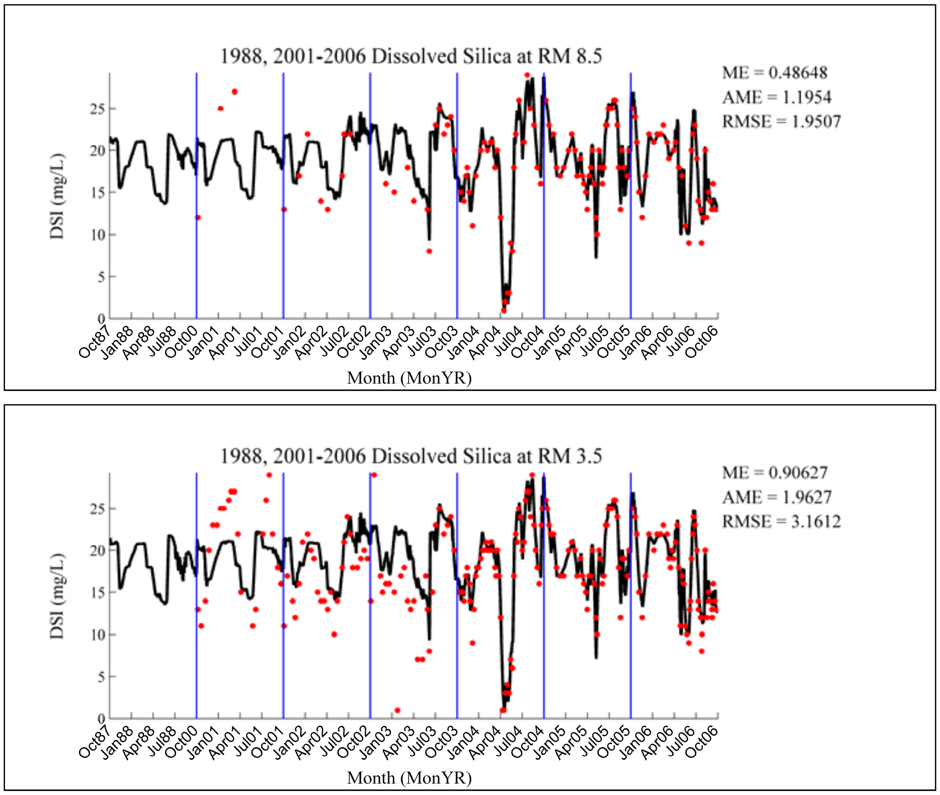


Figure 51. (concluded).

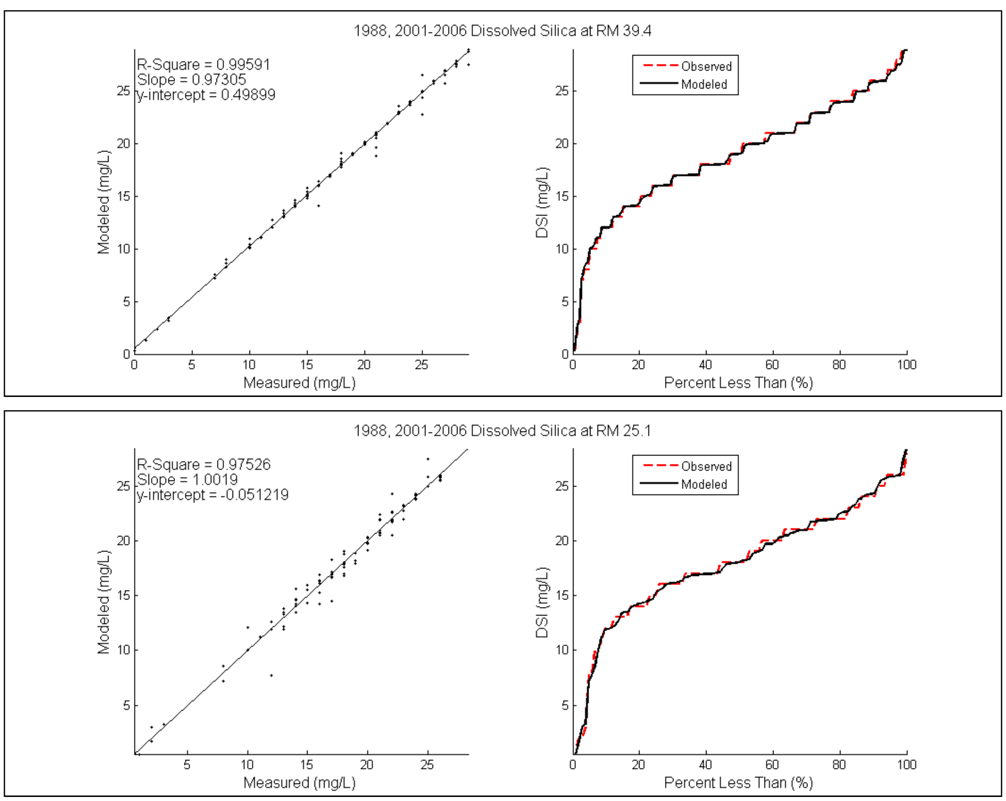


Figure 52. DSI linear and cumulative distribution plots at various calibration stations (continued).

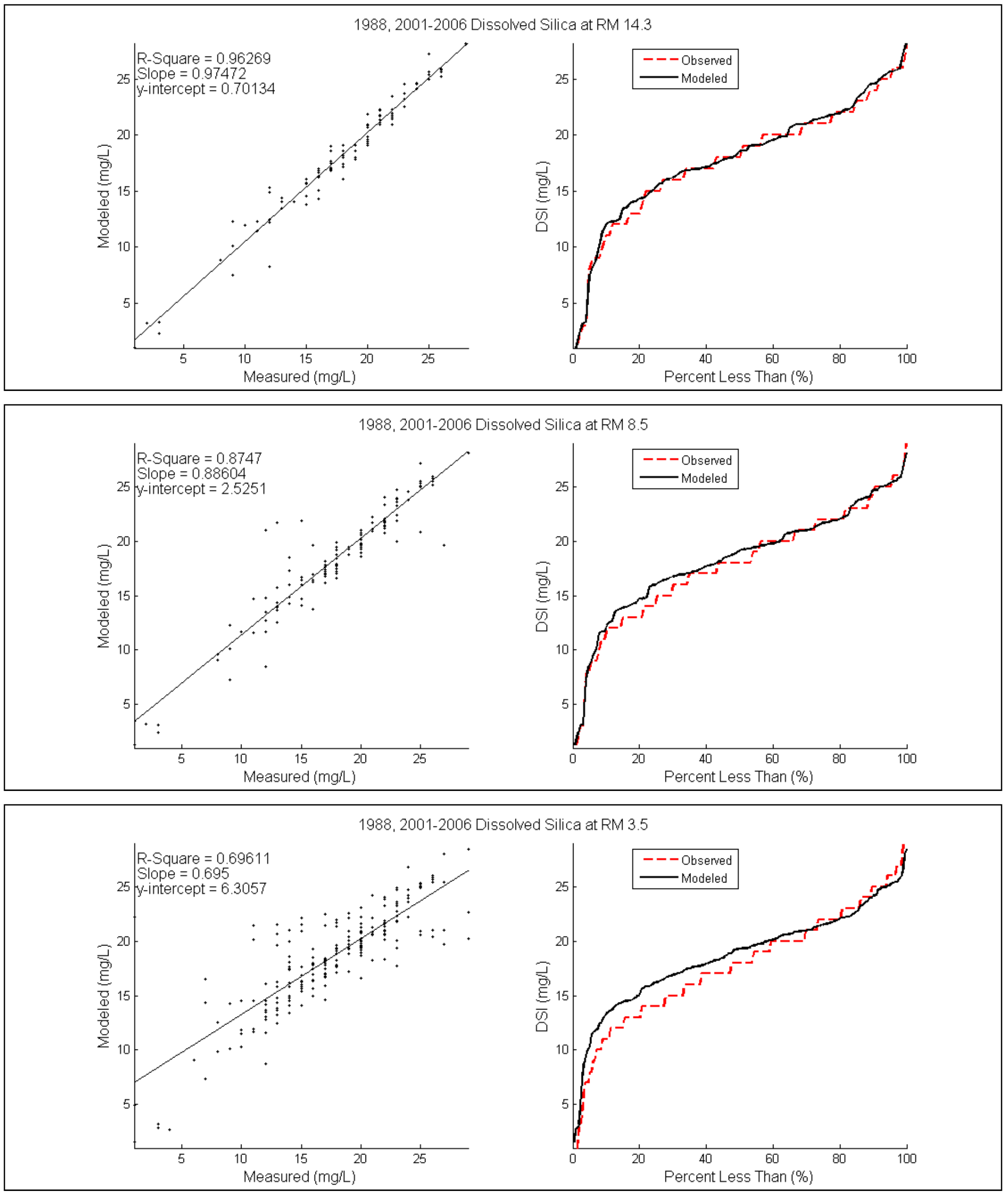


Figure 52. (concluded).

Inorganic suspended solids

Figures 53 and 54 show ISS time series plots and linear and cumulative distribution plots, respectively. According to these figures, the model tends to do well with inorganic suspended solids predictions. At Fort Snelling, the AME = 37.58 mg/L, which is well below the 10% AME target found in Table 28 (143.30 mg/L). At RM 3.5, the model overpredicts approximately the upper 40% of measured data by about 20.0 mg/L.

Table 28. 10% target for ISS (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	1.00	916.00	8.24	91.50
RM 25.1	1.00	654.00	24.88	65.30
RM 14.3	0.00	560.00	33.32	56.00
RM 8.5	0.00	808.00	30.58	80.80
RM 3.5	2.00	1435.00	37.58	143.30

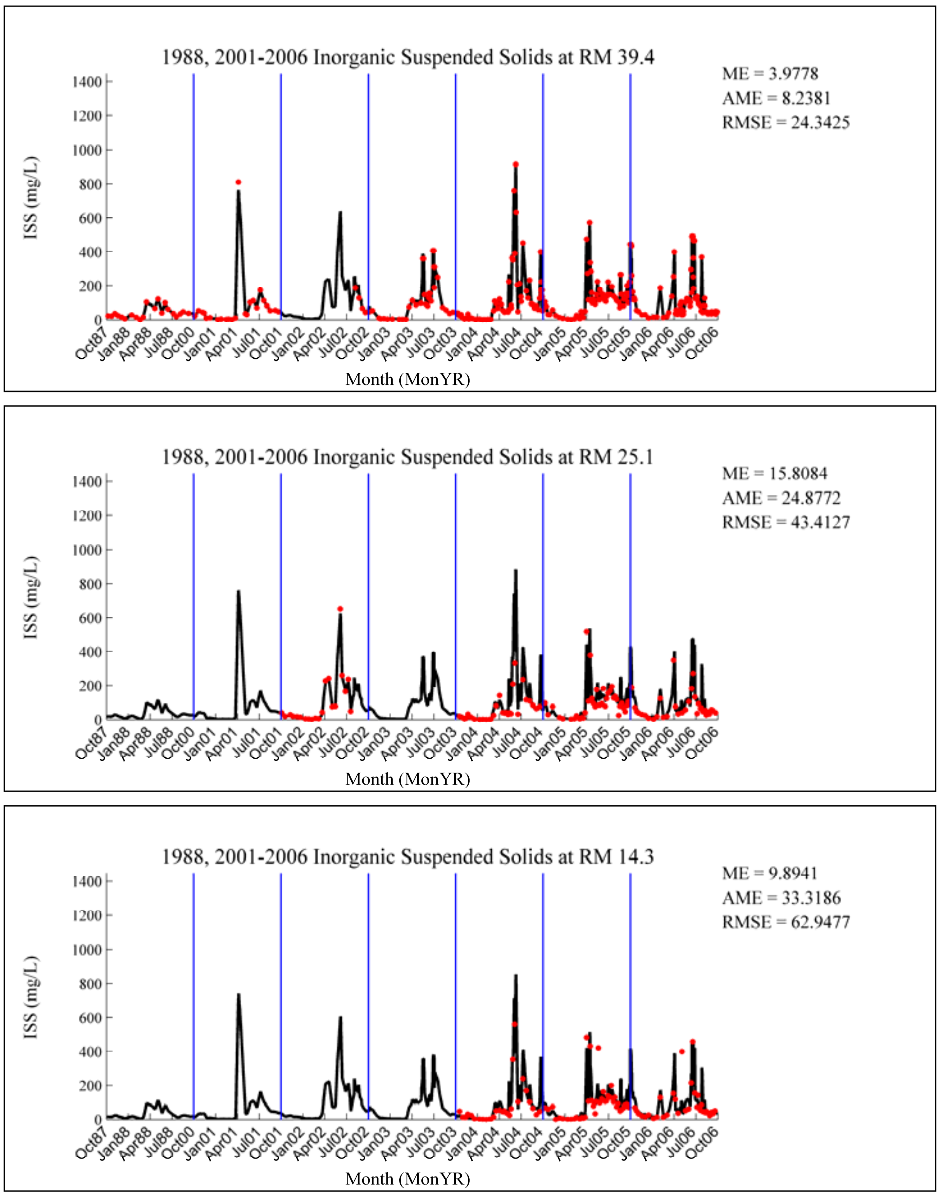


Figure 53. ISS at various calibration stations (continued).

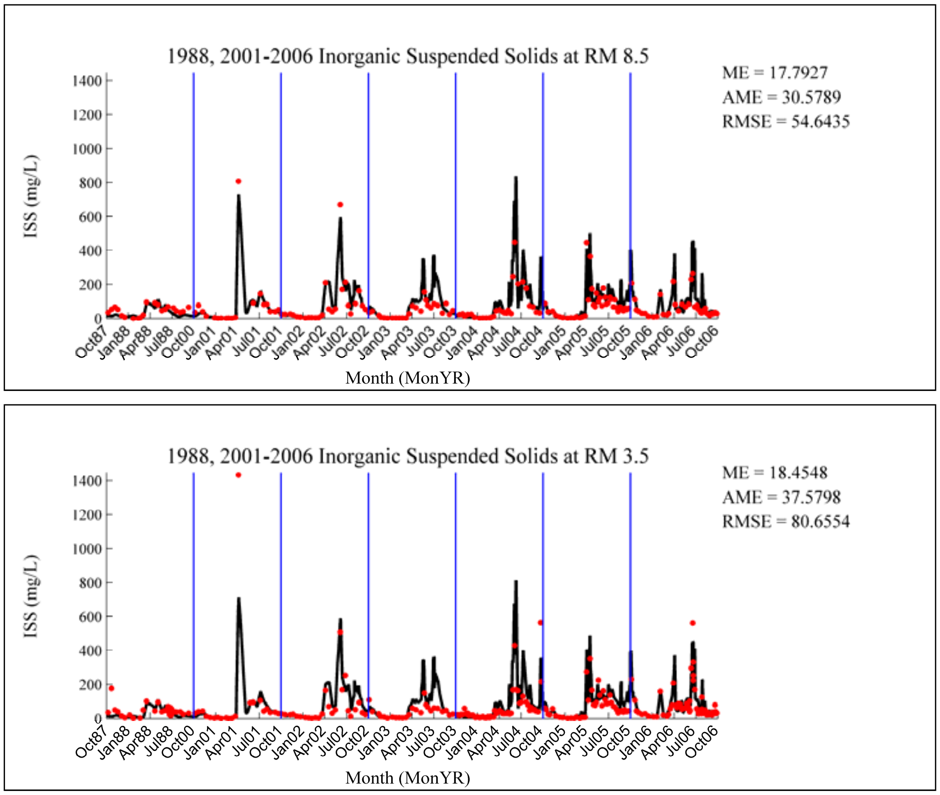


Figure 53. (concluded).

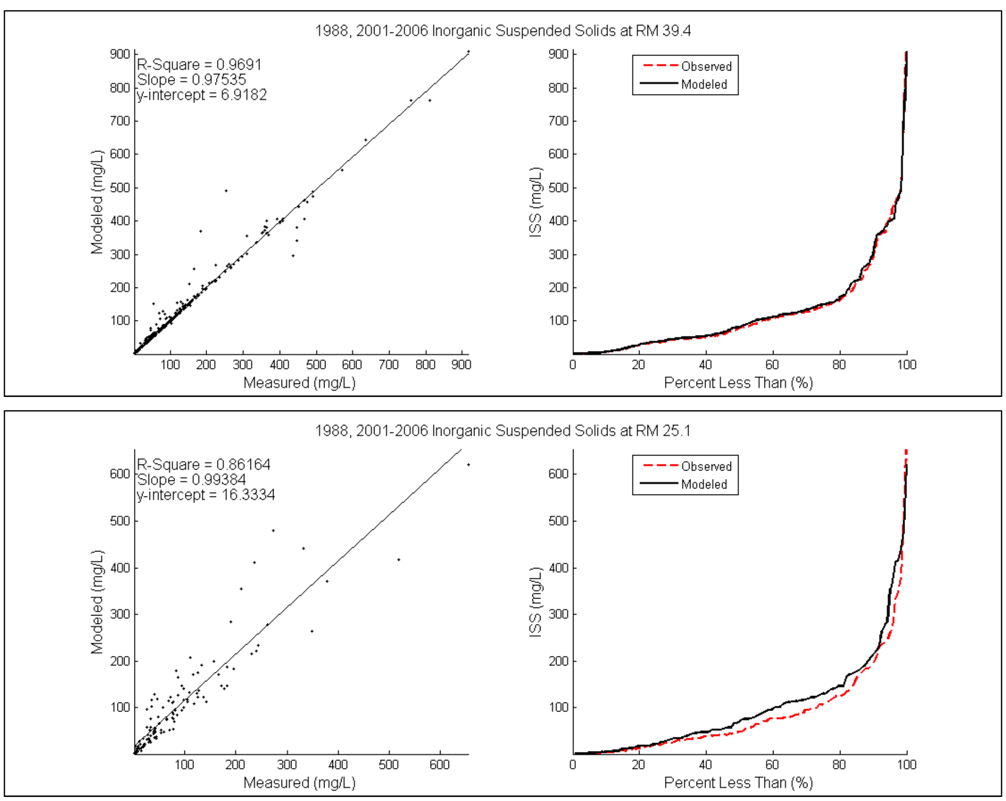


Figure 54. ISS linear and cumulative distribution plots at various calibration stations (continued).

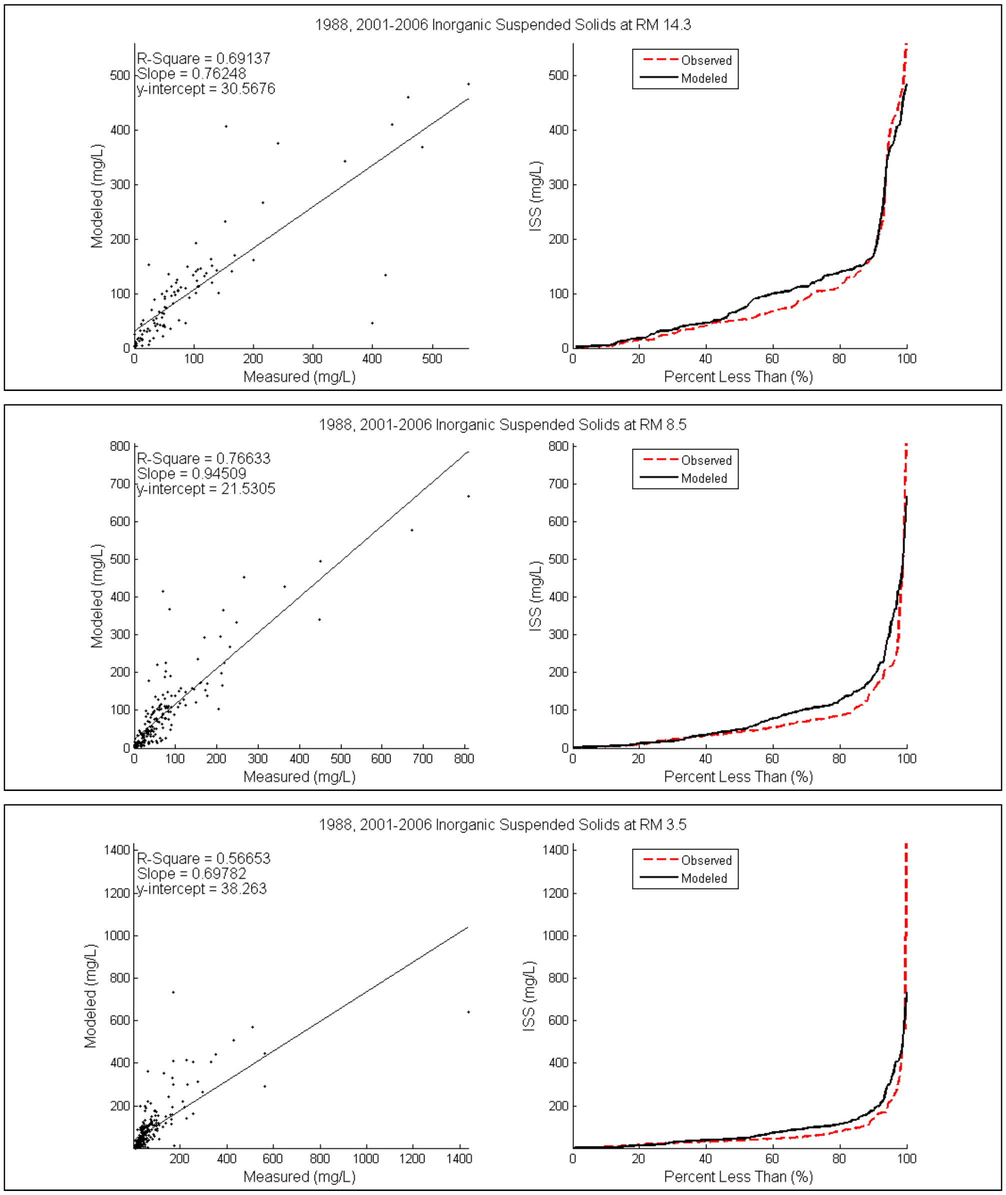


Figure 54. (concluded).

Nitrate + nitrite nitrogen

Figures 55 and 56 show NO3 time series plots and linear and cumulative distribution plots, respectively. According to these figures, the model tends to do very well with nitrate-nitrite predictions. At Fort Snelling, the AME = 0.62 mg/L, which is well below the 10% AME target found in Table 29 (1.46 mg/L).

Table 29. 10% target for NO3 (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	0.05	16.40	0.13	1.64
RM 25.1	0.05	14.70	0.32	1.47
RM 14.3	0.07	14.30	0.45	1.42
RM 8.5	0.06	14.00	0.57	1.39
RM 3.5	0.09	14.70	0.62	1.46

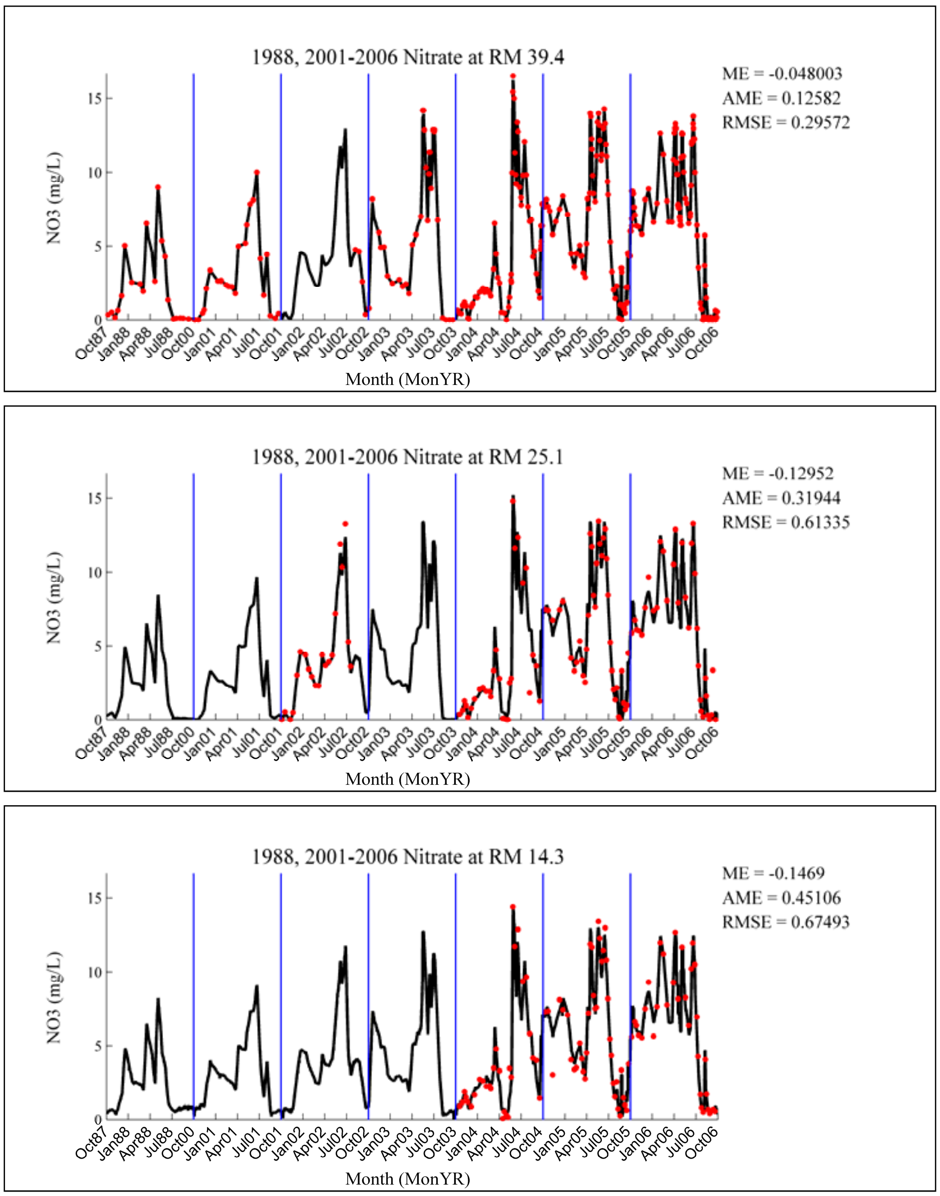


Figure 55. NO3 at various calibration stations (continued).

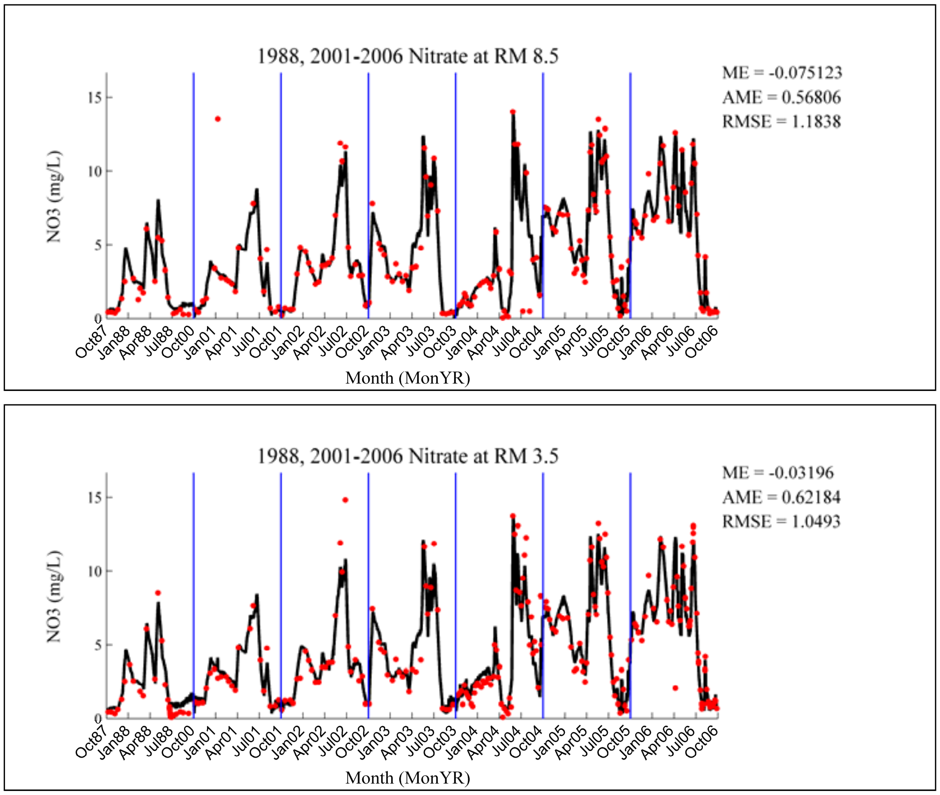


Figure 55. (concluded).

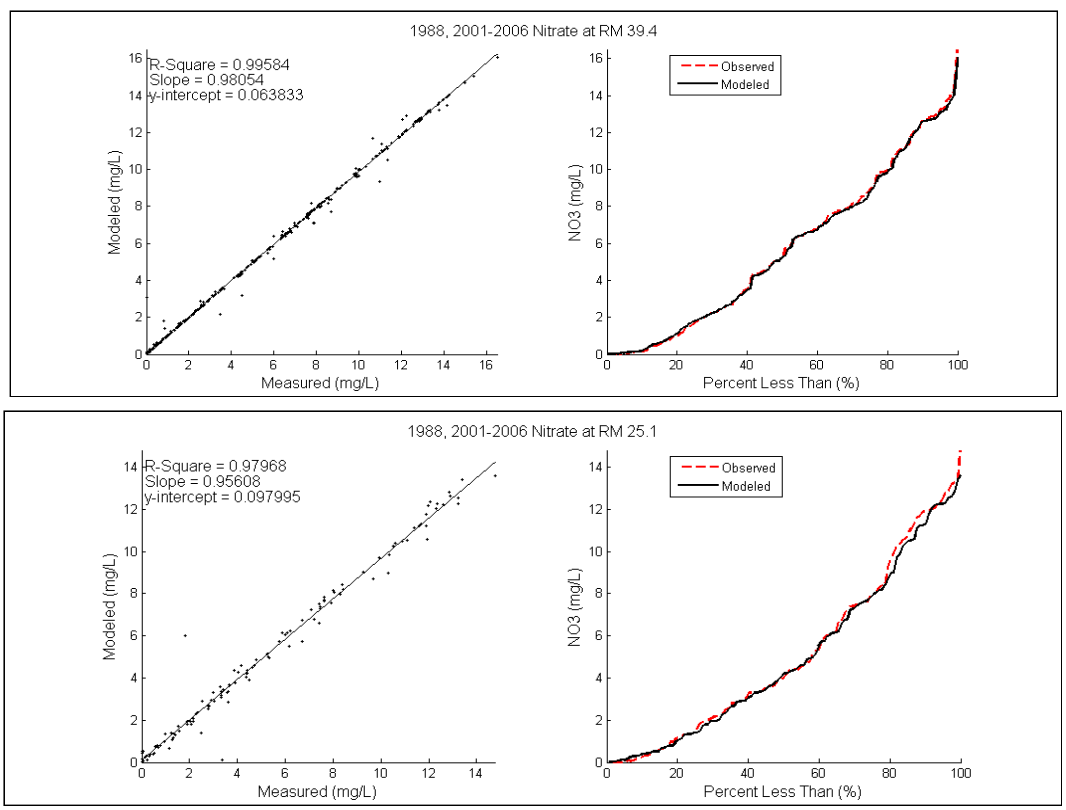


Figure 56. NO3 linear and cumulative distribution plots at various calibration stations (continued).

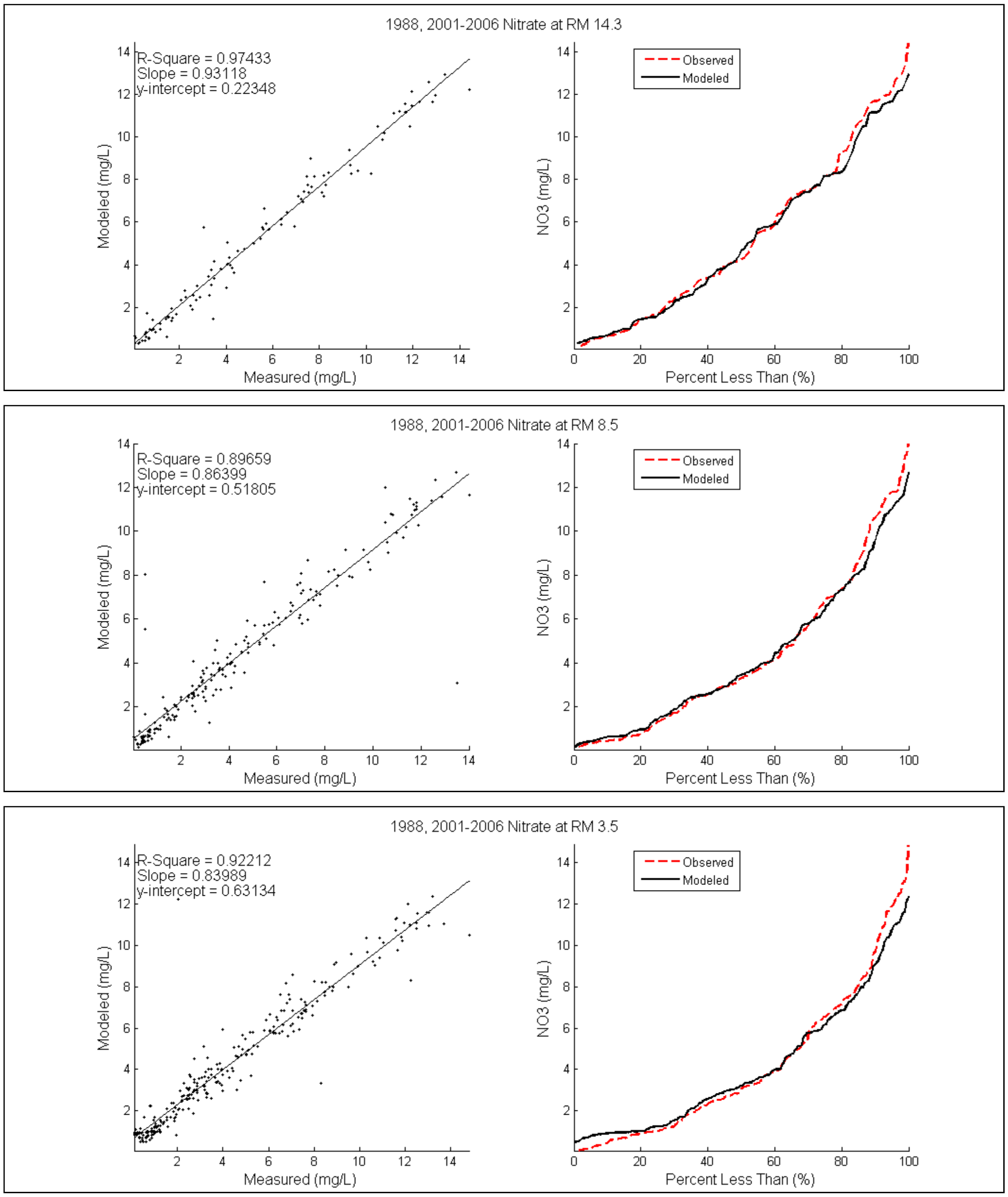


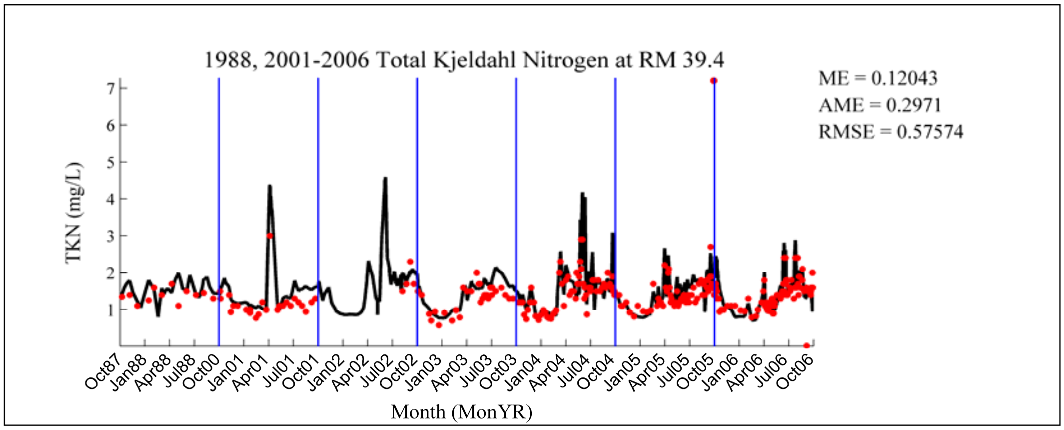
Figure 56. (concluded).

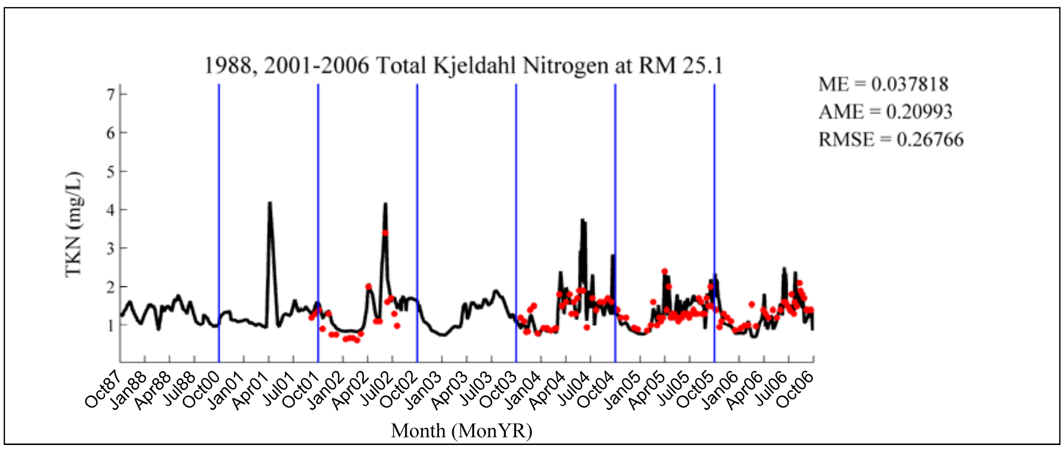
Total Kjeldahl nitrogen

Figures 57 and 58 show TKN time series plots and linear and cumulative distribution plots, respectively. According to these figures, the model tends to do very well with total Kjeldahl nitrogen predictions. At Fort Snelling, the AME = 0.32108 mg/L, which is below the 10% AME target found in Table 30 (0.47 mg/L).

Table 30. 10% target for TKN (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	0.04	7.20	0.30	0.72
RM 25.1	0.60	3.40	0.21	0.28
RM 14.3	0.78	2.50	0.20	0.17
RM 8.5	0.63	4.00	0.21	0.34
RM 3.5	0.03	4.70	0.32	0.47





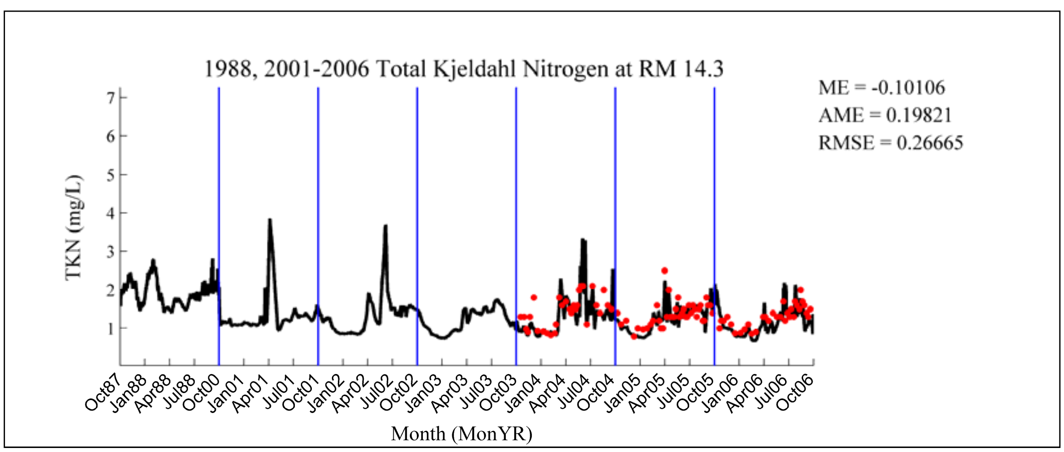


Figure 57. TKN at various calibration stations (continued).

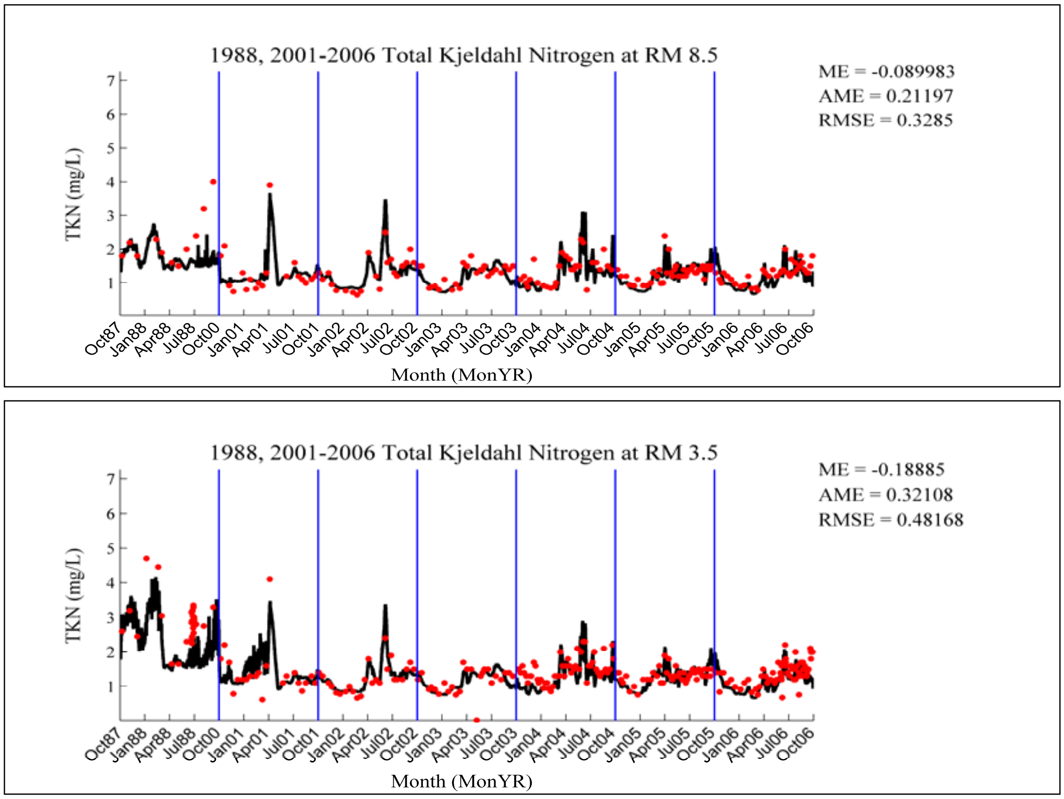


Figure 57. (concluded).

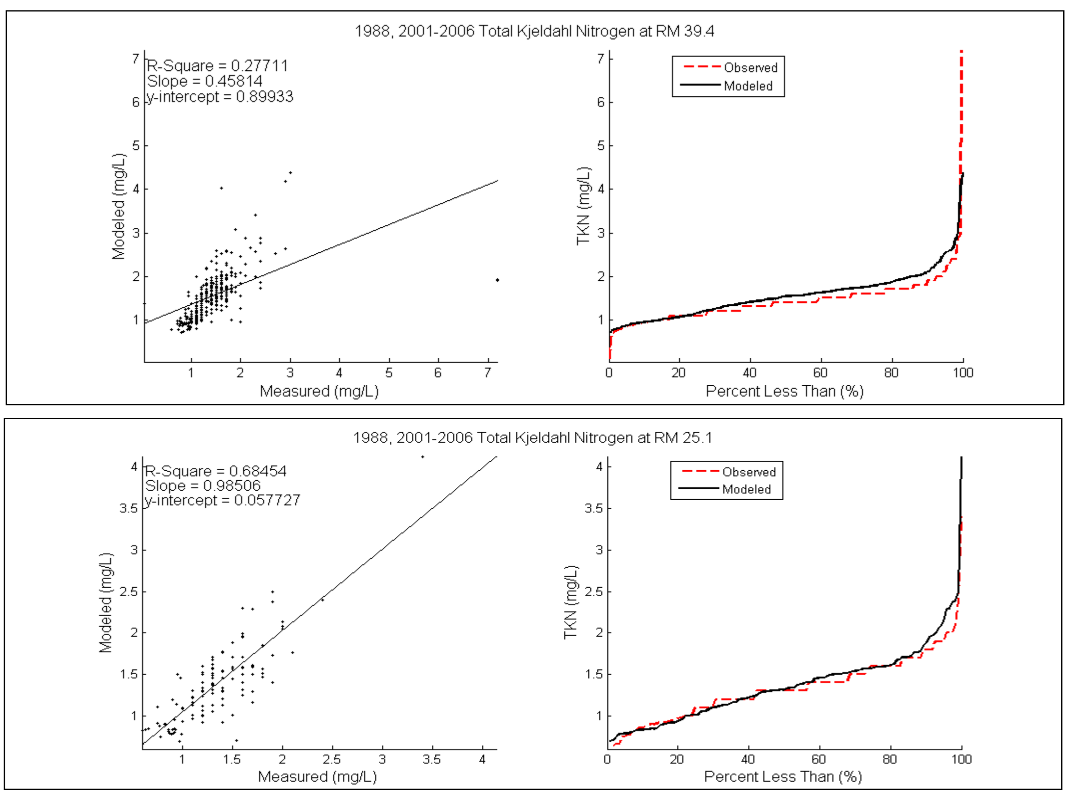


Figure 58. TKN linear and cumulative distribution plots at various calibration stations (continued).

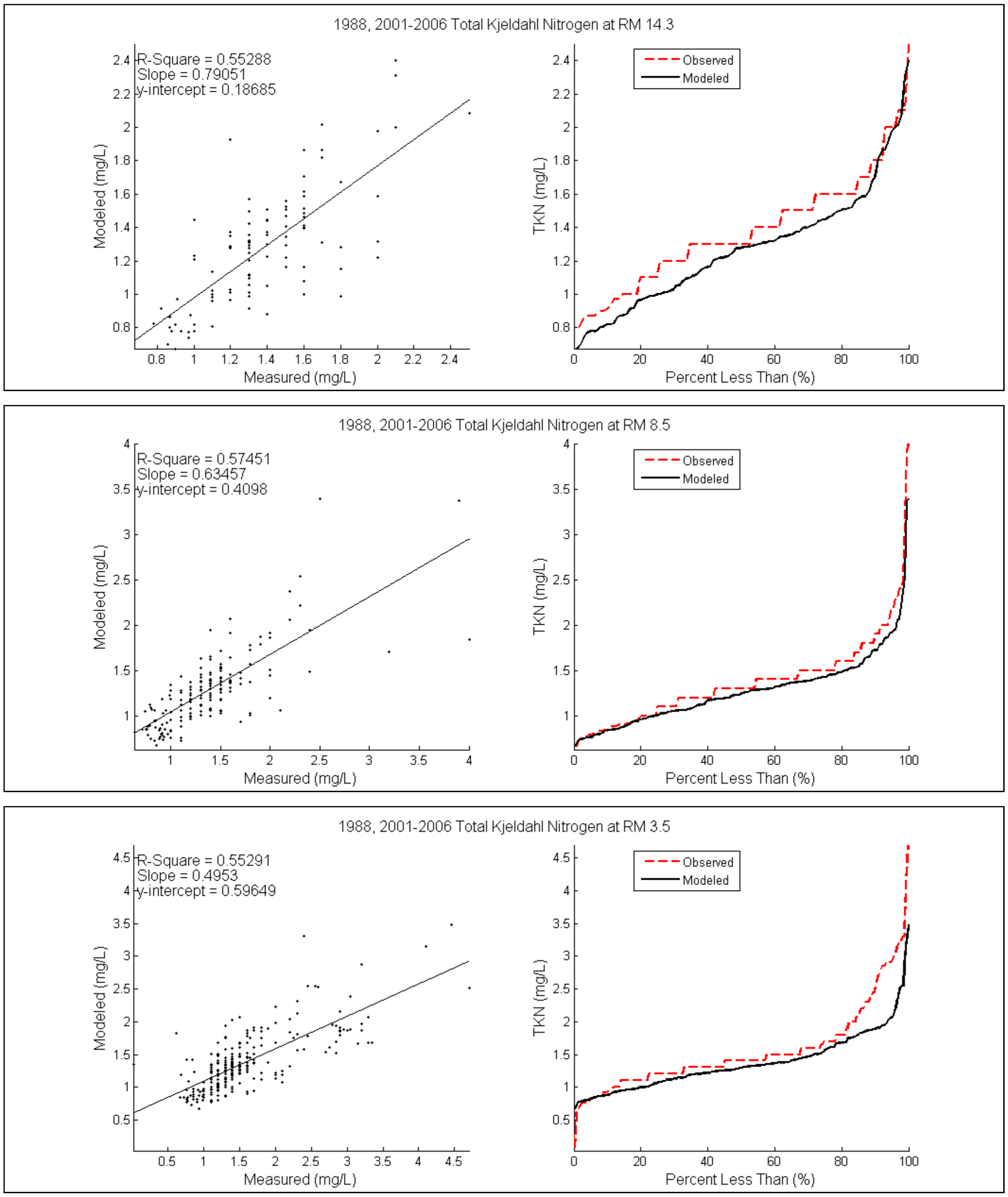


Figure 58. (concluded).

Bioavailable phosphorus

Figures 59 and 60 show PO_4 time series plots and linear and cumulative distribution plots, respectively. According to these figures, the model tends to do very well with PO_4 predictions. At Fort Snelling, the AME = 0.04 mg/L, which is below the 10% AME target found in Table 31 (0.06 mg/L). The model tends to overpredict orthophosphate downstream from Blue Lake beginning at RM 14.3.

Table 31. 10% target for PO4 (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	0.00	0.34	0.00	0.03
RM 25.1	0.01	0.31	0.01	0.03
RM 14.3	0.01	0.25	0.02	0.02
RM 8.5	0.01	0.43	0.02	0.04
RM 3.5	0.01	0.61	0.04	0.06

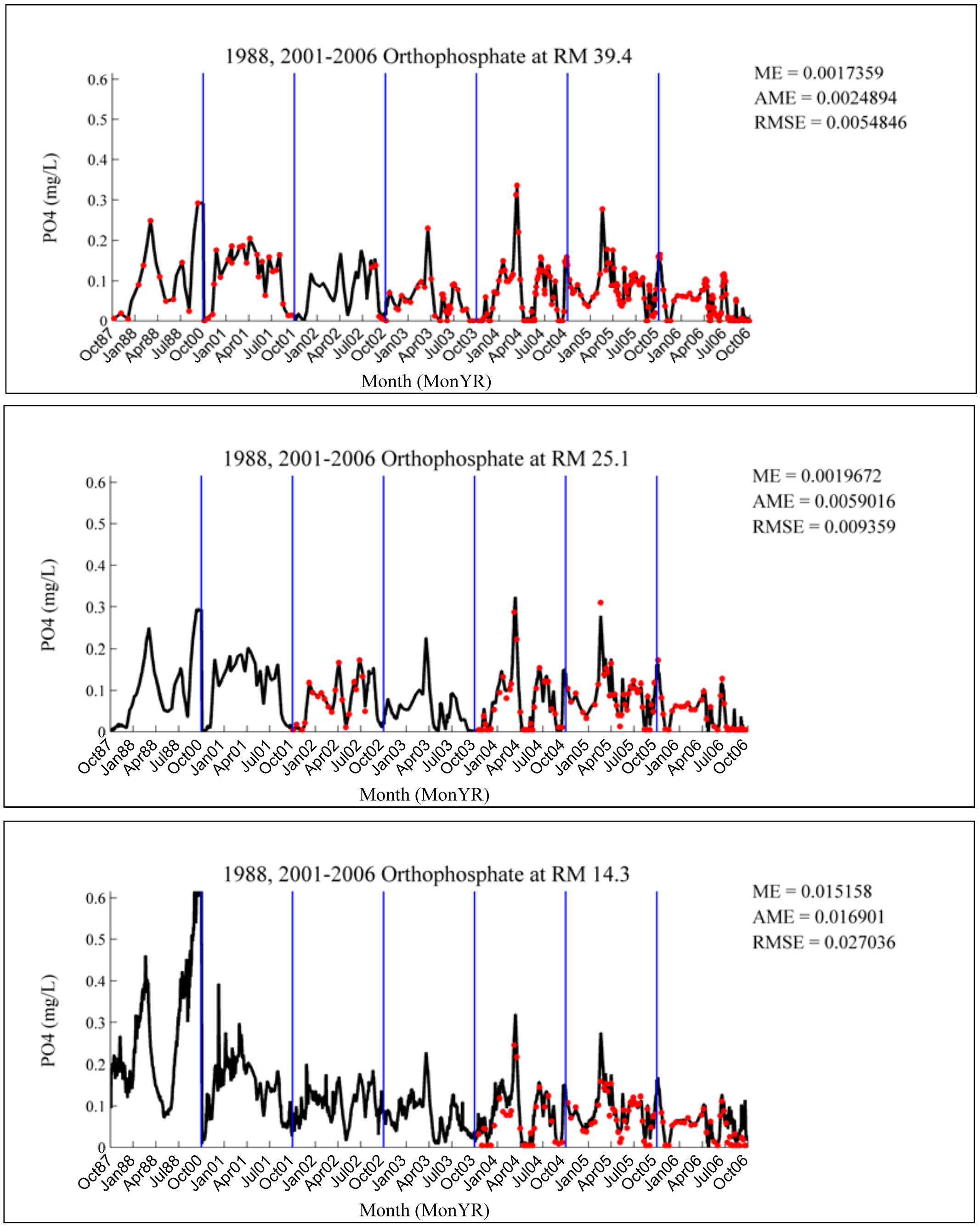


Figure 59. PO4 at various calibration stations (continued).

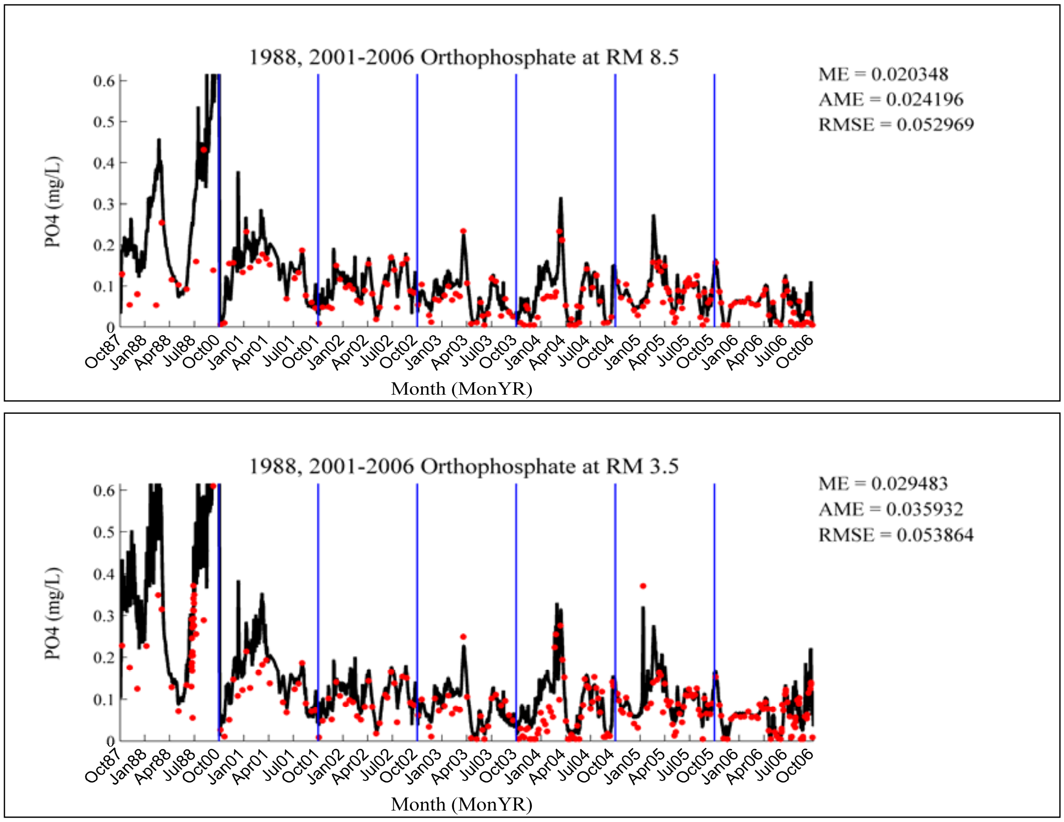


Figure 59. (concluded).

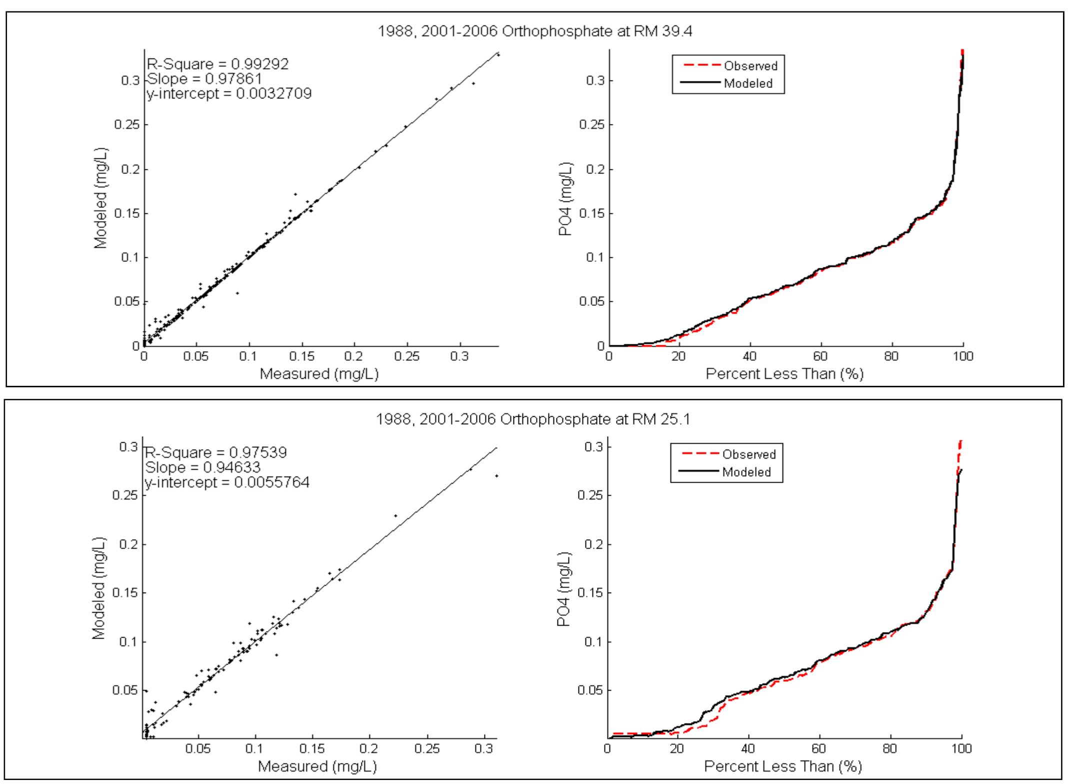


Figure 60. PO4 linear and cumulative distribution plots at various calibration stations (continued).

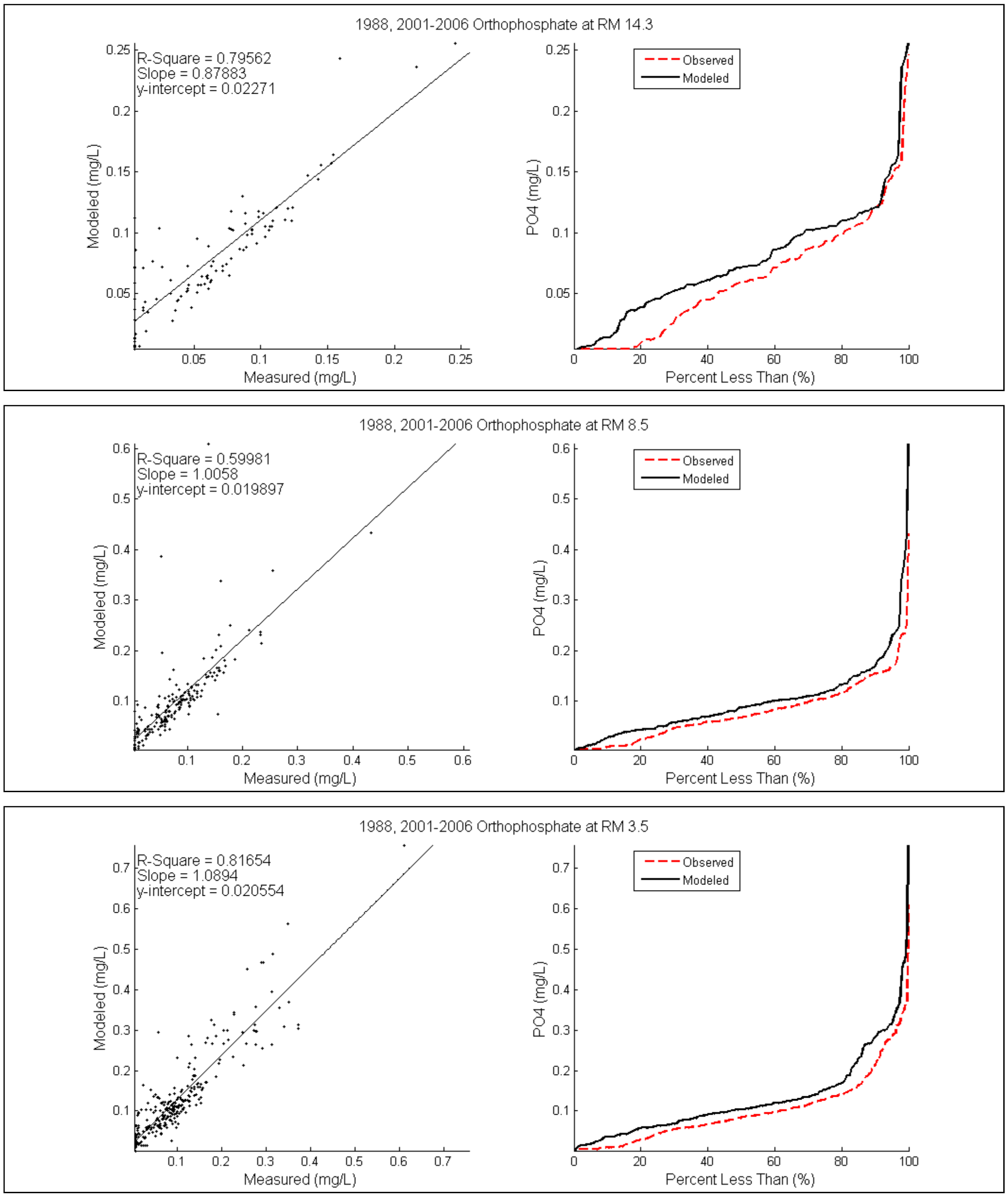


Figure 60. (concluded).

Total dissolved solids

TDS time series plots and linear and cumulative distribution plots are shown in Figures 61 and 62, respectively. According to these figures, the model tends to do very well with total dissolved solids predictions. At Fort Snelling, the AME = 31.95 mg/L, which is well below the 10% AME target found in Table 32 (79.10 mg/L). The model tends to slightly over-predict TDS downstream from the Black Dog Generating Plant withdrawal beginning at RM 8.5.

Table 32. 10% target for TDS (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	56.00	874.00	5.32	81.80
RM 25.1	361.00	836.00	13.47	47.50
RM 14.3	361.00	834.00	22.67	47.30
RM 8.5	57.00	842.00	27.44	78.50
RM 3.5	55.00	846.00	31.95	79.10

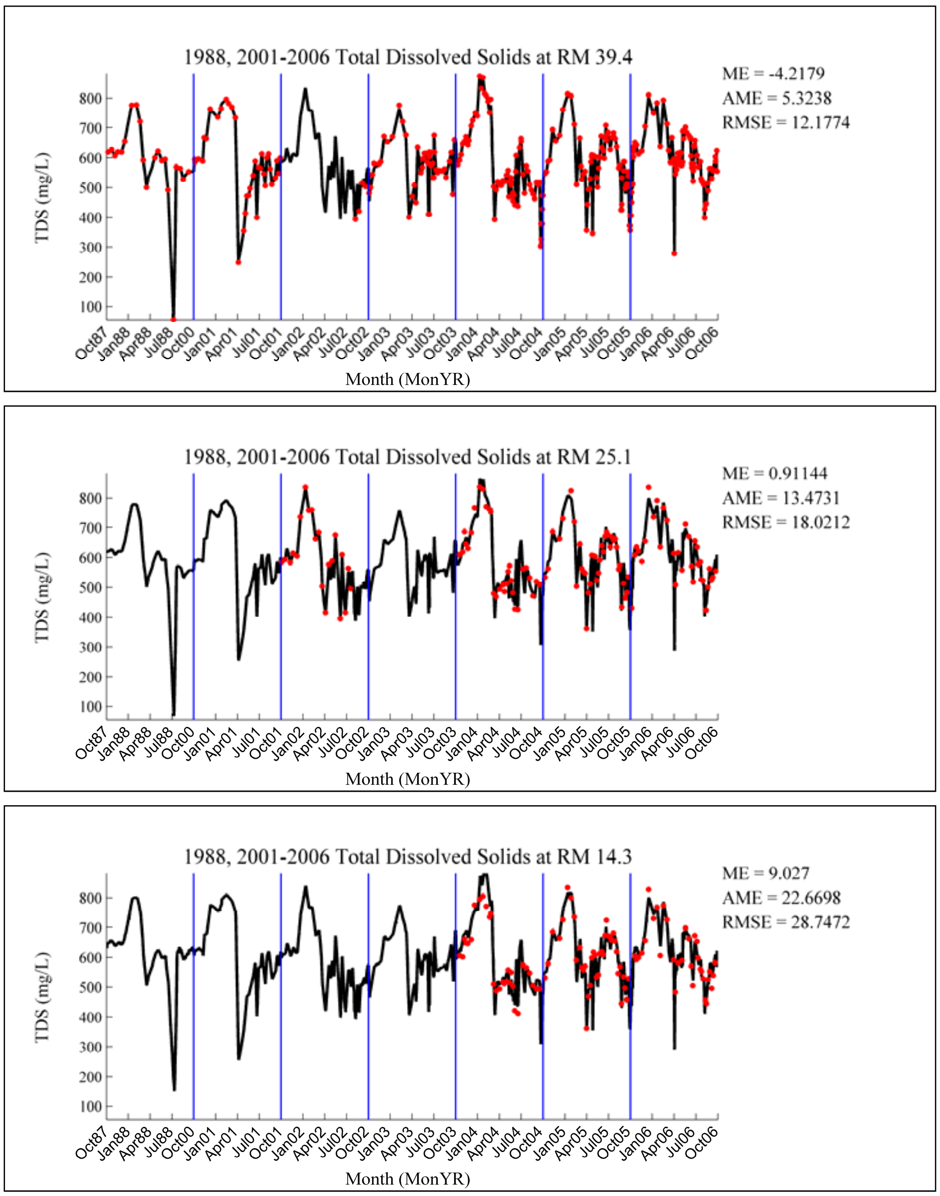


Figure 61. TDS at various calibration stations (continued).

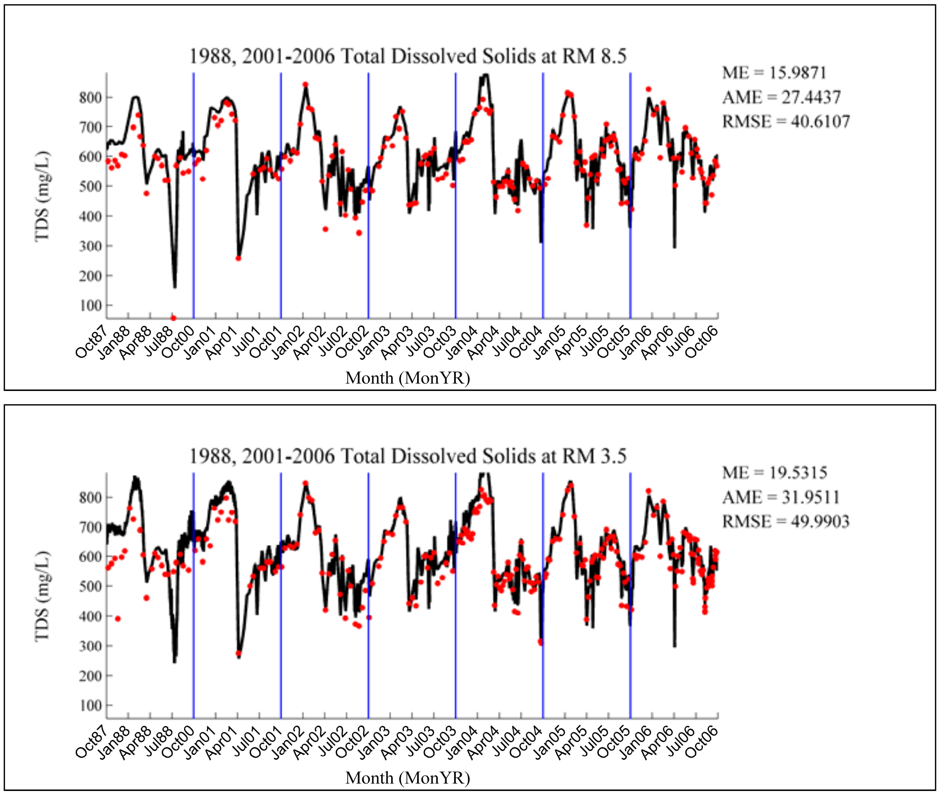


Figure 61. (concluded).

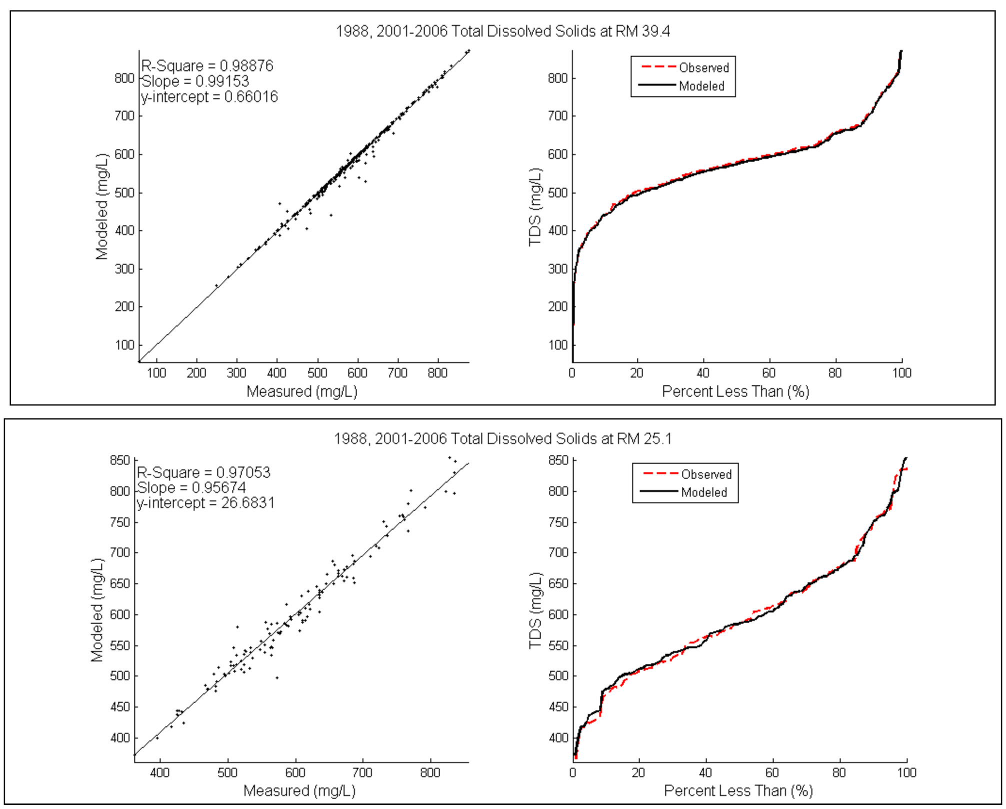


Figure 62. TDS linear and cumulative distribution plots at various calibration stations (continued).

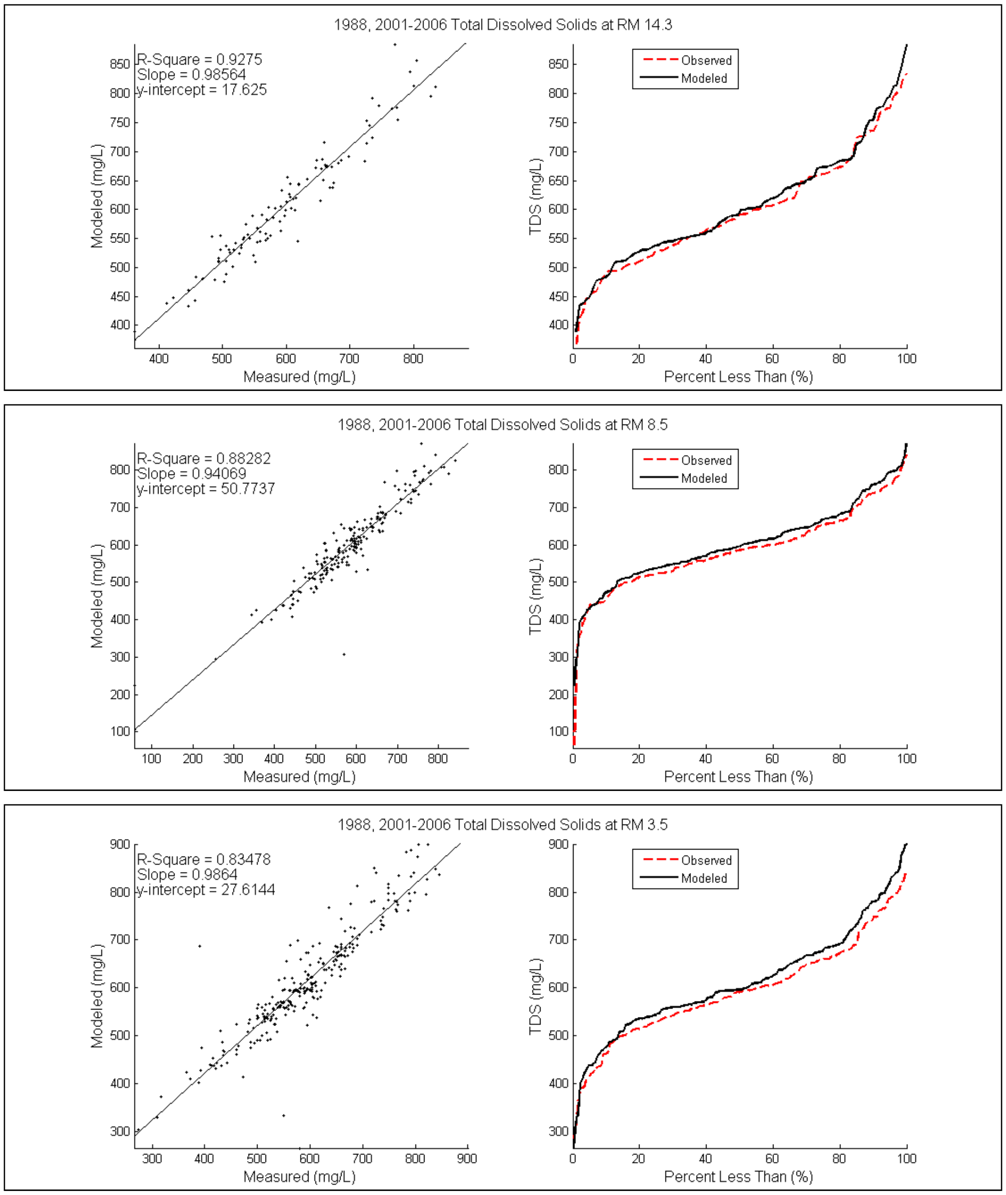


Figure 62. (concluded).

Total phosphorus

TP time series plots and linear and cumulative distribution plots are shown in Figures 63 and 64, respectively. According to these figures, the model tends to do very well with total phosphorus predictions. At Fort Snelling, the AME = 0.10 mg/L, which is below the 10% AME target found in Table 33 (0.11 mg/L). The model tends to slightly under-predict TP observed data throughout the entire reach of the river.

Table 33. 10% target for TP (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	0.01	2.22	0.07	0.22
RM 25.1	0.08	0.87	0.05	0.08
RM 14.3	0.07	0.74	0.05	0.07
RM 8.5	0.06	1.20	0.06	0.11
RM 3.5	0.01	1.10	0.06	0.11

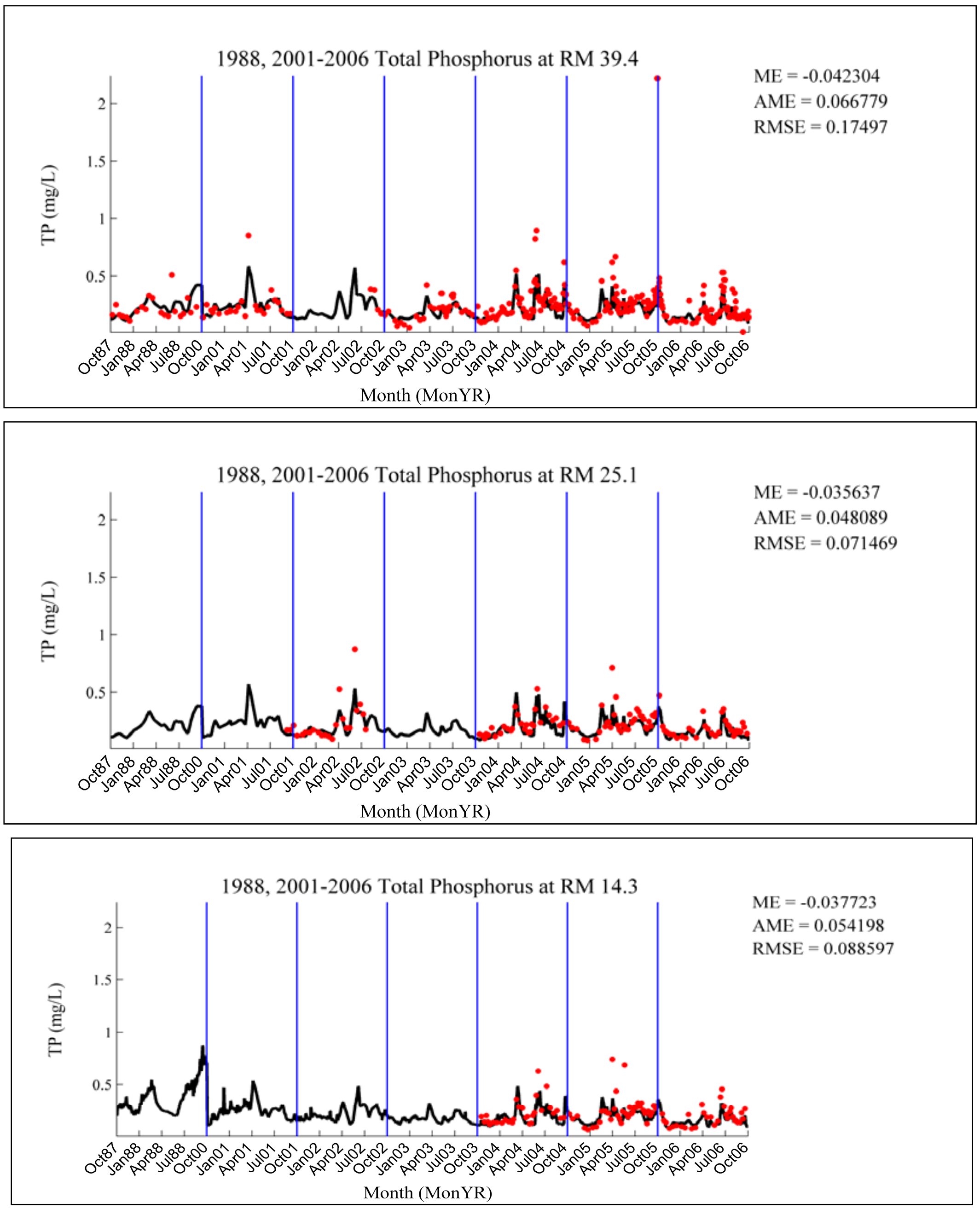


Figure 63. TP at various calibration stations (continued).

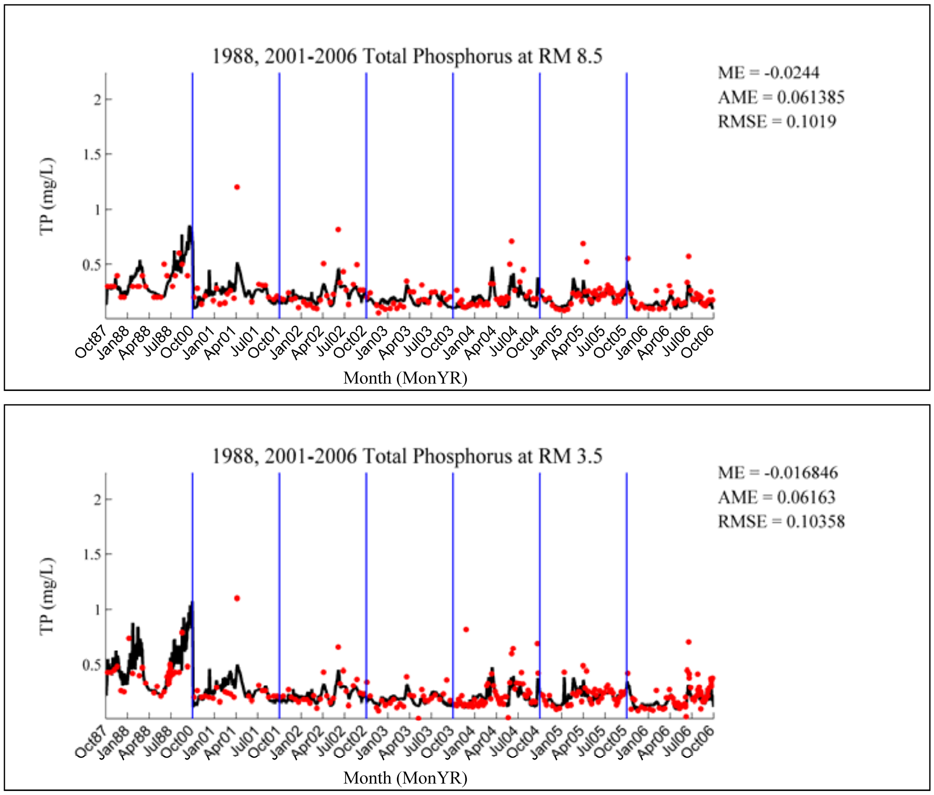


Figure 63. (concluded).

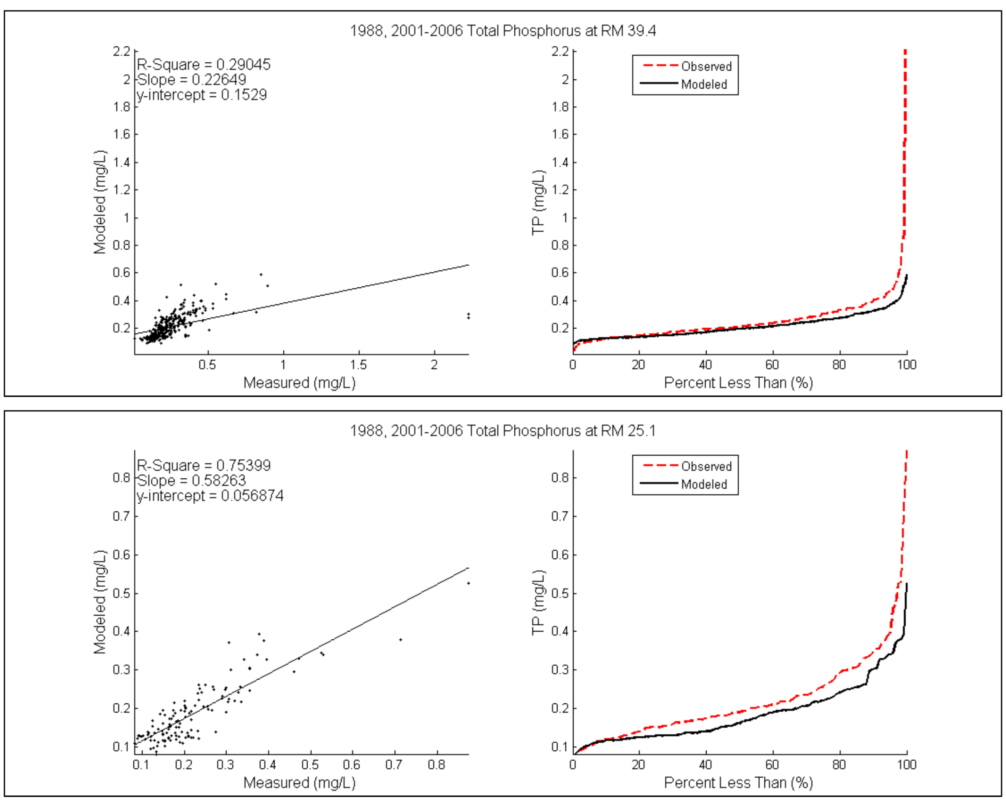


Figure 64. TP linear and cumulative distribution plots at various calibration stations (continued).

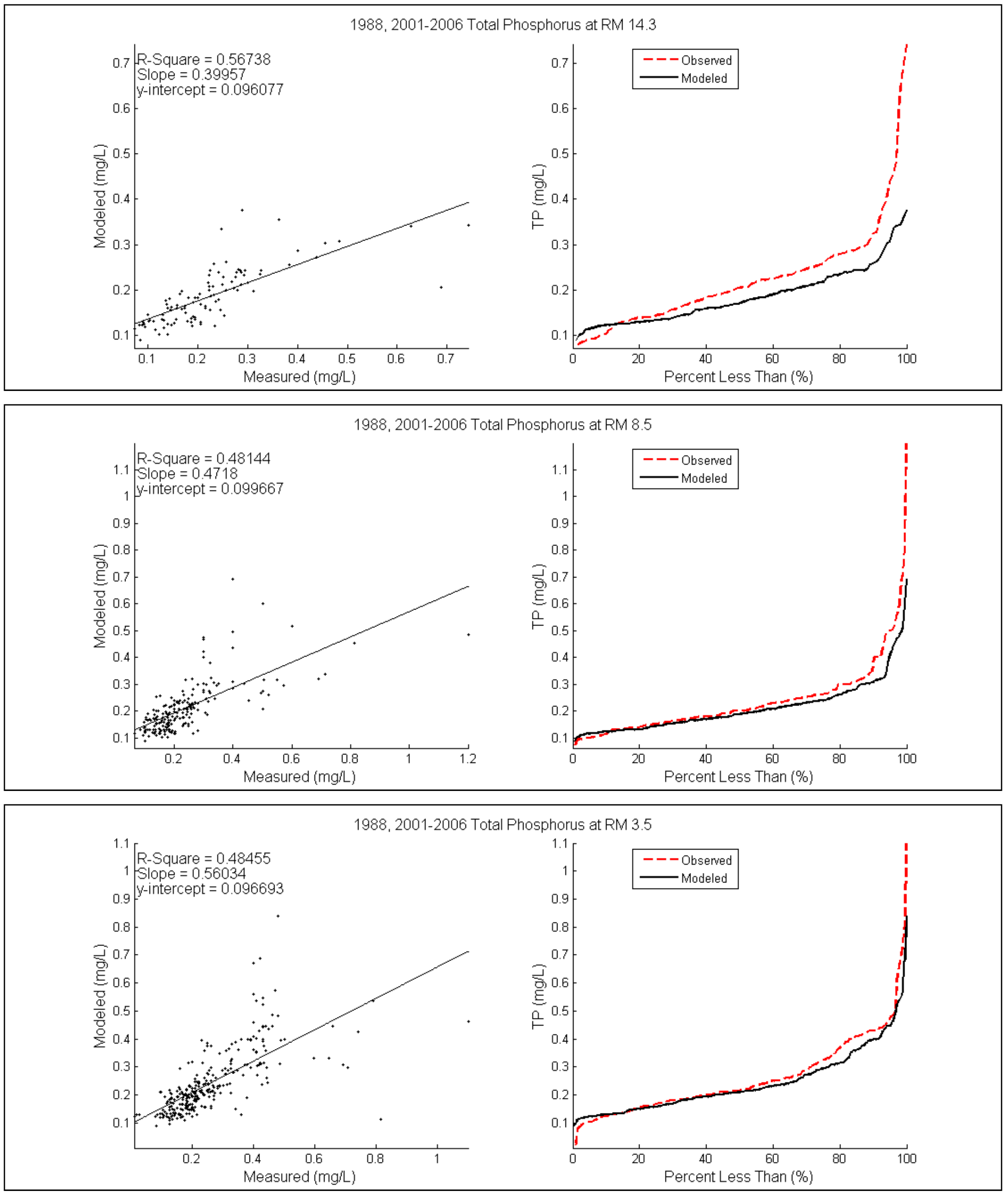


Figure 64. (concluded).

Biochemical oxygen demand

Time series plots and linear and cumulative distribution plots for BOD5 (total BOD5 representing both carbonaceaous and nitrogenous BOD5) are shown in Figures 65 and 66, respectively. The reader should be aware that the ERDC did not use BOD5 as a significant factor during the calibration process. Due to the error involved in measuring BOD5 and due to the fact that ERDC back-calculated BOD5 (see Appendix A -- Back-calculating BOD5 for model verification), using BOD is not the best water quality constituent to define the success of the model. According to these figures, the model tends to do very well with biochemical oxygen demand predictions. At Fort Snelling, the AME = 1.69 mg/L, which is below the 10% AME target found in Table 34 (2.00 mg/L). According to Figure 66, the model tends to overpredict the lower observed values and tends to underpredict the higher observed values for BOD5.

Table 34. 10% target for BOD5 (mg/L) for 1988, 2001-2006.

River Mile	Minimum	Maximum	AME	10% Target
RM 39.4	0.10	19.00	1.29	1.89
RM 25.1	1.00	7.80	1.00	0.68
RM 14.3	1.00	7.30	0.92	0.63
RM 8.5	1.00	15.30	1.11	1.43
RM 3.5	1.00	21.00	1.69	2.00

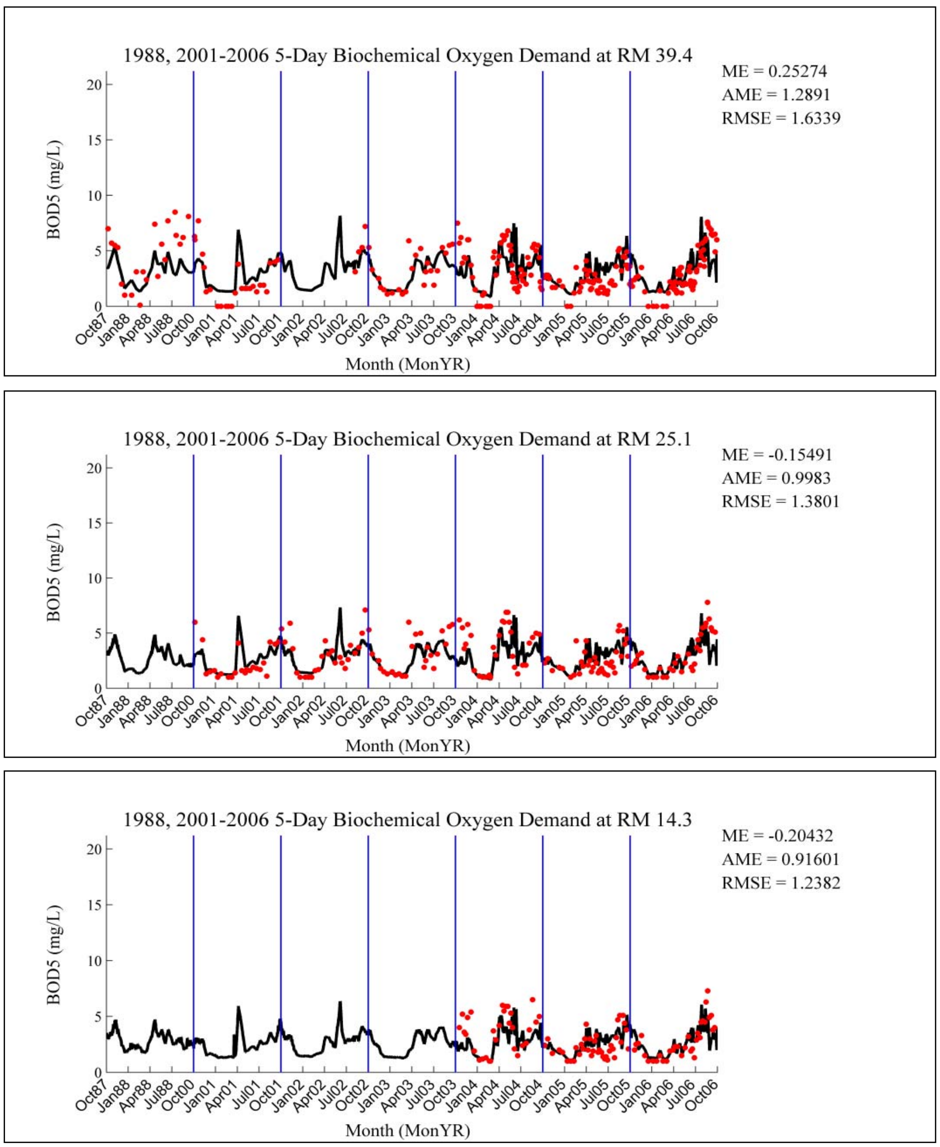


Figure 65. BOD5 at various calibration stations (continued).

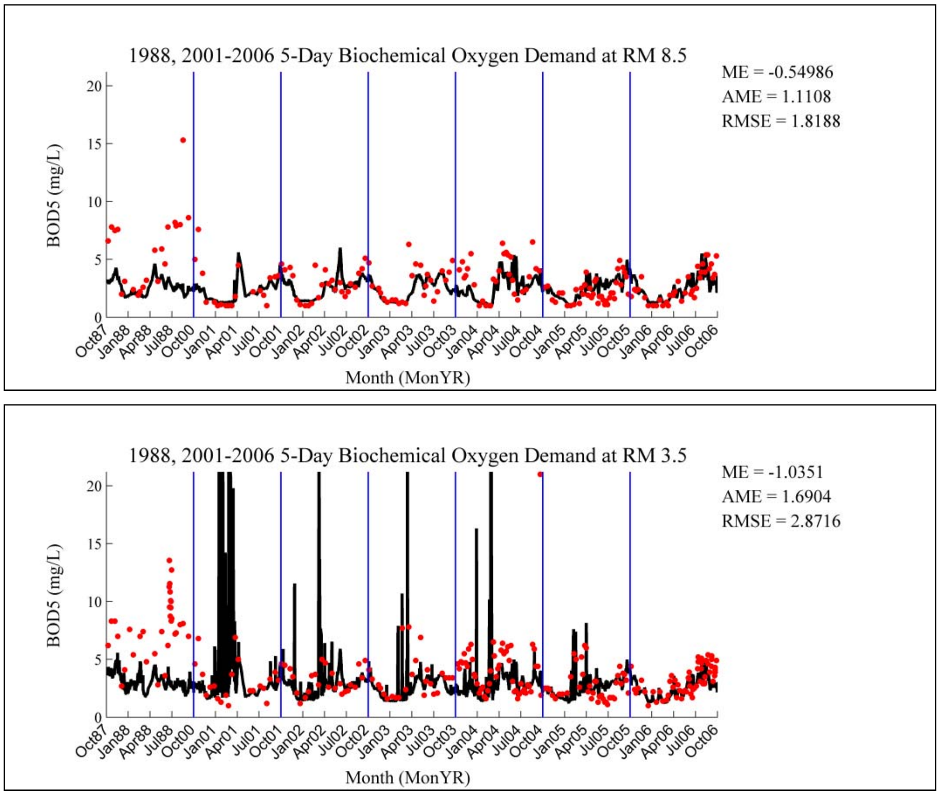


Figure 65. (concluded).

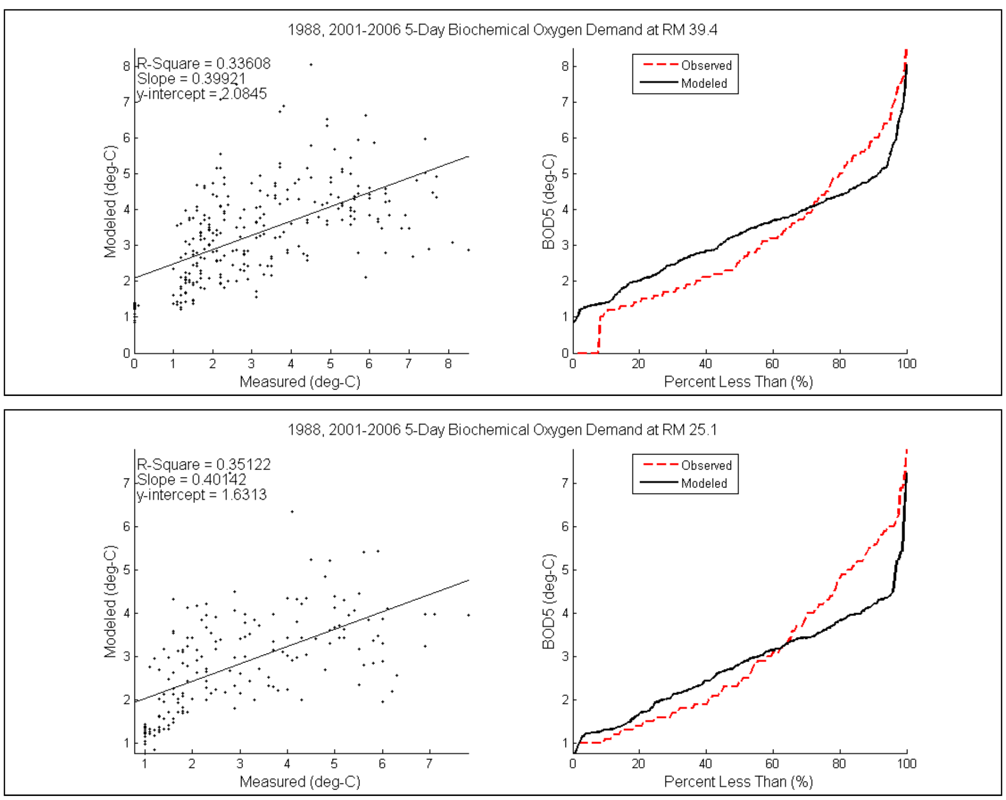


Figure 66. BOD5 linear and cumulative distribution plots at various calibration stations (continued).

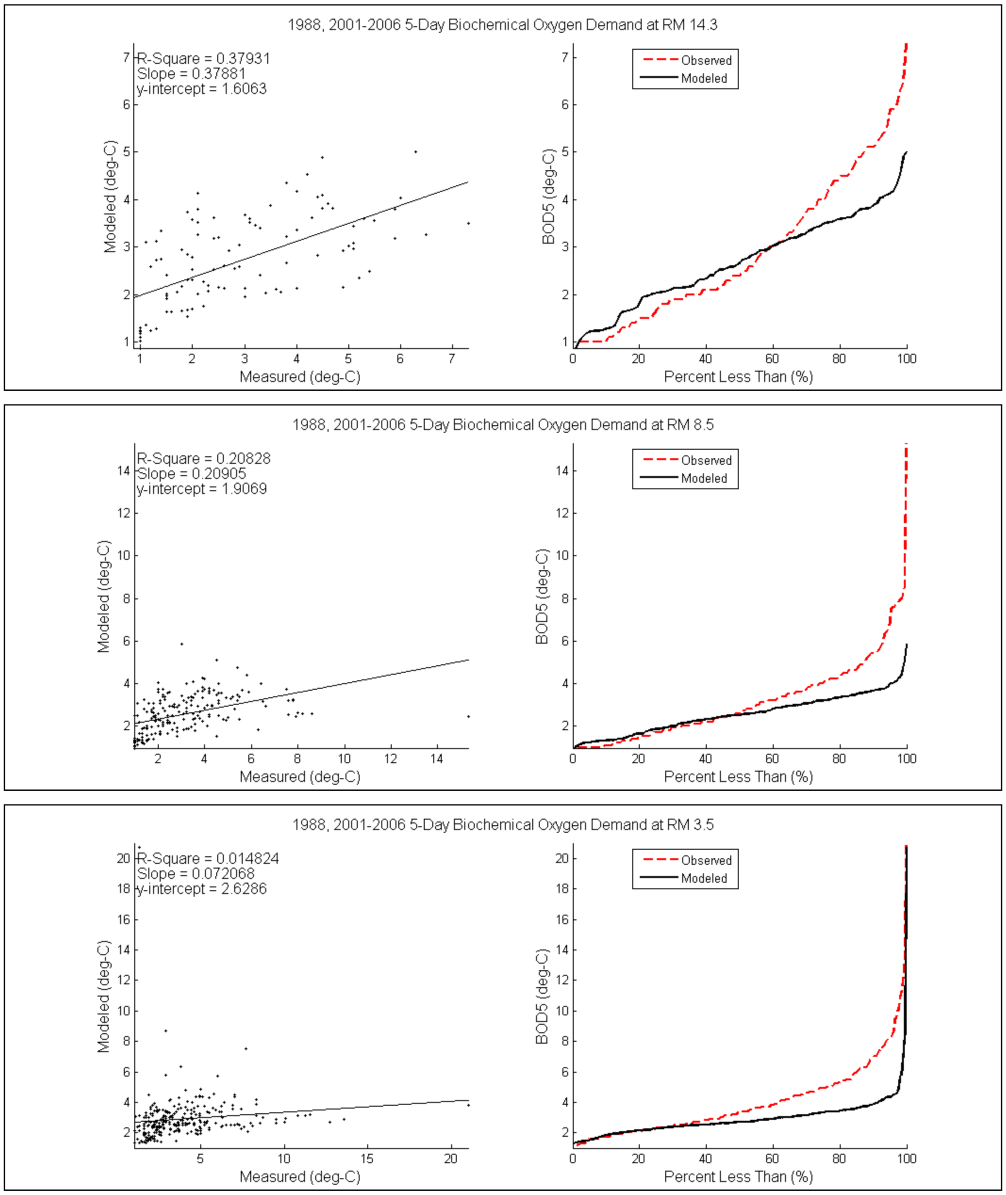


Figure 66. (concluded).

Statistical summary for Fort Snelling (RM 3.5)

In order to provide the reader with an overview of the calibration at Fort Snelling (the main calibration station), Table 35 presents the AME and R-squared values for all of the water quality constituents modeled. These statistics are presented for each individual water year and for the seven-year combined statistics.

Table 35. Overview of summary statistics for RM 3.5.

	1988		2001		2002		2003		2004		2005		2006		2006 (low flow)		7-year (combined)	
Constituent	R^2	AME	R^2	AME	R^2	AME	R^2	AME	R^2	AME	R^2	AME	R^2	AME	R^2	AME	R^2	AME
TEMP	0.97	1.91	0.99	0.91	0.99	0.64	0.99	0.74	0.99	1.01	0.99	1.00	0.99	0.61	0.98	0.87	0.99	1.34
FLOW	0.98	4.45	1.00	14.71	0.97	9.14	0.99	7.93	0.99	9.91	0.99	13.74	0.99	13.76	0.98	6.19	0.99	10.51
ELWS	0.84	0.03	0.99	0.11	0.97	0.06	0.99	0.05	0.99	0.07	0.99	0.15	0.99	0.16	0.94	0.05	0.98	0.09
DO	0.86	1.03	0.35	1.36	0.77	1.35	0.91	0.96	0.81	1.25	0.90	0.78	0.76	0.93	0.08	1.47	0.82	1.09
NH4	0.39	0.47	0.66	0.102	0.50	0.27	0.65	0.045	0.85	0.038	0.71	0.046	0.35	0.33	0.08	0.044	0.76	0.122
CHLA	0.34	20.72	0.59	16.48	0.82	11.15	0.59	14.85	0.50	32.07	0.60	12.93	0.64	22.46	0.30	33.98	0.54	20.33
PO4	0.58	0.093	0.77	0.043	0.85	0.022	0.76	0.023	0.72	0.036	0.85	0.018	0.56	0.022	0.57	0.045	0.82	0.036
ISS	0.12	17.3	0.95	49.0	0.80	43.1	0.42	40.9	0.46	52.1	0.84	31.0	0.78	36.9	0.66	36.3	0.57	37.58
TSS	0.12	19.11	0.96	52.10	0.82	43.57	0.42	41.31	0.48	53.64	0.85	30.07	0.79	36.86	0.63	36.20	0.58	38.31
TKN	0.28	0.92	0.49	0.37	0.71	0.19	0.17	0.22	0.52	0.25	0.56	0.13	0.32	0.26	0.01	0.28	0.55	0.32
TP	0.21	0.08	0.32	0.09	0.81	0.05	0.14	0.06	0.30	0.08	0.60	0.04	0.64	0.05	0.45	0.04	0.48	0.06
BOD	0.00	4.79	0.03	1.88	0.08	1.07	0.39	0.96	0.03	1.98	0.07	0.97	0.38	0.83	0.09	1.07	0.01	1.69
NO3	0.96	0.55	0.94	0.34	0.94	0.58	0.93	0.63	0.93	0.82	0.95	0.65	0.87	0.60	0.99	0.27	0.92	0.62
TDS	0.18	93.7	0.94	43.5	0.96	25.7	0.91	24.0	0.96	33.5	0.94	19.2	0.92	19.8	0.85	20.7	0.83	31.95
DOC			0.04	3.53	0.09	1.49	0.16	1.47	0.00	1.09	0.60	0.47	0.51	0.29	0.18	0.27		
DSI			0.18	4.66	0.46	3.02	0.29	4.44	0.96	1.07	0.94	0.71	0.94	0.97	0.87	1.36		
Diatoms									0.62	1.77	0.87	0.70	0.75	1.40	0.88	2.49		
Bluegreen									0.81	0.19	0.86	0.14	0.78	0.21	0.52	0.49		
Green									0.93	0.32	0.53	0.11	0.61	0.07	0.23	0.08		

Load comparisons – FLUX vs CE-QUAL-W2

Constituent loads at RM 39.4 and RM 3.5 were compiled from model results and compared to estimated loads reported for other projects using the FLUX program (Walker 1996). The load comparison between independent projects serves as an additional test of the calibration. See Figures 67-74 for comparisons of FLUX to W2 loads in the various years.

Estimated loads (in metric tons) for water years 2004-2006 provide the best data for comparison due to increased monitoring, identical estimation periods, and more constituents. In general, fewer samples were collected in 1988 than in 2001-2003, and fewer samples were collected in 2001-2003 than in 2004-2006. ERDC compiled FLUX loads at RM 39.4 and RM 3.5 for the three-year period as part of a budgetary analysis of the Lower Minnesota River (James 2007). Load estimates for TP, PO4, TKN, NO3, TSS, and NH4 were available. Both FLUX and model results for the period 2004-2006 were annual loads based on the water year. The coefficient of variation is included on plots for the FLUX-estimated loads.

FLUX-estimated loads for the years 2001-2003 were taken from two sources: *State of the Minnesota River* (MRBDC 2007) and *Regional Progress in Water Quality* (St. Paul Metropolitan Council 2004). The first report provided TP, PO4, NO3, and TSS loads for RM 39.4 and RM 3.5, and the second report provided TKN loads for RM 39.4 only. No NH4 loads were available. The MRBDC compiled loads for ice-out or April 1 through September 30. Ice-out generally occurs in mid-March. FLUX-estimated TKN loads were compiled for January 1 through December 31. Some FLUX loads could not be compiled for 2001 due to reduced sampling under flood conditions. All model-estimated loads for 2001-2003 were compiled for the period April 1 through September 30.

FLUX-estimated loads for the year 1988 were compiled for two reports by the St. Paul Metropolitan Council: *Regional Progress in Water Quality* (2004) and a report on loading sources to Lake Pepin by Meyer and Schellhaass (2002). TKN and NO3 loads were taken from the first source, and TP, PO4, TSS, and VSS loads were taken from the second source. No NH4 loads were available. FLUX results were annual loads based on the calendar year (January-December). Model results were also annual loads; however, they were based on the water year (October-September).

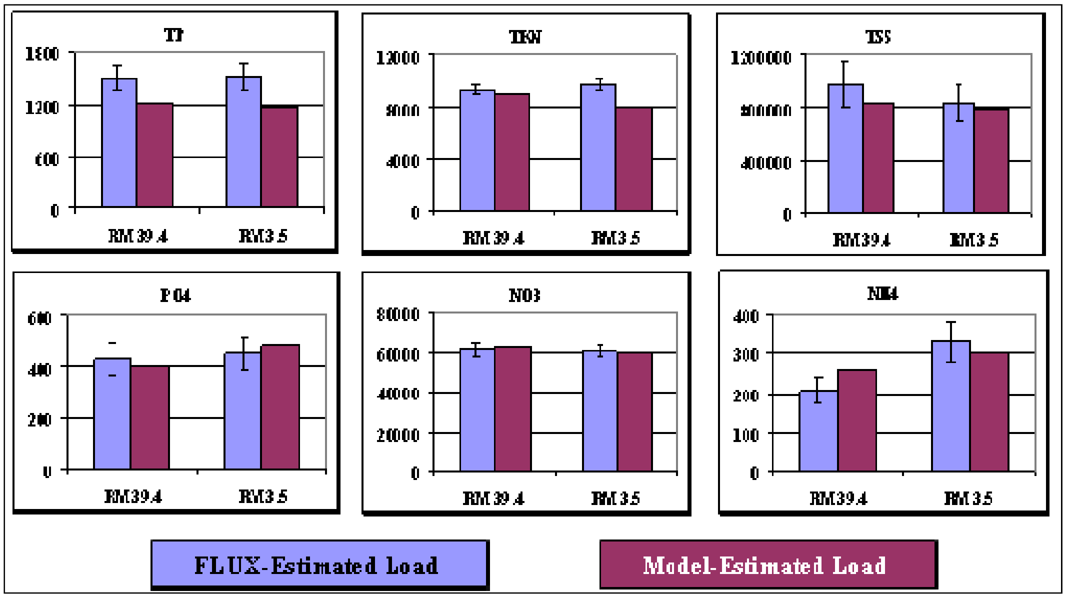


Figure 67. Comparison of annual loads (metric tons), FLUX and model, WY 2006.

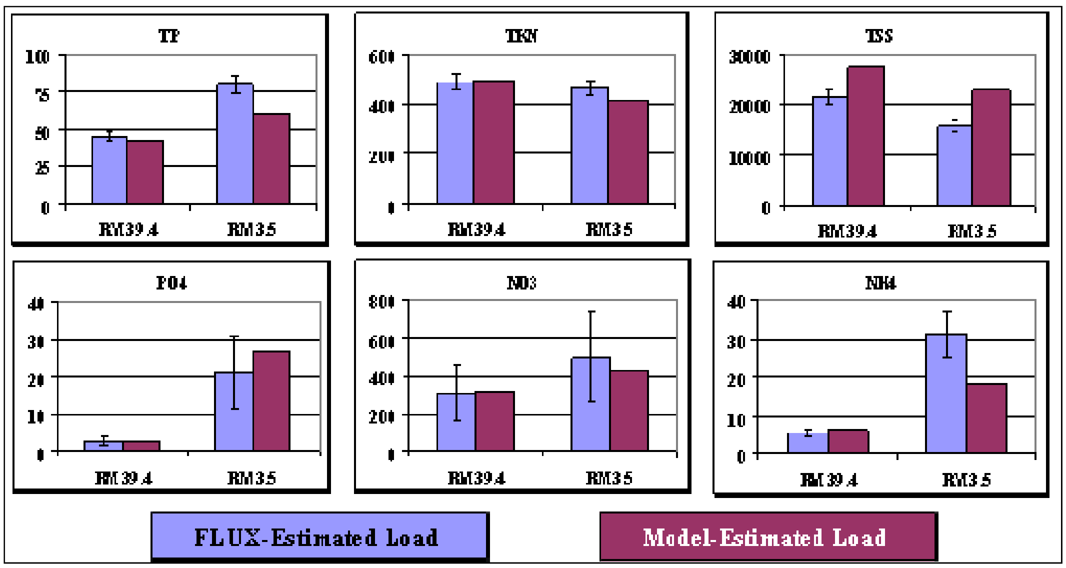


Figure 68. Comparison of loads (metric tons), FLUX and model, July 15-September 30, 2006.

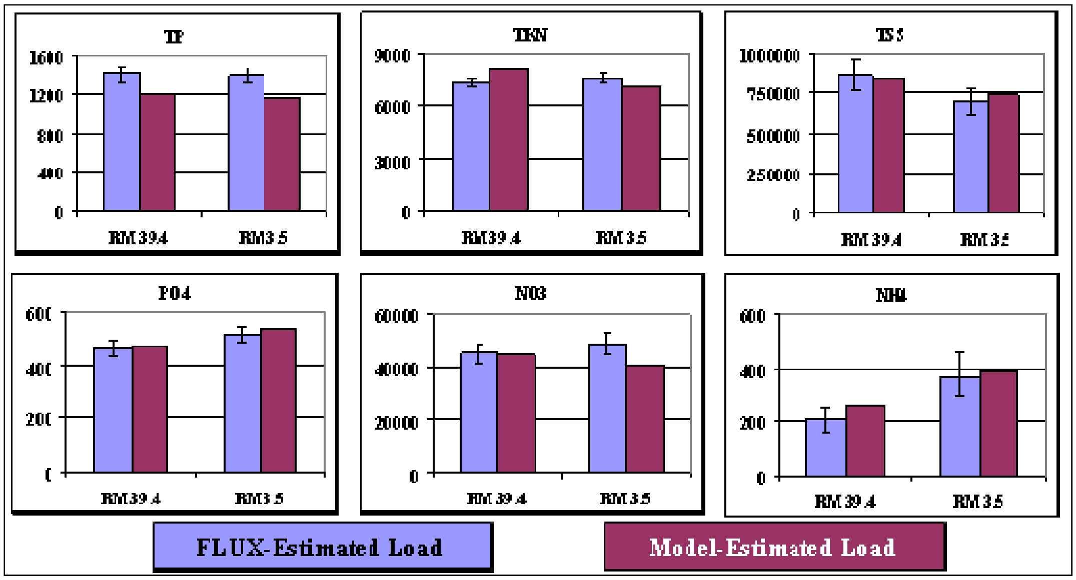


Figure 69. Comparison of annual loads (metric tons), FLUX and model, WY 2005.

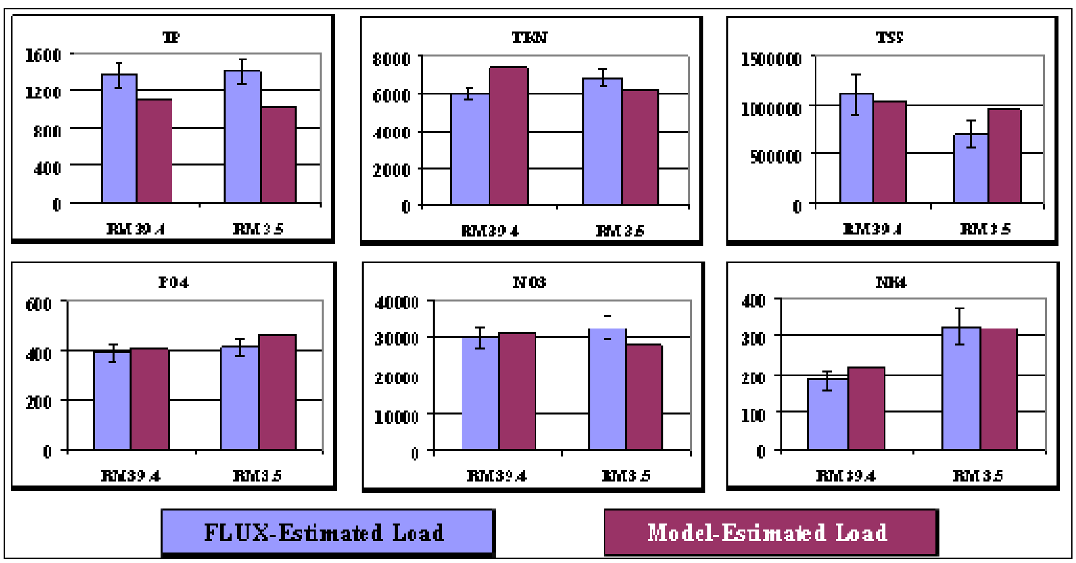


Figure 70. Comparison of annual loads (metric tons), FLUX and model, WY 2004.

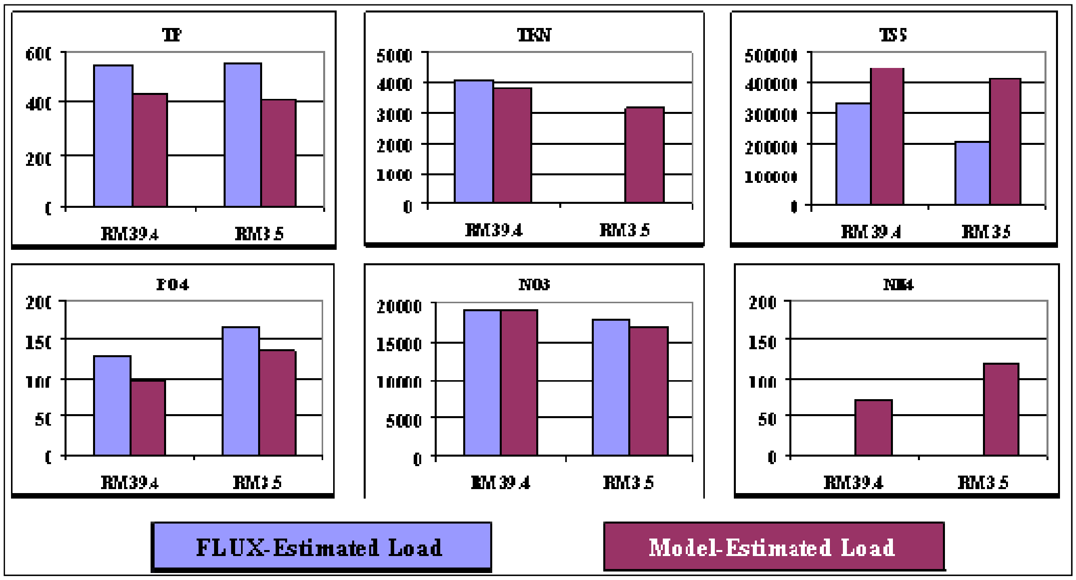


Figure 71. Comparison of loads (metric tons), FLUX and model, April-September, 2003.

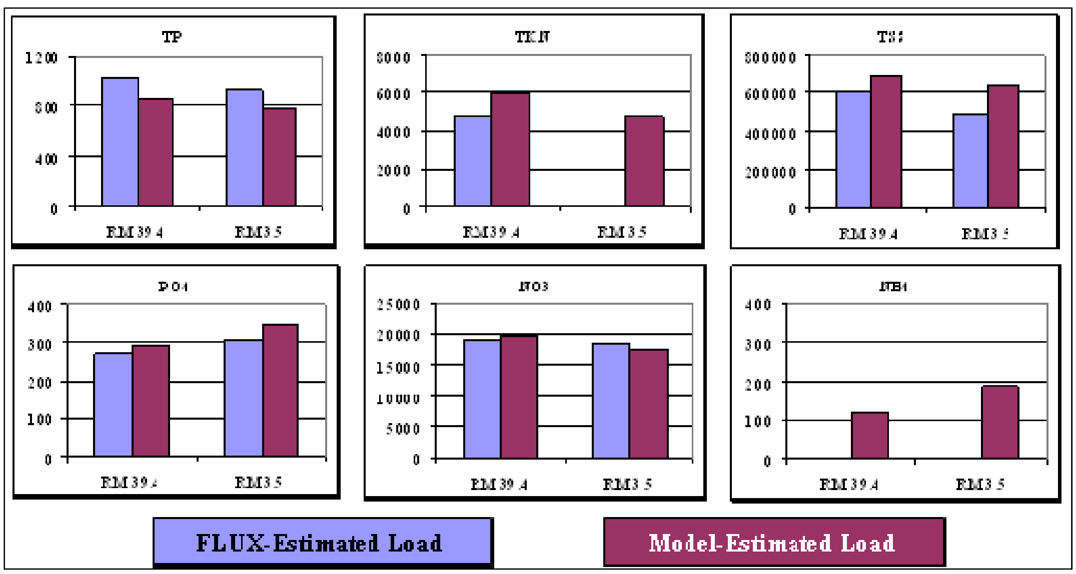


Figure 72. Comparison of loads (metric tons), FLUX and model, April-September, 2002.

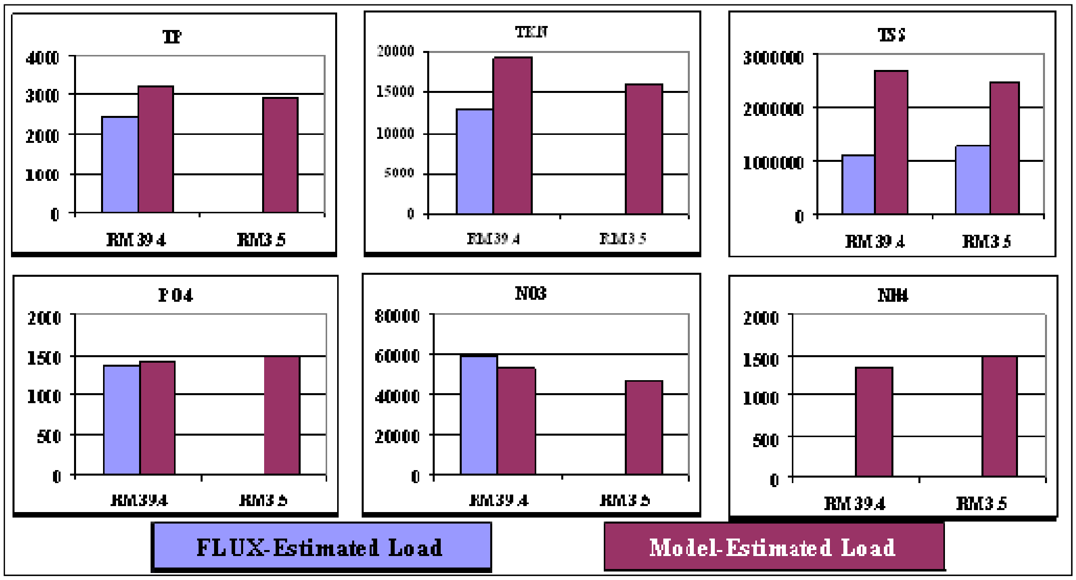


Figure 73. Comparison of loads (metric tons), FLUX and model, April-September, 2001.

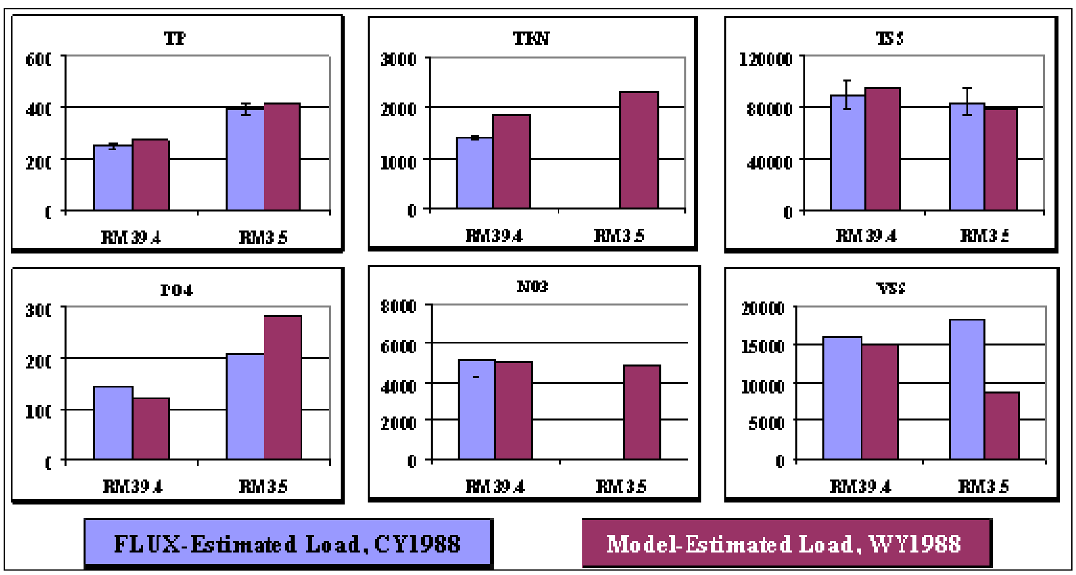


Figure 74. Comparison of loads (metric tons), FLUX and model, 1988.

For the extensively monitored years of 2004-2006, the model-estimated loads at RM 39.4 generally compare well with the FLUX-estimated loads. The exceptions are TP in all three years (W2 < FLUX), TKN in 2004 (W2 > FLUX), and TSS during the summer low-flow period in 2006 (W2 > FLUX). This served as a check on the model inputs for the upstream boundary. On an annual basis at RM 3.5, PO4 and NH4 loads compared well in all three years, and TKN and TSS compared well in two of three years. Model-estimated loads for NO3 were lower in two years, and TP loads were lower in all three years, likely carrying over from smaller loads at RM 39.4. During the summer low-flow period in 2006, TP and NH4 loads were lower in the model, while TSS loads were higher.

The load comparison was helpful when determining how to improve the October 2008 calibration. At the time, the load comparisons for dissolved constituents (PO4, NO3, and NH4) were good, but the load comparisons for constituents that included particulates (TSS, TP, TKN) were poor, with model-estimated loads much lower than FLUX-estimated loads. The solution was defining nonliving organic matter based on VSS and DOC. This increased the particulate matter and also helped lower DO concentrations to better match measured data.

5 Sensitivity and Component Analyses

Sensitivity and component analyses were conducted at different stages of model development. After the initial calibration (presented in October 2008), a narrow sensitivity analysis was performed on the model of WY 1988. It was limited in scope to target potential areas for improvement. Select model settings were changed one at a time, and the results were compared against a baseline and each other. The base run for WY 1988 in the initial calibration will be presented first, so the reader has a baseline to compare against the sensitivity results.

After the final calibration was presented (September 2009), a component analysis was performed on the models of WY 1988 and WY 2006. This analysis attempted to reveal the major sources and sinks of dissolved oxygen during low-flow periods in the summer. Sources, such as effluent CBOD loads, and sinks, such as algal respiration, were removed from the model one at time, and the results were compared. A similar analysis of DO components was presented in the waste load allocation study (MPCA 1985).

Finally, the model was tested for its sensitivity to the Black Dog GP. The final version of the model was run with and without the Black Dog GP withdrawal and discharges in WY 1988, WY 2003, and WY 2006. Good flow and temperature data were available for these years, but only limited water-quality data were available for 2005 and 2006. For periods without water-quality data, reflected inputs from an upstream segment were used. The sensitivity analysis tested whether model results were better with or without the reflected inputs.

1988 LMRM sensitivity analysis base run

Since the sensitivity analysis was performed before the final version of the model was approved, the results from the 1988 Sensitivity Analysis Base Run must be presented. Figures 75-77 and Table 36 present time series results and statistical results for RM 3.5 for NH4, CHLA, and DO. The statistics in Table 36 were calculated in Excel using the ANOVA statistical analysis. More information in interpreting the values in this table can be found at: http://cameron.econ.ucdavis.edu/excel/ex53bivariateregressionstatisticalinference.html.

Table 37 lists the rates that were tested in the sensitivity analysis along with the applied values and results. In each test, only a single rate was altered; all other inputs were unchanged. For example, the SOD rate was varied in eight different ways in eight model runs: decreased by 25%, 20%, 15%, 10%, and

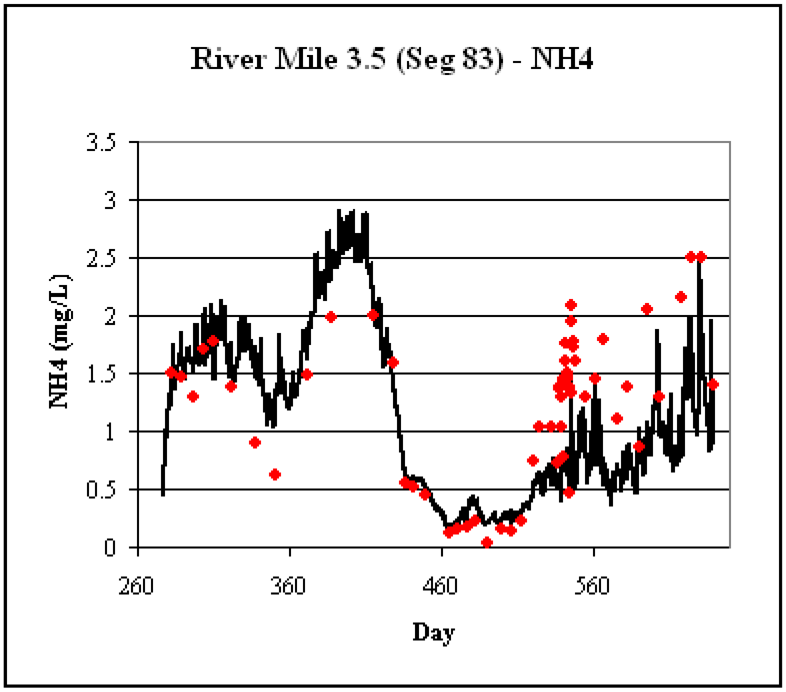


Figure 75. WY 1988 sensitivity analysis base run – ammonium.

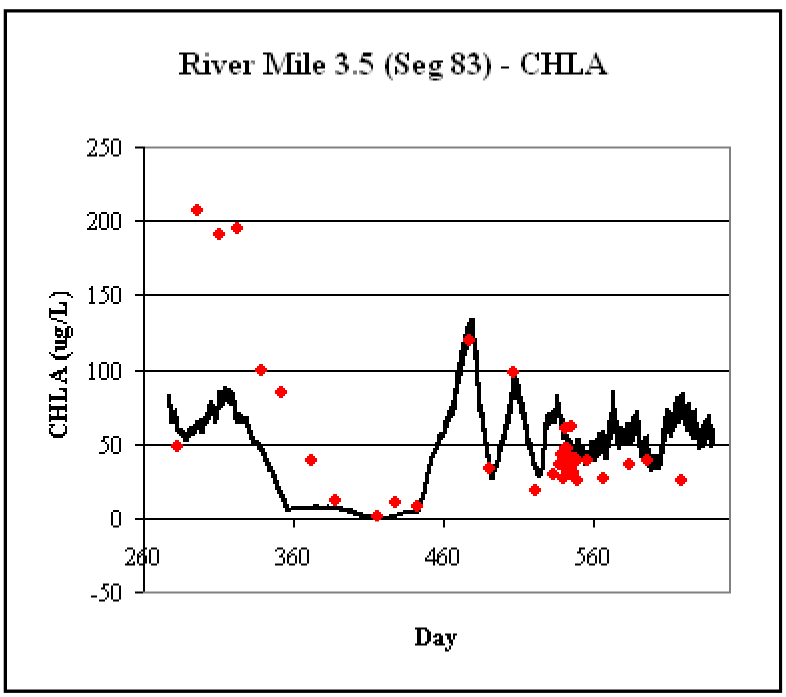


Figure 76. WY 1988 sensitivity analysis base run – chlorophyll a.

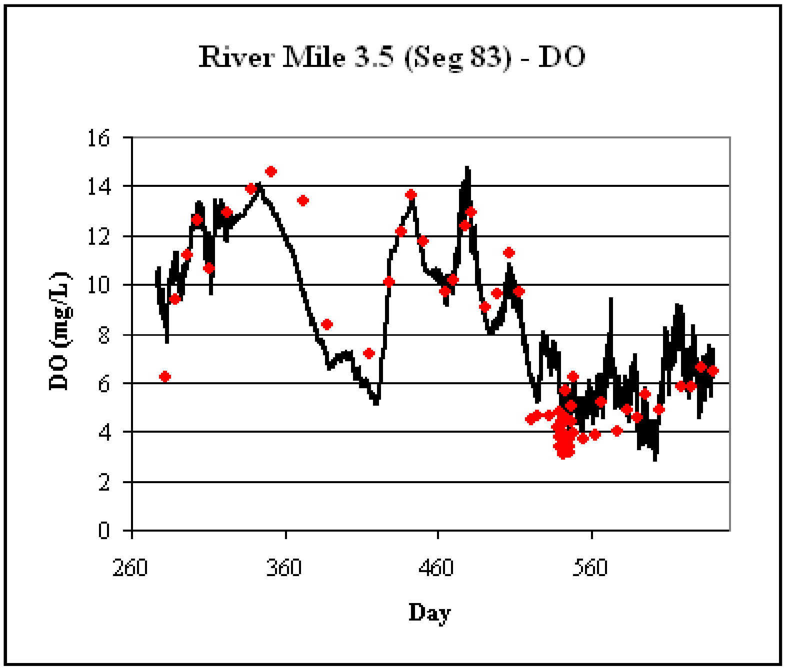


Figure 77. WY 1988 sensitivity analysis base run – dissolved oxygen.

Table 36. WY 1988 sensitivity analysis base run – statistics.

Constituent	β0-slope β1-y-intercept	Values	Standard Error	*t stat*	*P-value*	Lower 95%	Upper 95%	R^2	*N*	ME	AME	RMSE
NH_4	β0	0.28	0.12	2.32	0.02	0.04	0.53	0.35	60	-0.354	0.497	0.633
	β1	0.49	0.09	5.54	0.00	0.31	0.66					
DO	β0	2.16	0.30	7.12	0.00	1.55	2.77	0.87	60	0.42	1.13	1.41
	β1	0.75	0.04	19.37	0.00	0.67	0.83					
CHLA	β0	37.11	4.68	7.93	0.00	27.66	46.55	0.22	43	-2.44	24.92	40.86
	β1	0.23	0.07	3.39	0.00	0.09	0.37					

Table 37. Sensitivity analysis results.

Rate	Constituent	Values	% Change	DO		NH4		CHLA	
				R^2	AME	R^2	AME	R^2	AME
Zero-order Sediment Oxygen Demand Rates, g O_2 /(m^2 day)	SOD	0.165-3.0	-25%	0.87	1.01	0.66	0.32	0.44	32.84
		0.176-3.2	-20%	0.88	1.02	0.66	0.32	0.44	32.84
		0.187-3.4	-15%	0.88	1.03	0.66	0.32	0.44	32.84
		0.198-3.6	-10%	0.88	1.04	0.66	0.32	0.44	32.83
		0.209-3.8	-5%	0.88	1.06	0.66	0.32	0.44	32.83
		0.22-4.00	Base	0.88	1.09	0.67	0.32	0.44	32.85
		0.231-4.02	+5%	0.88	1.11	0.66	0.32	0.44	32.83
		0.242-4.4	+10%	0.88	1.12	0.67	0.32	0.44	32.84
		0.264-4.8	+20%	0.88	1.16	0.67	0.32	0.44	32.83

Rate	Constituent	Values	% Change	DO		NH4		CHLA	
				R^2	AME	R^2	AME	R^2	AME
Ammonium Decay Rate, 1/day	NH4DK	0.096	-20%	0.88	1.03	0.71	0.30	0.44	32.84
		0.102	-15%	0.88	1.04	0.69	0.30	0.44	32.82
		0.108	-10%	0.88	1.05	0.69	0.31	0.44	32.83
		0.114	-5%	0.88	1.07	0.67	0.31	0.44	32.83
		0.12	Base	0.88	1.09	0.67	0.32	0.44	32.85
		0.126	+5%	0.88	1.09	0.65	0.32	0.44	32.85
		0.132	+10%	0.88	1.11	0.64	0.33	0.44	32.84
		0.138	+15%	0.88	1.13	0.63	0.34	0.44	32.82
Particulate Organic Matter Settling Rate, m/day	POMS	0.60	-25%	0.88	1.14	0.67	0.32	0.45	32.92
		0.64	-20%	0.88	1.13	0.67	0.32	0.45	32.92
		0.72	-10%	0.88	1.10	0.67	0.32	0.44	32.89
		0.76	-5%	0.88	1.09	0.66	0.32	0.44	32.85
		0.80	Base	0.88	1.09	0.67	0.32	0.44	32.85
		0.84	+5%	0.88	1.07	0.66	0.32	0.44	32.81
		0.88	+10%	0.88	1.07	0.66	0.32	0.44	32.79
		0.96	+20%	0.88	1.05	0.66	0.32	0.43	32.74
		1.0	+25%	0.88	1.04	0.66	0.32	0.43	32.72
		1.2	+50%	0.88	1.03	0.66	0.32	0.43	32.65
Nitrate Decay Rate, 1/day	NO3DK	0.027	-10%	0.88	1.05	0.69	0.31	0.44	32.83
		0.03	Base	0.88	1.09	0.67	0.32	0.44	32.85
		0.033	+10%	0.88	1.11	0.64	0.33	0.44	32.84
Maximum Algal Growth Rate, 1/day	AG	1.615, 1.615, 1.955	-15%	0.87	1.29	0.68	0.31	0.45	33.53
		1.71, 1.71, 2.07	-10%	0.88	1.23	0.68	0.31	0.45	33.31
		1.805, 1.805, 2.185	-5%	0.88	1.16	0.67	0.32	0.45	33.09
		1.9, 1.9, 2.3	Base	0.88	1.09	0.67	0.32	0.44	32.85
		1.995, 1.995, 2.415	+5%	0.88	1.02	0.65	0.32	0.43	32.53
		2.09, 2.09, 2.53	+10%	0.88	0.98	0.65	0.32	0.42	32.47
		2.185, 2.185, 2.645	+15%	0.88	0.96	0.63	0.33	0.41	32.54

Rate	Constituent	Values	% Change	DO		NH4		CHLA	
				R^2	AME	R^2	AME	R^2	AME
Maximum Algal Respiration Rate, 1/day	AR	0.119, 0.17, 0.119	-15%	0.88	1.01	0.65	0.32	0.44	32.23
		0.126, 0.18, 0.126	-10%	0.88	1.03	0.65	0.32	0.44	32.45
		0.133, 0.19, 0.133	-5%	0.88	1.06	0.66	0.32	0.44	32.64
		0.14, 0.20, 0.14	Base	0.88	1.09	0.67	0.32	0.44	32.85
		0.147, 0.21, 0.147	+5%	0.88	1.12	0.67	0.32	0.44	33.03
		0.154, 0.22, 0.154	+10%	0.88	1.13	0.66	0.32	0.44	33.18
		0.161, 0.23, 0.161	+15%	0.88	1.17	0.67	0.32	0.43	33.35
Suspended Solids Settling Rate, m/day	SSS	0.12	-20%	0.88	1.09	0.66	0.32	0.44	32.84
		0.1275	-15%	0.88	1.10	0.67	0.32	0.44	32.84
		0.135	-10%	0.88	1.09	0.66	0.32	0.44	32.83
		0.1425	-5%	0.88	1.08	0.66	0.32	0.44	32.82
		0.15	Base	0.88	1.09	0.67	0.32	0.44	32.85
		0.1575	+5%	0.88	1.08	0.66	0.32	0.44	32.81
		0.165	+10%	0.88	1.08	0.66	0.32	0.44	32.84
		0.1725	+15%	0.88	1.07	0.66	0.32	0.44	32.84
		0.18	+20%	0.88	1.06	0.66	0.32	0.44	32.83

5%; increased by 5%, 10% and 20%. Table 37 contains a range of values for SOD because they apply to six reaches (Figure 27). The table presents the resulting R-squared and AME for DO, NH4, and CHLA at RM 3.5 for each change made. These three water quality parameters were selected because they were the focus in the 1985 waste load allocation study (MPCA 1985).

1988 dissolved oxygen component analysis

The final version of the 1988 LMRM was used to quantify the dissolved oxygen deficit contributions in July-September from the following components: wastewater CBOD loads, sediment oxygen demand, organic matter loads and OM contribution from algae, instream nitrification, algal respiration, and algal growth. For each of these model runs, loads and/or rates associated with each of these components were set to zero. For example, for the model run labeled ‘no OM,’ all OM input loads and all OM rates were set to 0.0. The predicted DO values from each model are shown in Figure 78. The solid black line represents the results from the final calibration for the 1988 LMRM. Similar to results presented in Lung (1996), algal respiration and algal growth represent the largest DO deficit components, and they more or less balance each other.

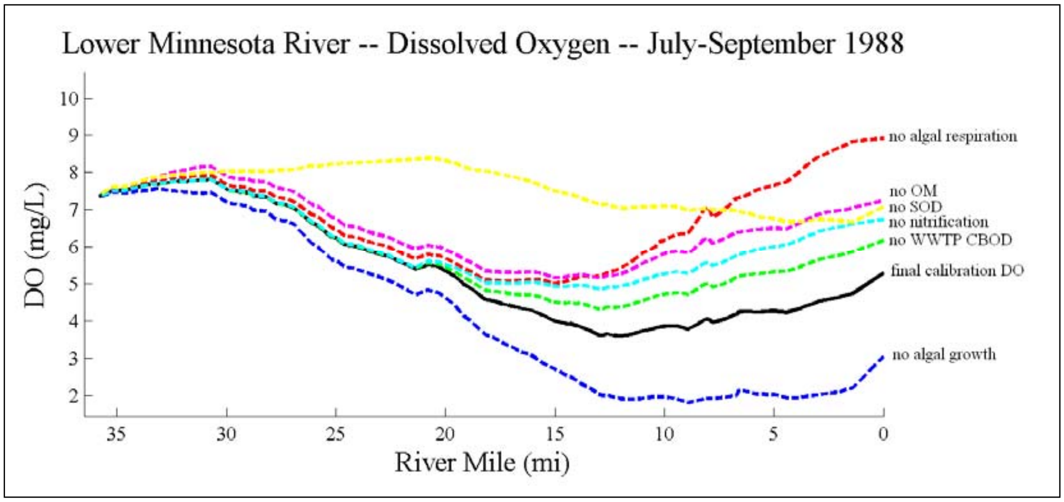


Figure 78. WY 1988 dissolved oxygen component analysis.

2006 dissolved oxygen component analysis

The final version of the LMRM for WY 2006 was also used to quantify the dissolved oxygen deficit contributions in July-September from the following components: wastewater CBOD loads, airport CBOD loads, sediment oxygen demand, organic matter loads and OM contribution from algae, instream nitrification, algal respiration, and algal growth. For each of these model runs, loads and/or rates associated with each of these components were set to zero. The predicted DO values from each model are shown in Figure 79. Note that for 2006, nitrification, wastewater CBOD loads, and airport CBOD loads had minimal to no impact on the DO deficit, which is why these three lines are plotted virtually on top of each other. Algal growth and algal respiration seemed to still have the largest overall impact on the deficit.

Impacts of Black Dog for 1988, 2003, and 2006

ERDC ran sensitivity tests for the Black Dog Generating Plant (GP) for water years 1988, 2003, and 2006. For each water year, the generating plant was completely removed from the model; that is, flows, temperature, or water quality were not used in the model for either outfall location or the

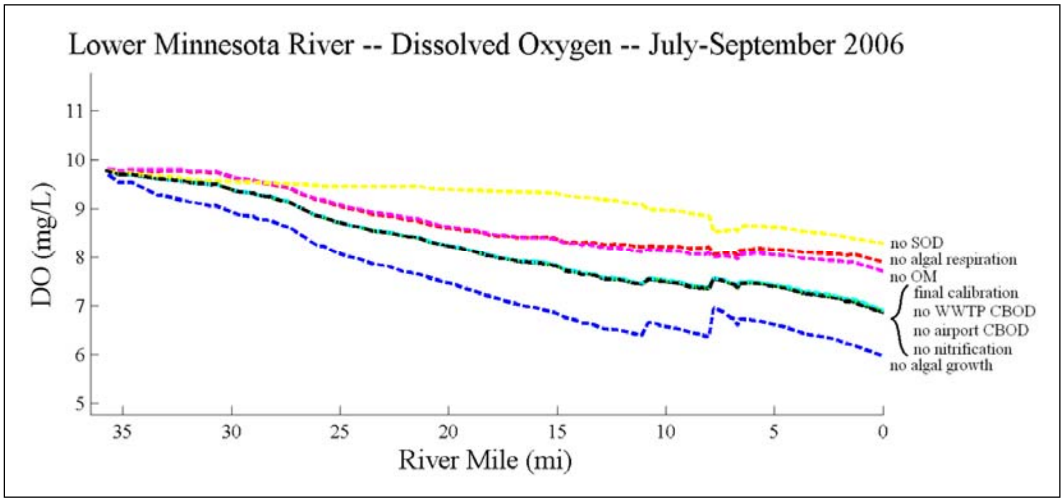


Figure 79. WY 2006 dissolved oxygen component analysis.

withdrawal. Figures 80-85 and Tables 38-43 present the results from model runs with and without the Black Dog Generating Plant. Recall that reflected input files from an upstream river segment were used to define water quality in the Black Dog outfalls in all years except late summer 2005 and 2006 when samples were collected. Measured flow and temperature were available for all years and were applied in the model. Model results for runs with and without the Black Dog plant are similar, but it was determined to include the outfalls and intake for completeness and future applications when more data are available.

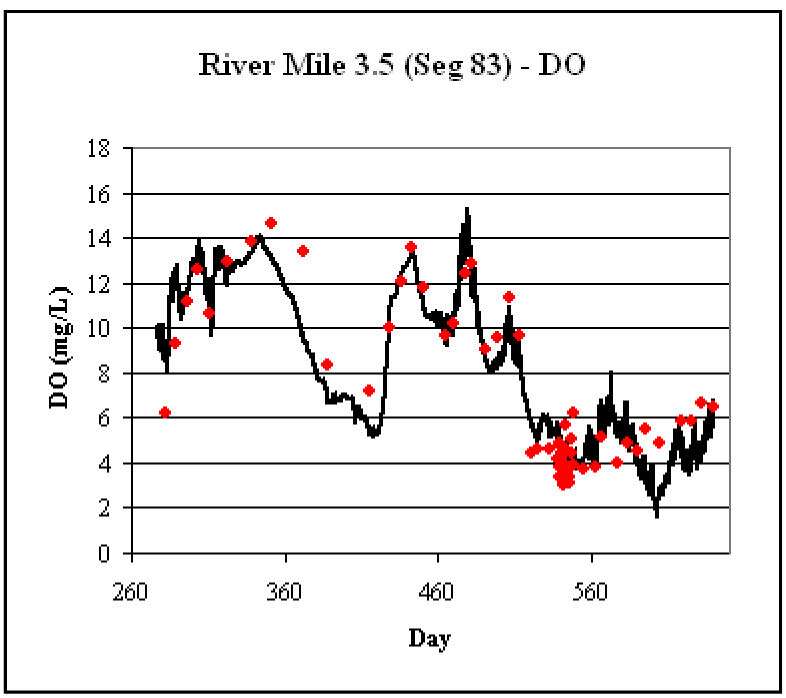


Figure 80. WY 1988 – no Black Dog GP.

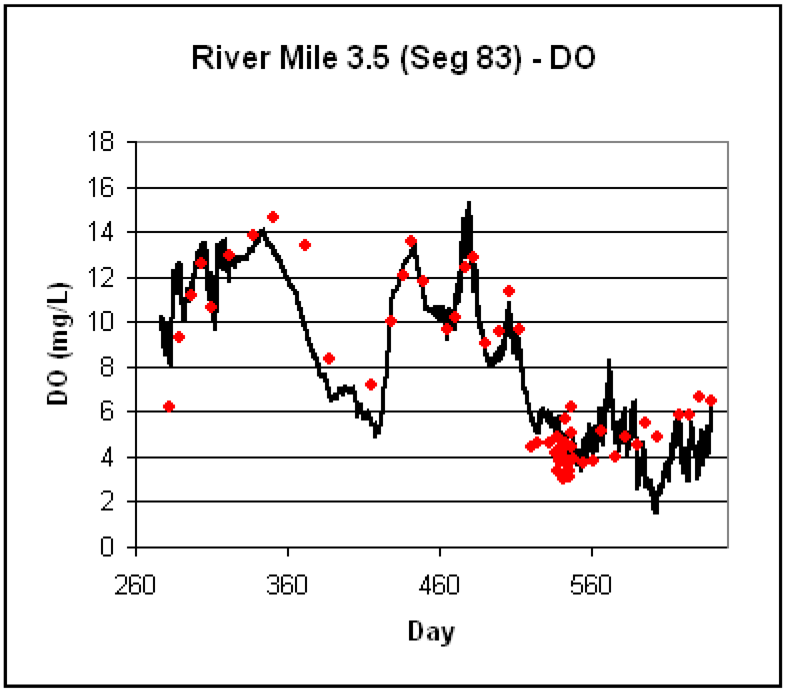


Figure 81. WY 1988 – Final calibration – with Black Dog.

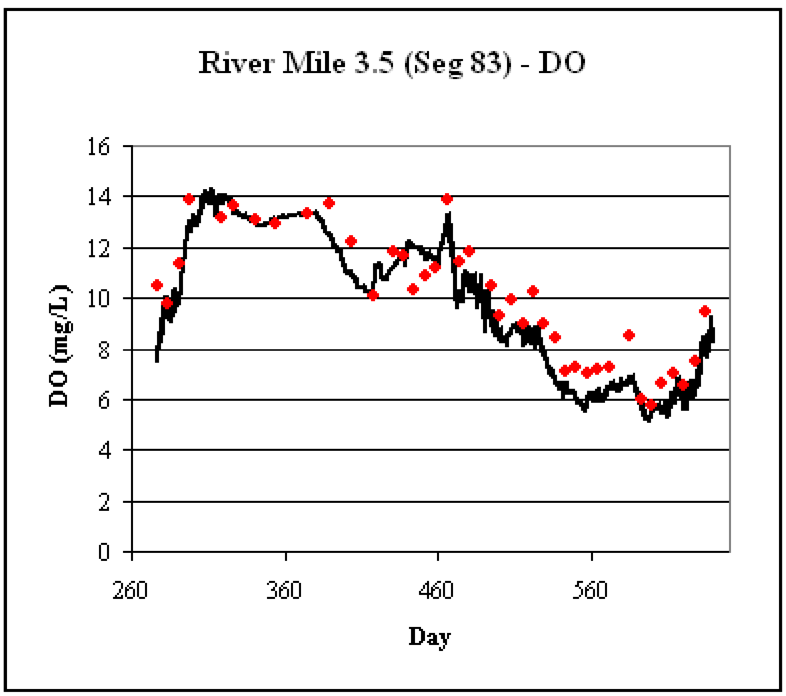


Figure 82. WY 2003 – no Black Dog GP.

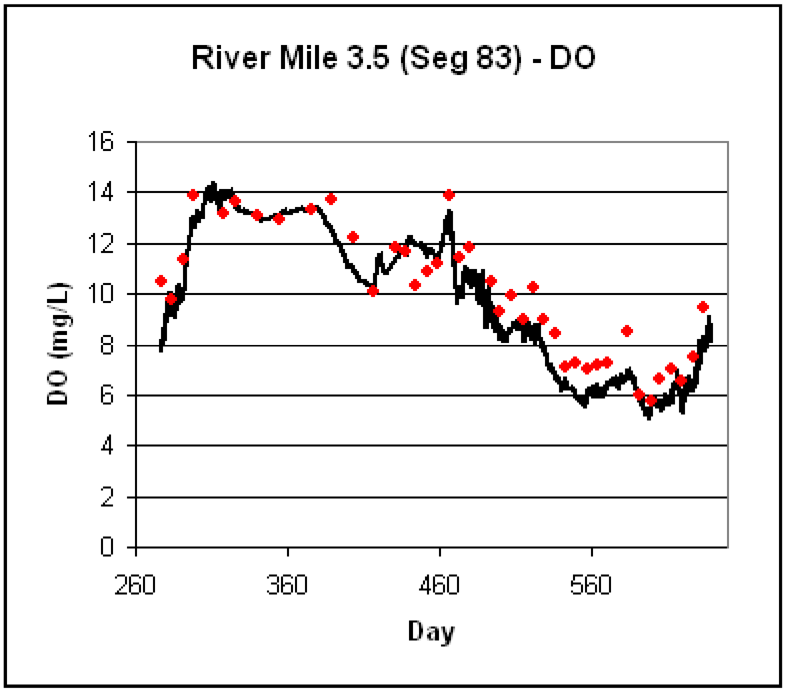


Figure 83. WY 2003 – Final calibration – with Black Dog.

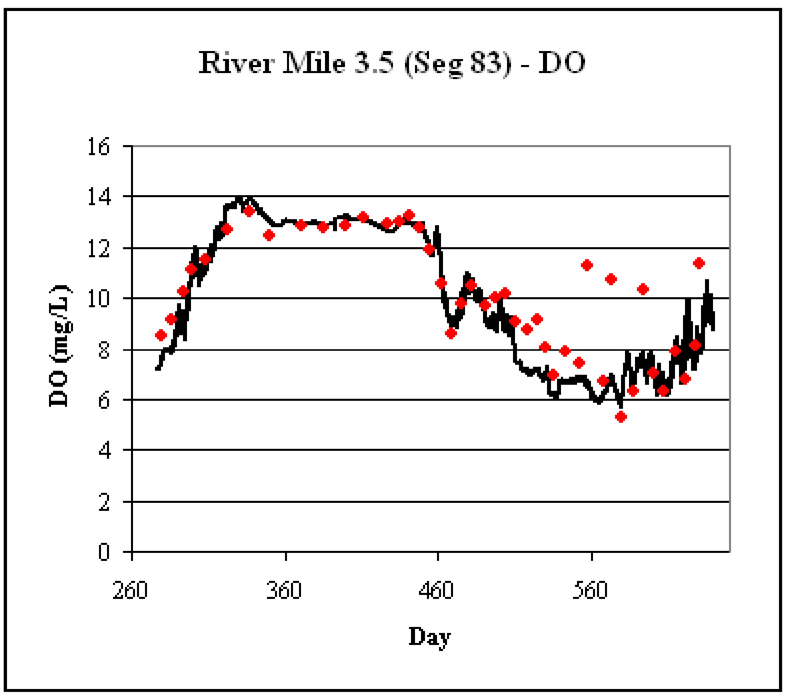


Figure 84. WY 2006 – no Black Dog GP.

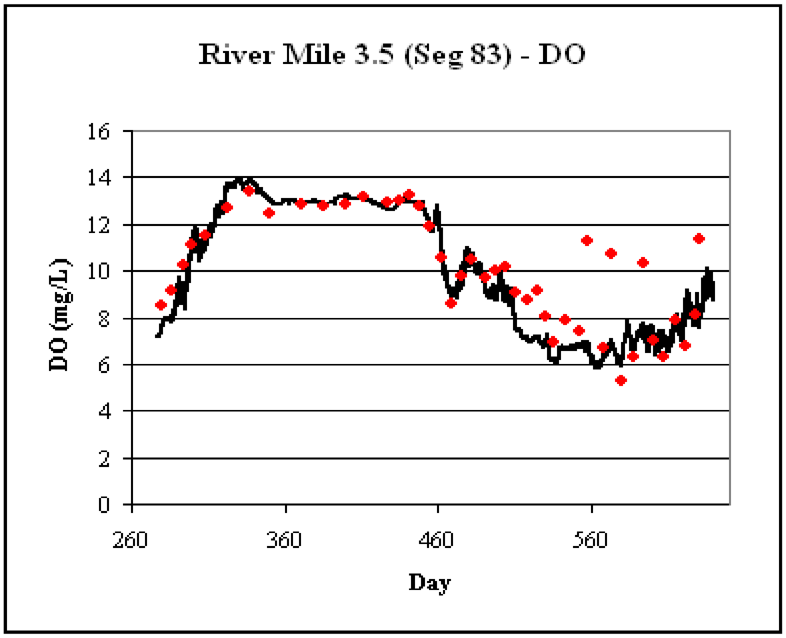


Figure 85. WY 2006 – Final calibration – with Black Dog.

Table 38. WY 1988 statistics – no Black Dog GP.

Constituent		Coefficients	Standard Error	*t stat*	*P-value*	Lower 95%	Upper 95%	R^2	*N*	ME	AME	RMSE
NH_4	β0	0.25	0.12	2.19	0.03	0.02	0.48	0.42	60	-0.322	0.459	0.583
	β1	0.54	0.08	6.51	0.00	0.37	0.70					
DO	β0	0.80	0.35	2.30	0.02	0.10	1.50	0.87	60	-0.12	1.00	1.28
	β1	0.87	0.04	19.46	0.00	0.78	0.96					
Temp	β0	0.86	0.12	6.98	0.00	0.61	1.10	0.98	346	-0.84	1.86	2.24
	β1	0.86	0.01	116.06	0.00	0.85	0.88					
CHLA	β0	28.64	4.11	6.96	0.00	20.33	36.94	0.33	43	-8.89	20.89	39.06
	β1	0.27	0.06	4.51	0.00	0.15	0.39					

Table 39. WY 1988 statistics – final calibration with Black Dog.

Constituent		Coefficients	Standard Error	*t stat*	*P-value*	Lower 95%	Upper 95%	R^2	*N*	ME	AME	RMSE
NH_4	β0	0.26	0.12	2.16	0.04	0.02	0.50	0.39	60	-0.331	0.473	0.604
	β1	0.53	0.09	6.06	0.00	0.35	0.70					
DO	β0	0.90	0.35	2.56	0.01	0.20	1.60	0.86	60	-0.10	1.03	1.29
	β1	0.86	0.04	19.13	0.00	0.77	0.95					
Temp	β0	1.11	0.13	8.63	0.00	0.86	1.36	0.97	346	-0.70	1.91	2.31
	β1	0.85	0.01	109.57	0.00	0.84	0.87					
CHLA	β0	29.11	4.08	7.13	0.00	20.87	37.36	0.34	43	-8.10	20.72	38.59
	β1	0.27	0.06	4.65	0.00	0.16	0.39					

Table 40. WY 2003 statistics – no Black Dog GP.

Constituent		Coefficients	Standard Error	*t stat*	*P-value*	Lower 95%	Upper 95%	R^2	*N*	ME	AME	RMSE
NH_4	β0	0.02	0.01	1.33	0.19	-0.01	0.05	0.66	40	-0.010	0.045	0.073
	β1	0.73	0.09	8.50	0.00	0.55	0.90					
DO	β0	-1.39	0.55	-2.52	0.02	-2.51	-0.27	0.91	40	-0.78	0.96	1.13
	β1	1.06	0.05	19.79	0.00	0.95	1.17					
Temp	β0	0.23	0.23	1.01	0.32	-0.23	0.69	0.99	40	-0.33	0.75	0.93
	β1	0.96	0.01	68.27	0.00	0.93	0.99					
CHLA	β0	6.64	6.88	0.97	0.34	-7.59	20.87	0.59	25	-4.42	14.95	20.11
	β1	0.75	0.13	5.77	0.00	0.48	1.02					

Table 41. WY 2003 statistics – final calibration with Black Dog.

Constituent		Coefficients	Standard Error	*t stat*	*P-value*	Lower 95%	Upper 95%	R^2	*N*	ME	AME	RMSE
NH_4	β0	0.02	0.01	1.34	0.19	-0.01	0.05	0.65	40	-0.010	0.045	0.073
	β1	0.73	0.09	8.47	0.00	0.55	0.90					
DO	β0	-1.47	0.55	-2.68	0.01	-2.57	-0.36	0.91	40	-0.79	0.96	1.14
	β1	1.07	0.05	20.11	0.00	0.96	1.17					
Temp	β0	0.65	0.24	2.68	0.01	0.16	1.15	0.99	40	0.16	0.74	0.92
	β1	0.96	0.02	63.90	0.00	0.93	0.99					
CHLA	β0	6.78	6.86	0.99	0.33	-7.41	20.96	0.59	25	-4.22	14.85	19.99
	β1	0.75	0.13	5.80	0.00	0.48	1.02					

Table 42. WY 2006 statistics – no Black Dog GP.

Constituent		Coefficients	Standard Error	*t stat*	*P-value*	Lower 95%	Upper 95%	R^2	*N*	ME	AME	RMSE
NH_4	β0	0.04	0.00	10.87	0.00	0.03	0.05	0.05	66	-0.017	0.039	0.062
	β1	0.08	0.04	1.82	0.07	-0.01	0.17					
DO	β0	-0.52	0.85	-0.61	0.55	-2.23	1.20	0.78	43	-0.62	0.91	1.37
	β1	0.99	0.08	11.91	0.00	0.82	1.16					
Temp	β0	-0.55	0.12	-4.41	0.00	-0.80	-0.30	0.99	81	-0.08	0.63	0.86
	β1	1.05	0.01	112.47	0.00	1.03	1.07					
CHLA	β0	15.03	4.47	3.36	0.00	6.10	23.95	0.68	66	-13.49	21.05	2945
	β1	0.61	0.05	11.65	0.00	0.50	0.71					

Table 43. WY 2006 statistics – final calibration – with Black Dog.

Constituent		Coefficients	Standard Error	*t stat*	*P-value*	Lower 95%	Upper 95%	R^2	*N*	ME	AME	RMSE
NH_4	β0	0.04	0.00	11.20	0.00	0.03	0.05	0.35	66	-0.009	0.033	0.051
	β1	0.24	0.04	5.87	0.00	0.16	0.32					
DO	β0	-0.31	0.87	-0.36	0.72	-2.07	1.45	0.76	43	-0.60	0.93	1.39
	β1	0.97	0.08	11.43	0.00	0.80	1.14					
Temp	β0	-0.31	0.12	-2.54	0.01	-0.55	-0.07	0.99	81	0.19	0.61	0.87
	β1	1.05	0.01	116.04	0.00	1.03	1.07					
CHLA	β0	15.97	4.47	3.57	0.00	7.03	24.90	0.64	66	-15.94	22.46	31.96
	β1	0.56	0.05	10.74	0.00	0.46	0.66					

6 Model Application

Once the model was fully tested and met performance targets, a number of loading scenarios were applied to demonstate the model's capabilities. The scenarios were based on current permit limitations and completed load allocation studies. In one scenario, results from the Minnesota River Basin Model were translated and input to the LMRM. The objectives were to show that the model produces reasonable results even when loads are greatly increased or decreased, the model can be linked to other models, and the model is suitable for application in future load allocation studies and facility or watershed planning.

Application targets and loading sources

Sources of information that were well defined and generally accepted were used for the model application. This was achieved by staying within the bounds of current standards and permits and completed load allocation studies. The constituents of interest, in order of priority, are dissolved oxygen, ammonia, nutrients, and sediment. Among these constituents, state standards currently exist for dissolved oxygen, ammonia, and turbidity in rivers.

A DO standard of 5 mg/L as a daily minimum concentration must be met in the Minnesota River upstream of RM 21.0 at least 50% of the days at which the river flow is equal to the lowest weekly flow with a once in 10-year recurrence interval ($7Q_{10}$). Between RM 21.0 and the mouth, the DO standard is 5.0 mg/L as a daily average concentration. The standard for un-ionized ammonia nitrogen is 0.04 mg/L as a 30-day average concentration. The percent un-ionized is calculated for the specific temperature and pH. The ammonia standard must be met at least 50% of the days at which the river flow is equal to the lowest 30-day flow with a once in 10-year recurrence interval ($30Q_{10}$). The turbidity standard is 25 NTU at all flows. The state's triennial water-quality rule revision, 2008-2011, will include new eutrophication standards for rivers, a modified turbidity standard, and a new nitrate standard based on aquatic life toxicity.

For the base model in the scenarios, one or more of the seven calibrated water years can be chosen: 1988 and 2001-2006. River flow is an important factor in the water quality of the lower Minnesota River so it played a large role in the choice of year. Figure 86 compares flows at RM 39.4 during the modeled years to percentile flows over the historic record. Annual average flows in the modeled years range from the lowest tenth percentile in 1988 to the highest tenth percentile in 2001.

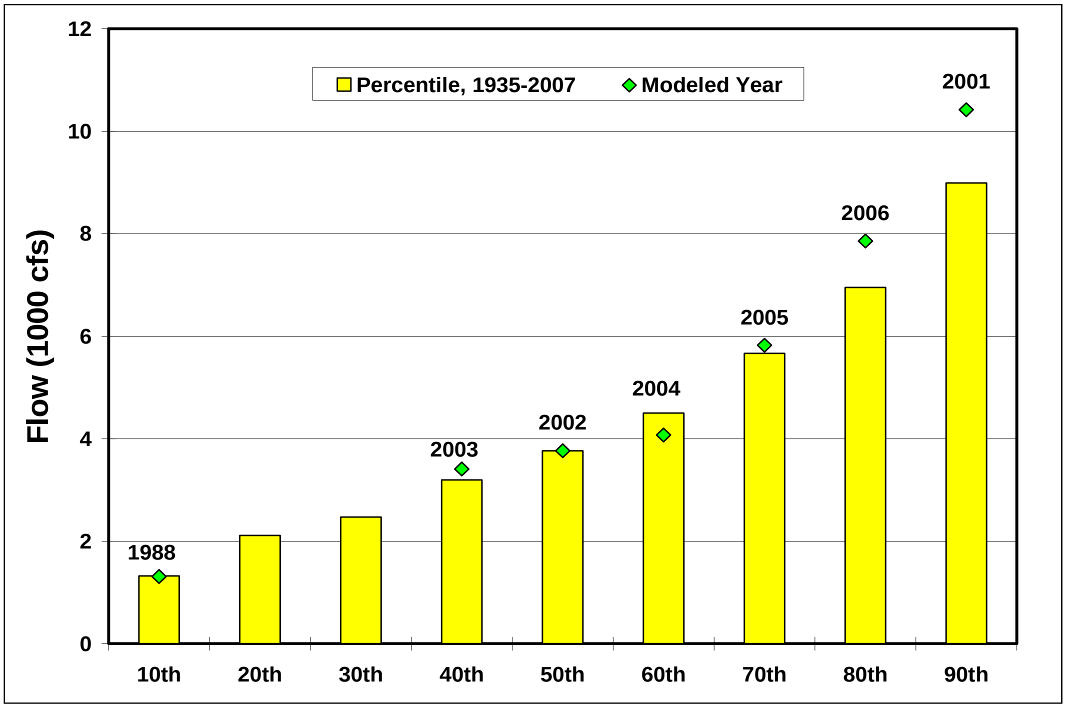


Figure 86. Mean annual flow, historic percentiles and modeled years, RM 39.4.

Critical conditions for DO, NH4, and eutrophication occur in the summer months, June through September, when river flows are low and water temperatures are high. For an extended period during the summer of 1988, river flows were near the $7Q_{10}$ statistic for the 70-year record and below the $7Q_{10}$ statistic for the more recent 30-year record (Figure 87). River flows fell below 2,000 cfs late in the summers of 2001, 2003, and 2006 but remained above the $7Q_{10}$ statistic. Diel DO fluctuation due to phytoplankton activity occurs more frequently at flows below 2,000 cfs as shown for 2003 in Figure 88, so this number provides a target flow for summer low-flow conditions.

Discharge permits are reissued by the MPCA on a five-year cycle under the National Pollution Discharge Elimination System (NPDES). They provide sources of maximum permitted loads for point-source discharges that could be applied in the model. Seven NPDES-permitted discharges are currently defined in the model: Blue Lake WWTP, Seneca WWTP, Black Dog Generating Plant (two outfalls), and Minneapolis-St. Paul International Airport (three stormwater outfalls).

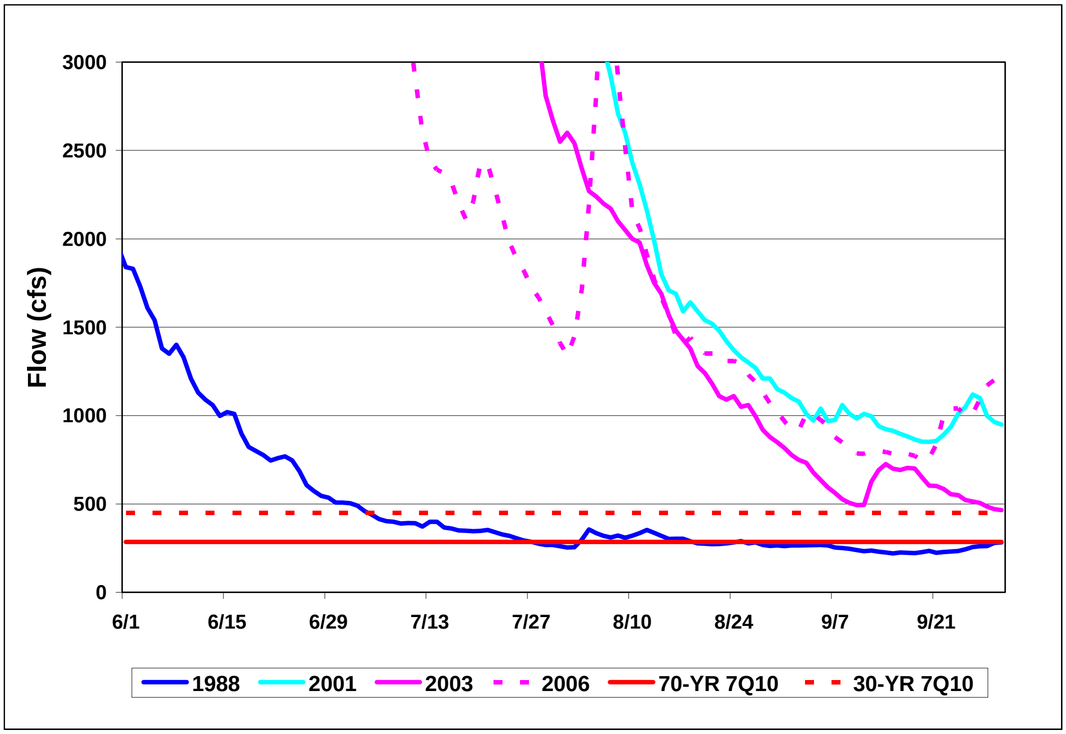


Figure 87. Mean daily flow compared to $7Q_{10}$ statistic, select summers at RM 39.4.

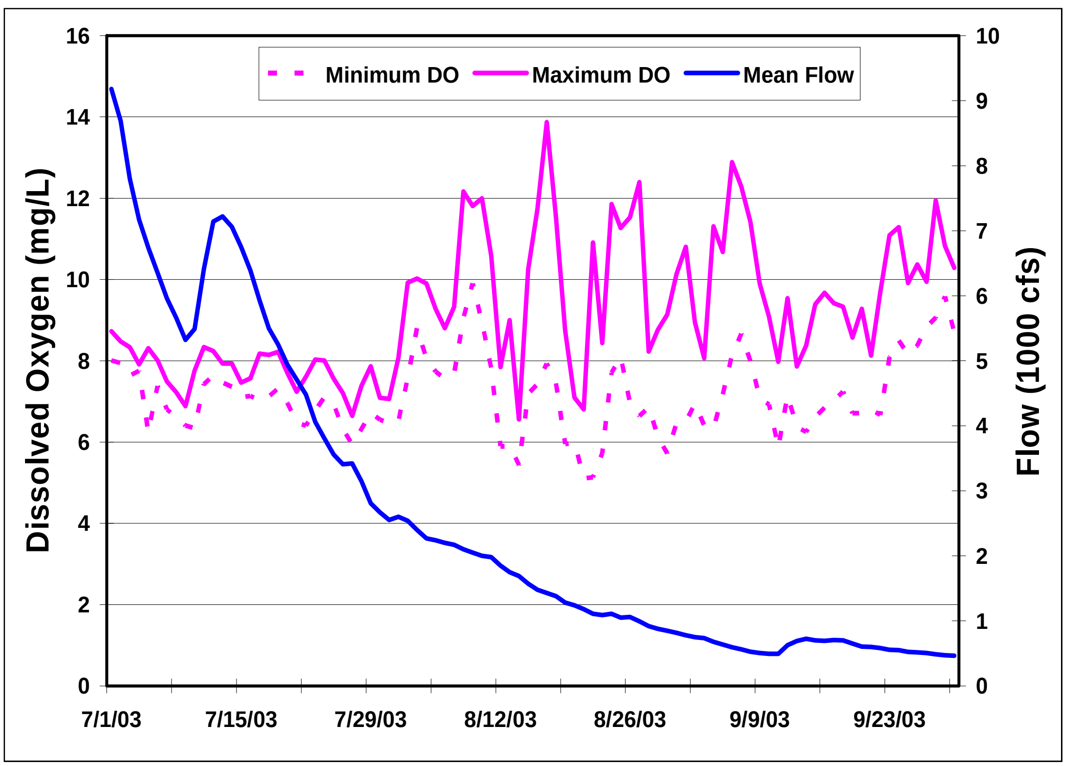


Figure 88. DO concentrations and flow, July-September 2003, RM 3.5.

Load allocation studies that have been completed and approved provide additional sets of well-defined loads and conditions to apply in the model (MPCA 2007). The most recent waste load allocation (WLA) study of the lower Minnesota River was completed in 1985 (MPCA 1985). In addition to recommending effluent BOD and NH4 limitations, the study concluded that a 40% reduction in BOD loads from nonpoint sources was needed to meet the DO standard. This resulted in a total maximum daily load (TMDL) study of BOD sources in the Minnesota River Basin from Lac Qui Parle to Jordan, Minnesota (MPCA 2004).

Tetra Tech (2003) developed a watershed model using the HSPF framework for application in the TMDL study. Resulting loads from scenarios run in the Minnesota River Basin Model for the DO TMDL study provide possible sets of inputs to apply in the Lower Minnesota River Model if an acceptable translation between the HSPF outputs and CE-QUAL-W2 inputs is developed. TMDL studies are currently underway for turbidity in the Minnesota River Basin and turbidity and nutrients in the Mississippi River and Lake Pepin. In this model application, the scenarios were limited to information from completed load allocation studies, but the model may prove useful for application in the current and future studies.

Scenario A: Apply maximum permitted WWTP loads

How different would the water quality of the river have been in 1988 or other low-flow periods if the point sources had been discharging at their maximum permitted loads? The model provides a tool to address this and other questions. In the first model application, labeled Scenario A, flow and concentration files for the Blue Lake and Seneca WWTPs were changed to reflect current permit limitations. The revised files were applied to the models for water years 1988, 2001, 2003, and 2006, which had periods of low river flows in the summer.

Permit limitations were not applied to other point sources. The Black Dog Generating Plant is regulated primarily for thermal effects on the river, so its flow and temperature files from the calibration were not modified. The reflected inputs for water quality at the two outfalls from Black Dog Lake were updated in Scenario A to account for changes at the Blue Lake WWTP, which is located upstream. Stormwater discharges from the international airport have an annual CBOD5 load limit of 810 metric tons (mt), but loading rates vary greatly with storm events and seasonal conditions. To simplify the model application, input files for the airport were also left unchanged from the calibration.

Note that the airport outfalls were not defined in the 1988 model due to insufficient data. Also, the airport underwent substantial stormwater improvements by 2004. Airport discharges exceeded NPDES permit limitations for CBOD5 in calendar years 2001 and 2002. The improvements have substantially lowered both the concentration and mass of CBOD5 discharged.

Table 44 summarizes how maximum permitted loads for the two WWTPs were defined in Scenario A. The same input files were applied in all years. Flows were based on the seasonal average flows used in the permits to calculate seasonal or monthly CBOD5 and TSS load limitations. To make it easy to change monthly limits in future applications, the input files contain records for the first and last days in each month (total of 24 records). In addition to CBOD5 and TSS, permit requirements for DO, NH4, and TP were applied with the exception of aeration requirements at Seneca (later tested in Scenario B). Values for unregulated variables were based on measured data. More details on the inputs follow.

CBOD is input to the model as ultimate CBOD. For both WWTPs, monthly average concentrations of 48 mg/L (June-September) and 80 mg/L (October-May) were assigned. These are roughly the same values applied in the WLA study (MPCA 1985), which were based on CBOD5 permit limitations of 12 mg/L (Blue Lake, summer), 15 mg/L (Seneca, summer), and 25 mg/L (both WWTPs, other months) multiplied by ultimate-to-5-day CBOD ratios (Blue Lake, 3.95 at 12 mg/L and 3.19 at 25 mg/L; Seneca, 3.17 at all concentrations). In the WLA study, CBOD decay rates (base *e*) were specified only for the river: 0.13/day for RM 25-17 and 0.11/day for RM 17-0. In CE-QUAL-W2, decay rates can be specified for individual CBOD sources. In Scenario A, CBOD decay rates for both WWTPs were set to 0.11/day to best reflect the WLA settings. In effluent samples collected in 1982, the median bottle decay rate was 0.07/day. In the calibration files, CBOD decay rates were set to .0322/day (Blue Lake, 2001-06), 0.0294 (Seneca, 2001-06), and 0.085/day (both WWTPs, 1988, pre-upgrade). All CBOD settings would require careful evaluation in a load allocation study.

Table 44. Definition of maximum permitted WWTP loads in Scenario A.

Model Input	Definition
Flow	Assign seasonal average flows used in the permits to calculate seasonal or monthly CBOD5 and TSS load limitations: Blue Lake: June-Sept, 37 mgd; Oct-Feb, 32 mgd; Mar-May, 42 mgd Seneca: June-Sept, 38 mgd; Oct-Feb, 34 mgd; Mar-May, 38 mgd
Temperature	Use calibration files, which were based on measured data.
Total Dissolved Solids	Assign median concentrations from 2004-06: Blue Lake, 1100 mg/L; Seneca, 1510 mg/L
Inorganic Suspended Solids	Assign 100% of the monthly average TSS limit of 30 mg/L.
Orthophosphate	Split the annual average TP limit of 1 mg/L between PO4 and the organic P associated with CBOD. Resulting PO4: June-Sept, 0.808 mg/L; Oct-May, 0.680 mg/L
Ammonium	Assign the monthly average permitted limits: May, 9 mg/L; June, 12 mg/L; July-Sept, 2 mg/L; Oct, 5 mg/L; Nov, 7 mg/L; Dec-Mar, 22 mg/L; April, none but use 22 mg/L
Nitrate	Subtract the monthly NH4 limits from the annual average concentrations of total inorganic nitrogen from 2004-06: Blue Lake, 11 mg/L; Seneca, 14 mg/L
Dissolved Silica	Assign the median concentrations from 2004-06: Blue Lake, 22 mg/L; Seneca, 17 mg/L
Carbonaceous Biochemical Oxygen Demand	Apply seasonal CBODU values from the WLA study (MPCA 1985): June-Sept, 48 mg/L; Oct-May, 80 mg/L
Dissolved Oxygen	Apply minimum required concentrations: Blue Lake: 7 mg/L in Dec-Mar and 6 mg/L in Apr-Nov Seneca: 6 mg/L in all months (aeration applied in Scenario B)
Organic Matter	Set all four groups to zero. Use CBOD instead.
Phytoplankton	Set all three groups to zero.

In CE-QUAL-W2, organic matter is represented by the CBOD groups, non-living organic matter (OM) groups, and phytoplankton (ALG) groups. Care must be taken not to double-count organic matter in the inputs for these groups. All organic matter from the Blue Lake and Seneca WWTPs was assigned to the CBOD groups in order to best reflect permit requirements, assign specific decay rates, and track these individual sources of oxygen demand. Labile organic phosphorus and nitrogen are associated with the CBOD groups via stoichiometric ratios (0.004 P:CBOD and 0.060 N:CBOD in this application). Organic P and N decay with CBOD, resulting in PO4 and NH4, and some portion settles with CBOD. A disadvantage of choosing the CBOD groups over the OM groups is less control over the labile/refractory and dissolved/particulate fractions.

Choosing CBOD over OM affects the assignment of permit limitations for TSS (monthly average of 30 mg/L) and TP (annual average of 1 mg/L). For this application, 100% of the TSS limit was assigned to ISS and the TP limit was split between PO4 and the organic P associated with CBOD. Organic P is 0.192 mg P/L when CBOD is 48 mg/L and 0.320 mg/L when CBOD is 80 mg/L. From special effluent samples collected during 2004-06, the percent organic/total SS averaged 66% at Blue Lake and 81% at Seneca, and the percent PO4/TP averaged 60%. For suspended solids, the assignment will result in overestimating the state variable ISS and underestimating the derived variable VSS, which in turn will affect the calculation of light attenuation and turbidity to some extent. The assignment overestimates PO4 by setting the effluent PO4 concentration to 0.808 mg/L in summer and 0.680 mg/L in other months when it is currently split 60/40 between PO4 and other P forms.

Effluent aeration to a DO concentration of 16 mg/L is required at the Seneca WWTP when river flows at Jordan are below 1200 cfs for seven consecutive days during June through September. This condition occurred during most of the summer in 1988 and for periods in late summer in 2001, 2003, and 2006. Aeration was not simulated in Scenario A, but it was tested in Scenario B.

The only changes from the calibration to Scenario A were revised input files for the Blue Lake and Seneca WWTPs and revised CE-QUAL-W2 control files to increase the CBOD decay rates for the WWTPs and adjust the time-step as needed. Figures 89-92 show the results for select variables (DO, NH4, PO4, and CHLA) in 1988 and 2003. These two years provided the lowest flows and largest contrast. Results for 2001 and 2006 were similar to those for 2003, but were more dampened due to higher river flows. Changes in TSS and NO3 were also plotted but were relatively minor.

The WWTPs currently perform at levels much below the maximum permitted limits for CBOD5 and NH4. In special effluent samples collected in 2004-06 for the modeling project, the average CBOD5 concentrations were 3.3 mg/L for Blue Lake and 4.3 mg/L for Seneca (86 samples each), and the average NH4 concentrations were 0.3 and 0.8 mg/L, respectively (88 samples each). Several values were recorded as below the detection limits for CBOD5 and NH4; these were set to the detection limit for calculating the average. In contrast, the summer permit limits are 12 and 15 mg/L for CBOD5 and 2.0 mg/L for NH4, and the winter permit limits are 25 mg/L for CBOD5 and 22 mg/L for NH4.

In 1988 the WWTPs had not yet upgraded to advanced secondary treatment with nitrification, so effluent CBOD5 and NH4 concentrations were substantially higher than current levels. CBOD5 and NH4 averaged 12 and 13 mg/L at Blue Lake and 16 and 15 mg/L at Seneca, respectively. The two WWTPs were upgraded in 1992.

As expected in response to the increased CBOD5 and NH4 effluent loads, DO concentrations decrease in Scenario A when compared to the calibration results (Figure 89). This is particularly evident in winter when river flows are low and effluent loads are high. Note that DO concentrations at RM 3.5 were often below 5 mg/L in the calibration results for June-September 1988. The additional effluent loads in Scenario A depress DO concentrations somewhat further in the summer of 1988. Setting effluent loads to their maximum permitted limits depresses DO levels to a greater degree (at times 1-2 mg/L) in August and September 2003 compared to 1988 because actual effluent loads in 2003 were much lower than permitted loads. In the calibration, DO concentrations stayed above 5 mg/L at RM 3.5 in 2003, while in Scenario A, DO concentrations fall below 5 mg/L for a period in August and September.

The CBOD loads represent the oxygen demand (carbonaceous) associated with the decomposition of organic matter in the effluent. The NH4 loads also represent a source of oxygen demand (nitrogenous), as NH4 is converted to NO3 in the river. Both combine to decrease DO concentrations in the river. Effluent DO concentrations can increase or decrease DO concentrations in the river, depending on which are higher. In Scenario A, river DO concentrations responded as expected in two contrasting years, lending added confidence in the model's ability to forecast.

The improved level of wastewater treatment between 1988 and 2003 is also apparent in the calibration results for NH4, with concentrations at RM 3.5 often in the 1-3 mg/L range in 1988 and most concentrations below 0.5 mg/L in 2003 (Figure 90). A portion of the decrease in 2003 is explained by higher flows affording more dilution.

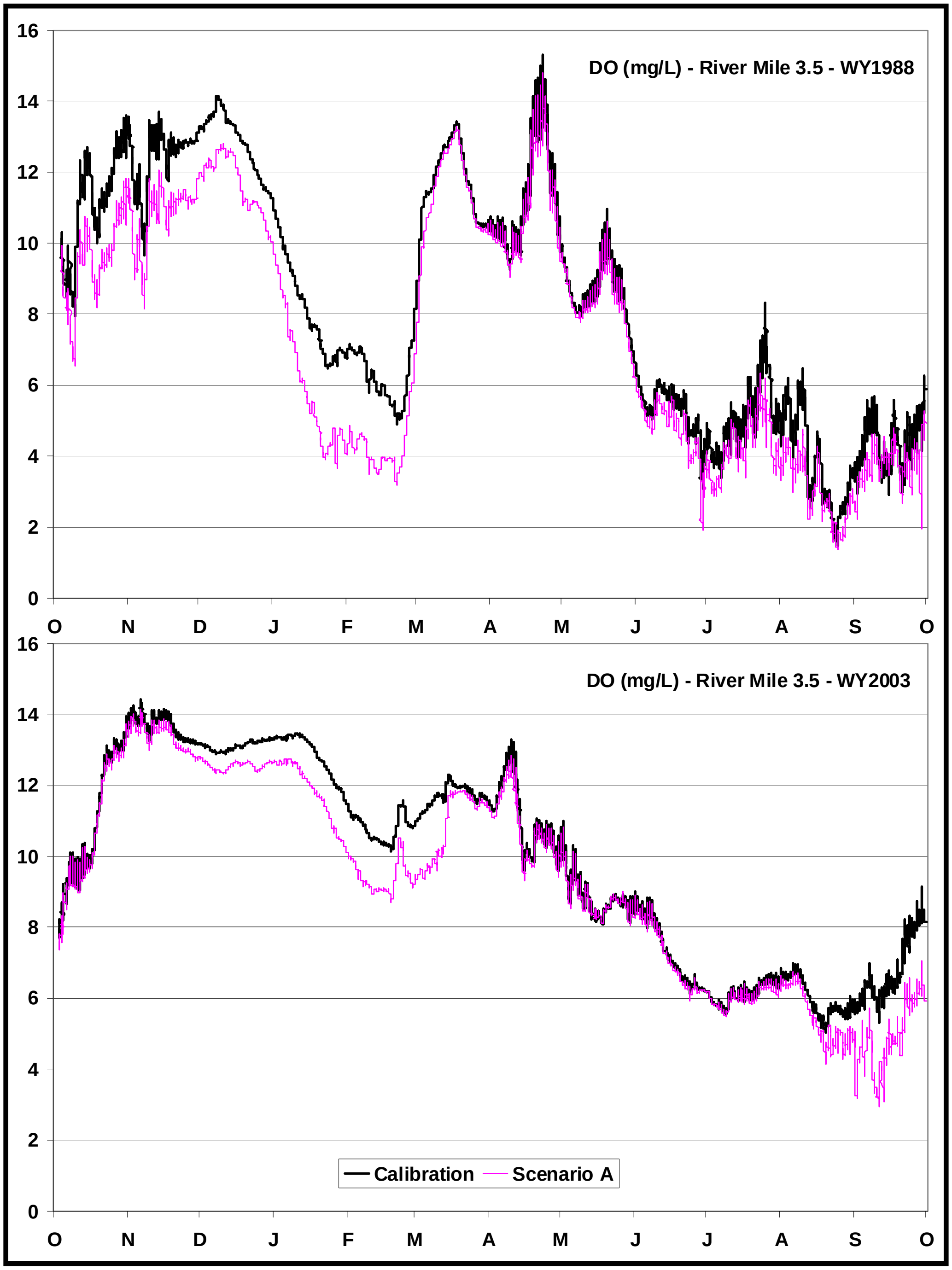


Figure 89. Scenario A results for DO at RM 3.5, 1988, and 2003.

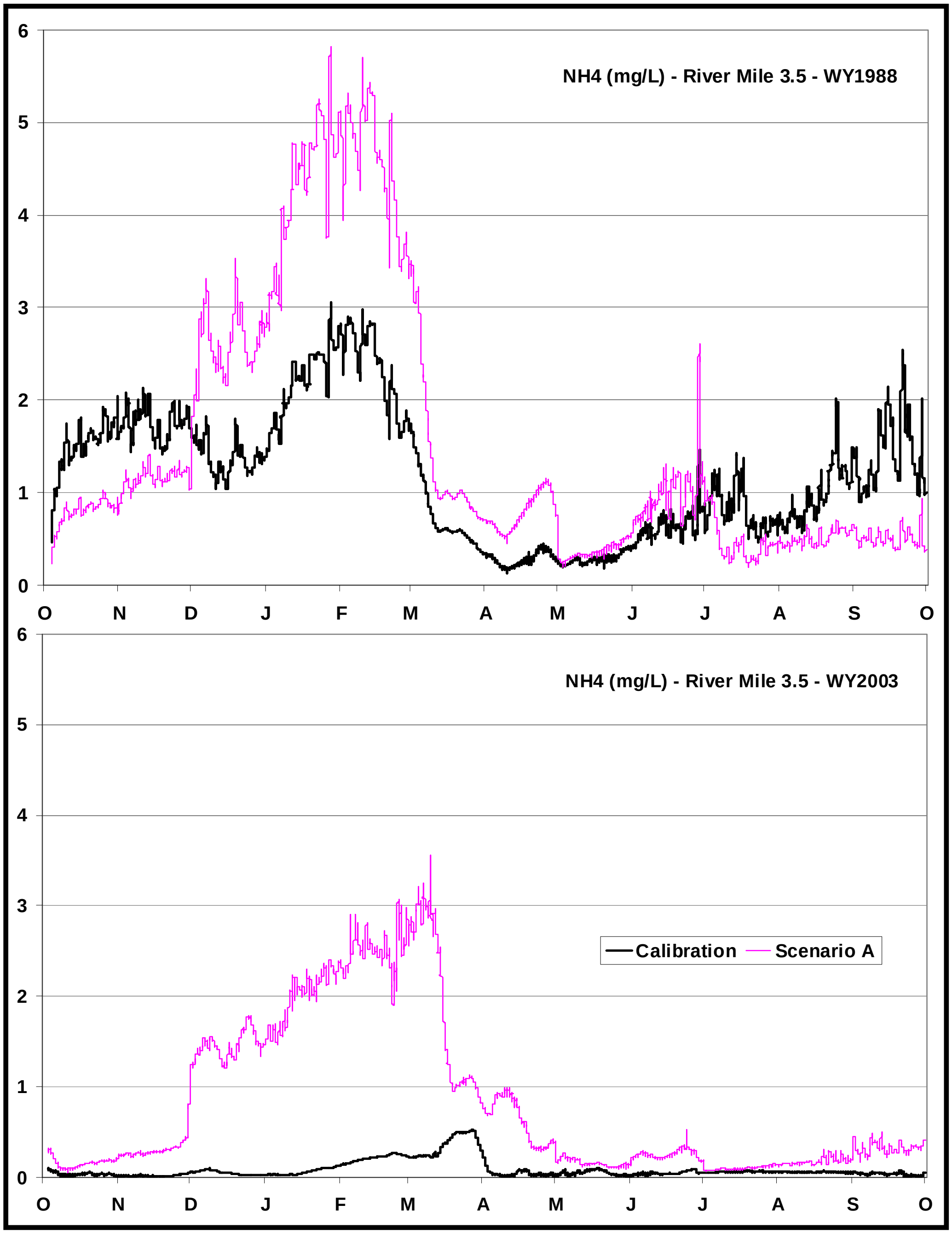


Figure 90. Scenario A results for NH4 at RM 3.5, 1988, and 2003.

Increasing the effluents to maximum permitted loads has the same effect during December to July in both years: river NH4 concentrations increase greatly over the calibration results. However, due to the treatment upgrade, Scenario A has opposite effects in 1988 and 2003 during the fall and summer periods. In 1988 maximum permitted NH4 loads decrease river concentrations at RM 3.5, while in 2003 the loads increase river concentrations over the calibration results.

To fully evaluate the toxic effects of ammonia on aquatic life, the un-ionized portion must be calculated using temperature and pH. pH was not simulated in the Lower Minnesota River Model due to inadequate data for alkalinity and total inorganic carbon. Temperature and pH measurements at the continuous monitor at RM 3.5 could be used with model results for NH4 to estimate un-ionized ammonia at this site. pH levels could change in the future under some scenarios (e.g., reduced phytoplankton levels).

Since the mid-1990s, operations at the Blue Lake and Seneca WWTPs were modified to optimize phosphorus removal, producing annual average effluent phosphorus concentrations below 1.5 mg/L. Biological phosphorus removal to 1.0 mg/L was fully implemented by the end of 2008. In contrast, the average effluent TP concentration was 3.5 mg/L at both facilities in 1988. The reduction in effluent TP loads likely contributed to the decrease in TP concentrations at RM 3.5 between 1988 and 2003 in the calibration results (Figure 91).

In Scenario A, effluent TP concentrations are set to 1.0 mg/L. This change yields different responses in the two years because phosphorus removal was partially implemented in 2003, often decreasing effluent TP below 1.0 mg/L. Effluent TP loads in Scenario A yield large decreases in river PO4 concentrations at RM 3.5 in 1988 with the exception of spring and early summer (Figure 91). In contrast, Scenario A generally yields a slight increase in river PO4 concentrations in 2003. The increase is more evident during low-flow periods in late winter and late summer in 2003.

In the results for 1988 in Scenario A, large decreases in PO4 concentrations at RM 3.5 translate to moderate decreases in CHLA concentrations in fall and summer (Figure 92). CHLA concentrations in the fall of 1987 ranged from 60 to 90 ug/L in the calibration results; they decrease by less than 5 ug/L in Scenario A. Summer concentrations in 1988 ranged from 10 to 70 ug/L in the calibration results; they decrease by varying amounts in Scenario A but as much as 10 ug/L or more at times. CHLA concentrations change very little in 2003 under the Scenario A effluent loads.

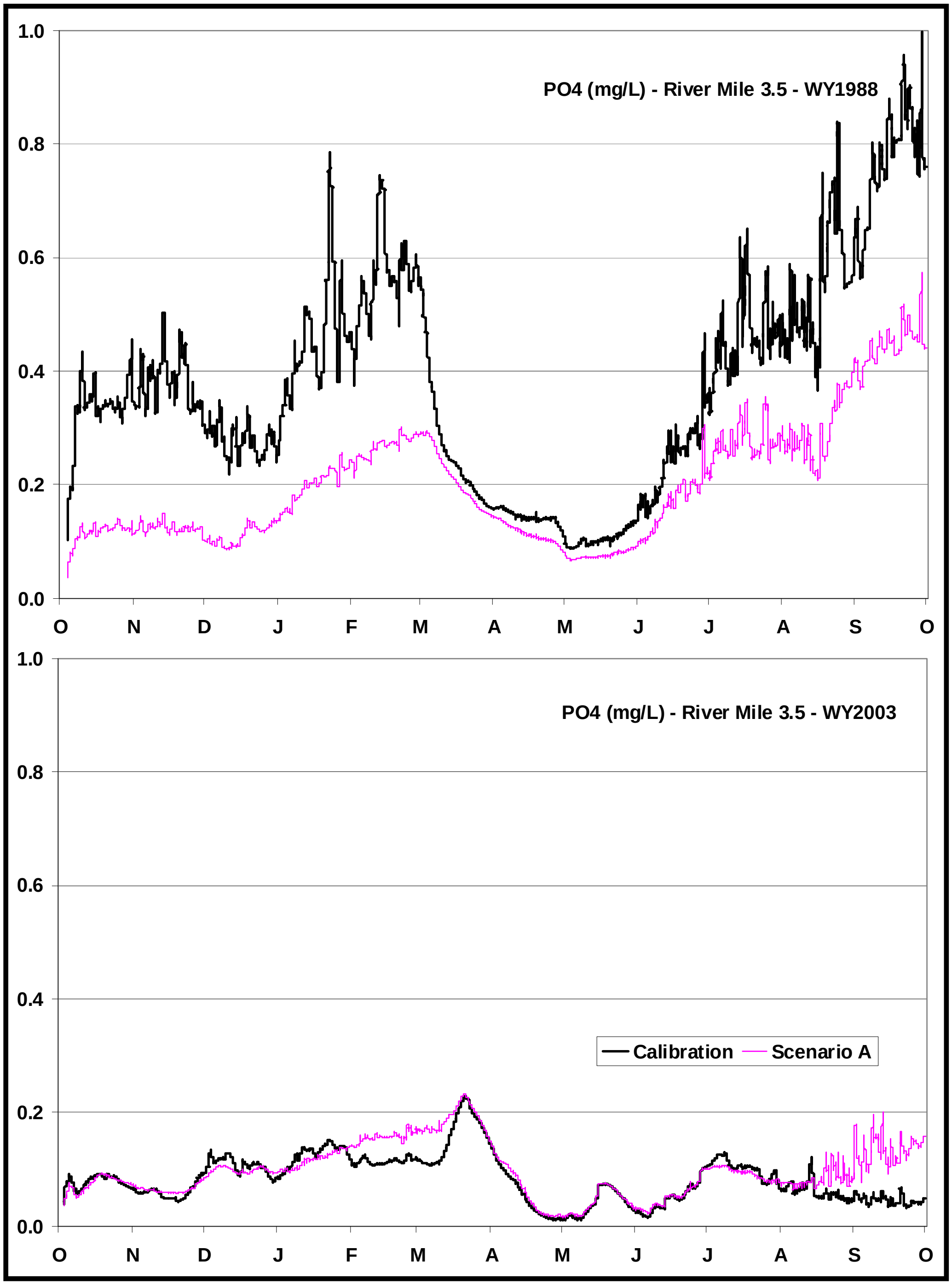


Figure 91. Scenario A results for PO4 at RM 3.5, 1988, and 2003.

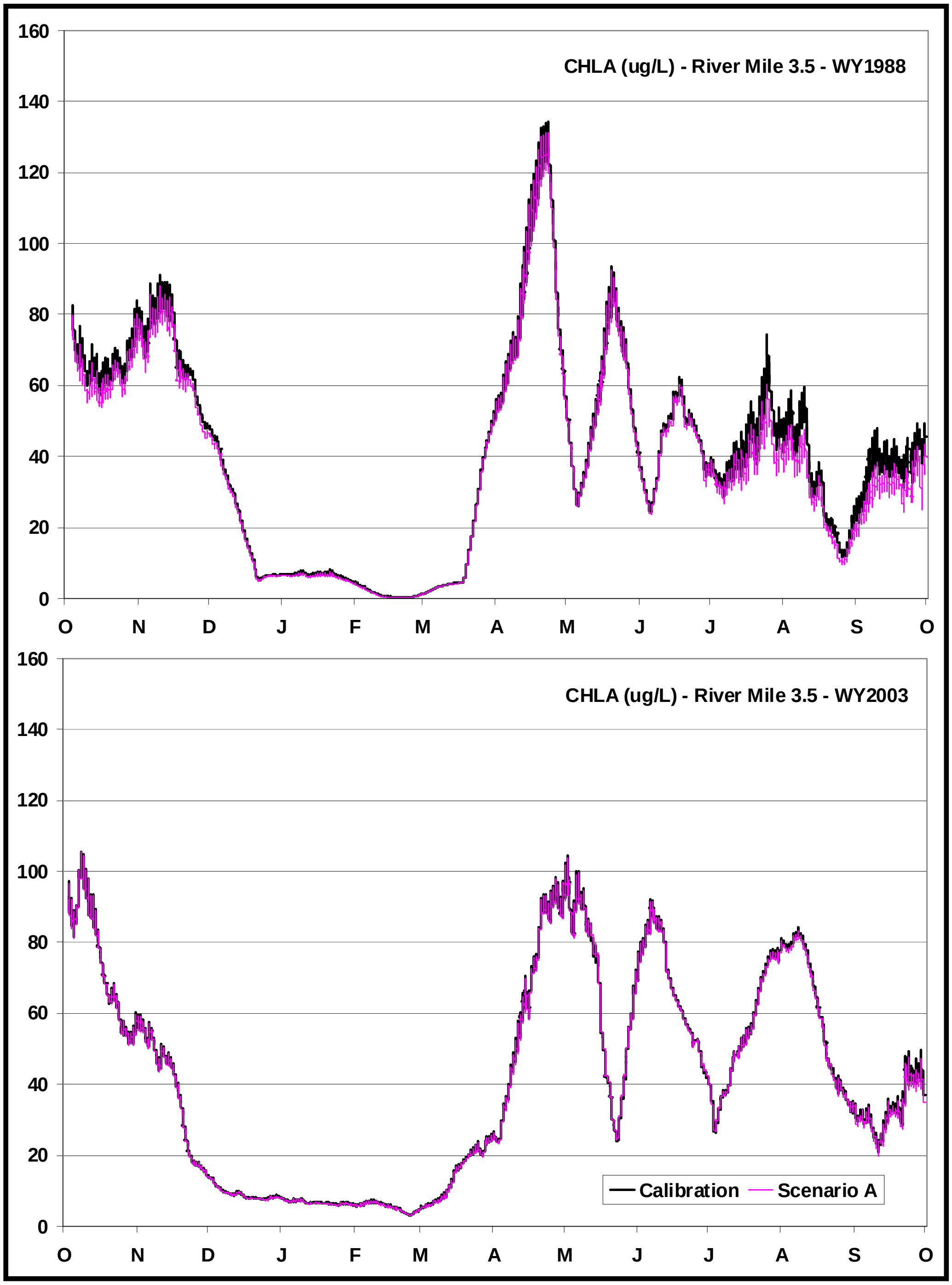


Figure 92. Scenario A results for CHLA at RM 3.5, 1988, and 2003.

Scenario B: Apply WWTP loads from waste load allocation study

The most recent WLA study by the MPCA (1985) recommended reductions in four sources of oxygen demand in order to meet DO standards in the lower Minnesota River in the future, which at the time was assigned to the year 2000:

1. CBOD and NH4 loads from the Blue Lake WWTP.
2. CBOD5 and NH4 loads from the Seneca WWTP in addition to effluent aeration when river flows were less than 1200 cfs for seven consecutive days in the summer.
3. CBOD5 loads in the Minnesota River at Shakopee (40% reduction).
4. Sediment oxygen demand and benthic ammonium release rates from the river bed (40% reduction).

Scenarios B, C, and D attempt to replicate the reduced loads and rates from the WLA study in the CE-QUAL-W2 model of water year 1988, when river flows were near the $7Q_{10}$ statistic for much of the summer. In Scenario B, WWTP inputs from Scenario A were adjusted to match the WLA settings for Blue Lake and Seneca. In Scenario C, the SOD rates, which are linked in CE-QUAL-W2 to ammonium release, were adjusted. In Scenario D, the CBOD loads at Jordan were reduced using output from the Minnesota River Basin Model as applied in the DO TMDL Study (MPCA 2004). The objective of this set of scenarios is to evaluate whether the CE-QUAL-W2 model generates results reasonably in line with the WLA study.

The current permit limitations for CBOD5 and NH4 effluent concentrations at the two WWTPs were established in the 1987 amendment to the WLA study. These are the concentrations in the input files for Scenario A. The loads differ, however, because the WLA study applied projected annual average flows of 32 mgd at the Blue Lake WWTP and 34 mgd at the Seneca WWTP. In Scenario B, effluent flows were revised to match flows in the WLA study. The concentration files were unchanged from Scenario A (Table 44) with one exception: DO concentrations at Seneca were increased to 16 mg/L during June-September to simulate effluent aeration. In 1988 flows at Jordan were below the aeration target of 1200 cfs from June 11 to September 30. Note that Scenario A included maximum permitted concentrations for TSS (30 mg/L, 100% assigned to ISS) and TP (1.0 mg/L, split between PO4 and the organic P associated with CBOD).

The results for DO concentrations at RM 3.5 under Scenario B are shown in Figure 93. The blue line shows the results of changes to the WWTP flows alone, while the pink line adds aeration at Seneca during the summer. The results are identical for October-May (shown in pink). The blue line tracks closely with the results for Scenario A despite the change in flows. For example, during summer, flows from Blue Lake decrease from 37 mgd in Scenario A to 32 mgd in Scenario B, and flows at Seneca decrease from

38 to 34 mgd. The model predicts that aeration at Seneca will increase river DO concentrations by a small margin at mile 3.5 but not enough to pull concentrations above 5 mg/L at all times. As in the WLA study, the CE-QUAL-W2 model shows that load reductions at the WWTPs alone are not enough to maintain water-quality standards.

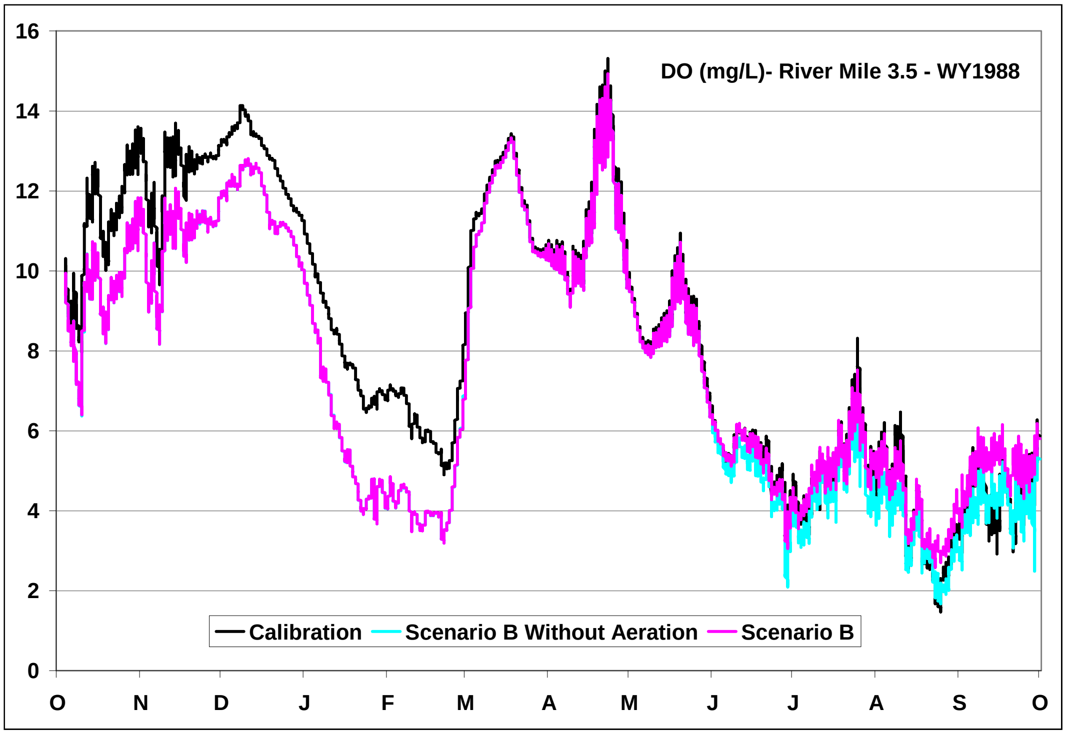


Figure 93. Scenario B results for DO at RM 3.5, 1988.

Scenario C: Apply SOD settings from waste load allocation study

In the WLA study (MPCA 1985), SOD rates were reduced by 40% from model calibration rates to match CBOD load reductions recommended for the Minnesota River at Shakopee in projections of future summer conditions (Table 45). Benthic NH4 release rates were also reduced by 40%. In Scenario C, the reduced SOD rates from the WLA study were applied along with the WWTP loads in Scenario B. SOD rates for RM 25-22 in the WLA study were extended upstream to RM 36.3 (near Jordan) in the CE-QUAL-W2 model. The user specifies SOD rates at 20 ◦C for individual segments, and the model adjusts the rates to the ambient temperature. In CE-QUAL-W2, the sediment release rate of NH4 under anaerobic conditions is specified as a fraction of SOD, which is 0.010 in this application.

Table 45. SOD rates applied in WLA Study (MPCA 1985).

River Miles	Calibration SOD Rates (gm/m²/day)	Projection SOD Rates (gm/m²/day)
25-22	0.60	0.36
22-17	2.83	1.70
17-11	1.42	0.85
11-7	1.25	0.75
7-0	1.32	0.79

As shown in Figure 94, the reduced SOD rates in Scenario C increased DO concentrations at RM 3.5 by roughly 1 mg/L in the summer months. The model appears sensitive to the settings for SOD rates, so the settings warrant a closer look. Figure 95 compares SOD rates applied in the WLA study to a steady-state model of a summer survey in 1980 and the SOD rates applied in the current study to the CE-QUAL-W2 model of water year 1988. While SOD rates applied in the two models are within 0.5 $gm/m^2/day$ in the critical lower reach, rates applied in the CE-QUAL-W2 model are generally 1.17 or 1.34 $gm/m^2/day$ higher over miles 22-13 and 3.4 $gm/m^2/day$ higher over miles 25-22 compared to rates in the WLA study.

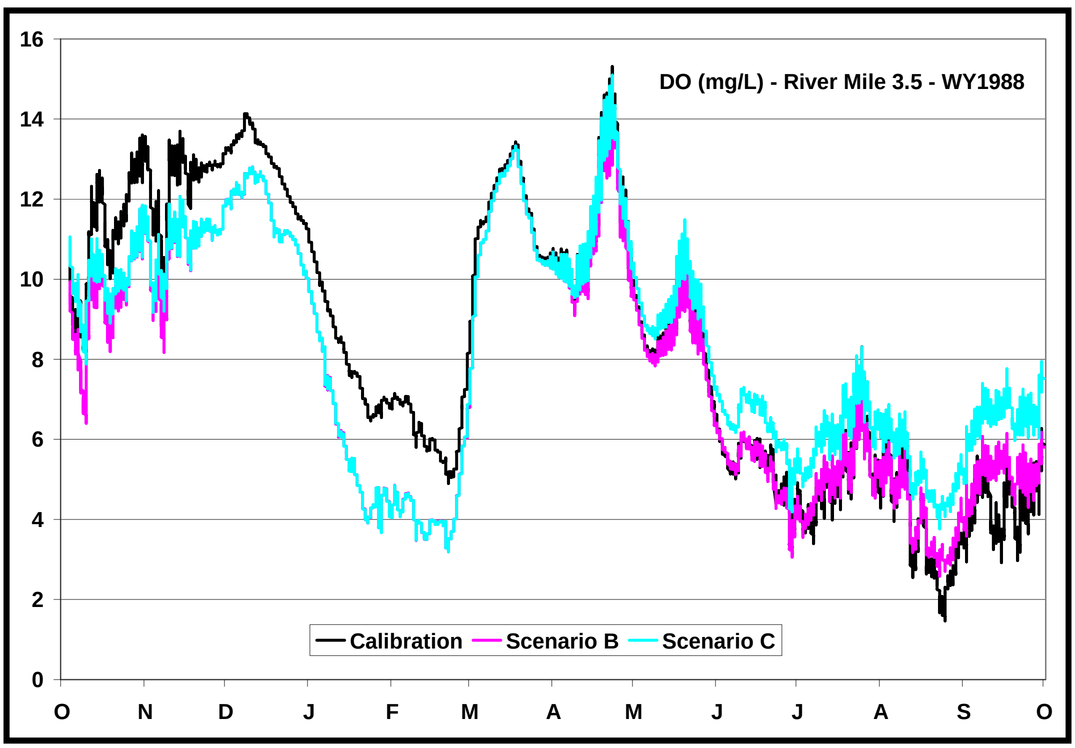


Figure 94. Scenario C results for DO at RM 3.5, 1988.

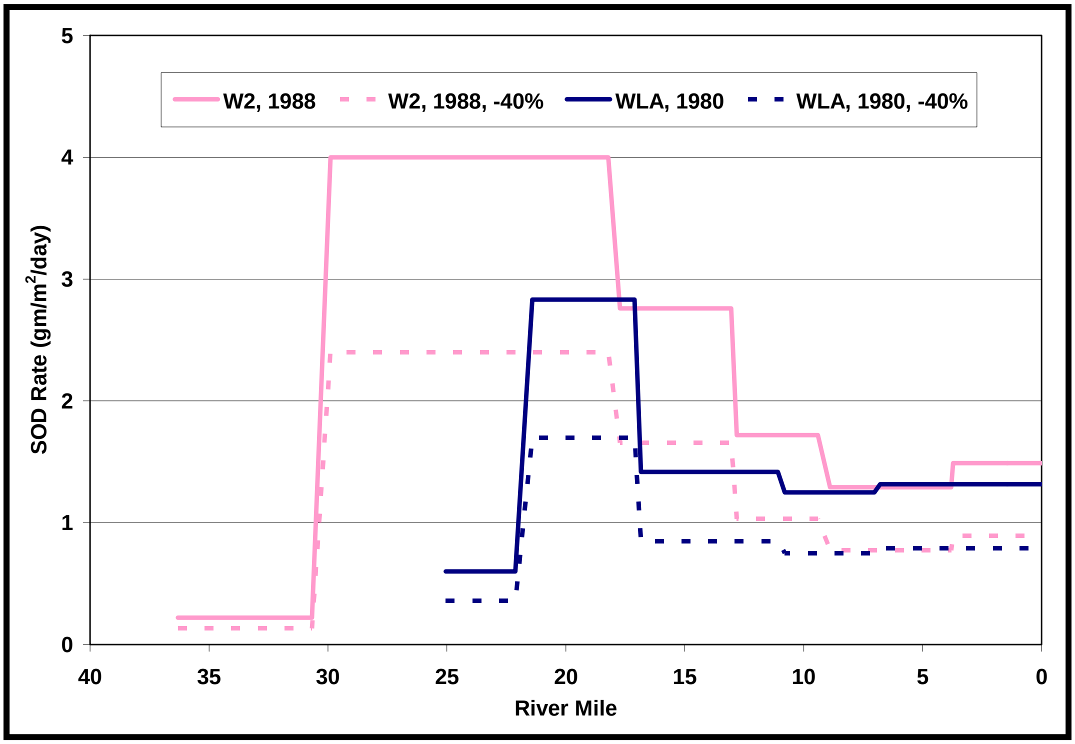


Figure 95. SOD rates applied in CE-QUAL-W2 model of 1988 and WLA model of 1980.

To evaluate the effects of different SOD rates on DO concentrations, the four sets of SOD rates in Figure 95 were applied to the CE-QUAL-W2 model of 1988. Figure 96 shows the results for June 1988. This month was selected because frequent DO measurements (shown as red triangles in Figure 96) were collected at RM 3.5 for a low-flow survey of Pool 2 of the Mississippi River. In general, SOD rates applied in the CE-QUAL-W2 calibration yield a better match to measured DO concentrations than rates applied in the WLA calibration. In Figure 96 the results for the CE-QUAL-W2 model with reduced SOD rates (dashed pink line, often covered) track closely with results for the WLA model with unreduced rates (solid blue line). This comparison demonstrates the importance of sensitivity analyses and the need for careful evaluation of important rates in future load allocation studies.

Scenario D: Apply results from Minnesota River basin model

In the final scenario, oxygen-demanding loads at Jordan were reduced. This was accomplished in different ways in the two models applied by the MPCA in the earlier WLA and TMDL studies. A steady-state model was applied in the WLA study, and only one number was changed to simulate the 40% reduction in the future summer run: summer average CBODU

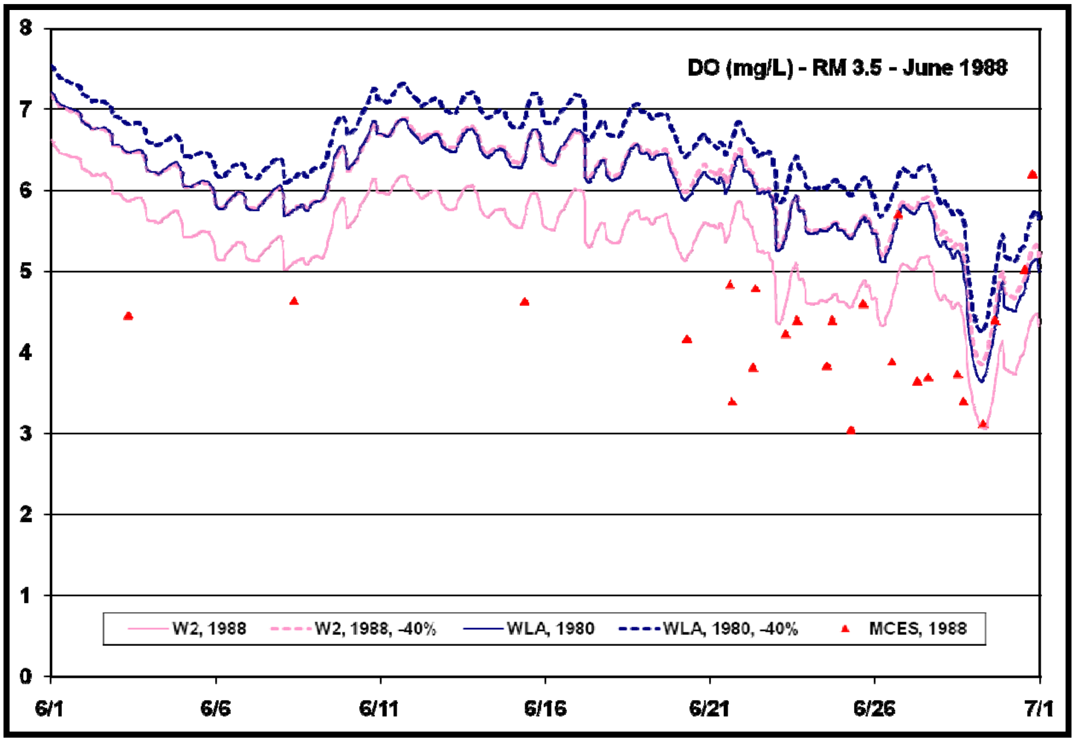


Figure 96. DO concentrations at RM 3.5 in June 1988 using different sets of SOD rates.

concentration at RM 25.0 (Shakopee). In the subsequent TMDL study of BOD sources upstream of Jordan, a large time-variable watershed model was developed using the HSPF framework, and several management scenarios were applied to find a combination of practices that would meet the BOD reduction target at Jordan or Shakopee (MPCA 2004; Tetra Tech 2003). The net effect of the final HSPF scenario (#7) was to reduce BOD loads at Jordan by 40% during summer low-flow conditions, but results for nearly all variables changed. In particular, suspended solids, phosphorus, and phytoplankton concentrations were reduced.

Scenario D took advantage of the TMDL work and input results from the final load-reduction scenario of the Minnesota River Basin Model (HSPF) into the Lower Minnesota River Model (CE-QUAL-W2). The HSPF results at Jordan were applied along with WWTP loads from Scenario B. To offer a clearer picture of the revised Jordan inputs, reduced SOD rates from Scenario C were not applied in Scenario D. While the HSPF model was developed for multiple years, TMDL projections focused on 1988 due to flow conditions near the $7Q_{10}$ statistic cited in the DO standard. Mean daily flows and concentrations for water year 1988 from the HSPF model, Scenario 7, were provided by the MPCA.

The state variables in the two models are equivalent in some cases but more often differ, so it was necessary to translate some HSPF outputs into CE-QUAL-W2 inputs. A translation table was compiled with assistance from the MPCA and Tetra Tech (Table 46). Some translations warrant additional notes:

- CE-QUAL-W2 supports any number of ISS groups, but only one group was defined in the Lower Minnesota River Model due to inadequate data on particle sizes. Concentrations for the three HSPF groups (sand, silt, and clay) were combined and input to the CE-QUAL-W2 model. An alternative was to add two ISS groups and matching sediment characteristics (e.g., settling rates). Note that CE-QUAL-W2, V3.6, does not support sediment transport as in the HSPF framework (Cole and Wells 2008).
- Three forms of particulate PO4 and NH4 sorbed to the three ISS groups are defined in HSPF, but they were minor fractions in the 1988 model. In CE-QUAL-W2, they were combined with dissolved PO4 and NH4. CE-QUAL-W2, V3.6, allows P sorption to suspended particles and settling but does not support desorption (Cole and Wells 2008).
- The OM groups in CE-QUAL-W2 were approximately converted from the BOD and organic carbon groups in HSPF. Note that the HSPF model was calibrated for BOD and not organic matter. In particular, the refractory portion of OM in HSPF may not be well represented.
- The single group of phytoplankton in HSPF was split into three groups in CE-QUAL-W2 using monthly average percentages from all available biomass data at RM 3.5.

Table 46. Translation table from HSPF to CE-QUAL-W2, Minnesota River at Jordan.

W2 ID	CE-QUAL- W2 Description (mg/L unless specified)	HSPF Approximate Equivalent
Q	Flow (cms)	Calculate from QVOL (flow in acre-foot/day). Calculate water-quality concentrations in CE-QUAL-W2 from loads and flows in HSPF.
TMP	Temperature (deg C)	Use measured temperature, not HSPF results.
TDS	Total dissolved solids	None in HSPF but used only as a tracer in CE-QUAL-W2. Use mean daily TDS from CE-QUAL-W2 results for Jordan (segment 2).
ISS	Inorganic suspended solids	Combine SAND, SILT, and CLAY.
PO4	Bioavailable phosphorus	Combine PO4 (dissolved PO4) and PO4A-C (sorbed PO4). The annual average sorbed/total PO4 was 4% in WY 1988 (HSPF at Jordan).
NH4	Ammonium nitrogen	Combine NH3 (dissolved NH3) and NH3A-C (sorbed NH3). The maximum sorbed/total NH4 was 1% in WY 1988 (HSPF at Jordan).

W2 ID	CE-QUAL- W2 Description (mg/L unless specified)	HSPF Approximate Equivalent
NO3	Nitrate-nitrite nitrogen	Use NO3.
DSI	Dissolved silica	None in HSPF but not often limiting in CE-QUAL-W2. Use mean daily DSI from CE-QUAL-W2 results for Jordan (segment 2).
LDOM	Labile dissolved organic matter (dry wt)	Calculate LOM as BOD*1.252, which converts BOD in mg/L to biomass in mg/L (dry wt). OM is non-living biomass in CE-QUAL-W2, while ALG is living biomass.
RDOM	Refractory dissolved organic matter (dry wt)	Calculate ROM as ORGC*2.041, which converts carbon in mg/L to biomass in mg/L (dry wt). Ignore ORGP and ORGN as W2 estimates these from OM using fixed stoichiometry. Note: The HSPF model was calibrated for BOD not OM.
LPOM	Labile particulate organic matter (dry wt)	Split LOM into LDOM and LPOM using monthly mean percentages from water years 2004-06.
RPOM	Refractory particulate organic matter (dry wt)	Split ROM into RDOM and RPOM using monthly mean percentages from water years 2004-06.
CBOD1-CBOD6	Carbonaceous biochemical oxygen demand	Set to zero at Jordan. Only used for Blue Lake, Seneca, and airport.
ALG1	Diatoms, biomass (dry wt)	Calculate from PHYT, which is also biomass in mg/L (dry wt). Split off three algal groups using monthly splits from historical measured data (1988, 1996, 2004-06 at MI 3.5).
ALG2	Blue-green algae	Split from PHYT.
ALG3	Other algae	Split from PHYT.
DO	Dissolved oxygen	Use DO.

In Scenario D, only the inputs at Jordan and the WWTPs were changed; other model inputs and settings remained the same as in the calibration. No changes were made to reconcile differences in coefficients between the two models, such as settling and decay rates, but this is an area worth further evaluation. The HSPF-to-W2 translation in this application should be considered preliminary. Scenario D offers a demonstration of what might be possible with linking models in future applications.

To test the translation, CE-QUAL-W2 inputs at Jordan for the calibration and Scenario D (not shown) were plotted and compared. Allowing for differences in how both sets of inputs were derived and how the HSPF scenario reduces BOD loads, the results seemed reasonable. Table 47 compares annual and summer loads at Jordan in the calibration and Scenario D. In addition to reductions in CBODU loads (derived from the OM and ALG groups), reductions in TSS and TP loads are simulated in Scenario D. Nitrogen loads, however, increase in this scenario compared to the calibration.

Table 47. Comparison of loads at RM 39.4 in Calibration and Scenario D runs, 1988.

W2 ID	Calibration Annual Load (mt)	Scenario D Annual Load (mt)	Calibration Summer Load (mt)	Scenario D Summer Load (mt)
ISS	79573	65898	7945	795
PO4	121	80	18	1
NH4	174	223	4	40
NO3	5065	7854	184	408
LDOM	2332	4873	286	241
RDOM	13203	5243	1625	4
LPOM	1488	3078	227	309
RPOM	8395	2850	1274	5
ALG 1-3	5071	3085	736	267
DO	12270	16153	1169	832
TN	6915	9126	418	498
TP	274	176	39	5
TSS	94526	74911	10181	1376
CBODU	42684	26780	5806	1156

Figure 97 shows Scenario D results for DO concentrations at RM 3.5 in water year 1988. Results from the calibration and Scenario B were plotted for comparison. As in the WLA study, BOD reductions at Jordan and the two WWTPs (Scenario D) increase DO concentrations at RM 3.5 in July-September over BOD reductions at the WWTPs alone (Scenario B). However, DO concentrations in June and October-January are projected to be lower in Scenario D than in Scenario B. DO concentrations decrease below 5 mg/L during periods in June and August. Reduced BOD (organic matter) loads might lead to decreased SOD rates and increased DO concentrations; however, zero-order SOD rates in this sample application were left at calibration rates.

Note the marked difference between Scenarios B and D in winter DO concentrations with much higher DO (>12 mg/L) at RM 3.5 in Scenario D (Figure 97). This difference originates at RM 39.4 where HSPF results for DO were greater than 16 mg/L during January-March compared to measured DO concentrations below 9 mg/L in 1988. The W2 calibration inputs at RM 39.4 were based on measured data. With winter DO concentrations largely controlled by temperature and reaeration and biological activity at a minimum, the handling of ice formation in the HSPF model

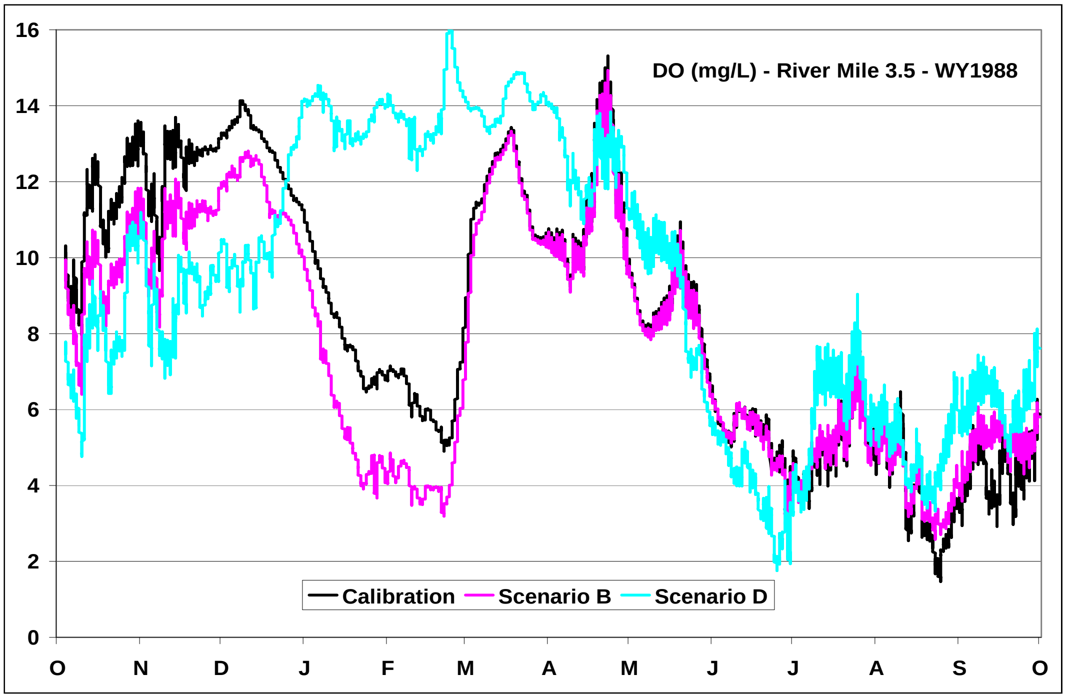


Figure 97. Scenario D results for DO at RM 3.5, 1988.

might explain the difference. W2 simulates ice formation and results show ice cover during the period January-March 1988. Ice cover affects reaeration rates.

Figure 98 shows the results for NH4 concentrations, which are similar to the results for Scenario B during the critical summer period of July-September. However, annual and summer NH4 loads are higher at Jordan in Scenario D than in the calibration (Table 47), which results in higher NH4 concentrations at RM 3.5 particularly in the winter and June. Concentrations even exceed Scenario B results in December and June. To evaluate toxicity, un-ionized NH4 concentrations would need to be estimated using temperature and pH. Benthic NH4 release rates were not adjusted in Scenario D.

PO4 load reductions at Jordan and the two WWTPs combine to reduce PO4 concentrations at RM 3.5, with the exception of elevated levels in November and December in response to loads at Jordan (Figure 99). The decrease is especially apparent in late August through September. Lower summer PO4 concentrations in Scenario D do not translate to lower summer CHLA concentrations (Figure 100) despite lower loads of phytoplankton biomass at Jordan (Table 47). The model actually predicts higher CHLA concentrations at RM 3.5 in July-September under Scenario D. Lower CHLA concentrations are predicted for early summer and fall.

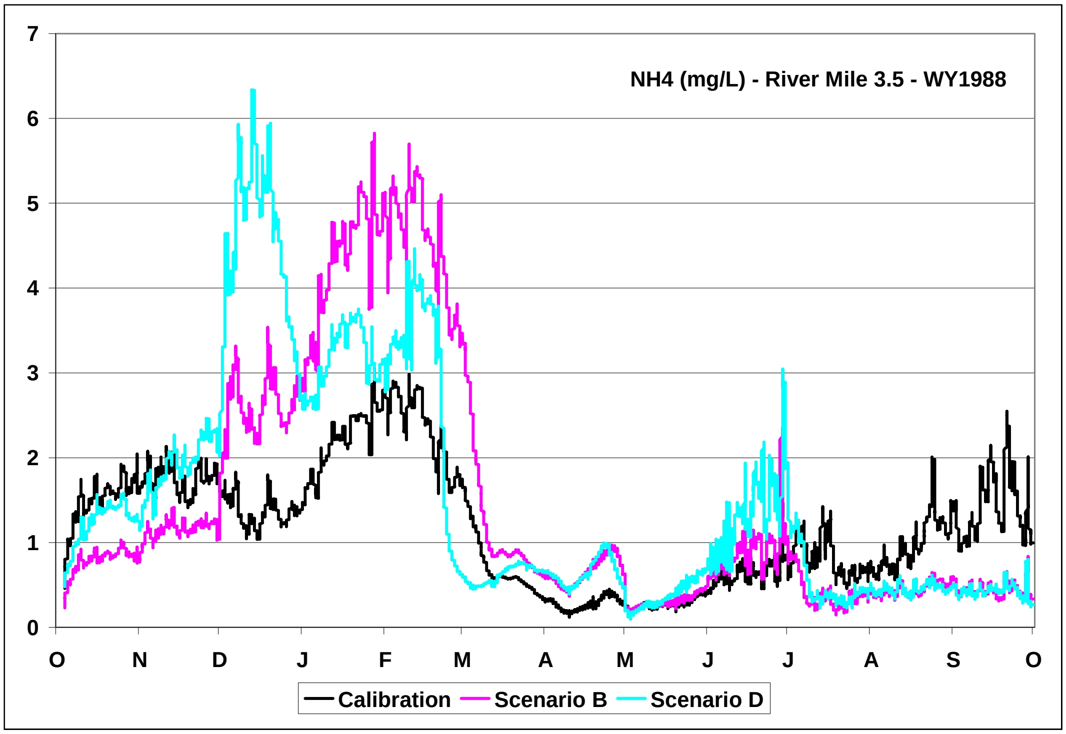


Figure 98. Scenario D results for NH4 at RM 3.5, 1988.

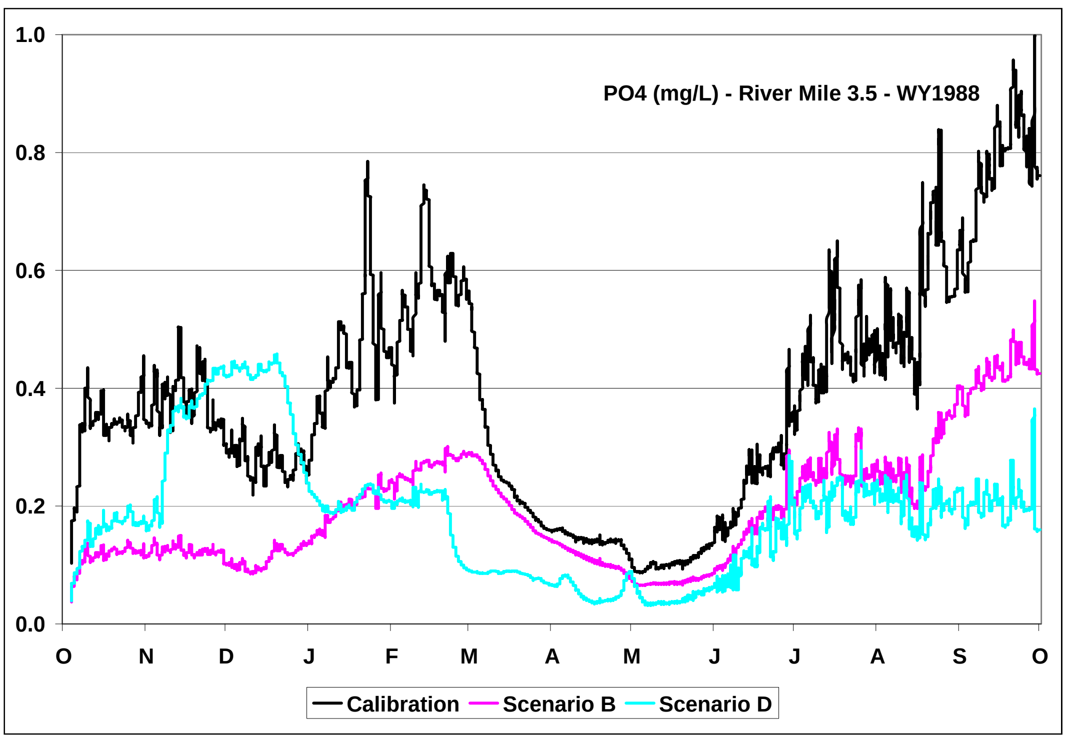


Figure 99. Scenario D results for PO4 at RM 3.5, 1988.

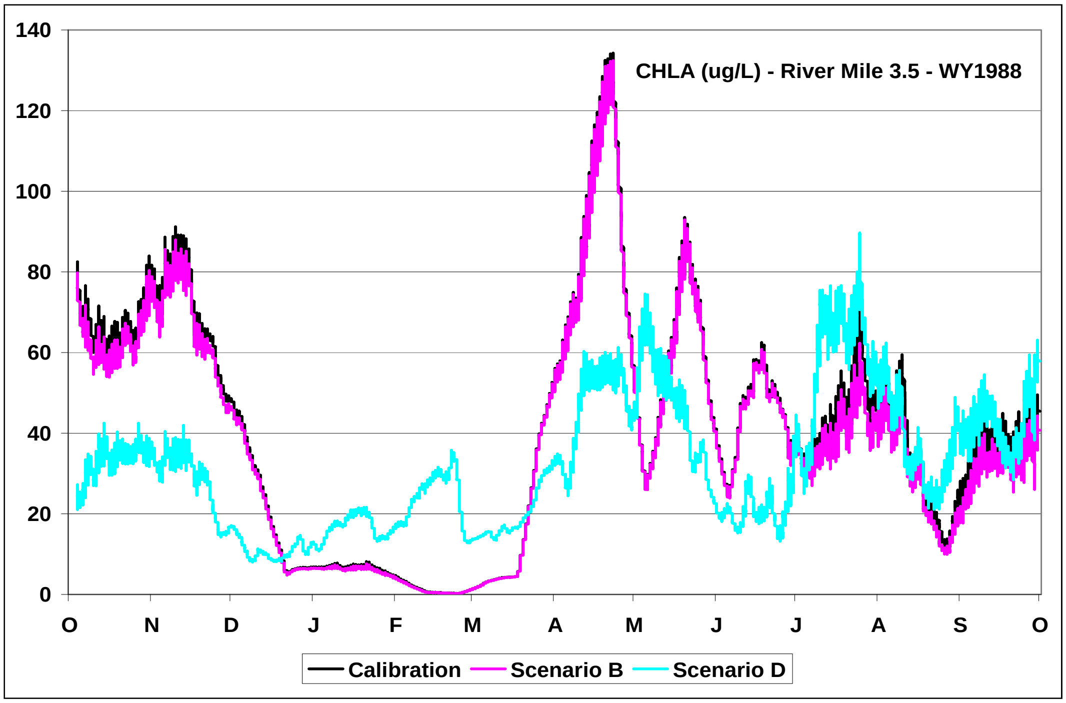


Figure 100. Scenario D results for CHLA at RM 3.5, 1988.

The phytoplankton response in Scenario D warrants further exploration. Phytoplankton growth is dependent on flow, temperature, light, and nutrients. The loading rates of suspended solids, nutrients, and also phytoplankton biomass were changed in Scenario D; all of these changes may affect CHLA concentrations at RM 3.5. Decreased PO4 and ALG loads may have been offset by increased light due to fewer suspended solids. Note that the CHLA concentrations at RM 3.5 are much higher during July and late August/early September in Scenario D than in the calibration and Scenario B.

Figure 101 shows the results for TSS concentrations at RM 3.5 in Scenario D compared to the calibration and Scenario B results, and Figure 102 shows the results for estimated turbidity. TSS is a derived variable that the model calculates from results for the state variables ISS, LPOM, RPOM, and ALG1-3. Organic (volatile) suspended solids (VSS) are represented by LPOM, RPOM, and ALG1-3. Turbidity was estimated from the model results for ISS and VSS using a formula provided by Dr. Robert Megard[1] and based on measured data from the lower Minnesota River:

$$\text{Turbidity (NTU)} = 0.80*\text{VSS} + 0.46*\text{ISS (mg/L)}$$

[1] Personal Communication. 2008. Dr. Robert O. Megard, Professor Emeritus, Department of Ecology, Evolution, and Behavior, University of Minnesota, St. Paul, MN.

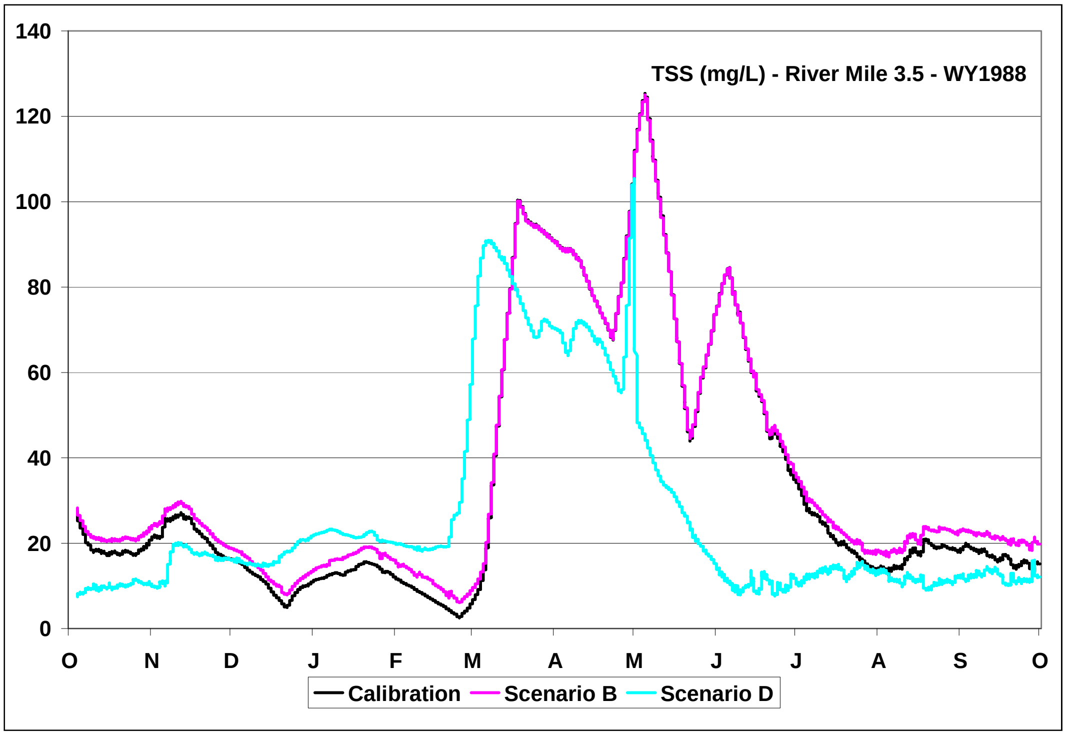


Figure 101. Scenario D results for TSS at RM 3.5, 1988.

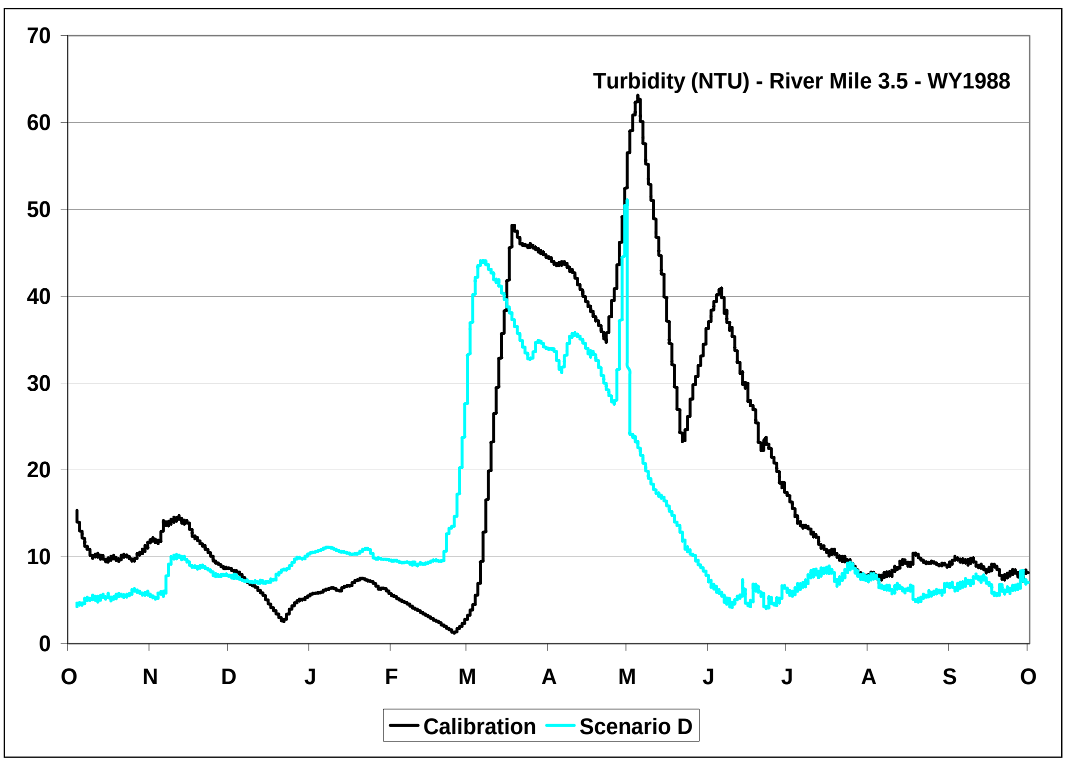


Figure 102. Scenario D results for Turbidity at RM 3.5, 1988.

Light extinction coefficients in the Lower Minnesota River Model are also based on Dr. Megard's work.

With the exception of winter, TSS concentrations at RM 3.5 are lower in Scenario D than in the calibration and Scenario B (Figure 101) due to decreased TSS loads at Jordan (Table 47). The timing and magnitude of peak concentrations change somewhat in Scenario D due to changes in the timing and magnitude of flow at Jordan predicted by the HSPF model. Turbidity results closely mirror the TSS results. Note the decreased turbidity and TSS in July and late August/early September, which correspond to periods of increased CHLA concentrations in Scenario D (Figure 100). Increased light offers a possible explanation for the increased phytoplankton levels at RM 3.5 in Scenario D, but there may be other factors. For example, the downstream boundary was not adjusted in any of the scenarios to account for water-quality changes in Pool 2 of the Mississippi River, and backwashing from Pool 2 can affect water quality in the Minnesota River, especially in the lower reach at low flows.

CE-QUAL-W2 provides the option to output selected advective, diffusive, and kinetics fluxes in order to evaluate their influence on phytoplankton biomass, DO, and other state variables. This option was not explored in this application but is available for future applications.

Application summary

Various loading scenarios were applied to the CE-QUAL-W2 model of the lower Minnesota River to demonstrate its potential use in load allocation studies, facility and watershed planning, and other applications. The four scenarios were designed around current NPDES permit limitations and approved WLA and TMDL studies:

- Scenario A: Set the Blue Lake and Seneca WWTPs to their maximum permitted limits.
- Scenario B: Use the effluent concentrations in Scenario A, but change the effluent flows to average annual and apply aeration at Seneca as in the WLA study (MPCA 1985).
- Scenario C: Use the Scenario B settings, but reduce SOD rates to those applied in the WLA study to meet DO standards in the future.
- Scenario D: Use the Scenario B settings, but reduce BOD loadings at Jordan by applying the results of the HSPF model used in the DO TMDL study (MPCA 2004).

Scenario A was applied to the CE-QUAL-W2 models of water years 1988, 2001, 2003, and 2006 because river flows decreased below 2,000 cfs during the summer. Scenarios B-D were applied to the model of 1988 because summer flows were near the $7Q_{10}$ statistic used to determine BOD load allocations.

Figure 103 summarizes the results for Scenarios B, C, and D in a longitudinal plot of average DO concentrations from RM 36 to the mouth under summer low-flow conditions in August and September 1988. These two months were the focus of management scenarios run for the DO TMDL study. As in the WLA study, the model predicts that DO concentrations will fall below 5 mg/L with BOD reductions only at the Blue Lake and Seneca WWTPs. BOD reductions at Jordan and associated reductions in SOD rates are also needed to meet DO standards under summer low-flow conditions. Agreement with the WLA study provides additional confidence in the model's usefulness in future load allocation studies and other applications. Scenario D demonstrated the ability to translate and transfer results from another model into the CE-QUAL-W2 model for use in management decisions. Output from the CE-QUAL-W2 model may also be translated and input to a Mississippi River model.

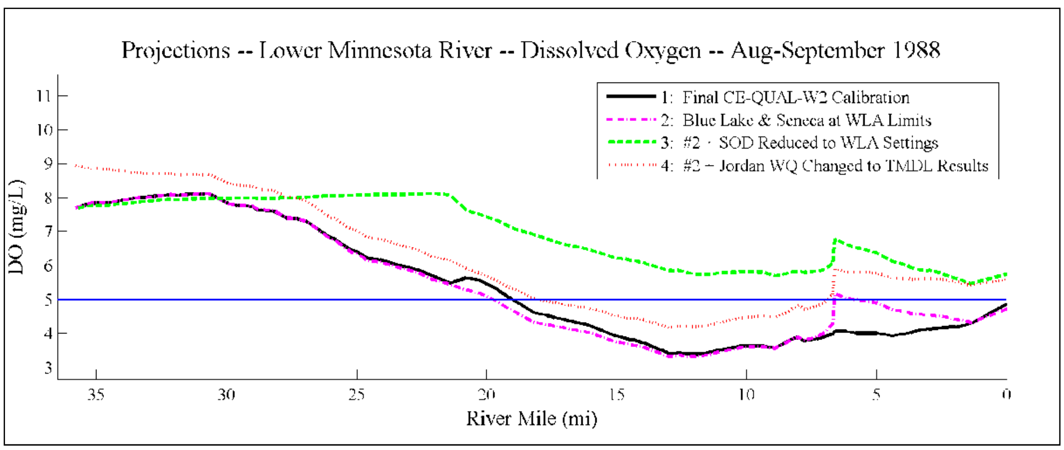


Figure 103. Scenario B-D results for DO, RM 36-0, August-September 1988.

Recommendations for future applications include the following:

- Conduct additional sensitivity analyses on model settings that may be important. For example, the model sensitivity to SOD rates was demonstrated in this application.

- Use the tools provided by the CE-QUAL-W2 model, such as the ability to output advective, diffusive, and kinetic fluxes, to evaluate how they influence water quality.
- Consider whether additional features are needed in the CE-QUAL-W2 model for studies of the lower Minnesota River and recommend enhancement to the model developers. Sediment transport, sediment diagenesis, phosphorus sorption, and variable algal stoichiometry were noted as potentially useful in this project.
- Consider applying and testing additional state variables in the model. For example, multiple ISS groups would have been helpful in the HSPF-to-W2 linkage, and the use of OM instead of CBOD groups for the discharges is worth exploring. Additional monitoring data may be needed to support new model variables.
- When linking to another model, be aware of differences in the state variables, coefficients, and capabilities.

The results of the calibration and application of the CE-QUAL-W2 model of the lower Minnesota River show that it is an acceptable tool for studying dissolved oxygen, nutrient, phytoplankton, and turbidity levels in the lower Minnesota River under a variety of conditions.

7 Summary and Conclusions

The seven water years modeled (1988, 2001-2006) provided a wide range of hydrologic variability. The fact that one calibration was developed that captures the trends in water quality suggests that this is a useful model for scenario analysis. Over the seven-year span, flows were high enough that the natural levees of the Minnesota River were overtopped (2001) and low enough that the 7Q10 flow was realized (1988). Higher or lower flows are possible but statistically rare. Thus, confidence in the model's ability to account for water quality impacts related to the hydrograph is high.

Across all years the model captured the quantitative and qualitative trends in all modeled parameters. With rare exceptions, the statistical measures of model performance were excellent and better than many other studies (Arhonditsis and Brett 2004). Qualitatively, trends were consistent with measured data. This is noteworthy because the model performance statistics were paired temporally and spatially closely with the measured data. Temporal comparisons between model output and measured data were made within 0.02 day or approximately 28 minutes. Spatially, all measured data were compared to the surface layer (0.4 m depth) and to the nearest model segment over the river length (approximately 0.2 mile). This is conservative but results in more certainty about the model statistics.

A significant trend measurable because of the five data collection stations in the model domain was a linear decrease in model performance from the upstream boundary (near Jordan) to the downstream boundary (Mississippi River). Nearly every modeled constituent when compared to data tended to become less comparable to the data as distance from Jordan increased. This is not surprising because as distance from the upstream boundary increases, the river biogeochemical environment becomes more complex, and uncertainty with model travel time estimates contributes more error to the computed mass-balance.

Linear models of measured to modeled data highlight (Chapter 4) the decrease in model performance with distance from Jordan. Close to Jordan, most linear models have an intercept of zero and a slope of one with narrow confidence intervals. Further downstream, the linear models may still have an excellent fit, but the intercept may no longer be zero and the slope may be less than one. A model slope less than one with a good fit suggests that factors other than the modeled dependent variable are contributing to the measured variable. A good fit is a consequence of the calibration and because there is no longer a one-to-one relationship between measured and modeled data, interpretation of model results requires caution. However, this condition is common in many models. All linear models and supporting statistical information have been provided.

The calibration process highlighted several general factors that contribute to water quality modeling challenges in the Minnesota River that reflect variable hydrology, downstream boundary conditions in the Mississippi River and operation of the Black Dog Generating Plant.

Variable hydrology

River discharge is a dominant driver of water quality for the majority of the year. When flows are above approximately 50 m^3/s, constituent transport dominates water quality. As flows decline below this point, travel times are reduced and oxygen sinks (organic matter, algal respiration, sediment oxygen demand) are able to act over a longer time period.

Black Dog Generating Plant

Black Dog Generating Plant withdraws water from the Minnesota River. As a percentage of river flow, the Black Dog withdrawal is routinely 50% and can be as high as 300%. Thus, a significant amount of the river is cycled through Black Dog Lake. The lake has short retention times and limited exposure to the cycling water. Data for the Black Dog Generating Plant operations were limited to daily or hourly estimates of flow and temperature in all years but only 15 samples of water quality during low flow periods in 2005 and 2006. Data were insufficient to model Black Dog Lake or its effects on water quality. However, the limited data indicate that there can be differences between river water quality and cooling-pond water quality.

Black Dog Generating Plant can also influence Minnesota River water quality when withdrawal rates are greater than river flow because the water supply is then supplemented by the Mississippi River. When this happens, the Mississippi River can be drawn upstream and mix with the Minnesota River. Because the downstream boundary condition is not well characterized, upstream flow is a source of uncertainty.

Downstream boundary conditions

The downstream boundary condition allows water from the Mississippi River to move into the Minnesota River as stage changes or as Black Dog Generating Plant water withdrawal rate changes. The impact of the downstream boundary condition on model calibration is small because upstream flow was also relatively limited. However, because data were limited for the downstream boundary compared to the upstream boundary at Jordan, the impact of the downstream boundary is uncertain.

Data

Data collection on the Minnesota River may be unparalleled in terms of temporal and spatial resolution. However, the model calibration process resulted in several observations to improve the data collection in future efforts.

Continuous monitoring

Continuous data were available for flow, water surface, temperature, dissolved oxygen, and turbidity for portions of the study. High frequency flow and stage data improved the flow and stage calibration compared to a daily average flow or stage. However, the impact was small.

Continuous temperature (15-minute) data were, in comparison, important in representing the dissolved oxygen and were superior to daily average temperature. Temperature calibrations also improved when 15-minute data were used. High-frequency temperature data should be incorporated in future studies.

This study also had 15-minute dissolved oxygen data. Unlike flow, temperature, and stage, which have their own separate and respective input files, dissolved oxygen (like other constituents) is input in a common file. This requires that the input time-step be exactly the same for each water quality constituent. Thus, 15-minute frequency data for dissolved oxygen input requires that all other input constituents be input at 15-minute intervals, but since 15-minute data were not available for all constituents, estimates had to be interpolated. Not surprisingly, interpolating all of the other constituent concentrations to fit the 15-minute dissolved oxygen data decreased model performance. In addition to the increased interpolation error of the other constituents, continuous dissolved oxygen data are subject to greater measurement error than laboratory or calibrated field probe measurement. This contributed to decreased model performance when using the 15-minute data; however, this impact was not quantified.

Currently, the W2 model requires that the user develop one constituent concentration input file that contains a Julian date and a corresponding value for each modeled constituent. For this reason, it was not practical to use the continuous DO data because all of the other water quality constituent data would have needed to be linearly interpolated. Given that real time measurement of dissolved oxygen and other constituents is becoming increasingly more common, the CE-QUAL-W2 model should be modified to support separate input files for water-quality variables measured at different frequencies. This would eliminate the need for interpolation.

Organic matter

Organic matter was a critical aspect of producing a good simulation of dissolved oxygen and ammonia. In the model, organic matter is input at the upstream boundary, tributaries, and point sources. Organic matter is specified as labile dissolved and particulate, and refractory dissolved and particulate in CE-QUAL-W2. Data from other constituents had to be used to develop the required CE-QUAL-W2 inputs (see p. 34). An approach was developed to accomplish this. However, assumptions had to be made about the proportions of labile and refractory organic matter. Data to develop these splits would have aided the calibration procedure. In effect, because there was no information on this split, literature values were used (LimnoTech 2007, 2008, 2009; Kim et al. 2006), but this had an unknown uncertainty associated with it. This aspect of the model may be considered a "tunable" state variable (Arhonditsis and Brett 2004), and, as such, additional data to support better parameterization are warranted.

Algae and chlorophyll *a*

The algae:chlorophyll *a* ratio (ACHLA) was used as a calibration parameter due to the lack of measured algal data for the historical years. This calibration parameter is specified once in the control file and is not spatially or time varying in the model. Reducing this parameter from 0.135 to 0.0675 mg algae/ug chla greatly helped to improve the model calibration (see p. 7). Chlorophyll *a* data were available for the historical years, so this ratio was applied to the CHLA data to produce total algal biomass. As discussed in Chapter 3, the specific algal groups were then calculated based on suggested splits from Table 12. Due to the temporal variation of the algal: chlorophyll *a* ratio, the original value was reduced until a better model-to-data fit was achieved.

Additional modeling

Barge movement

Barge traffic is a potential complicating factor to water quality in the Minnesota River, but the magnitude of impact is unknown. Barge traffic can suspend sediment and alter mixing, which in turn may influence dissolved oxygen, ammonia, and nutrients. The impact is difficult to quantify because barge traffic is transitory. The measured water quality data and calibrated model reflect the influence of barge movement, but the magnitude of the impact cannot be separated from the measured data or the model.

The USACE-ERDC has a barge movement algorithm operating with the 2D depth average finite element Adaptive Hydraulics (ADH) code that has potential to assist with addressing the influence of barge movement. The barge movement model allows specification of barge and tow type, size, speed, and travel path. Barge movement and water quality are coupled. The model is available but should be considered under development. In addition, data to support model validation have not been collected beyond a few measurements.

Navigation channel

The navigation channel extends from Savage downstream to the river mouth. The channel may be related to river dissolved oxygen limitations. Because of the 2D lateral averaging in CE-QUAL-W2, the local bathymetric change associated with the navigation channel is eliminated. A different model (2D depth averaged using ADH) or even a 3D application may be required to estimate the contribution of the navigation channel to dissolved oxygen limitations. Existing water quality data would be adequate to calibrate 1) a new model with the navigation channel, and then 2) a scenario with the navigation channel absent.

Post Black Dog Generating Plant modifications

The impact of the Black Dog Generating Plant on the model has been previously discussed. Plans to modify the cooling water ponds to increase retention time to allow more time for cooling will also alter the water volume removed and returned to the river for cooling. The reduced volume of water being recirculated through the cooling water ponds will simplify modeling the Minnesota River because the potential impact to the model mass-balance is reduced. However, the increased retention time in the ponds will also change the water quality. This potentially increases the data needed to represent the outflows.

Near-time forecasting

Recent advances in data fusion, network communication, and sensor technology suggest that it is now possible to model in near-time. Because of the excellent data set available on the Minnesota River, it is probable that good statistical models that relate flow to various water quality parameters could be developed that would allow parameterization of input files for CE-QUAL-W2 or other models. This would allow the model to be used as a prediction tool to forecast dissolved oxygen and other water quality constituents. This might have an immediate practical application to the decision to oxygenate effluent at treatment plants.

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Appendix A: Defining Organic Matter

W2 background on BOD groups and ultimate BOD analyses

The CE-QUAL-W2 model allows any number of different CBOD groups that can be assigned individual U:5 ratios and decay rates. The default CBODU:CBOD5 ratio is 1.85 and the default CBOD5 decay rate is 0.1/day. The ratio and rate can be varied by CBOD group (that is, source), but they cannot vary over time.

Initially, six CBOD groups were defined in the W2 input files:

- CBOD1 Minnesota River at mile 39.4 (Jordan)
- CBOD2 Blue Lake WWTP
- CBOD3 Seneca WWTP
- CBOD4 All airport stormwater outfalls
- CBOD5 All tributaries
- CBOD6 Minnesota River at mile 3.5 (Fort Snelling)

Later, in the final calibration, all CBOD defined for the Minnesota River and tributaries were shifted to nonliving organic matter or phytoplankton biomass. Values for the CBOD1, CBOD5, and CBOD6 groups were set to zero, while the values for the CBOD2-CBOD4 groups were converted from 5-day to ultimate CBOD.

As part of the enhanced monitoring program in WY 2004-2006, MCES conducted 5-day and 70-day BOD tests on samples collected from the river, tributaries, and discharges. Total and carbonaceous BOD tests were run on unfiltered and filtered samples. MCES runs the ultimate BOD test for 70 days. The number of days was based on long-term tests conducted in the late 1980s to determine when oxygen demand plateaus.

Table A1 summarizes U:5 ratios and linear regressions from unfiltered samples. Table A2 summarizes the bottle CBOD decay rates from all samples. In general, median values were applied in the model.

Table A1. Ultimate to 5-day results for unfiltered CBOD tests.

	CBODU	U:5 Ratio				Linear Thru 0	
Site	N	Mean	Median	Min	Max	Coef	R^2
MI 39.4	20	3.3	3.2	2.6	4.5	3.0	0.84
MI 3.5	29	3.1	2.9	2.0	4.2	3.0	0.65
Blue Lake	13	7.5	7.4	4.5	12	6.9	0.35
Seneca	14	10.2	9.5	5.5	23	9.1	-0.03
Airport 020	9	5.2	3.8	1.4	14	8.9	0.64
Airport 020[1]	19	3.7	2.5	1.0	17	8.1	0.53
Airport 030[1]	19	11.6	7.1	2.4	62	13	0.68
Airport 040[1]	12	5.9	5.3	1.5	12	4.8	0.99
Tributaries	25	4.7	4.6	2.7	8.8	4.1	0.64

[1] Source: Metropolitan Airports Commission, 2001-2004.

Table A2. Bottle decay rates for CBODU tests.

	Bottle Decay Rates (/day, base e) for CBODU Tests							
	Unfiltered Samples				Filtered Samples			
Site	Mean	Median	Min	Max	Mean	Median	Min	Max
MI 39.4	0.0604	0.0610	0.0330	0.0929	0.0339	0.0345	0.0146	0.0553
MI 3.5	0.0530	0.0522	0.0315	0.0768	0.0323	0.0315	0.0138	0.0606
Blue Lake	0.0339	0.0322	0.0196	0.0622	0.0295	0.0272	0.0177	0.0461
Seneca	0.0341	0.0294	0.0219	0.0729	0.0279	0.0249	0.0100	0.0660
Airport 020	0.0482	0.0495	0.0238	0.0698	0.0398	0.0384	0.0184	0.0576
Tributaries	0.0354	0.0347	0.0251	0.0437	0.0263	0.0257	0.0234	0.0324

Determining organic matter inputs

Since dissolved organic carbon (DOC), volatile suspended solids (VSS), and algae were monitored by MCES, dissolved organic matter (DOM) can be estimated as:

$$DOM = \frac{DOC}{\delta_C} \tag{1}$$

where DOC is the measured DOC concentration in mg/L and $\delta_C = 0.45$, carbon-organic matter ratio (specified in the control file as ORGC). Once

DOM is estimated, it can be assumed that VSS ~ POM (particulate organic matter) in the system, so total organic matter (TOM) can be estimated.

$$TOM = DOM + POM \approx DOM + VSS \tag{2}$$

Algae are tracked separately in the LMRM, so care must be taken not to double count it in the organic matter budget. The algal contribution must be calculated and subtracted from the TOM in the system:

$$NA\_TOM = TOM - (ALGBIOMASS) \tag{3}$$

Once the non-algal portion of TOM (NA_TOM) is known, the non-algal portion of particulate organic matter (NA_POM) can be estimated as:

$$NA\_POM = NA\_TOM - DOM \tag{4}$$

Finally, a 15%:85% Labile:Refractory split is assumed for the organic matter and the following can be calculated:

$$LDOM = 0.15 * DOM \tag{5}$$

$$RDOM = (1 - 0.15) * DOM \tag{6}$$

$$LPOM = 0.15 * NA\_POM \tag{7}$$

$$RPOM = (1 - 0.15) * NA\_POM \tag{8}$$

Note: If, NA_POM < 0, it is assumed that LPOM, RPOM = 0.10 mg/L per recommendation from Chris Berger (PSU).

Back-calculating BOD_5 for model verification

Once the model results are output, those results must be compared with actual measured data. BOD5 is back-calculated based on model output as:

$$BOD5 = \frac{BOD2}{U:5_2} + \frac{BOD3}{U:5_3} + \frac{BOD4}{U:5_4} + \frac{(AOD + OMOD)}{U:5_{RIVER}} \tag{9}$$

where $U:5_x$ is the ultimate:5-day ratio as reported in Table A3, AOD is the algal biochemical oxygen demand in mg/L, calculated as:

$$AOD = \delta_O * (ALGBIOMASS) \tag{10}$$

Table A3. U:5 ratios used in the LMRM.

BOD group	Site	U:5 Ratios	
		2001-2006	1988
BOD_River	Approximate Mean from River Data	4.5	4.5
BOD2	Blue Lake (Chaska & Savage in 1988)	7.4	4.0
BOD3	Seneca	9.5	3.5
BOD4	Airport Stormwater Outfalls	4.96	NA

where δ_O = 1.4; ratio of O_2 consumed (g), per OM (g). OMOD is the biochemical oxygen demand due to the organic matter in mg/L, calculated as:

$$OMOD = \delta_O * (LDOM + LPOM + 0.15 * (RDOM + RPOM)) \qquad (11)$$

where δ_O = 1.4; ratio of O_2 consumed (g), per OM (g).

Appendix B: External Peer Review

This appendix contains the external peer review of the LMRM Project, followed by the modelers' comments. The review was conducted by Dr. Wu-Seng Lung. Dr. Lung is a professor in the Civil and Environmental Engineering Department at the University of Virginia and currently serves as the Assistant Chair and Director of the Graduate Program. At the end of his peer review is a section titled 'The Reviewer' in which Dr. Lung explains his areas of expertise.

Peer Review Memo: Dr. Wu-Seng Lung

Review Summary of CE-QUAL-W2 Modeling of the Minnesota River

Introduction and Purpose

A review of the modeling analysis of the Minnesota River was requested jointly by the Metropolitan Council and Army Engineer Research and Development Center (AERDC). The review is designed to address a number of questions:

1. Is this model adequately calibrated for use in load allocation studies for BOD/DO and ammonia?
2. Is the model adequately calibrated for use in load allocation studies for nutrients and turbidity?
3. Any recommended future work?

Model results were made available in Excel files from the AERDC ftp site. They are in the form of time-series plots and one-to-one statistics of model results vs. data for 1988 and 2001-2006. A series of email communications between the reviewer and the modeling team members were made from March through May to discuss and clarify the model results. This brief report summarizes the findings and recommendations following the review.

Technical Review of Model Results

The CE-QUAL-W2 code was configured in a 2-D fashion for the Minnesota River from Jordan to the junction with the Upper Mississippi River. This portion of the river is divided into 88 longitudinal segments and each segment is further sliced into multiple 1-m layers. Substantial field data was used to support the modeling analysis. The Minnesota River model was first calibrated using the 1988 (a low flow year) data for a whole year simulation. Then the model was run time variably from October 1, 2001 to September 30, 2006 for continuous simulations, yielding a very robust model calibration and verification analysis.

The Minnesota River model is found to be well calibrated using the data of 1988 and 2001-2006. Mass transport modeling of the Minnesota River was performed by reproducing total dissolved solids levels in the water column from 2001 to 2006 (e.g. model results at RM3.5 in Figure 1 below). Subsequently, total suspended solids concentrations are also reproduced by the model.

The model with calibrated mass transport was then used to perform water quality simulations. Again, the model was run for 1988 and for a 6-year period from October 1, 2001 to September 30, 2006. A number of key water quality kinetic coefficients and parameters were examined to scrutinize the model results:

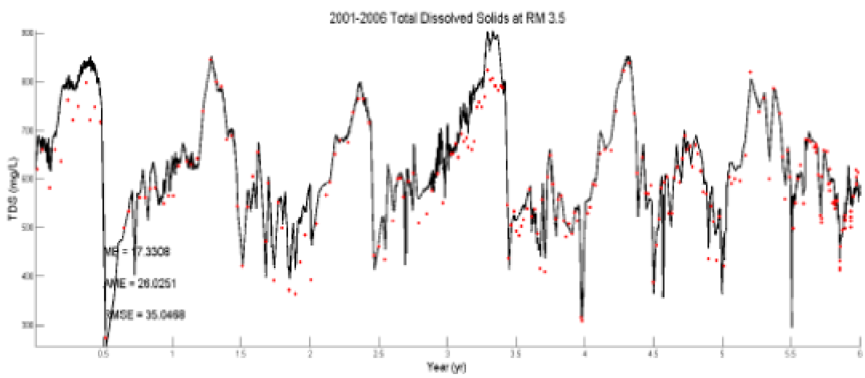


Figure 1. Time-Variable Modeling Using Total Dissolved Solids as a Conservative Tracer (2002-2006)

1. Continuous time-variable water quality simulations
2. CBOD deoxygenation coefficient in the water column
3. Sediment oxygen demand assigned in model input
4. Diurnal DO fluctuation

Continuous Model Runs

While the 1988 run is a stand-alone simulation, the other model runs from 2001 to 2006 were conducted on an individual year basis. That is, the initial conditions on October 1 during this 6-year period were reset based on the field data collected closest to day 275. Subsequent tests were made to perform this 6-year run on a continuous basis and the new results are very close to the original model results (email between Lung and Smith on May 5, 2009). Such a test is critical to insure the model integrity. The outcome indicates that the model runs passed this test.

CBOD Deoxygenation Coefficient in Water Column

This is a key model kinetic coefficient in BOD/DO modeling of the Minnesota River and is almost always derived from model calibration. The difficulty of deriving this coefficient value is that it reflects the wastewater discharge characteristics, which could vary from time to time following treatment plant upgrades. A good example is the case of the Upper Mississippi River. Lung (1996) reported incremental decreasing of this coefficient value in the Upper Mississippi River following the treatment upgrades at the Metro Plant from primary treatment, secondary, to advanced secondary (secondary with nitrification). Compounding this issue is that while the model simulates $CBOD_u$, its results must be translated back to $CBOD_5$ for comparison with the field data in model calibration. In addition, CBOD loads from wastewater treatment plants are measured and reported as $CBOD_5$, which must be converted to $CBOD_u$ for model input. As such, the ratio of $CBOD_u$ to $CBOD_5$ of loads and ambient water is crucial to this exercise. Table 1 lists the ratios used in the model, resulting in spatially variable CBOD deoxygenation rates in a range between 0.0257 day^{-1} and 0.085^{-1} day.

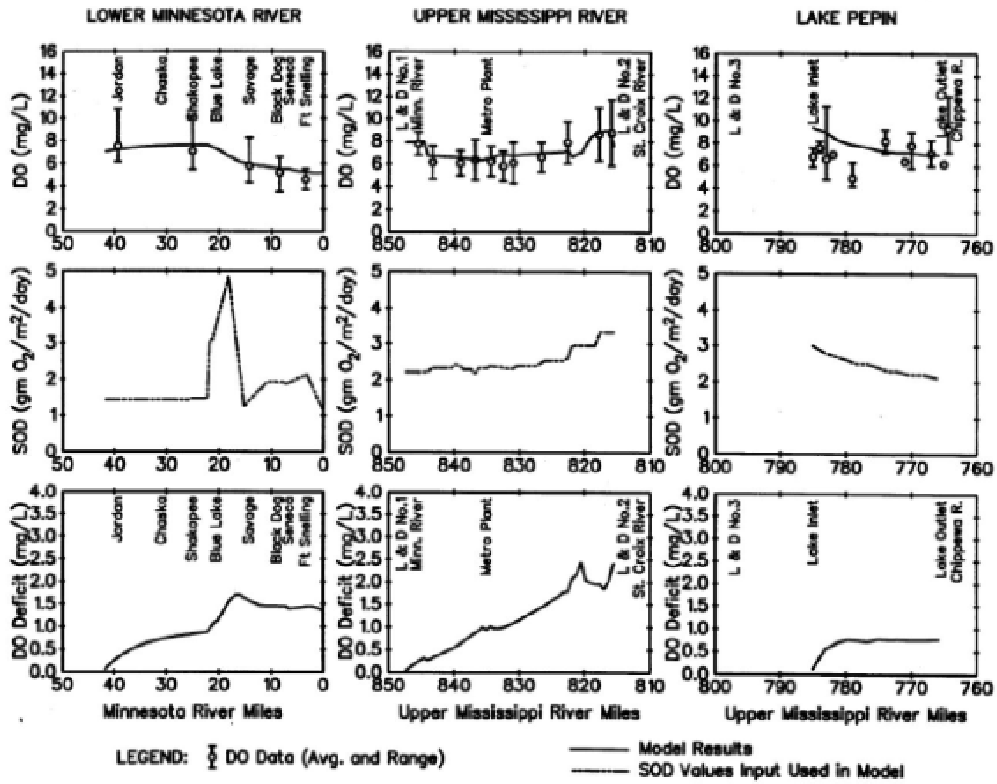
Table 1. $CBOD_u$ to $CBOD_5$ Ratios Used in the Minnesota River Model

Site	N	Mean	Median	Min	Max	Coef	R2
MI 39.4	20	3.3	3.2	2.6	4.5	3.0	0.84
MI 3.5	29	3.1	2.9	2.0	4.2	3.0	0.65
Blue Lake	13	7.5	7.4	4.5	12	6.9	0.35
Seneca	14	10.2	9.5	5.5	23	9.1	-0.03
Airport 020	9	5.2	3.8	1.4	14	8.9	0.64
Airport 020*	19	3.7	2.5	1.0	17	8.1	0.53
Airport 030*	19	11.6	7.1	2.4	62	13	0.68
Airport 040*	12	5.9	5.3	1.5	12	4.8	0.99
Tributaries	25	4.7	4.6	2.7	8.8	4.1	0.64

In an earlier steady-state modeling study of the Minnesota River, Lung calibrated the CBOD deoxygenation rates of 0.11 day^{-1} and 0.073 day^{-1} for 1980 and 1988 model calibration, respectively. Comparing these rates with those used in this current modeling analysis indicates that they are close and consistent. The rates also suggest that the CBOD currently existing is mainly of refractory nature, thereby yielding low rates and high $CBOD_u$ to $CBOD_5$ ratios. It also indicates that the model is not sensitive to this kinetic coefficient.

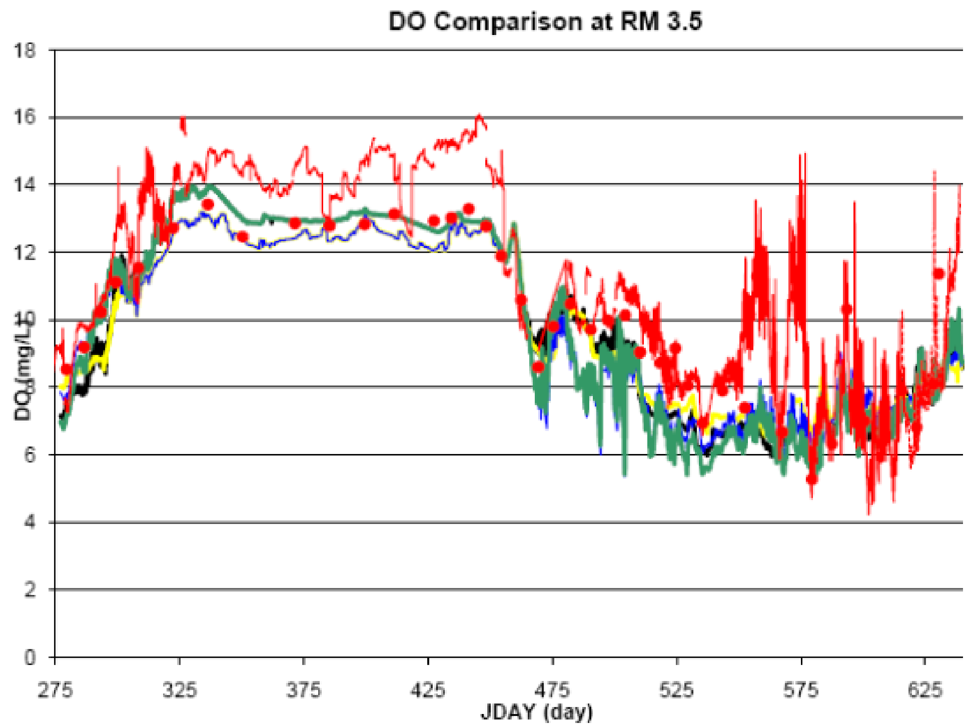
Sediment Oxygen Demand

The SOD values used in the current model are found to be comparable with those used by Lung (2001) shown in the following figure for the Minnesota River, Upper Mississippi River and Lake Pepin. The difficulty lies in the model prediction for wasteload allocations when future values of SOD is not available *a prior*.



Diurnal DO Fluctuation Calculation

The W2 model calculates photosynthetic DO production on a daily average basis and therefore is not capable of yielding hourly DO fluctuations (as shown in the summer of 2006 at RM3.5 in the following figure). Yet, the field data clearly show significant diurnal DO fluctuations. This should not be construed as an issue as the DO standard in the lower 21 miles of the Minnesota River is 5 mg/L daily average. The field data at RM3.5 indicates only on very rare occasions that the absolute minimum DO could be lower than 5 mg/L.



Summary and Conclusions

The model calibration analysis is quite robust and technically sound, supported by comprehensive statistical analysis of the model results vs. data. While the model is ready for use in wasteload allocations studies for BOD and ammonia, it must be pointed out that one of the uncertainties in the model is assigning the SOD values in model predictions. The W2 model used in this study does not have a sediment diagenesis module directly linked with the water column processes. As such, caution must be exercised in assigning the SOD values in model predictions.

Another issue with model prediction is related to the downstream boundary conditions, which should be free of any influence from the system response in the Minnesota River. It is understood that if the downstream boundary conditions are compromised by the system response, they are not qualified to serve as boundary conditions.

Is the model ready for use in nutrient and turbidity load allocation studies? It is recommended that the W2 model be modified to treat hourly photosynthetic DO production and respiration to address the significant diurnal DO fluctuations. It is clear that the point source related BOD/DO problem has been well addressed and removed in the Minnesota River. Like many water systems, the rising problem is eutrophication/nutrient related DO problems (i.e. diurnal fluctuations).

Finally, long term CBOD tests of wastewater and ambient samples at select locations are recommended such as the results from the Upper Mississippi River and the Metro Plant in 1980's (Lung, 2001). These test results will validate the spatially variable CBOD deoxygenation rates in the river.

The Reviewer

Dr. Wu-Seng Lung received his MS degree in Hydrology/Hydraulics from the University of Minnesota in 1970 and PhD degree in Environmental and Water Resources Engineering from the University of Michigan in 1975, specializing in water quality modeling. Between 1975 and 1983, he worked at environmental consulting firms applying modeling to various water quality studies for regulatory agencies, industries, and law firms. Dr. Lung joined the Civil and Environmental Engineering Department at the University of Virginia in 1983, currently serving as the Assistant Chair and Director of Graduate Program.

He has over 35 years of experience in modeling natural water systems. At Virginia, he has been working on estuarine modeling of eutrophication and toxic substances. In 1990, he completed a modeling study for EPA on the eutrophication potential in coastal embayments in Prince William Sound, Alaska as part of the effort to clean up the contaminated beaches by spraying fertilizers and chemicals following the EXXON VALDEZ oil spill. In 1991, he was named by EPA to a review panel of model evaluation group for the EPA Chesapeake Bay Program, providing guidance to water quality modeling work on the Chesapeake Bay watershed. His work on estuarine modeling has been synthesized into a book entitled, *Water Quality Modeling: Application to Estuaries*, published by CRC Press in 1993. His recent work is eutrophication and metals modeling of the Patuxent Estuary, Maryland supported by the National Oceanic and Atmospheric Administration (NOAA). He is currently involved in a study of assessing the impact of increased release from Lake Okeechobee on the water quality of the Caloosahatchee Estuary in Florida. Dr. Lung also has extensive experience in lake and reservoir modeling, addressing eutrophication, acidification and hydrothermal problems. He completed a study for South Florida Water Management District on integrating a hydrodynamic model with the WASP/EUTRO5 model for Lake Okeechobee in September 1997. In 1998, he was appointed by the 30th Circuit Court of Michigan to serve as Court Master on the water quality issues of Lake Platte. His modeling results led to an out-of-court settlement in 2000. His modeling experience has been put together in a book entitled, Water Quality Modeling for Wasteload Allocations and TMDLs, published by John Wiley & Sons in May 2001.

He has served as consultant on water quality modeling to a number of organizations including the U.S. EPA, Metropolitan Waste Control Commission (St. Paul, MN), Environmental Research & Technology, Inc., Normandeau Associates, Inc., HydroQual, Inc., Limno-Tech, Inc., Dames and Moore, the Soap and Detergent Association, Tetra Tech, Inc., the Procter & Gamble Company, ASci Corporation, Whitman, Requardt and Associates, Montgomery Watson (Hong Kong. Taiwan), Virginia Dominion Power, and URS Corporation. He performed modeling work on rivers, reservoirs, and estuaries in Slovenia and Baltic states under NATO sponsorship from 1998 to 2003. Dr. Lung served as Associate Editor for *Journal of Environmental Engineering* from 1994 to 1998, responsible for the area of water quality modeling for the journal. He was the Editor-in-Chief for *Water Quality and Ecosystem Modelling* from 2000 to 2002. He is currently serving on the editorial board for *Journal of Hydro-Environment Research*. Since 1998, Dr. Lung has been a member of the EPA Science Advisory Board, serving on three committees: Ecological Processes and Effects (EPEC); Environmental Modeling; and Radiation.

Dr. Lung is currently leading an International Collaboration of Multidisciplinary Research under the U21 (Universitas 21) Water Future Network. The focus of the research is developing a modeling framework to track the fate and transport of EDCs (Endocrine Disrupting Chemicals) and PPCPs (Pharmaceutical and Personal Care Products) in receiving water ecosystems. He is coordinating this international research effort with colleagues of U21 institutions on four selected sites: (University of Virginia) - a freshwater stream in Virginia, USA; (University of Birmingham) - an urban river near Birmingham, UK; (University of Hong Kong) - a coastal bay in Hong Kong; and (University of Queensland) - a reservoir in Australia.

From 1992 to 1994, Dr. Lung performed a number of water quality modeling studies on the Upper Mississippi River and the Minnesota River for the Metropolitan Council:

1. A post-audit study of the advance secondary treatment at the Metro Plant – This work resulted in two publications: in Lung (1996), Post-Audit of the Upper Mississippi River BOD/DO Model. *J. Environ. Eng.*, 122(5):350-358 and Lung (1998), Trends in BOD/DO Modeling for Wasteload Allocations. *J. Environ. Eng.*, 124(10):1004-1007.
2. A time-variable mass transport modeling of the Upper Mississippi River
3. A time-variable water quality modeling the Upper Mississippi River and Lake Pepin to address phosphorus removal at the Metro Plant
4. A numerical tagging study of the Upper Mississippi River – This work is published as: Lung (1996), Fate and Transport Modeling Using a Numerical Tracer. *Water Resources Res.*, 32(1):171-178.
5. Development of a mass transport model for the Minnesota River
6. Steady-State Water Quality Modeling of the Minnesota River Using WASP/EUTRO.

Modelers' comments

At the suggestion of Dr. Lung during the review process, the ERDC performed a test run concerning the continuous model runs. Figure B1 plots the results from both the individual years and the continuous run on one chart for five constituents: temperature, chlorophyll-a, nitrate, ammonium, and dissolved oxygen. Note that the only difference between the two lines occurs at the beginning of a water year. This change has minimal impact on the final results of the model.

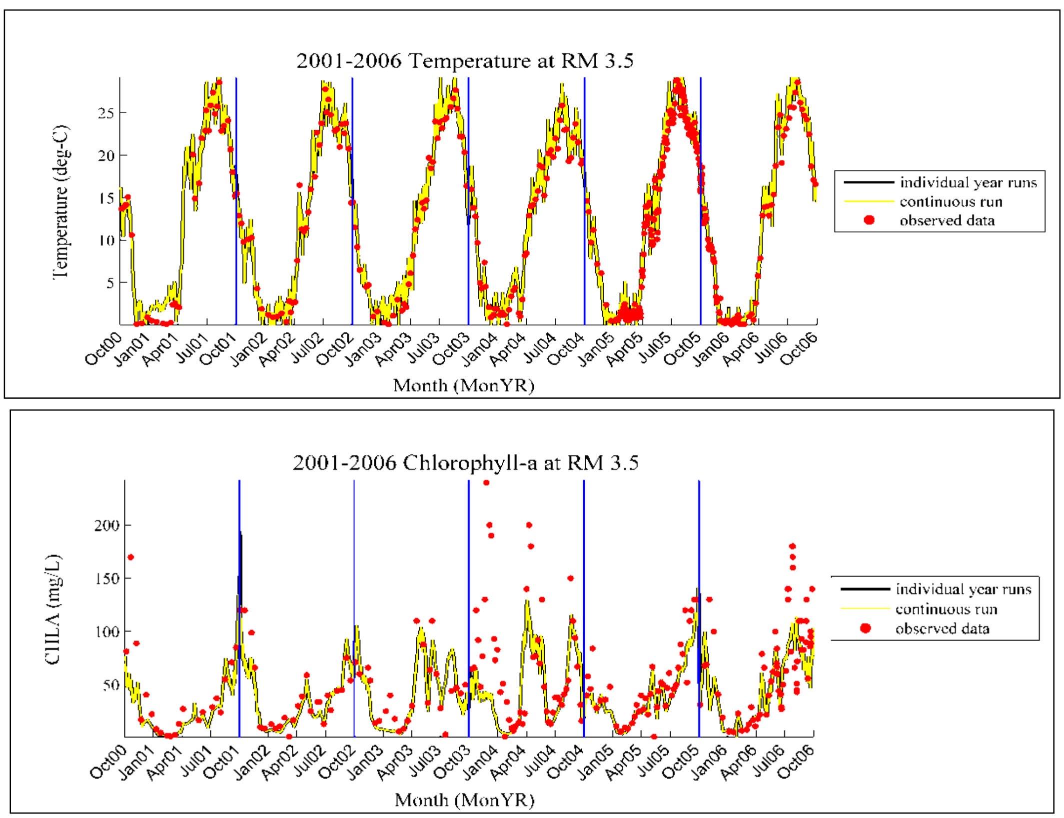


Figure B1. Continuous run vs. individual year runs for 2001-2006 (continued).

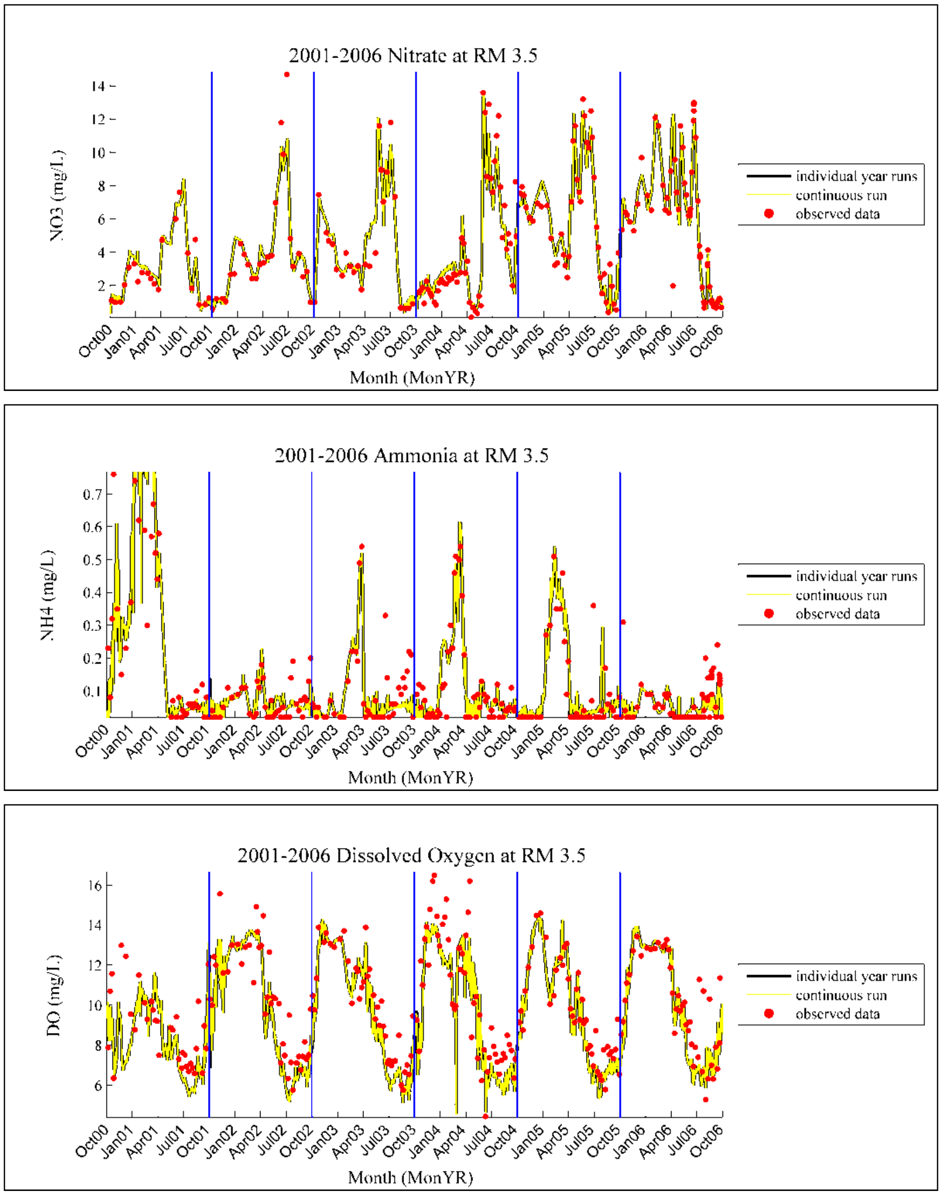


Figure B1. (concluded).

Appendix C: Dr. R.O. Megard's Research

Turbidity and Transparency of the Lower Minnesota River
December 2007
Robert O. Megard
Professor Emeritus

Department of Ecology, Evolution and Behavior
University of Minnesota
1985 Upper Buford Circle
St. Paul MN 55108

With data provided by MCES, I calculated the effects of suspended and dissolved materials on the turbidity and transparency of the Lower Minnesota River during 2006. Underwater light is attenuated (scattered and absorbed) by suspended solids (SS), which are separated analytically into organic solids (VSS) and inorganic (nonvolatile) solids (NVSS). Light also is attenuated by dissolved organic carbon (DOC).

Concentrations of NVSS in this section of the river are consistently higher than those of VSS and DOC (Figure 1). Concentrations of both NVSS and VSS increase as the total concentration of attenuators (SS + DOC) increase, although NVSS increases faster than VSS. The concentration of DOC is nearly constant. At highest concentrations, NVSS is about 10 times more concentrated than VSS and 100 times more concentrated than DOC.

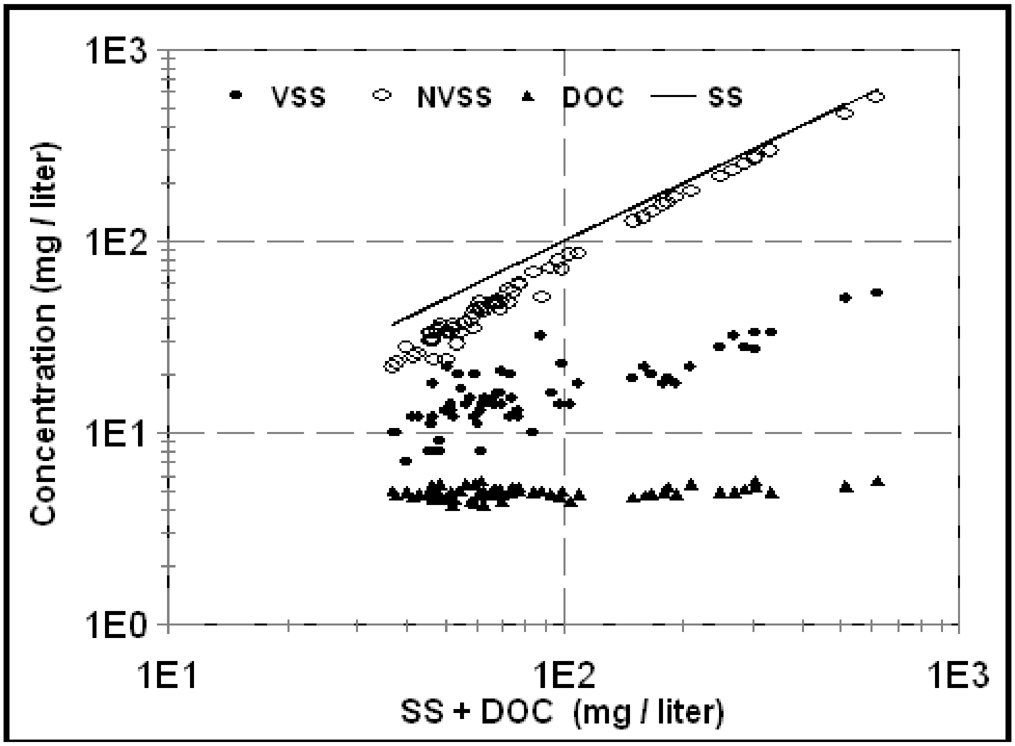


Figure 1

Turbidity

Nephelometric turbidity (TURB) depends on scattering by VSS and NVSS according to

$$\text{Turb} = 0.80(\text{VSS}) + 0.46(\text{NVSS}), \quad (1)$$

which indicates that suspended organic particles (VSS) are somewhat stronger scatterers of underwater light than suspended inorganic particles (NVSS) (Fig. 2). Dissolved organic carbon apparently has no effect on Nephelometric turbidity, which measures light scattering but not absorption.

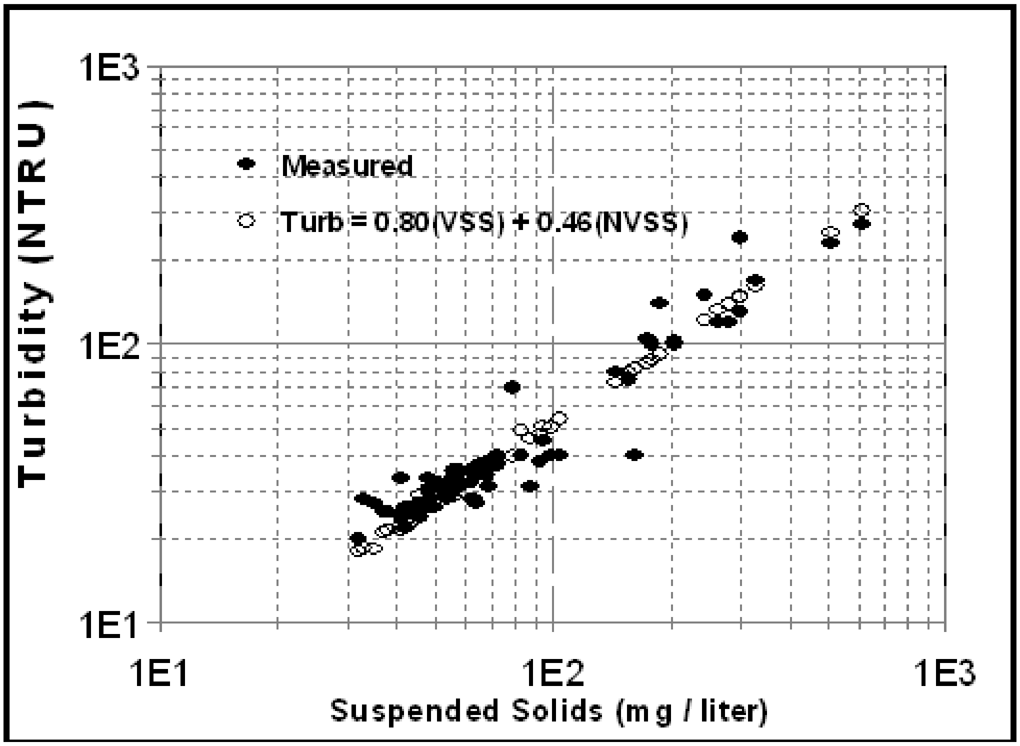


Figure 2

The scattering coefficient for VSS in the Lower Minnesota River apparently is somewhat less than in Lake Pepin (Pool 4), where it was found in an earlier analysis that

$$\text{Turb} = 1.3(\text{VSS}) + 0.6(\text{NVSS}). \quad (2)$$

Transparency

The effect of light attenuators on Secchi transparency was calculated with an equation for reciprocal transparency (1/S) in terms of SS + DOC that is shown in Figure 3. The equation indicates that the partial coefficient for VSS (0.15) is much larger than the partial coefficient for NVSS (0.01). The coefficient for DOC (0.1) also is larger then the NVSS coefficient.

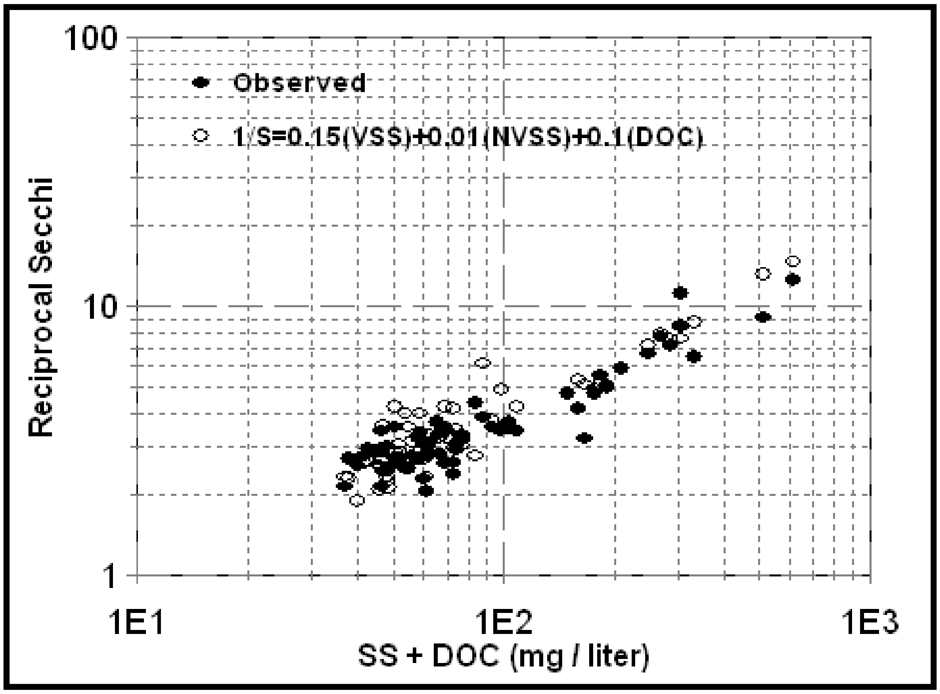


Figure 3

These values were used to predict Secchi transparency (S, measured in meters) from concentrations of VSS, NVSS, and DOC (Fig. 4). For this prediction, I assumed a numerical value of A = 1.46 for the Secchi constant, which I evaluated independently with data from the Mississippi River.

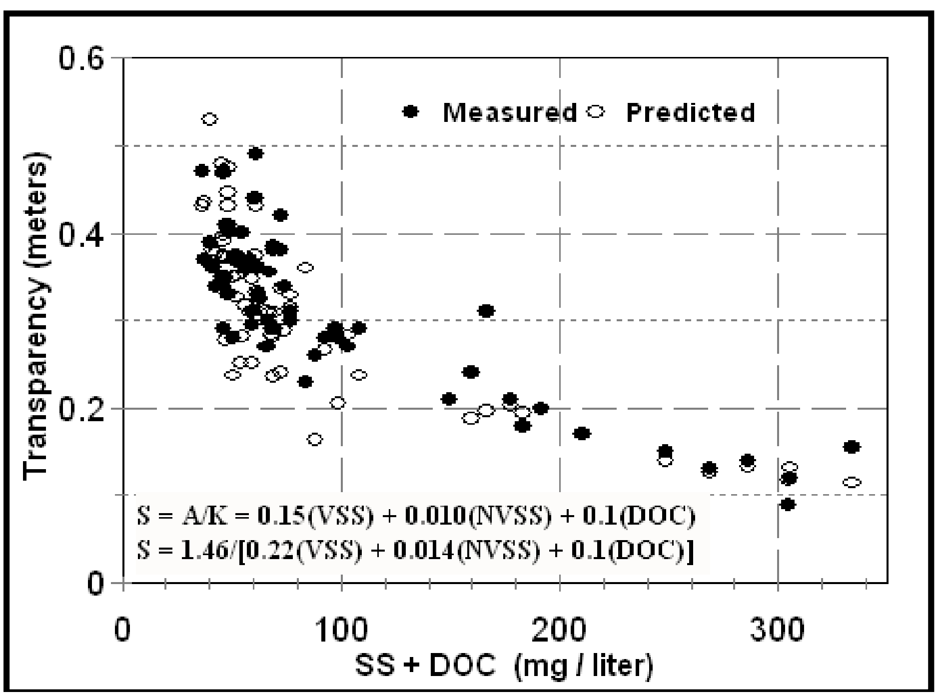


Figure 4

This value of A was used to estimate K, the attenuation coefficient for diffuse underwater light, in terms of the three attenuators: VSS, NVSS and DOC (Fig. 5).

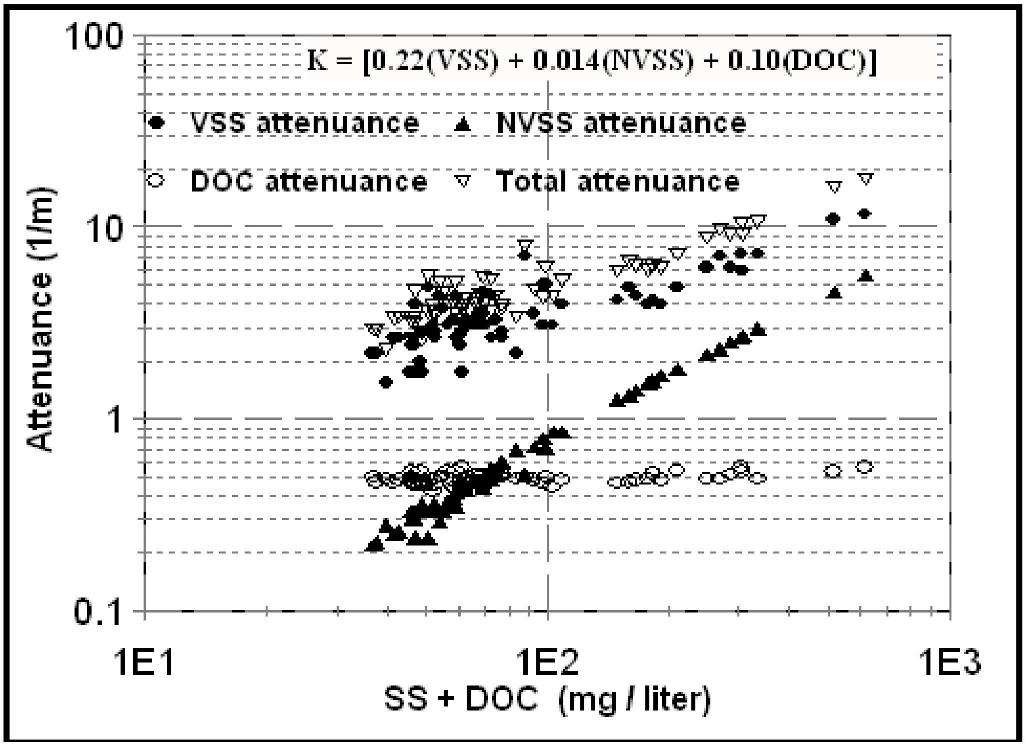


Figure 5

Organic particles (VSS) are the dominant light attenuators in this section of the river (Fig 5); VSS concentrations are lower than those of NVSS (Fig. 1), but the large partial attenuation coefficient for VSS compensates for its lower concentration.

The organic particles in VSS are derived from watershed soils and from river plankton. They probably are strong attenuators because they scatter and absorb light; in contrast, non-volatile suspended particles, which are probably clay minerals derived from the watershed and the river channel, probably scatter light but do not absorb it.

Appendix D: Table of Coefficients

Table D1 lists the coeffiecients used in the final calibration for the LMRM. (**) in the CBOD category denotes that there was a slight difference in the control file for the 1988 LMRM. In 1988, the treatment plants had not been upgraded, so the KBOD values were set to 0.0850 day^{-1}. This is the only difference in any of the control files.

Table D1. Coefficients used in LMRM.

Category	Coefficient	Description	Units	Default	LMRM
Extinction	EXH20	Extinction in pure water	m^{-1}	0.25 or 0.45	0.581
	EXSS	Extinction due to inorganic SS	m^{-1}	0.010	0.014
	EXOM	Extinction due to organic SS	m^{-1}	0.100	0.220
	BETA	Fraction of incident solar radiation absorbed at surface	-	0.450	0.450
	EXC	Read extinction coefficients	ON/OFF	OFF	OFF
	EXIC	Interpolate extinction coefficients	ON/OFF	OFF	OFF
	EXA1	Algal light extinction for ALG1	m^{-1}/gm^{-3}	0.200	0.220
	EXA2	Algal light extinction for ALG2		0.200	0.220
	EXA3	Algal light extinction for ALG3		0.200	0.220
Suspended Solids	SSS	SS settling rate	m^{-1}	1.000	0.150
	SEDRC	SS	ON/OFF	OFF	OFF
	TAUCR	critical shear stress	dynes/cm^2	0.000	0.000
Algal Rates – ALG1	AG	Maximum algal growth rate for ALG1	day^{-1}	2.000	1.900
	AR	Maximum algal respiration rate for ALG1	day^{-1}	0.040	0.140
	AE	Maximum algal excretion rate ALG1	day^{-1}	0.040	0.040
	AM	Maximum algal mortality rate ALG1	day^{-1}	0.100	0.050
	AS	Algal settling rate ALG1	m day^{-1}	0.100	0.250
	AHSP	Algal half saturation for P limited growth ALG1	g/m^3	0.003	0.001
	AHSN	Algal half saturation for N limited growth ALG1	g/m^3	0.014	0.050
	AHSSI	Algal half saturation for silica limited growth ALG1	g/m^3	0.000	0.002
	ASAT	Light saturation intensity at maximum photosynthetic rate ALG1	W m^2	75.000	72.640
Algal Rates – ALG2	AG	Maximum algal growth rate ALG2	day^{-1}	2.000	1.900

Category	Coefficient	Description	Units	Default	LMRM
	AR	Maximum algal respiration rate ALG2	day^{-1}	0.040	0.200
	AE	Maximum algal excretion rate ALG2	day^{-1}	0.040	0.040
	AM	Maximum algal mortality rate ALG2	day^{-1}	0.100	0.100
	AS	Algal settling rate ALG2	m day^{-1}	0.100	0.000
	AHSP	Algal half saturation for P limited growth ALG2	g/m^3	0.003	0.001
	AHSN	Algal half saturation for N limited growth ALG2	g/m^3	0.014	0.005
	AHSSI	Algal half saturation for silica limited growth ALG2	g/m^3	0.000	0.002
	ASAT	Light saturation intensity at maximum photosynthetic rate ALG2	W m^2	75.000	48.430
Algal Rates – ALG3	AG	Maximum algal growth rate ALG3	day^{-1}	2.000	2.300
	AR	Maximum algal respiration rate ALG3	day^{-1}	0.040	0.140
	AE	Maximum algal excretion rate ALG3	day^{-1}	0.040	0.040
	AM	Maximum algal mortality rate ALG3	day^{-1}	0.100	0.100
	AS	Algal settling rate ALG3	m day^{-1}	0.100	0.200
	AHSP	Algal half saturation for P limited growth ALG3	g/m^3	0.003	0.001
	AHSN	Algal half saturation for N limited growth ALG3	g/m^3	0.014	0.005
	AHSSI	Algal half saturation for silica limited growth ALG3	g/m^3	0.000	0.002
	ASAT	Light saturation intensity at maximum photosynthetic rate ALG3	W m^2	75.000	72.640
Algal Temp – ALG1	AT1	Lower temperature for algal growth ALG1	°C	5.000	0.500
	AT2	Lower temperature for maximum algal growth rate ALG1	°C	25.000	10.000
	AT3	Upper temperature for maximum temperature growth rate ALG1	°C	35.000	25.000
	AT4	Upper temperature for algal growth ALG1	°C	40.000	35.000
	AK1	Fraction of algal growth rate AT1 ALG1	-	0.100	0.100
	AK2	Fraction of maximum algal growth rate at AT2 ALG1	-	0.990	0.990
	AK3	Fraction of maximum algal growth rate at AT3 ALG1	-	0.990	0.990
	AK4	Fraction of algal growth rate at AT4 ALG1	-	0.100	0.100
Algal Temp – ALG2	AT1	Lower temperature for algal growth ALG2	°C	5.000	16.000
	AT2	Lower temperature for maximum algal growth rate ALG2	°C	25.000	25.000

Category	Coefficient	Description	Units	Default	LMRM
Algal Temp – ALG2	AT3	Upper temperature for maximum temperature growth rate ALG2	°C	35.000	27.000
	AT4	Upper temperature for algal growth ALG2	°C	40.000	30.000
	AK1	Fraction of algal growth rate AT1 ALG2	-	0.100	0.100
	AK2	Fraction of maximum algal growth rate at AT2 ALG2	-	0.990	0.990
	AK3	Fraction of maximum algal growth rate at AT3 ALG2	-	0.990	0.990
	AK4	Fraction of algal growth rate at AT4 ALG2	-	0.100	0.100
Algal Temp – ALG3	AT1	Lower temperature for algal growth ALG3	°C	5.000	12.000
	AT2	Lower temperature for maximum algal growth rate ALG3	°C	25.000	17.000
	AT3	Upper temperature for maximum temperature growth rate ALG3	°C	35.000	32.000
	AT4	Upper temperature for algal growth ALG3	°C	40.000	36.000
	AK1	Fraction of algal growth rate AT1 ALG3	-	0.100	0.100
	AK2	Fraction of maximum algal growth rate at AT2 ALG3	-	0.990	0.990
	AK3	Fraction of maximum algal growth rate at AT3 ALG3	-	0.990	0.990
	AK4	Fraction of algal growth rate at AT4 ALG3	-	0.100	0.100
Algal stoichiometry	AP	Fraction P for all groups	-	0.005	0.005
	AN	Fraction N for all groups	-	0.080	0.080
	AC	Fraction C for all groups	-	0.450	0.450
	ASI	Fraction Si for all groups	-	0.000	0.180
	ACHLA	Chlorophyll-algae ratio for all groups	-	0.05	0.0675
	APOM	Fraction algae lost by mortality to POM for all groups	-	0.800	0.800
	ANEQN	NH4 preference factor for all groups	1 or 2	2	2
	ANPR	NH4 half saturation coefficient for NH4-NO3 for all groups	-	0.001	0.001
Dissolved organic matter	LDOMDK	Labile DOM decay rate	day^{-1}	0.100	0.080
	RDOMDK	Labile to refractory decay rate	day^{-1}	0.001	0.001
	LRDDK	Maximum refractory decay rate	day^{-1}	0.010	0.001
Particulate organic matter	LPOMDK	Labile POM decay rate	day^{-1}	0.080	0.080
	RPOMDK	Labile to refractory decay rate	day^{-1}	0.001	0.001
	LRPDK	Maximum refractory decay rate	day^{-1}	0.010	0.010
	POMS	Settling rate	m day^{-1}	0.100	0.100
Organic matter	ORGP	Fraction P		0.005	0.005

Category	Coefficient	Description	Units	Default	LMRM
stoichiometry	ORGN	Fraction N		0.080	0.050
	ORGC	Fraction C		0.450	0.450
	ORGSI	Fraction Si		0.180	0.180
	OMT1	Lower temperature for OM decay	°C	4.000	4.000
	OMT2	Upper temperature for OM decay	°C	25.000	25.000
	OMK1	Fraction of OM decay rate at OMT1	°C	0.100	0.100
	OMK2	Fraction of OM decay at OMT2	°C	0.990	0.990
Carbonaceous BOD1	KBOD	5-day decay rate at 20 °C for BOD1	day^{-1}	0.100	0.0345
	TBOD	Temperature coefficient for BOD1	-	1.020	1.020
	RBOD	Ratio of CBOD to ultimate CBOD for BOD1	-	1.850	1.000
	CBODS	CBOD settling rate for BOD1	m day^{-1}	0.000	0.000
Carbonaceous BOD2	KBOD	5-day decay rate at 20 °C for BOD2	day^{-1}	0.100	0.0322**
	TBOD	Temperature coefficient for BOD2	-	1.020	1.020
	RBOD	Ratio of CBOD to ultimate CBOD for BOD2	-	1.850	1.000
	CBODS	CBOD settling rate for BOD2	m day^{-1}	0.000	0.000
Carbonaceous BOD3	KBOD	5-day decay rate at 20 °C for BOD3	day^{-1}	0.100	0.0294**
	TBOD	Temperature coefficient for BOD3	-	1.020	1.020
	RBOD	Ratio of CBOD to ultimate CBOD for BOD3	-	1.850	1.000
	CBODS	CBOD settling rate for BOD3	m day^{-1}	0.000	0.000
Carbonaceous BOD4	KBOD	5-day decay rate at 20 °C for BOD4	day^{-1}	0.100	0.0495
	TBOD	Temperature coefficient for BOD4	-	1.020	1.020
	RBOD	Ratio of CBOD to ultimate CBOD for BOD4	-	1.850	1.000
	CBODS	CBOD settling rate for BOD4	m day^{-1}	0.000	0.000
Carbonaceous BOD5	KBOD	5-day decay rate at 20 °C for BOD5	day^{-1}	0.100	0.0257
	TBOD	Temperature coefficient for BOD5	-	1.020	1.020
	RBOD	Ratio of CBOD to ultimate CBOD for BOD5	-	1.850	1.000
	CBODS	CBOD settling rate for BOD5	m day^{-1}	0.000	0.000
Carbonaceous BOD6	KBOD	5-day decay rate at 20 °C for BOD6	day^{-1}	0.100	0.0315
	TBOD	Temperature coefficient for BOD6	-	1.020	1.020
	RBOD	Ratio of CBOD to ultimate CBOD for BOD6	-	1.850	1.000
	CBODS	CBOD settling rate for BOD6	m day^{-1}	0.000	0.000
CBOD stoichiometry	CBODP	P stoichiometry for CBOD decay	-	0.004	0.004
	CBODN	N stoichiometry for CBOD decay	-	0.060	0.060
	CBODC	C stoichiometry for CBOD decay	-	0.320	0.320
Inorganic	PO4R	Sediment release rate of P, fraction of SOD	-	0.001	0.001

Category	Coefficient	Description	Units	Default	LMRM
Phosphorus	PARTP	P partitioning coefficient for suspended solids		0.000	0.000
Ammonium	NH4REL	Sediment release rate, fraction of SOD	-	0.001	0.010
	NH4DK	NH4 decay rate	day^{-1}	0.120	0.120
	NH4T1	Lower temperature for NH4 decay	°C	5.000	5.000
	NH4T2	Lower temperature for maximum NH4 decay	°C	25.000	25.000
	NH4K1	Fraction of nitrification rate at NH4T1	-	0.100	0.100
	NH4K2	Fraction of nitrification rate at NH4T2	-	0.990	0.990
Nitrate	NO3DK	Nitrate decay rate	day^{-1}	0.030	0.030
	NO3S	Denitrification rate from sediments	m day^{-1}	1.000	0.300
	NO3T1	Lower temperature for NO3 decay	°C	5.000	5.000
	NO3T2	Lower temperature for maximum NO3 decay	°C	25.000	25.000
	NO3K1	Fraction of denitrification rate of NO3T1	-	0.100	0.100
	NO3K2	Fraction of denitrification rate at NO3T2	-	0.990	0.990
Silica	DSIR	Dissolved silica sediment release rate, fraction of SOD	-	0.100	0.100
	PSIS	Particulate biogenic settling rate	m sec^{-1}	1.000	1.000
	PSIDK	Particulate biogenic silica settling rate	day^{-1}	0.300	0.300
	PARTSI	Dissolved silica partitioning coefficient	-	0.000	0.000
Iron	FEREL	Fe sediment release rate, fraction of SOD	-	0.500	0.500
	FESETL	Fe settling velocity	m sec^{-1}	2.000	2.000
Sediment CO2 release	COR2REL	Sediment CO2 release rate, fraction of SOD	-	1.200	1.200
O2 stoichiometry	O2NH4	O2 stoichiometry for nitrification	-	4.570	4.570
	O2OM	O2 stoichiometry for organic matter decay	-	1.400	1.400
	O2AR	Oxygen stoichiometry for algal respiration for all groups		1.100	1.100
	O2AG	Oxygen stoichiometry for algal primary production for all groups	-	1.400	1.400
O2 limit	KDO (O2LIM)	Dissolved O2 half saturation constant or concentration at which aerobic processes are at 50% of their maximum	g m^{-3}	0.100	0.100
Sediment	SEDC	First order sediment decay	ON/OFF	OFF	OFF
	PRNSC	Print to snp.opt file	ON/OFF	OFF	OFF
	SEDCI	Initial sediment concentration	g m^{-2}	0.000	0.000
	SEDK	Sediment decay rate	day^{-1}	0.100	0.100
	SEDS	Sediment settling rate	m day^{-1}	0.100	0.080
	FSOD	Fraction of zero order decay used	-	1.000	1.000
Sediment	FSED	Fraction of 1st order decay used	-	1.000	1.000

Category	Coefficient	Description	Units	Default	LMRM
	SEDBR	sediment burial rate	day^{-1}	0.010	0.010
	SODT1	Lower temperature for 0 order SOD decay	°C	4.000	8.000
	SODT2	Upper temperature for zero order SOD decay	°C	25.000	12.000
	SODK1	Fraction of SOD at lower temperature	-	0.100	0.100
	SODK2	Fraction of SOD at upper temperature	-	0.990	0.990
SOD	SOD	Zero order decay rate per segment	$g\ m^{-2}\ day^{-1}$		varies

Appendix E: LMRM W2 Control Files by Water Year

To obtain this information, contact:

David L. Smith (CEERD-EP-W)
601-634-4267
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
David.L.Smith@usace.army.mil

Tammy Threadgill (CEERD-EP-W)
601-634-3870
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
Tammy.L.Threadgill@usace.army.mil

Appendix F: Time Series Plots

To obtain this information, contact:

David L. Smith (CEERD-EP-W)
601-634-4267
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
David.L.Smith@usace.army.mil

Tammy Threadgill (CEERD-EP-W)
601-634-3870
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
Tammy.L.Threadgill@usace.army.mil

Appendix G: Cumulative Distribution Plots

To obtain this information, contact:

David L. Smith (CEERD-EP-W)
601-634-4267
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
David.L.Smith@usace.army.mil

Tammy Threadgill (CEERD-EP-W)
601-634-3870
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
Tammy.L.Threadgill@usace.army.mil

Appendix H: Scatter Plots

To obtain this information, contact:

David L. Smith (CEERD-EP-W)
601-634-4267
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
David.L.Smith@usace.army.mil

Tammy Threadgill (CEERD-EP-W)
601-634-3870
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
Tammy.L.Threadgill@usace.army.mil

Appendix I: Tabular Statistics

To obtain this information, contact:

David L. Smith (CEERD-EP-W)
601-634-4267
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
David.L.Smith@usace.army.mil

Tammy Threadgill (CEERD-EP-W)
601-634-3870
U.S. Engineer Research and Development Center
3909 Halls Ferry Road Vicksburg, MS 39180
Tammy.L.Threadgill@usace.army.mil

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14. ABSTRACT

The U.S. Army Engineer Research and Development Center (USACE ERDC) Environmental Lab (EL) and Metropolitan Council Environmental Services (MCES) developed an advanced water quality model of the Lower Minnesota River (Jordan, Minnesota, to the mouth) using the CE-QUAL-W2 modeling framework. This portion of the river is a highly impaired system with a very rich set of monitored data. Model development consisted of calibration and validation of seven water years: 1988 (low flow) and 2001-2006. Data from 2006 were first used to calibrate the model, and the same parameter values were applied to all other years for validation. The 2006 parameter set worked well for all years except 1988. The model was then recalibrated using data from 1988 and verified by applying the revised parameter set to the other six years. The model output agrees to an acceptable level with observed data for every water year simulated. The Lower Minnesota River Model (LMRM) provides a tool for load allocation studies and facility or watershed planning, in addition to providing a bridge to other water quality modeling efforts in the area.

15. SUBJECT TERMS

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