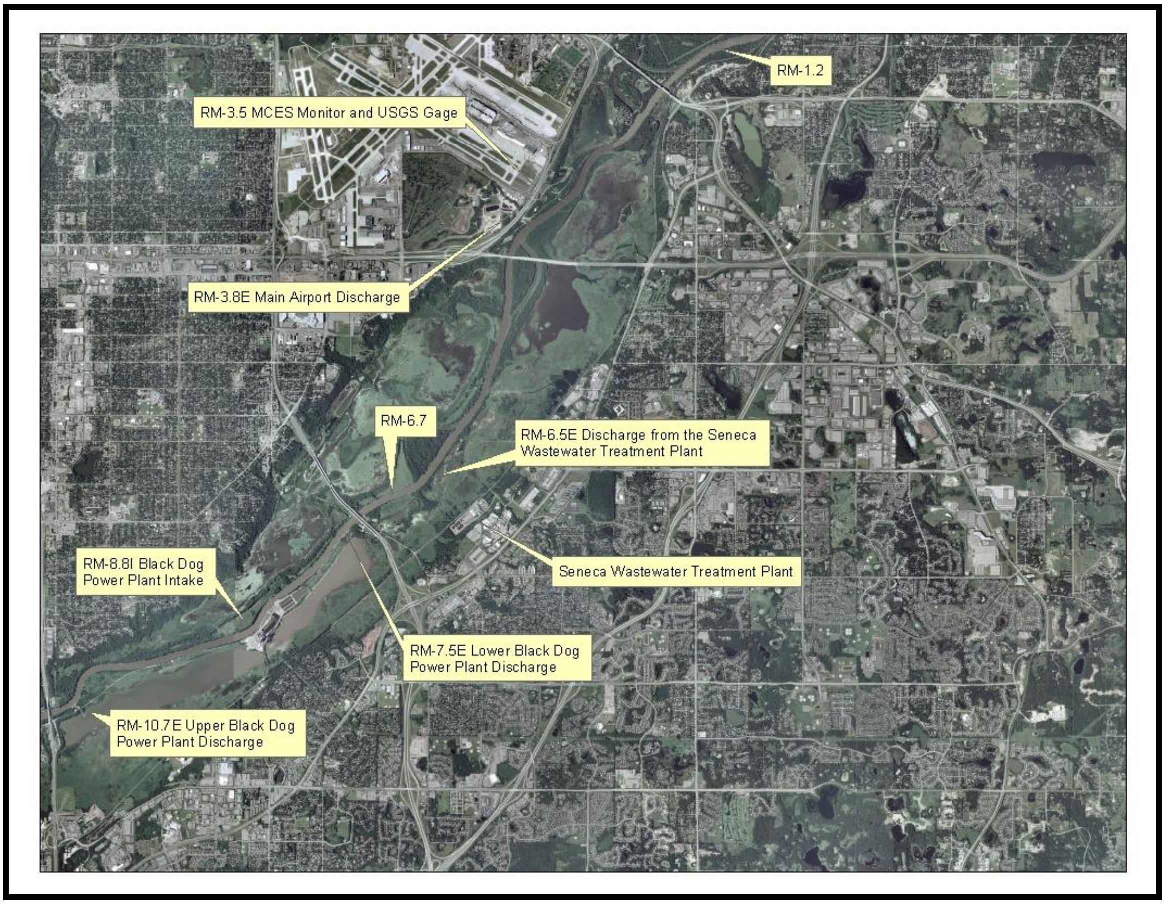
Lower Minnesota River Study

Monitoring and Modeling Water Quality From Jordan, Minnesota, to the Mouth



Aerial Photograph of Lower Reach of 40-Mile Study Area (MPCA, 2007c)

Metropolitan Council

In Cooperation With
U.S. Army Corps of Engineers
Minnesota Pollution Control Agency
U.S. Geological Survey
Lower Minnesota River Watershed District
Metropolitan Airports Commission

Final Report
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ABBREVIATIONS AND ACRONYMS

AME	Absolute mean error
CHLA	Chlorophyll *a* associated with living algae (i.e., phaeophytin-corrected)
BOD	Biochemical oxygen demand
CBOD	Carbonaceous biochemical oxygen demand, 5-day ($_5$) or ultimate ($_U$)
CE-QUAL-W2	Modeling framework supported by the USACE
DO	Dissolved oxygen
ECOMSED-RCA	Modeling framework applied to the Mississippi River and Lake Pepin
ERDC	U.S. Army Engineer Research Development Center
GP	Generating Plant (Black Dog)
GPP	Gross primary production in the water column
HSPF	Modeling framework applied to the Minnesota River Basin, Lac Qui Parle to Jordan
ISS	Inorganic suspended solids
MAC	Metropolitan Airports Commission
MCES	Metropolitan Council Environmental Services
MDNR	Minnesota Department of Natural Resources
ME	Mean error
Metro Area	Seven-county metropolitan area of Minneapolis and St. Paul, Minnesota
MPCA	Minnesota Pollution Control Agency
MSP	Minneapolis-St. Paul International Airport
N	Nitrogen
NH4	Ammonium nitrogen (as mg N/L)
NO3	Nitrate nitrogen (as mg N/L)
NPDES	National Pollutant Discharge Elimination System
NTU	Nephelometric turbidity units
NVSS	Nonvolatile (inorganic) suspended solids
OM	Organic matter, nonliving or living
P	Phosphorus
R	Respiration in the water column
RMSE	Root mean square error
SOD	Sediment oxygen demand
SRP	Soluble reactive phosphorus (as mg P/L)
Summer	June through September, unless otherwise noted
TMDL	Total maximum daily load; refers to load allocations for both point and nonpoint sources
TKN	Total Kjeldahl nitrogen (as mg N/L)
TP	Total phosphorus (as mg P/L)
TSS	Total suspended solids
USACE	U.S. Army Corps of Engineers
USGS	U.S. Geological Survey
VSS	Volatile (organic) suspended solids
WLA	Waste load allocation; refers to load allocations for point sources
WWTP	Wastewater treatment plant
WY	Water year, October 1 through September 30

MEASUREMENT UNITS AND CONVERSION FACTORS

Metric Measurements	Multiply By	To Yield U.S. Equivalents
Centimeters (cm)	0.3937	inches (in)
Meters (m)	3.281	feet (ft)
Kilometers (km)	0.6214	miles (mi)
Metric tons (mt)	1.102	short tons
Square meters (m^2)	10.76	square feet (ft^2)
Hectare (ha)	2.471	acres (ac)
Square kilometers (km^2)	0.3861	square miles (mi^3)
Cubic hectometers (hm^3)	811.0	acre-feet (acre-ft)
Milligrams per gram (mg/g)	1	part per thousand
Milligrams per liter (mg/L)	1	parts per million (ppm)
Micrograms per liter (µ/L)	1	parts per billion (ppb)
Meters per second (m/s)	3.281	feet per second (fps)
Cubic meters per second (m^3/s, cms)	35.31	cubic feet per second (cfs)
Cubic meters per second (m^3/s, cms)	22.82	million gallons per day (mgd)
Kilograms per hectare (kg/ha)	0.8921	pounds per acre (lb/ac)
Degrees Centigrade (ºC)	(9/5 * ºC) + 32	degrees Fahrenheit (ºF)

ABSTRACT

A partnership of federal, state, and local agencies developed a water-quality model of the lower 40 miles of the Minnesota River. This reach lies at the juncture of two contrasting landscapes: a predominantly agricultural watershed to the west and an expanding metropolitan area to the east. It is listed as impaired for not meeting water-quality standards for dissolved oxygen, turbidity, bacteria, PCBs, and mercury. Excessive nutrients and sediment are also concerns. The lower Minnesota River receives pollutant loads from point sources, including two major wastewater treatment facilities, and loads from nonpoint sources, such as rural and urban runoff. A water-quality model was needed for facility and watershed planning. The partners chose the CE-QUAL-W2 model framework and designed a three-year monitoring program to support it.

Hydrodynamics play a prominent role in the water quality of rivers; however, the lower Minnesota River differs from most in being part of a navigation system. For the majority of the year, river discharge is the main driver of water quality in the lower 40 miles. At flows greater than approximately 2,000 cubic feet per second, transport dominates water quality. Hydrodynamics become more complex at lower flows, including cooling-water withdrawals and pooling effects. At lower flows in summer, greater depths and slower velocities in the navigation channel increasingly affect sediment, light, nutrient, phytoplankton, and oxygen dynamics.

Despite efforts to reduce sediment loads to the Minnesota River, suspended-solids concentrations remain high. During 2000-2009, the median concentration of total suspended solids at river mile 3.5 was 47 mg/L. Suspended solids affect transparency and turbidity, which in turn impact phytoplankton and other aquatic life. While inorganic solids dominate river concentrations, organic solids play an important role in attenuating light. Light through its effect on phytoplankton is an important factor in oxygen metabolism in the lower Minnesota River. At lower river flows, fine materials settle and deposit on the river bed. Measured rates of sediment oxygen demand were low to moderate but remain an important component of oxygen dynamics in the river.

Nutrient levels are also high. During 2000-2009, the median total nitrogen concentration at river mile 3.5 was 4.81 mg/L with nitrate representing the largest portion. The median total phosphorus concentration was 0.19 mg/L with orthophosphate representing roughly a third. Approximately one-half of the particulate phosphorus is biologically labile or easily recycled to orthophosphate under certain conditions. Phosphorus dynamics are complex with physical factors dominating at higher flows and phytoplankton playing an increasing role at lower flows. The upper reach shows the most potential for phosphorus limitation of algal growth.

During 2004-2006, the headwaters near Jordan, Minnesota, contributed over 88% of the suspended-solids and nutrient loads to the lower 40 miles of the Minnesota River. Local tributaries and dischargers contributed the remainder. At lower flows, the portion of nutrient loads contributed by the two major wastewater treatment plants increased; for example, during late summer 2006, the facilities contributed 34, 46, and 75% of the ammonia, nitrate, and orthophosphate loads, respectively. The lower Minnesota River was a deposition zone for suspended solids, with annual retention of 22-39% of total loads in 2004-2006. During these three years, the reach was a sink for phosphorus with 5-11% of the load retained, but it was a source of ammonium with 28-50% more load exported than received.

High levels of nutrients in the Minnesota River support high levels of phytoplankton. During 2003-2009, median summer chlorophyll *a* concentrations were 87 and 58 μg/L at river miles 39.4 and 3.5, respectively. Concentrations at mile 39.4 exceeded those at mile 3.5 during the ice-free months and especially in late summer. Under summer low-flow conditions in 2006, viable chlorophyll *a* concentrations decreased from mile 39.4 to the mouth as phaeophytin *a* concentrations increased, suggesting algal die-off in the lower reaches. Increased water-column depths and lower current velocities in the navigation channel may settle or mix phytoplankton out of the narrow photic zone, leading to senescence. The decomposition of phytoplankton contributes to ammonium and orthophosphate concentrations in the lower reach, as well as oxygen demand.

Phytoplankton production and respiration are strong components of oxygen dynamics in the lower Minnesota River, especially during summer low-flow conditions. Oxygen dynamics are a complex mixture of physical and biochemical factors. Physical factors include temperature, flow, wind, ice, and light. At lower flows, biochemical factors become more important. These include phytoplankton activity, decomposition of nonliving organic matter, and sediment oxygen demand. During two surveys in 2006, phytoplankton respiration generally exceeded production, showing the river to be predominantly hetereotrophic during summer low-flow conditions.

Effluent quality at the Blue Lake and Seneca WWTPs has improved greatly since the 1980s. CBOD, ammonia, and phosphorus concentrations and loads are consistently well below permit limitations. Effluent characteristics have changed as well, with organic matter becoming slower to decay and phosphorus becoming less biologically available. While current effluent CBOD and ammonia loads have little effect on river dissolved-oxygen concentrations, the two WWTPs continue to enrich the river with phosphorus and nitrogen. Discharges from the Black Dog Generating Plant and international airport were challenging to monitor and model. Additional work is needed to understand their impact on water quality.

The Lower Minnesota River Model was calibrated against seven years of data, 1988 and 2001-2006, with a variety of flows ranging from drought to flood. The calibration strategy focused on performance during summer low-flow conditions. A set of parameters was developed to meet performance targets in 1988, and then this set was applied to the other six years. Across all years the model captured the quantitative and qualitative trends in all modeled parameters. With rare exceptions, the statistical measures of model performance met calibration targets. Qualitatively, trends were consistent with measured data.

That one calibration captured trends in water quality over a range of flows suggests that this is a useful tool for predicting future conditions. Four loading scenarios were applied to the model to demonstrate its potential use in facility and watershed planning. In one scenario, output from the Minnesota River Basin Model was translated and used as input to the CE-QUAL-W2 model, showing the ability to link modeling efforts. Scenario results were reasonable, adding confidence in the model's performance and utility. The results of the calibration and application of the Lower Minnesota River Model show that the model is an acceptable tool for studying dissolved oxygen, nutrients, phytoplankton, and turbidity under a variety of conditions.

The purpose of this report is to summarize and integrate all aspects of the Lower Minnesota River Study. Readers should refer to individual project reports for more information.

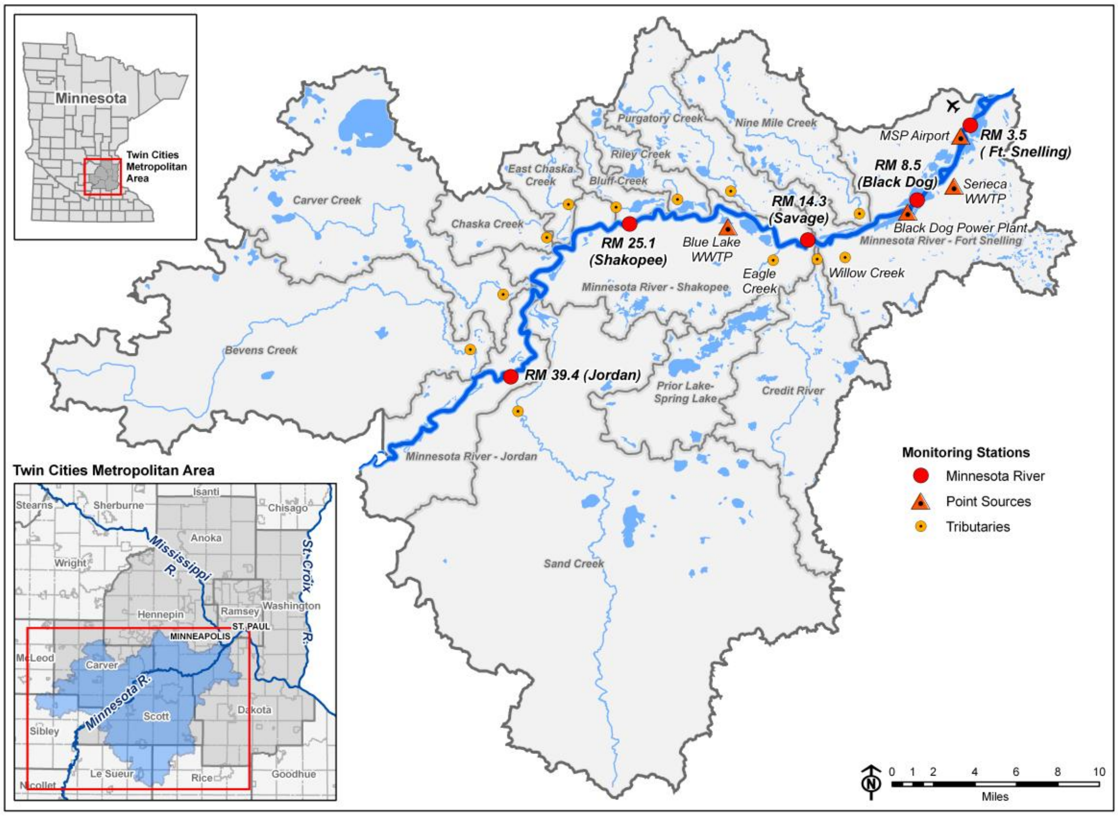
1 INTRODUCTION

The twin cities of Minneapolis and St. Paul, Minnesota, started in the 1820s as small settlements near Fort Snelling at the confluence of the Minnesota and Mississippi Rivers (Figure 1). Today the Twin Cities Metropolitan Area (Metro Area) encompasses seven counties with a combined area of 3000 mi^2 (7700 km^2) and population of 2.6 million (2000 census). The Metropolitan Council is a regional planning agency that provides wastewater and transit services, coordinates development, and assists communities as they plan for growth. The region expects an additional one million people over thirty years with the most rapid growth in the southwestern counties of Scott and Carver where population is projected to increase by 147% and 183%, respectively (Metropolitan Council, 2009). The Minnesota River forms the boundary between the two counties and receives discharges from the state's third and fourth largest wastewater treatment plants (WWTP) with a combined average flow of 51 mgd (2.2 m^3/s) during 2000-2009. Wastewater flow to the two facilities is projected to increase to an estimated 64 mgd (2.8 m^3/s) by 2030 (Bryce Pickart, Metropolitan Council Environmental Services). Regional planning and state regulatory agencies need updated information and a forecasting tool to assess the impact of increased wastewater discharges on the water quality of the Minnesota River.

The lower Minnesota River lies at the juncture of two contrasting landscapes: a predominantly agricultural watershed to the west and an expanding metropolitan area to the east. Pollutant loads contributed from rural and urban sources and hydrodynamics altered by a navigation system and water appropriations combine to impact water quality in this reach. The lower 22 miles of the Minnesota River appear on the state's list of impaired waters for not meeting water-quality standards for dissolved oxygen, turbidity, bacteria, PCB, and mercury (Minnesota Pollution Control Agency, 2008). 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, 2007). In turn, the Minnesota River contributes the highest sediment and nutrient loads to the Mississippi River upstream of Lake Pepin, a natural impoundment in Navigation Pool 4 (LimnoTech, 2009; Kloiber, 2004; Larson et al., 2002).

The lower Minnesota River also lies at the juncture of four pollutant load allocation studies that affect land and water resources management over large portions of the state. The first began as a waste load allocation (WLA) study of the lower 22 miles of the Minnesota River by the Minnesota Pollution Control Agency (MPCA, 1985). The study established effluent limits for carbonaceous biochemical oxygen demand (BOD) and ammonia for the two major WWTPs and other dischargers. The study found, however, that point-source controls were not enough to maintain dissolved-oxygen (DO) standards in the river, so it recommended a 40% reduction in BOD loads from nonpoint sources. For this reason and subsequent load allocations to headwater and tributary nonpoint sources, the WLA study of the lower Minnesota River (MPCA, 1985) is considered one of the earliest total maximum daily load (TMDL) studies in the nation (U.S. Environmental Protection Agency, 1992). TMDL studies determine pollutant load allocations for both point and nonpoint sources.

Figure 1 - Monitoring Stations on the Lower Minnesota River and Tributaries



The WLA/TMDL study of the lower Minnesota River generated a second-phase TMDL study of upstream sources of BOD from the Minnesota River Basin. To evaluate pollutant sources and management alternatives, a watershed model using the Hydrologic Simulation Program-Fortran (HSPF) framework was developed (Tetra Tech, 2003). The model covers a large portion of the Minnesota River Basin from Lac Qui Parle Dam at river mile (RM) 284 to the city of Jordan at RM 39.4. The MPCA completed a TMDL report in 2004 and an implementation plan in 2006 (MPCA, 2004; MPCA, 2006). The report attributed BOD loads at Jordan to upstream phosphorus loads and resulting phytoplankton production. With DO concentrations expected to decrease below the standard of 5.0 mg/L most frequently during the summer under low river flows, the implementation plan identified phosphorus loads from 40 of 143 permitted municipal and industrial WWTPs as having the greatest impact while runoff from agricultural cropland as having minimal impact. The MPCA implemented a basin-wide phosphorus permit for dischargers with opportunities for trading among facilities (MPCA, 2007a).

In 2004 the MPCA initiated large TMDL studies of turbidity in the Minnesota River Basin (MPCA, 2005) and nutrients and turbidity in the Lake Pepin watershed (MPCA, 2007b). The Lake Pepin watershed covers an area of 47,100 mi^2 (122,000 km^2), encompassing more than half of the land area of Minnesota and portions of Wisconsin, Iowa, and South Dakota. The Minnesota River Basin Model previously applied in the DO/BOD TMDL study was further improved for application to the turbidity and nutrient TMDL studies (Tetra Tech, 2008). A linked model of hydrodynamics, sediment transport, and water quality using the ECOMSED and RCA frameworks was developed for the main stem of the Mississippi River from Lock and Dam No. 1 through Lake Pepin. The Mississippi River model was first built for the Lake Pepin Phosphorus Study (HydroQual, 2002) and later expanded and improved for the TMDL studies of the Lake Pepin watershed (LimnoTech, 2009). The downstream boundary of the Minnesota River Basin Model is the city of Jordan, while the upstream boundary of the Upper Mississippi River – Lake Pepin model is Lock and Dam No.1, located approximately four miles upstream of the confluence with the Minnesota River. This left a gap of 40 miles in the lower Minnesota River between the two models.

Four TMDL studies with wide-ranging implications hinge on a good understanding of water-quality dynamics in the lower Minnesota River, increasing the need for current and reliable data and modeling for this reach. Since the WLA/TMDL study of the lower Minnesota River in 1985, the two major WWTPs have upgraded to advanced secondary treatment with nitrification and phosphorus removal, and changes have occurred at other dischargers and in land management practices. Trend analyses have shown significant changes in the water quality of the Minnesota River at Jordan and Fort Snelling since the 1970s and 1980s (Johnson et al., 2009; Kloiber, 2004; MPCA, 2002; Kroening and Andrews, 1997). These changes justify updating the WLA/TMDL study of the lower Minnesota River. The DO/BOD TMDL implementation plan also recommended an update of the 1985 study (MPCA, 2006). Further, monitoring and modeling of the lower Minnesota River is needed to inform and to some degree link the Minnesota River Basin and Mississippi River models for the large turbidity and nutrient TMDL studies. James (2007) and earlier studies (e.g., MPCA, 1985; Kroening and Andrews, 1997) have demonstrated changes in the amounts of nutrients, suspended solids, and phytoplankton between RM 40 and the mouth of the Minnesota River that should be considered in the Lake Pepin TMDL study. These factors provided justification for the Lower Minnesota River Study.

2 STUDY AREA DESCRIPTION

The Minnesota River has a watershed area of 16,900 mi^2 (43,800 km^2) and drains much of southwestern Minnesota and minor portions of Iowa and South Dakota. The river runs 330 miles from its origin in Big Stone Lake on the South Dakota border to its confluence with the Mississippi River in St. Paul, Minnesota. The river valley was formed by the Glacial River Warren, which flowed from the southern end of Glacial Lake Agassiz at the end of the last glacial period roughly ten thousand years ago (Waters, 1977). Today's river is much smaller and under fit compared to the wide glacial river valley across which it now meanders. The basin has relatively flat topography and rich soils—both well suited to agriculture. In 1997, 73 percent of the areal coverage in the Minnesota River Basin was classified as cultivated cropland (National Resources Inventory). The great majority (~90%) of original wetlands in the basin have been tiled and drained for agricultural uses.

The study area for this project was the lower 40 miles of the Minnesota River, beginning near the city of Jordan, Minnesota (Figure 1). While the river enters the Metro Area some distance upstream, Jordan is the best location for a model boundary because the U.S. Geological Survey (USGS) maintains a long-term stream-flow gaging station (USGS Station #05330000) on a bridge crossing the river near this city. Also, Metropolitan Council Environmental Services (MCES) maintains a long-term water-quality monitoring station at this location. During water years (WY) 1935 through 2008, the annual mean flow of the Minnesota River at Jordan was 4,551 cfs (129 m^3/s). Ten percent of daily mean flows exceeded 12,100 cfs (343 m^3/s), 50 percent exceeded 1,910 cfs (54.1 m^3/s), and 90 percent exceeded 345 cfs (9.77 m^3/s). During the ten-year period of 1998-2007, mean annual flow-weighted concentrations for the Minnesota River at Jordan were 187 mg/L for total suspended solids, 0.286 mg/L for total phosphorus, 7.03 mg/L for nitrate nitrogen, and 1.42 mg/L for total Kjeldahl nitrogen (Kloiber, 2004).

Since 1937 the National Weather Service has operated a meteorological station at the Minneapolis-St. Paul (MSP) International Airport, which is located on a bluff near the mouth of the Minnesota River (Figure 1). Mean daily maximum temperatures range from 21.7°F (-5.7°C) in January to 83.4°F (28.6°C) in July (National Climate Data Center, 1971-2000). Normal annual precipitation is 29.41 inches (74.7 cm). The wettest months are generally May-August (three to four inches per month), and the driest months are generally December-February (an inch or less per month). Normal annual snowfall is 55.9 inches (142.0 cm).

Numerous lakes and wetlands are located in the wide floodplain of the lower Minnesota River. In 1976 Congress established the Minnesota Valley National Wildlife Refuge to protect the floodplain between Belle Plaine, Minnesota, and Fort Snelling State Park. It is one of only four national urban wildlife refuges. The refuge contains 14,000 acres (57 km^2) of forest, wetlands, and wet meadows that are managed to provide habitat for migratory waterfowl, fish, and other wildlife species. Management includes dikes and other water control structures on many lakes and wetlands in the floodplain. The final four miles of the Minnesota River and its floodplain are part of the Mississippi National River and Recreation Area, established by Congress in 1988.

A nine-foot deep, 100-foot wide channel is maintained by the U.S. Army Corps of Engineers (USACE) for commercial barge navigation from the mouth of the Minnesota River to RM 14.7 at Savage, Minnesota. A grain company maintained a nine-foot channel to a barge terminal at RM 21.8 until the early 1980s (MPCA, 2007c). The USACE stopped maintaining a four-foot channel to Shakopee sometime prior to 2001. Lock and Dam No. 2 on the Mississippi River near Hastings, Minnesota, became operational in 1931, raising the water surface at the mouth of the Minnesota River about 1.0 ft (0.3 m) and at the city of Shakopee (RM 25.6) about 0.2 ft (0.06 m). The combined effects of a dredged channel in the lower Minnesota River and the backwater pool created by Lock and Dam No. 2 transform the river from a relatively shallow, free-flowing stream in the upper reach to a deeper, low-velocity channel maintained for commercial navigation in the lower reach (MPCA, 1985).

An average of 21,000 cubic yards (16,000 m^3) of dredged material are removed from the bed of the navigation channel each year, with the most frequently dredged areas at RM 1-2, 4-5, and 12-13 (Lower Minnesota River Watershed District, 1999). Before recent growth in the corn-based ethanol industry, 50 percent of the grain exiting Minnesota was loaded on barges in Savage. The standard barge is 35 by 195 ft (10.7 by 59.4 m) and carries 1500 tons (1361 mt) of cargo. From 2000 to 2008, barges transported an annual average 3.4 million mt/yr or an average of 9.9 barge loads/day assuming a 250-day shipping season (Richard Lambert, Minnesota Department of

Transportation). Over the nine-year period, barge traffic ranged from 5.0 million mt/yr and 14.6 loads/day in 2002 to 1.5 million mt/yr and 4.5 loads/day in 2008.

Table 1 lists the major tributaries that enter the lower 40 miles of the Minnesota River, and Figure 1 displays the hydrologic boundaries of the watersheds. Land use is primarily agricultural in the western watersheds but becomes increasingly developed as the river flows north and east toward its confluence with the Mississippi River. MCES, Carver County, and local watershed organizations have monitored these 11 tributaries for various lengths of time. The MCES stream monitoring program started in 1989 to measure nonpoint-source loads in response to the WLA/TMDL study in 1985.

Table 1 – Descriptions of Monitored Tributaries

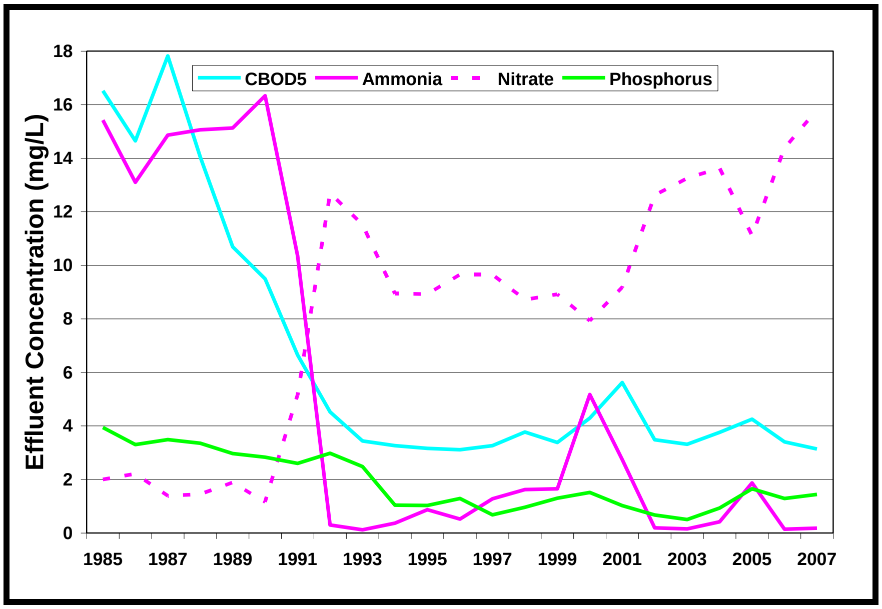
Tributary	Confluence (river mile)[1]	Area (mi^2)[2]	Dominant Land Use	Monitoring (agency, year started)
Sand Creek	35.5	260	Rural	MCES, 1990
Carver Creek	34.1	83	Rural	MCES, 1989
Chaska Creek	31.6	16	Mixed	Carver County, 1998
East Chaska Creek	30.3, 30.0	12	Mixed	Carver County, 2003
Bluff Creek	22.5	9	Mixed	MCES, 1991
Riley Creek	22.3	13	Mixed	MCES & partners, 1999
Purgatory Creek	19.6	36	Mixed	Watershed District, 2003
Eagle Creek	15.8	7	Mixed	MCES & partners, 1999
Credit River	13.7	51	Mixed	MCES, 1989
Nine Mile Creek	11.0 & 12.5	38	Urban	MCES, 1989
Willow Creek	11.0	42	Urban	MCES & partners, 1999

[1] (Larson, 2006) [2] (Larson, 2004)

In 2002 the MPCA compiled a preliminary list of 36 permitted discharges to the lower 40 miles of the Minnesota River (Larson, 2004). The list included direct discharges to the river and indirect discharges to unmonitored tributaries. The following paragraphs describe the four major dischargers that were defined in the model: Blue Lake WWTP, Seneca WWTP, MSP airport, and Black Dog Generating Plant (Figure 1).

The Blue Lake and Seneca WWTPs are owned and operated by the Metropolitan Council and discharge to the Minnesota River at RM 20.5 and RM 6.5, respectively. In 1992 both facilities were expanded and upgraded, providing advanced secondary treatment with nitrification, chlorination, and dechlorination. The average wet weather design flows are 42 mgd (1.84 m^3/s) at the Blue Lake WWTP and 38 mgd (1.66 m^3/s) at the Seneca WWTP. Monthly average effluent limitations for 5-day carbonaceous BOD ($CBOD_5$) in the summer are 12 and 15 mg/L, respectively, but both facilities consistently produce summer average concentrations below 5 mg/L. Since the mid-1990s, the two facilities have been operated to optimize phosphorus (P) removal, producing annual average effluent P concentrations below 1.8 mg/L. Biological P removal to 1.0 mg/L as an annual average concentration was fully implemented in 2008. Figure 2 displays changes in effluent concentrations for $CBOD_5$, ammonia, nitrate, and total P at the Seneca WWTP from 1985 to 2007. Staged expansion of the capacity at the Blue Lake WWTP to accommodate regional growth is planned.

Figure 2 - Mean Annual Effluent Concentrations at the Seneca WWTP, 1985-2007



Stormwater discharges from the MSP airport are regulated under the National Pollutant Discharge Elimination System (NPDES). Airport stormwater is collected and discharged to the Minnesota River at two outfalls near RM 3.8 and 3.0. Airport stormwater contains CBOD loads from de-icing and anti-icing alcohols (propylene and ethylene glycol). Before recent upgrades, the Metropolitan Airports Commission (MAC) recovered approximately 40% of the total glycol used for deicing; the remainder was lost to the environment. The MSP airport is permitted to discharge 900 short tons (816 metric tons) of $CBOD_5$ per calendar year but exceeded this limit in 2001 and 2002. In 2005 the MAC completed an airfield improvement and expansion plan, which included a number of construction projects to improve the recovery of aircraft de-icing fluids. $CBOD_5$ loads to the river have decreased, and the $CBOD_5$ mass load limit has not been exceeded since 2002 (Section 11.3).

The Black Dog Generating Plant (GP) is a 538-megawatt facility that withdraws cooling water at a maximum permitted rate of 268,175 gpm (16.9 m^3/s) from the Minnesota River near RM 8.8 (see photograph on title page). The facility has an open-cycle cooling system: water is pumped from the river, passed through the facility once, and discharged to Black Dog Lake (500 acres, 2 km^2). Black Dog Lake functions as a shallow cooling lake to reduce water temperature before discharging to the Minnesota River. Cooling water flows by gravity to the west and east ends of the lake. Each end has a controlled weir outlet structure to manage water retention and cooling. The lake discharges to the Minnesota River at RM 10.7 and RM 7.5. Cooling-water requirements vary with energy demand. For example, during April through September 2006, the percent of river flow at RM 39.4 withdrawn by the Black Dog GP at RM 8.8 varied from 1% in early April to 72% in mid-September. A higher portion of the river is withdrawn during hot or cold weather when river flows are low and electrical power demands for heating or air conditioning are greater; these conditions typically occur in winter and late summer.

3 MODEL FRAMEWORK

In 1999 the MPCA and MCES began meeting to share plans and discuss needs for water-quality modeling in the Metro Area. A joint workgroup identified the need to update the WLA/TMDL 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, with the Lower Minnesota River Watershed District, MAC, MPCA, USACE, and USGS as co-sponsors. An interagency group formed to guide the technical aspects. In the first year the group selected a model framework and designed a three-year monitoring program to support it (Larson, 2006). A larger group of stakeholders was engaged to track the progress and provide feedback.

The project proposal outlined the features and capabilities of the water-quality model needed to meet the objectives and priorities (Larson, 2004). The top priority was developing a tool for setting effluent limitations for wastewater treatment facilities and other point sources. Second was determining pollutant load reductions from headwaters and tributaries 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.

As proposed, the model domain would extend from the USGS and MCES monitoring stations at RM 39.4 to the mouth of the Minnesota River. An advanced eutrophication model with good hydrodynamics was recommended to simulate oxygen, nutrient, phytoplankton, and sediment dynamics. The river warranted a time-variable model with the ability to simulate storm events and diel effects, and the model would need two and possibly three dimensions to capture vertical and lateral differences. The model would be calibrated against multiple years representing all seasons and various flow regimes to determine seasonal effects and understand conditions under low, normal, and high flows. The proposal also recommended a model that was well tested, flexible, accepted, suitable, and versatile.

The joint workgroup discussed whether to convert the WLA model (RMA-12, a version of QUAL-II) to a comparable current platform (QUAL2E or WASP) or to extend the HSPF or ECOMSED-RCA model. By 2003 the choice had narrowed to an extension of the ECOMSED-RCA model of the Mississippi River or a new CE-QUAL-W2 model of the lower Minnesota River. CE-QUAL-W2 is a two-dimensional, laterally averaged hydrodynamic and water-quality model supported by the USACE (Cole and Wells, 2008). Both models offered advanced water-quality features and strong hydrodynamics. Extending the HSPF model from Jordan to the mouth was not chosen because it was not well suited to urban watersheds, and the lower Minnesota River displays some of the complex hydrodynamics of an impounded system that the other two models could better address. The group decided that the simpler CE-QUAL-W2 model was appropriate for the lower Minnesota River and the ECOMSED-RCA model was more complex than necessary. Also, federal assistance was available to develop a CE-QUAL-W2 model.

The CE-QUAL-W2 model is well suited for application to the lower Minnesota River because of the following characteristics (Smith et al., 2010):

1. Appropriate for modeling long, narrow water bodies with spatially varying depths

2. Capable of modeling all constituents of concern in the river, including dissolved oxygen, nutrients, phytoplankton, organic matter, and suspended solids

3. Applied successfully to hundreds of aquatic systems

4. Well known, understood, and widely accepted

5. Capable of providing a wide variety of model output for comparison to observed data

6. Capable of simulating various responses due to changes in loads and rates

CE-QUAL-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. Version 3.6 supports a total of over 28 state variables and over 60 derived variables.

In 2005 the Metropolitan Council entered a cost-sharing agreement with the U.S. Army Corps of Engineer (USACE) 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 (Smith et al., 2010). The Lower Minnesota River Model simulates discharge, water elevation, temperature, DO, total dissolved solids, inorganic suspended solids, phosphate, ammonium, nitrate, silica, three groups of BOD, three groups of phytoplankton, and four forms of organic matter. The model grid includes 90 longitudinal segments, with lengths ranging from 134 to 2,321 meters, and accommodates up to 111 vertical layers, with heights ranging from 0.4 to 0.6 meters. In the model of WY 2006, the number of active cells ranges from 507 to 1023.

The user manual for CE-QUAL-W2 recommends calibration against multiple data sets representing a wide variety of conditions (Cole and Wells, 2008). The Lower Minnesota River Model was calibrated against three years with enhanced monitoring, WY 2004-2006, and four earlier years, WY 1988 and WY 2001-2003. In the original scope, model development for the four historical years was scheduled early in the project to inform the monitoring program. As it happened, models for the three more recent years, 2004-2006, were first developed. The recent period did not include an extended period of low river flows in summer, so the drought year of 1988 was selected from the historic record. Water years 2001-2003 were also selected to provide a continuous record and a variety of flows, including a flood in 2001 and summer low flows in 2003. The Minnesota River Basin and Mississippi River models were calibrated against longer periods (1986-2006 and 1985-2006, respectively), but seven years are common among the three models, and the high flow year 2002 and low flow year 2006 were specially targeted in the Lake Pepin TMDL study (LimnoTech, 2009).

Seven monitoring stations were used for evaluating model performance during calibration (Smith et al., 2010). Locations with long-term monitoring data are RM 39.4, RM 25.1, RM 14.3, RM 13.0 (elevation only), RM 11.7 (temperature only), RM 8.5, and RM 3.5. RM 39.4 near Jordan represents the inflow boundary condition, and RM 3.5 near Fort Snelling contains the most complete calibration data set. RM 3.5 was used as the primary calibration site because it is near the mouth of the river, below all major point sources, and in the reach of expected lowest DO concentrations.

To calibrate the model, the strategy was to first develop a set of parameters that produced good model-to-data agreement in WY 2006 and then apply the same set to the other six years (Smith et al., 2010). The parameter set for WY 2006 worked well for all years except the drought year 1988 and summer periods with low river flows in other years. The calibration strategy was changed to focus on performance during summer low-flow conditions. A second set of parameters was developed to meet performance targets in the model of WY 1988 and then this set was applied successfully to WY 2001-2006. The settings for model coefficients are identical in all years with the exception of faster CBOD decay rates for the WWTPs in 1988, which reflect measured changes in effluent characteristics before and after treatment upgrades in 1992.

Calibration results for key variables are presented in Sections 5 to 10. Four model applications are demonstrated in Section 12. For complete information on the modeling project, see Smith et al. (2010). Spreadsheets containing calibration results, post-processing macros, statistical results, and plots are available for each of the seven modeled years. Two-dimensional, time-variable animations of key variables are also available for each year.

4 MONITORING PROGRAM

The Metropolitan Council operates a long-term monitoring program for water quality in Metro-Area rivers with five stations on the Minnesota River at RM 39.4 (Jordan), 25.1 (Shakopee), 14.3 (Savage), 8.5 (Black Dog), and 3.5 (Fort Snelling) (Figure 1). The five stations were initially positioned to monitor water quality upstream and downstream of WWTPs. In addition to the river stations, MCES frequently monitors effluent at the Blue Lake and Seneca WWTPs and the outlets of nine tributaries to the Minnesota River (Figure 1).

The project proposal for the Lower Minnesota River Model recognized the need for additional monitoring to develop the model (Larson, 2004). For example, different sampling stations, variables, and frequencies were needed to better support the model. More intensive monitoring during low river flows in summer would provide data to calibrate the model for critical DO conditions. Special studies were recommended to define key model parameters, such as reaeration rates, and to gauge the importance of potential factors, such as ground water.

In 2003 MCES and partners designed an enhanced monitoring program to support the CE-QUAL-W2 model of the lower Minnesota River (Larson, 2006). The program was implemented over three water years, 2004-2006. Multiple years were chosen to increase the probability of capturing a range of flows, in particular summer low-flow conditions. The program was designed using the following sources:

- General sampling guidelines in the user manual for CE-QUAL-W2 (Cole and Wells, 2008)

- Specific recommendations for the Minnesota River from CE-QUAL-W2 developers and modelers at the ERDC and USGS

- MCES experience with monitoring and modeling rivers for more than 30 years, including a monitoring program to support the ECOMSED-RCA model of the Mississippi River

- Advice from technical partners, especially the MPCA, ERDC, and USGS

- Previous studies such as the plots and sensitivity analysis in the WLA study, which provided guidance on important locations, inputs, and parameters (MPCA, 1985)

The partners reviewed the program as the project progressed, and the monitoring plan was periodically updated.

The monitoring program consisted of three basic elements: 1) base monitoring program, 2) summer low-flow monitoring program, and 3) special monitoring and field studies. The base monitoring program consisted of routine monitoring of the river, discharges, and tributaries year-round over three years to meet model-recommended data requirements. Table 2 summarizes the CE-QUAL-W2 model recommendations and how they were applied to the Minnesota River at RM 39.4 and 3.5 (upstream and downstream model boundaries), 12 tributaries, seven discharges, and one intake.

At the three intermediate river stations (RM 25.1, 14.3, and 8.5), a smaller set of variables were monitored weekly or twice per month. Historic loads from the tributaries were evaluated, and four of the largest contributors were selected for enhanced monitoring. The selected watersheds represented different land uses: two rural (Sand and Carver Creeks) and two urban (Nine Mile Creek and Credit River). Discharge monitoring beyond that required by permits focused on frequent monitoring at the two WWTPs with limited monitoring at the Black Dog GP and MSP airport outfalls. For estimating ratios of ultimate to 5-day CBOD and rates of CBOD decay, 70-day CBOD tests were conducted seasonally from samples collected at the five river stations, two WWTPs, one airport outfall, and four tributaries.

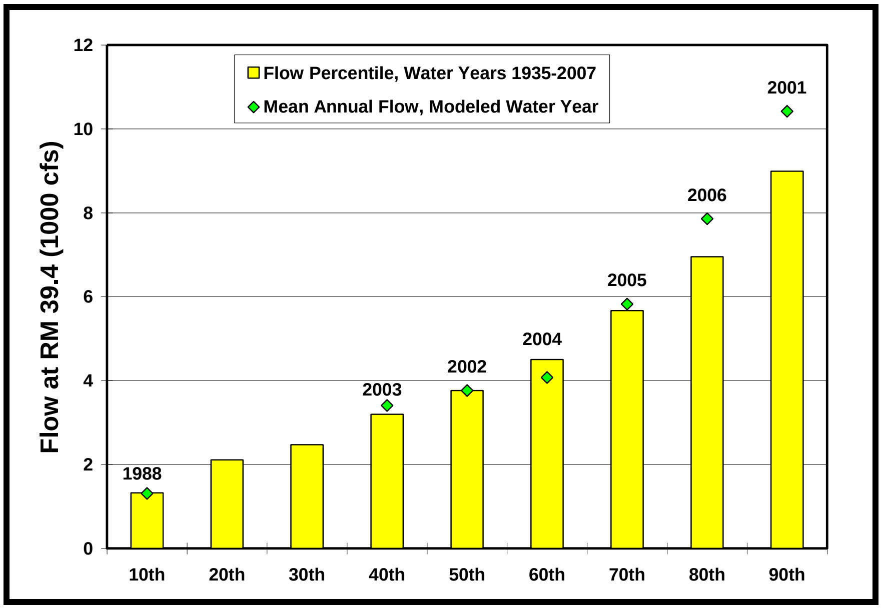
Table 2 - Sampling Stations, Variables, and Frequencies in the Base Monitoring Program

CE-QUAL-W2 Recommendations (Cole and Wells, 2008)			Minnesota River, Tributary, and Discharge Monitoring, Water Years 2004-2006						
			River Boundaries (2 Stations)		River In-Pool (3 Stations)	Tributaries (11 Outlets)	Discharges (7 Outfalls)		
Parameter	**Level[1]**	**Freq[2]**	**Mile 39.4**	**Mile 3.5**	**River Miles[3] 25.1, 14.3, 8.5**	**Full Set at 4; Subset at 7**	**WWTPs[4] (2)**	**Black Dog GP (2)**	**MSP Airport (3)**
Flow	1	D or C	C	C		C	C	C + intake	C
Temperature	1	D or C	C	C	W	C	D	C	W
Conductivity	2	D or C	C	C	W	C	2/M	C, L	
Dissolved oxygen	2	D or C	C	C	W	2/M + S	C	C, L	L
pH	2	D or C	C	C	W	2/M + S	D	C, L	D
Total dissolved solids	2	D or C	2/M + S	2/M	2/M	2/M + S	2/M	L	L at 1
Total organic carbon	1	W + S							
Dissolved organic carbon	2	W + S	W + S	W	2/M	2/M + S	2/M	L	L at 1
BOD and CBOD, 5-day	2	W + S	W ± S	W	2/M	2/M ± S	2/M[4]	L	D
BOD and CBOD, 70-day			4/yr	4/yr	4/yr	4/yr at 4	4/yr		L at 1
Total phosphorus (P)	1	W + S	W + S	W	2/M	2/M + S	2/M[4]	L	L at 1
Soluble reactive P	1	W + S	W ± S	W	2/M	2/M ± S	2/M	L	L at 1
Total dissolved P	2	W + S	2/M + S	2/M	2/M	2/M + S	2/M	L	L at 1
Total and diss. inorganic P	2	W + S							
Total reactive P			2/M + S	2/M	2/M	2/M + S	2/M	L	L at 1
Nitrite-nitrate nitrogen (N)	1	W + S	W + S	W	2/M	2/M + S	2/M[4]	L	L at 1
Ammonium N	1	W + S	W + S	W	W	2/M + S	2/M[4]	L	W
Total Kjeldahl N	2	W + S	2/M + S	2/M	2/M	2/M + S	2/M[4]	L	L at 1
Dissolved Kjeldahl N	2	W + S	2/M + S	2/M	2/M	2/M + S	2/M	L	L at 1
Total suspended solids	2	W + S	W + S	W	2/M	2/M + S	2/M[4]	L	W or M
Volatile suspended solids	2	W + S	W + S	W	2/M	2/M + S	2/M	L	L at 1
Dissolved silica	2	W + S	2/M ± S	2/M	2/M	2/M + S	2/M	L	L at 1
Chlorophyll *a*	2	W + S	2/M ± S	2/M	2/M	2/M ± S	2/M	L	L at 1
Total alkalinity	2	W + S							
Phytoplankton biomass	Pool	M	1-2/M	1-2/M					
Light and vertical profiles	Pool	M	Low flow	Low flow	Low flow				

[1] Level recommended for boundary conditions: 1 minimum parameter, 2 additional parameter.
[2] Frequency: D daily, C continuous, W weekly, + S and storm sampling, 2/M twice a month, M monthly, L limited to < 20 samples or days.
[3] At in-pool stations, monthly sampling is recommended as a minimum, and field measurements and chlorophyll move to level 1 parameters.
[4] These parameters are monitored more frequently (3-5/week) for process control and discharge permits.

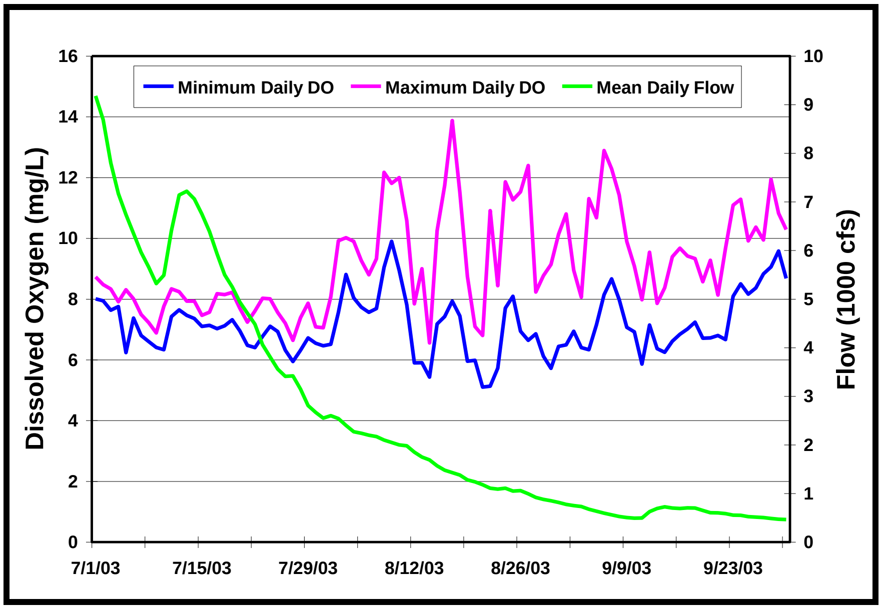
Water years 2004, 2005, and 2006 produced a variety of river flows with mean annual flows ranking near the 50th, 70th, and 85th percentiles, respectively, for the Minnesota River near Jordan, 1935-2007 (Figure 3). Flows during the four earlier years selected for model development (WY 1988 and WY 2001-2003) provided a good complement with mean annual flows ranging from the 10th to 90th percentile (Figure 3).

Figure 3 – Flow Percentiles for 1935-2007 and Mean Annual Flows for Modeled Years



The summer low-flow monitoring program initially targeted a period of 8 to 12 weeks in summer (June 1 through September 15) when river flows near Jordan decreased below 1,000 cfs (28.3 m^3/s). The flow target was based on residence time, effluent-to-river flow ratio at the WWTPs, withdrawal-to-river flow ratio at the Black Dog GP, and expected occurrence of low DO concentrations. The target was later doubled to 2,000 cfs (56.6 m^3/s) based on evidence of increased diel DO fluctuation, indicating phytoplankton activity, and decreased velocities, indicating fine particle settling, under this flow (Figure 4). Velocities at different depths were available from the USGS gaging station at RM 3.5. River flows did not approach the target until late in the third year. The low-flow monitoring program was implemented for seven weeks over the period July 24 – September 15, 2006. Sampling frequency increased to weekly, the number of river stations doubled from 5 to 10, and river samples were collected from a boat. In addition to the Blue Lake and Seneca WWTPs, MCES monitored the airport's main stormwater outfall and Black Dog GP's two cooling-lake outfalls.

Figure 4 – Daily DO Fluctuation at RM 3.5 and Flow at RM 39.4, July-September 2003



The following special monitoring tasks and field studies were considered priorities to be initiated or completed before the three-year monitoring program began in October 2003:

1. Meteorological station in the valley near the river at Fort Snelling
2. Continuous monitoring station for the Minnesota River near Jordan
3. Stream-flow gaging station for the Minnesota River at Fort Snelling
4. Study of mixing characteristics at the five long-term river monitoring stations
5. Rapid assessment of the sediment bed in the lower Minnesota River
6. Determination of ground-water flows to the lower Minnesota River

Items 1-3 were needed to meet basic model data requirements; items 4-6 were needed early in the project to better define the modeling approach and monitoring program. Later, during 2005-2007, the following special studies were completed:

7. Assessment of oxygen dynamics, including major sources and sinks
8. Synoptic survey of diel fluctuations in DO and other parameters
9. Research on nutrient dynamics, including P kinetics, fluxes, and bioavailability
10. Analysis of factors controlling transparency and turbidity
11. Comparison of integrated and discrete water samples

The special projects are summarized in later sections. See the monitoring program (Larson, 2006) or individual project reports for more information (HydrO_2, 2007; James, 2007; and MPCA, 2007c).

4.1 Monitoring Program and Model Performance

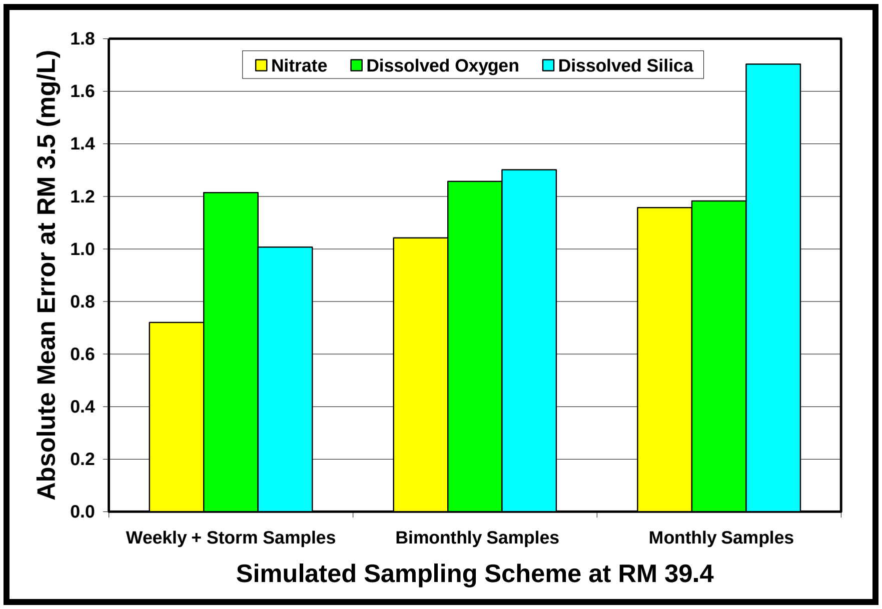
In the case of the lower Minnesota River, did a monitoring program customized to a specific model framework, CE-QUAL-W2, result in better model performance? The ERDC used the model to simulate different monitoring schemes and measure resulting changes in performance. In one test, they measured the effect of sampling frequency on goodness-of-fit statistics in WY 2004. The CE-QUAL-W2 user manual recommends collecting weekly plus storm samples at the model boundaries for analytical tests (Cole and Wells, 2008). At the upstream boundary at Jordan (RM 39.4), MCES collected weekly plus storm samples of the minimum parameters and at least bimonthly (twice per month) plus storm samples of additional parameters listed in the manual (Table 2). Weekly and bimonthly samples are grabs, while storm samples are event composites. Model inputs at the upstream boundary reflected all available water-quality data except continuous DO. The ERDC simulated reduced sampling frequencies at Jordan by first removing all storm samples and then reducing the number of grab samples to two per month in one test and one per month in another test. All other model inputs were left unchanged.

In general, removing inputs from storm samples at the upstream boundary (RM 39.4) did not substantially change model performance at the downstream station near Fort Snelling (RM 3.5). Few storm samples were collected in WY 2004 due to low flows and equipment problems related to reconfiguration after bridge construction. Reducing the frequency of upstream inputs to bimonthly or monthly, however, increased model error for most water-quality parameters at the downstream boundary. Changes in error varied greatly by parameter.

Figure 5 shows the results for three parameters in WY 2004. Decreasing the frequency of upstream inputs to bimonthly and monthly increased the absolute mean error for nitrate nitrogen at the downstream station by 37% and 47%, respectively. Similarly, the error increased by 26% and 51% for dissolved silica under the reduced sampling schemes. In contrast, DO concentrations at RM 3.5 were insensitive to the frequency of water-quality inputs at RM 39.4; the absolute mean error remained about 1.2 mg/L in all three monitoring schemes. On an annual basis, DO was more sensitive to temperature, which was defined at the upstream boundary every 15 minutes. Temperature inputs were not changed in the sampling simulations. Continuous temperature monitoring at Jordan was another element of the enhanced monitoring program.

Other parameters that were relatively insensitive to the upstream sampling scheme in WY 2004 were phosphorus (total and soluble reactive), ammonium, BOD, and chlorophyll. Total dissolved solids were sensitive with the error increasing by 23% when the inputs decreased to bimonthly. Compared to weekly plus storm samples, the error for total suspended solids actually decreased by 20% with bimonthly samples, possibly due to the loss of storm samples, and increased by 25% with monthly samples.

Figure 5 - Model Error at RM 3.5 under Three Sampling Schemes at RM 39.4, WY 2004



(ERDC)

Overall, following model recommendations to increase sampling frequency for key variables resulted in less error in the model results. The addition of model-recommended variables, such as dissolved silica and dissolved organic carbon, also benefitted model performance.

5 HYDRODYNAMICS

Features dominating the hydrodynamics of the lower Minnesota River are flows from the greater watershed upstream of Jordan, channel morphology including modifications for a navigation channel, and pooling effects behind Lock and Dam No. 2 in the Mississippi River. From the modeling effort, Smith et al. (2010) found that river discharge is the main driver of water quality for the majority of the year. At flows greater than approximately 50 m^3/s (1800 cfs), transport dominates water quality. At lower flows, greater depths and slower velocities in the navigation channel increasingly affect sediment, phosphorus, phytoplankton, and oxygen dynamics in the lower Minnesota River, as will be demonstrated in subsequent sections. Impacts of the withdrawal and discharges at the Black Dog GP vary with energy demand and river flow.

The cooperating agencies conducted studies on the river's hydrodynamics to inform the modeling and monitoring efforts. The following sections contain information from studies on mixing characteristics, ground-water inflows, and current velocity and direction. In a separate environmental study over the same period, 2004-2006, Xcel Energy (2007) examined the thermal effects of the Black Dog GP on the Minnesota River.

5.1 Mixing Characteristics

At the beginning of the project, it was not known whether the river was well mixed at different locations and flows or whether dissolved and particulate matter was unevenly distributed across the water column and channel. Information on mixing can help determine an appropriate sampling protocol and modeling approach, such as the number of dimensions and depths of vertical layers. In 2003 and 2004, the USGS conducted a study of mixing characteristics at the five MCES long-term monitoring stations on six dates representing a variety of river flows. Specific conductance, temperature, pH, DO, and turbidity were measured. At each station, parameters were monitored at five points across the channel with measurements taken within a meter of the river surface and river bottom at each point. When differences of a chosen magnitude were noted between the top and bottom measurements, the parameters were measured at 3-ft (1-m) increments along a vertical profile.

In 2006 the mixing study was complemented by a comparison of discrete and integrated samples at two of the monitoring stations. The USGS collected paired discrete and integrated samples at Jordan (RM 39.4) and Fort Snelling (RM 3.5) on eight dates during June-September, 2006. River flows at Jordan ranged from 837 to 12,600 cfs (23.7- 356.8 m^3/s) on the sampling dates. Discrete samples were collected at one meter below the water surface according to MCES protocols. Integrated samples were composited using the equal-width increment method described in the USGS field manual (USGS, 2006). MCES conducted laboratory analyses for suspended solids, nutrients, chlorophyll, and BOD. Along with the paired samples, the USGS took vertical profiles of field measurements along transects at the two stations as in the earlier study.

On the six dates in 2003-04 and eight dates in 2006, vertical differences greater than 20% in turbidity were observed in only 6 of 78 profiles at the two upstream stations, RM 39.4 and RM 25.1 (Table 3). In contrast, vertical differences in turbidity occurred in 52 of 108 profiles at the three downstream sites located within the navigation channel (RM 14.3, 8.5, and 3.5). Differences were measured at all flows at one or more stations; however, they occurred more frequently at RM 3.5 on dates when flows were between 1640 and 4880 cfs (46.4 to 138.2 m^3/s).

From this study limited to 12 dates and 2 or 5 stations, the river was well mixed with respect to turbidity in over 90% of profiles taken upstream of the navigation channel, but vertical differences in turbidity occurred in nearly half of the profiles taken within the navigation channel. Lateral differences in turbidity also occurred more frequently in the navigation channel. Turbidity results may have implications for the mixing characteristics of suspended particulate matter, including phytoplankton, detritus, and inorganic solids.

The MPCA examined the turbidity measurements taken by the USGS on six dates in 2003-2004 (Patrick Baskfield, personal communication). For each date and site, the turbidity measured at the point closest to the center of the channel and near the surface was compared to the mean of all turbidity measurements across the channel. On all dates at RM 39.4 and RM 25.1, turbidity at the center/surface point was within 7.3% of the mean turbidity of all points. On five of six dates at RM 3.5, the center/surface point was within 16% of the mean of all points; however, on one date the difference was 32.2%.

Table 3 - Number of Profiles with Vertical Differences Greater Than 20% in Turbidity

Date	Mean Daily Discharge (cfs) RM 39.4	Monitoring Station, Lower Minnesota River				
		Jordan RM 39.4	Shakopee RM 25.1	Savage RM 14.3	Black Dog RM 8.5	Fort Snelling RM 3.5
9/24/03	554	0 of 5	0 of 5	1 of 5	1 of 5	0 of 5
9/21/06	837	0 of 3				1 of 5
8/21/03	1,160	1 of 3	0 of 2	3 of 3	1 of 5	2 of 5
4/22/04	1,640	0 of 5	2 of 5	3 of 5	4 of 5	5 of 5
7/26/06	1,820	1 of 3				4 of 5
7/12/06	2,600	0 of 5				6 of 6
7/29/03	2,880	0 of 4	0 of 3	2 of 4	3 of 5	3 of 5
8/8/06	2,910	0 of 5				
8/11/04	4,880	0 of 5	0 of 5	3 of 5	3 of 5	4 of 5
6/8/06	6,650	2 of 5				
6/22/06	12,600	0 of 5				1 of 5
6/2/04	16,900	0 of 5	0 of 5	0 of 5	0 of 5	2 of 5

(More shading indicates higher number of profiles with differences, and
Dates and discharges in 2006 are for samples at RM 39.4.)

Vertical differences greater than 0.5 mg/L in DO concentrations were observed at only the three downstream sites at flows less than 3,000 cfs (85 m^3/s) (Table 4). At RM 14.3 vertical DO differences were observed in only 2 of 27 profiles (7%); while at RM 3.5, they occurred in 28 of 71 profiles (39%). Lower river flows and sites with greater water-column depths generally favored vertical DO differences; however, low flows in 2006 produced fewer differences at RM 3.5 than in 2003-2004. MCES later recorded some vertical DO differences at RM 3.5 during limited monitoring in August 2007 and August 2009 at low flows.

During the USGS study in 2003-2004, lateral DO differences occurred most frequently at RM 8.5, which is located near the intake to the Black Dog GP. DO differences of greater than 0.5 mg/L across the channel were measured on at least one date at all stations, especially at lower flows. The DO results may have implications for other dissolved constituents, especially those affected by phytoplankton activity or sediment-bed diagenesis. The only vertical or lateral temperature difference of greater than 2^{o} C was measured in April 2004 at RM 8.5 (Black Dog). No vertical or lateral differences greater than 10% in conductivity or greater than 0.5 in pH were recorded.

While conducting the mixing study in 2003-2004, the USGS took advantage of opportunities to measure changes in turbidity over time as a towboat and barge passed the field crew on five occasions. Responses varied by date, location, and depth; however, turbidity generally increased greatly with barge passage, but the effects subsided within five minutes. On one occasion, turbidity actually dropped from 100 to 80 NTU as the towboat and barge passed.

Table 4 - Number of Profiles with Vertical DO Differences Greater Than 0.5 mg/L

Date	Mean Daily Discharge (cfs) RM 39.4	Monitoring Station, Lower Minnesota River				
		Jordan RM 39.4	Shakopee RM 25.1	Savage RM 14.3	Black Dog RM 8.5	Fort Snelling RM 3.5
9/24/03	554	0 of 5	0 of 5	1 of 5	5 of 5	4 of 5
9/21/06	837	0 of 3				0 of 5
9/6/06	941	0 of 5				0 of 5
8/21/03	1,160	0 of 3	0 of 2	1 of 3	3 of 5	2 of 5
8/22/06	1,350	0 of 3				5 of 5
4/22/04	1,640	0 of 5	0 of 5	0 of 5	5 of 5	4 of 5
7/26/06	1,820	0 of 3				5 of 5
7/12/06	2,600	0 of 5				0 of 6
7/29/03	2,880	0 of 4	0 of 3	0 of 4	1 of 5	1 of 5
8/8/06	2,910	0 of 5				2 of 5
8/11/04	4,880	0 of 5	0 of 5	0 of 5	0 of 5	0 of 5
6/8/06	6,650	0 of 5				0 of 5
6/22/06	12,600	0 of 5				0 of 5
6/2/04	16,900	0 of 5	0 of 5	0 of 5	0 of 5	0 of 5

(More shading indicates higher number of profiles with differences, and Dates and discharges in 2006 are for samples at RM 39.4.)

Two synoptic surveys by the MPCA during summer low-flow conditions in 2006 also provided information on mixing characteristics of the lower Minnesota River (MPCA, 2007c). Field crews deployed sondes to continuously measure DO, temperature, pH, and specific conductance at two depths (centers of the photic zone and water column) at six stations during July 18-24, 2006. Diel DO plots showed differences of less than 0.5 mg/L between the two sondes at RM 39.4. Differences at RM 1.2 varied greatly with cloud cover and other conditions but ranged up to 2 mg/L. The survey was repeated over a longer period during August 30-September 13, 2006. Because the upstream sites were generally well mixed during the first survey, the MPCA moved the uppermost site from RM 39.4 to RM 22.6 and deployed sondes at two depths at only two sites. Differences in DO concentrations between the upper and lower sondes varied over time but were as high as 2 mg/L at RM 11 and 1 mg/L at RM 6.7. Differences between the two sondes were generally less at RM 6.7 possibly due to hydrodynamic effects of the Black Dog GP.

That same summer, June-September, 2006, the USGS collected pairs of discrete and integrated samples at RM 39.4 and RM 3.5 every other week. MCES performed laboratory and statistical analyses to compare water quality in the paired samples. Significant differences can indicate that the river was not well mixed for a particular variable and site on the eight dates. Concentrations of total suspended solids (TSS) and chlorophyll *a* (CHLA) were significantly different between sampling protocols at both sites; measurements from discrete samples were on average less than those from integrated samples. While TSS and CHLA concentrations differed between protocols at RM 39.4, the average ratio between concentrations in the discrete and integrated samples was 0.93 for both constituents, indicating that the difference was small. At RM 3.5, the ratio was similar for TSS (0.92) but lower for CHLA (0.76). Buoyancy or settling may affect phytoplankton distribution at this site. The findings of the USGS-MCES study may have implications for assessments that require representative data for the entire channel cross-section. In the eight paired samples at the two sites in summer 2006, BOD_5 and nutrient concentrations were not significantly different between discrete and integrated samples. Total phosphorus (TP), soluble reactive phosphorus (SRP), total Kjeldahl nitrogen (TKN), nitrate nitrogen (NO3), and ammonium nitrogen (NH4) were analyzed.

As part of a 316(a) demonstration environmental study during 2003-2006, Barr Engineering conducted a detailed study of near-field thermal mixing in the vicinity of the Black Dog GP (Appendix D in Xcel Energy, 2007). It included field measurements during 10 events to support near-field modeling. The goal was to characterize the thermal plume under different conditions. Only two events had measurable plumes (i.e., 5º F isotherm) at the RM 10.7 outfall with lengths of 158 and 246 ft (48 and 75 m). Three events had measurable plumes at the RM 7.6 outfall with lengths of 30, 500, and 4724 ft (9, 152, 1440 m). An event in April 2004 had the greatest thermal effect. As part of their synoptic sonde survey, the MPCA examined near-field mixing of DO below the Seneca WWTP on four dates in late summer 2006 when effluent aeration was required. Bank-to-bank mixing occurred in approximately one mile (MPCA, 2007c).

5.2 Ground-Water Inflows

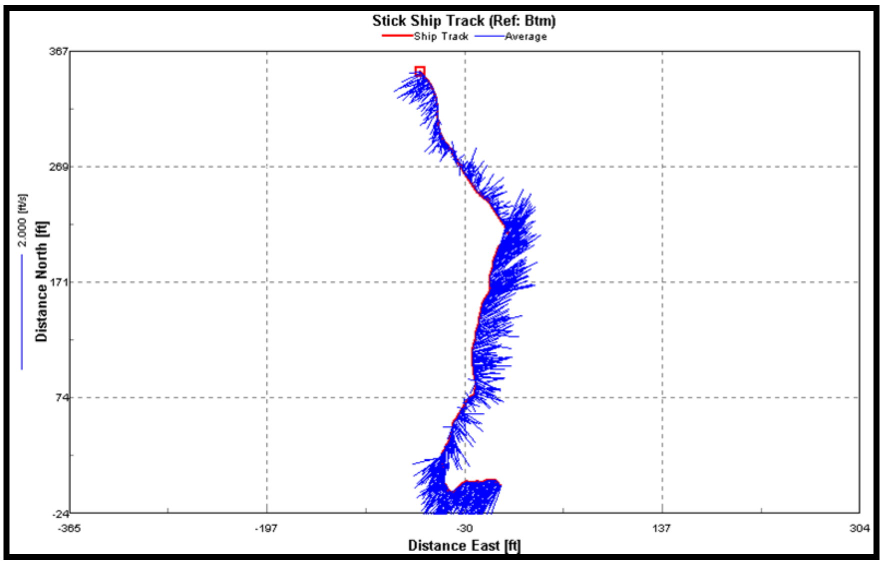
In 2003 project partners recommended an early study of ground-water flows to determine whether this source was important enough to warrant further studies to quantify flows and loads. On September 8 and 9, 2003, the USGS conducted a study of ground-water flows when river flows near Jordan were around 500 cfs (14 m^3/s). An acoustic Doppler current profiler was used to measure flow at 12 locations in the lower 40 miles of the Minnesota River. Sixteen tributaries to this reach were surveyed, and where streams were running, a current meter was used to estimate flow. Flow data for large permitted discharges were measured or obtained. By subtracting upstream river flow plus the sum of tributary and point-source flows from the downstream river flow for each reach, the amount of direct ground-water inflow or outflow was estimated.

The USGS combined these results with those from earlier studies in 1968 and 1997 and found that most gains and losses in flows over the monitored reaches were within measurement error (5-10%). Among the three studies, only five reaches experienced gains over 5%, and all but one gain could be partly explained by other factors. Based on these results, ground water appears to be a minor source of flow to the lower Minnesota River. The partners decided not to conduct further field studies of ground water, and the budgetary analysis later confirmed that ungaged inflows, presumably ground water, were minor (James, 2007). The importance of ground water to water quality in the river was not examined in this project.

5.3 Current Velocity and Direction

During the ground-water study by the USGS in September 2003, conditions were ideal for studying the river at steady low flows; however, the pooling effect behind Lock and Dam No. 2 made flow measurements difficult and unreliable in the lower 20 miles. Figure 6 shows a sample ship track across the river channel near RM 7.3 and the magnitudes and directions of stream velocity measured by the crew. The current direction was east (downstream) in the middle of the channel, but it was west (upstream) near the banks. This may be evidence of backwashing from Pool 2 of the Mississippi River. The field crew noted that the current direction varied from minute to minute and could also be caused by wind or boat traffic. While the USGS failed to measure river flows in the lower 20 miles under these difficult conditions, they succeeded in demonstrating the complex hydrodynamics of this reach at low flows. During their survey at low flows in September 2006, the MPCA crew noted days when even a moderate wind was strong enough to blow their anchored boat upstream against the river's current in the navigational channel (MPCA, 2007c).

Figure 6 - Current Velocity and Direction Along a Transect Near RM 7.3, 9/9/2003



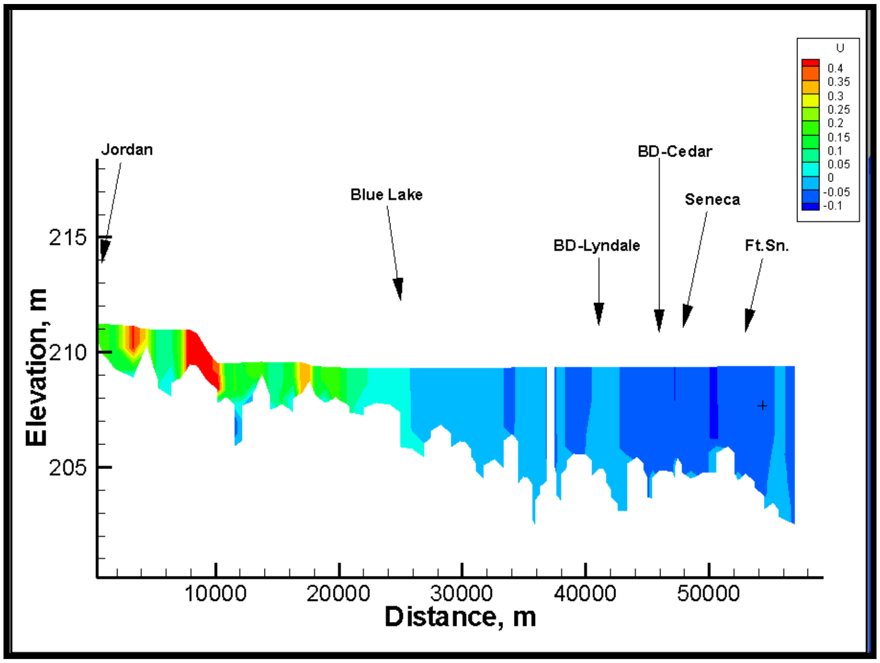
(USGS)

To better understand the navigation channel's effects on current velocities at low flows, the MPCA plotted velocities measured by the USGS as they moved down the river during the ground-water study (MPCA, 2007c). This chart was paired with a plot of channel width and depth at each station. A distinct break occurred at Port Cargill, which is located at the head of the navigation channel. Upstream of Port Cargill on September 8-9, 2003, current velocity was generally greater than 0.3 fps (0.09 m/s), where channel width and depth were less than 275 and 11 feet, respectively. The reverse was generally true downstream of the port where the navigation channel was wider and deeper, producing slower velocities.

Smith et al. (2010) recognized the potential for occasional backwashing from the Mississippi River and, for this reason, defined a downstream model boundary using elevation and water-quality data from mid-Pool 2. Backwashing may occur when there is little or no fall in surface water elevations of the Minnesota River or when there are higher surface water elevations in the Mississippi River. At high withdrawal rates and low river flows, the Black Dog GP can also draw water upstream from the Mississippi River. Figure 7 provides a model snapshot of one backwashing event in September 1988 in which negative or near-zero current velocities are shown in blue as far upstream as the Blue Lake WWTP. This event and others were transitory. According to model results, backwashing occurred most frequently in the low-flow year of 1988.

From results output at RM 3.5 every 0.02 day during WY 1988, negative velocities occurred at a frequency of 2.4%, and velocities less than 0.02 fps (0.006 m/s) occurred at a frequency of 7.7%.

Figure 7 - Backwashing Simulated as Negative Velocity (U, fps) in Model, 9/26/1988



(ERDC)

Responding to energy demands, the Black Dog GP withdraws and discharges various amounts of flow from and to the Minnesota River. This exchange affects the hydrodynamics of the river to different degrees. The Black Dog GP and its effects on the river are further discussed in Section 11.2.

5.4 Model Results for Flow and Temperature

Final calibration results for flow and temperature at RM 3.5 are presented in Figure 8 and Figure 9. In the time series plots, black solid lines represent model output, solid red circles represent measured data, and blue vertical lines represent divisions between water years. The plots present all model output and measured data for seven water years. Three statistics are also provided: mean error (ME), absolute mean error (AME), and root mean square error (RMSE). These statistics were calculated as shown in Equations 1-3 and represent seven-year statistics, not an average of statistics for individual years.

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

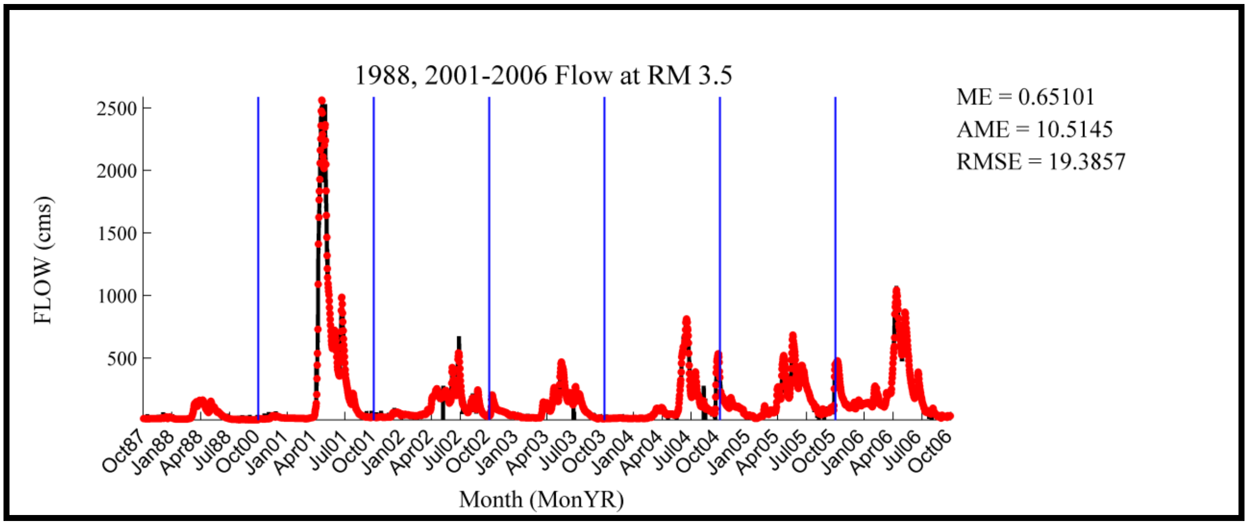
$$AME = \frac{\sum_{1}^{n} abs(model - data)}{n} \quad (2)$$

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

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. A higher target of 1% was set for flow. Results for other locations, one-to-one plots, cumulative distribution plots, and statistics for individual years are provided in Smith et al. (2010).

Figure 8 and Figure 9 show that the model does a very good job of predicting flow and temperature. Both are important factors affecting water quality in rivers. The AME for flow data pairs at RM 3.5 during the seven years is 10.51 cubic meters per second (cms or m^3/s) (371 cfs), which is less than 0.5% of the measured range of flows. Note that the gaging station at RM 3.5 was not installed until January 2004. Before this date, mean daily flows at this location were estimated by lagging flows at Jordan by one day and multiplying by 1.05. Therefore, the better test is how the model performs against measured flows after January 2004. The model also predicts water elevation levels very well. The AME for water elevation at RM 3.5 is 0.09 m (0.3 ft), which is less than 2% of the range of measured surface levels for the seven years. Before January 2004, water elevation at RM 3.5 was estimated from water elevation levels in Pool 2.

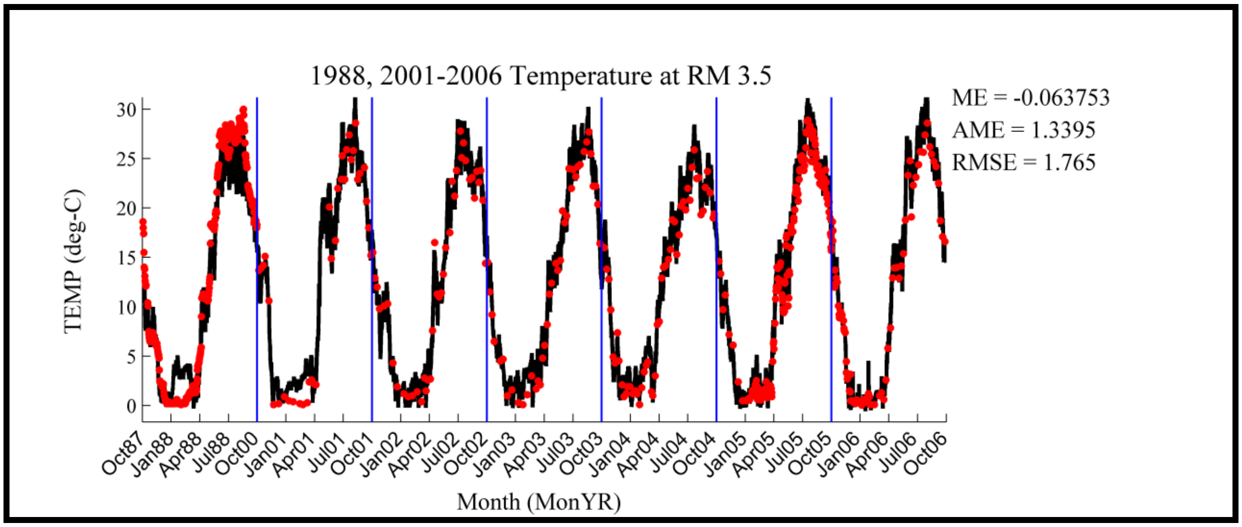
Figure 8 – Model Results for Flow at RM 3.5, 1988 and 2001-2006



(Smith et al., 2010)

The AME for all temperature data pairs at RM 3.5 during the seven years is 1.34º C, which is less than 5% of the measured range of temperatures. During all seven years, temperature was monitored continuously at RM 3.5; however, only a partial record of continuous temperature data was available to define the upstream model boundary at RM 39.4. Xcel Energy provided mean hourly and daily temperatures at RM 11.5 that were used to define the upstream boundary when continuous data were not available at RM 39.4. The CE-QUAL-W2 model also simulates ice formation and breakup, which may help estimate reaeration and DO concentrations in winter. Model results for ice cover generally matched field notes by MCES staff.

Figure 9 - Model Results for Temperature at RM 3.5, 1988 and 2001-2006



(Smith et al., 2010)

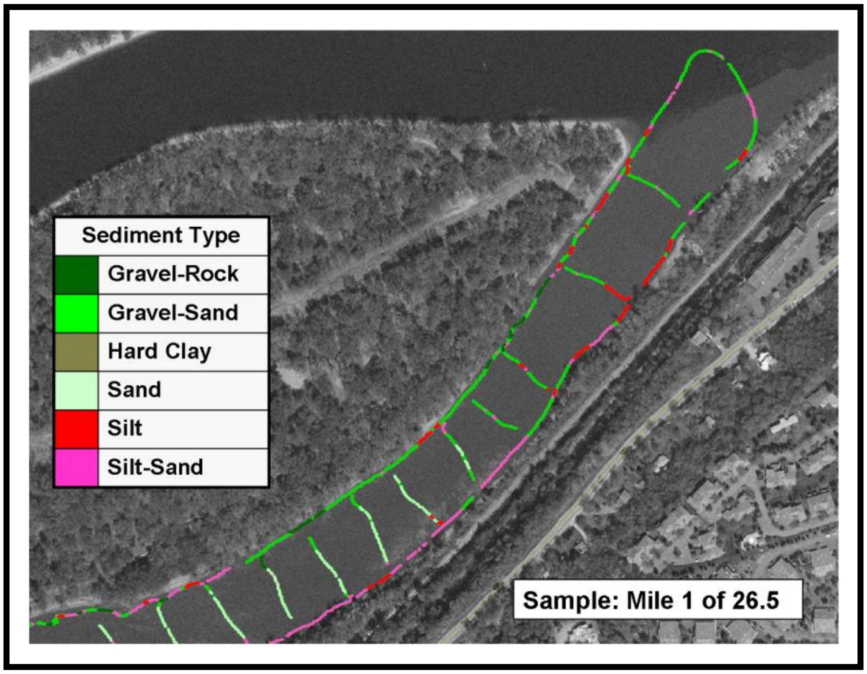
6 SEDIMENT BED

The Minnesota River has been generally characterized as having a sand bed, but fine materials are deposited at reduced flows at various locations. Knowing the distribution of sediment types in the river bed offers a valuable guide for planning other work such as assessments of sediment oxygen demand and nutrient release rates. To determine the locations and depths of deposits of different sediment types, the partners recommended conducting a sediment-bed survey using a rapid assessment technique within the first year of the project (2003).

The USGS executed a sediment-bed survey using continuous seismic-profiling equipment during the week of September 22, 2003, when river flows near Jordan were between 500 and 600 cfs (14 and 17 m^3/s). Conditions were ideal for describing sediment deposition under low flows; however, they limited the survey to the lower 26 miles of the river because the upper reach was not navigable. Profiles were taken along the right and left shorelines and on 720 transects every 200 ft (61 m) in the river channel. Identification of the bed material was field verified by collecting sediment cores and visually comparing the material to samples of known grain size. At these low river flows, a thin layer of silt was observed on the surface of the river bed, but it was too thin for the profiler to detect. Profiling data for the top two detectable sediment layers were later processed by the USGS. No physical or chemical analyses were conducted in the laboratory.

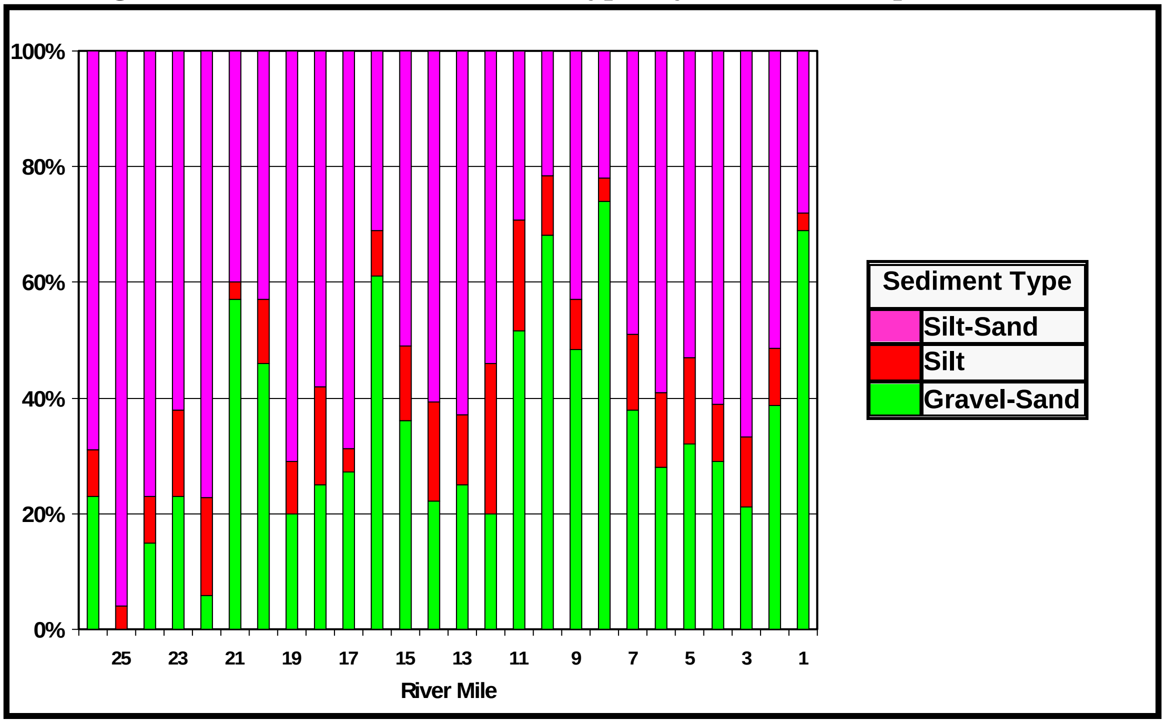
Figure 10 provides a sample of the sediment-bed map created by MCES from the profiling data. MCES compiled summary statistics on the major categories of particle sizes by river mile. The main sediment types in the top layer were silt-sand (57%) and sand-gravel (30%). The thickness of the top sediment layer in the lower 26 miles ranged from 0.02 meter to more than 6.0 meters and averaged around 0.3 meter. MCES also produced maps of channel depths from data provided by the USGS and USACE.

Figure 10 - Sample Map of Sediment-Bed Types, RM 1 to Mouth, September 2003



No strong longitudinal patterns emerged from the sediment-bed assessment; however, some slight spatial differences appeared (Figure 11). Some of the highest percentages of silt (12-26%) occurred at the upper end of the navigation channel (RM 11-15). Greater depths and slower velocities at the head of the channel may have increased settling of fine particles. Near Black Dog Lake (RM 8-11), the percentage of coarse material (sand-gravel) increased. Turbulence from navigation traffic or the Black Dog withdrawal and outfalls may have played a role. The percentage of silt-sand generally decreased between miles 25 and 8 from over 70% to less than 30%, and then it rebounded to 50-70% between miles 7 and 2. In the final mile, the amount of gravel increased, which was corroborated by the MPCA and LMRWD.

Figure 11 - Surficial Sediment-Bed Types by River Mile, September 2003



During the same week as the USGS sediment-bed assessment, the MPCA conducted a one-day visual survey of the sediment bed at six sites between RM 21 and 3.5. They collected grab samples, took photographs, and recorded observations. At most sites, they found a mixture of sand and silt, which agreed with the USGS profiles. At sites between RM 15.3 and 3.5, this mixture was covered by a thin layer, roughly one-eighth inch thick, of "gelatinous pudding-like silt" (Figure 12). One person on the MPCA crew observed little change in the visual appearance of the sediment compared to conditions during a survey in 1980.

Figure 12 - Photograph of Clamshell-Dredge Sediment Sample at RM 3.5, 9/24/2003

(MPCA)

Similar conditions at slightly higher flows were observed during two assessments of oxygen dynamics in 2006. $HydrO_2$ (2007) observed an increase in soft substrate at the sediment-water interface between July and September. River flow at Jordan averaged 2220 cfs (62.3 m^3/s) during the July survey and 966 cfs (27.4 m^3/s) during the September survey. From a preliminary analysis of velocity measurements from the USGS gaging station at RM 3.5, velocity decreases from ~0.6 fps (0.18 m/s) at a flow of 2200 cfs to ~0.2 fps (0.06 m/s) at a flow of 1000 cfs. During two sonde surveys in 2006, MPCA (2007c) also observed increased settling at RM 3.5 and 1.2 in September with "watery fines" deposited on the surface of the river bed. Velocities in the 0.1 to 0.2 fps range result in increased settling of small organic particles in rivers (MPCA, 2007c). High amounts of fine organic material may generate sediment oxygen demand and result in higher phosphate and ammonium releases from the sediment bed.

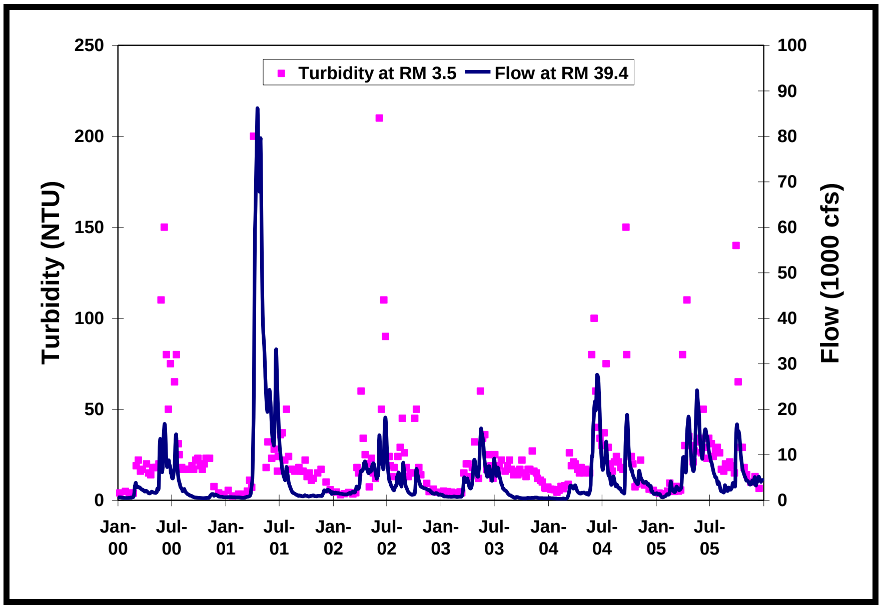
The ERDC studied sediment-bed characteristics when researching nutrient dynamics (James, 2007). They examined the upper 10 cm of 48 sediment cores collected from the lower 26 miles of the Minnesota River in September 2005 and October 2006. The sediment-bed map compiled by the USGS and MCES guided the selection of sites. As found in the previous studies, sand and silt particles dominated the top layer of the sediment bed. Across all cores, the mean percentages of sand, silt, and clay particles were 59.3%, 32.2%, and 8.5%, respectively, but the composition varied from core to core. Organic content was low, averaging only 2.5%. Moisture content and sediment density proved to be good predictors of sediment composition, with moisture content positively correlated with clay and silt and negatively correlated to sand. Sediment density had strong relationships in the opposite direction.

7 TURBIDITY, TRANSPARENCY, AND SUSPENDED SOLIDS

For waters classified as 2B or 2C, the state turbidity standard is 25 nephelometric turbidity units (NTU) (Minnesota State Rules, Chapter 7050). In the Metro Area, the Minnesota River is classified as 2B upstream of RM 22 and 2C from RM 22 to the mouth. Class 2B standards protect cool or warm water fish and associated aquatic life and habitats, while Class 2C standards protect indigenous fish and associated aquatic life and habitats. All forms of recreation, including bathing, are protected in Class 2B waters, while all forms except bathing are protected in Class 2C waters. The Class 2C reach of the lower Minnesota River was defined before the Blue Lake and Seneca WWTPs were upgraded in the mid-1990s. The turbidity standard is intended to protect aquatic life.

In 1996 the lower Minnesota River was added to the state's list of impaired waters for exceeding the turbidity standard. Two large TMDL studies of turbidity in the Minnesota River, Lac Qui Parle to Jordan, and in the Mississippi River, St. Paul to upper Lake Pepin, are currently underway (MPCA, 2005; Tetra Tech, 2008; MPCA, 2007c; LimnoTech, 2009). Figure 13 shows turbidity at MI 3.5 frequently exceeding the standard during six recent years, 2000-2005. During this period, MCES Laboratory Services analyzed turbidity in NTU with a HACH 2100A meter, standard equipment used in Minnesota at the time for assessing waters. The yearly percentage of readings exceeding 25 NTU at RM 3.5 ranged from 11% in 2003 to 47% in 2005. In each year, turbidity exceeded 50 NTU on one or more occasions, usually in concert with peaks in the hydrograph as runoff events delivered suspended-sediment loads.

Figure 13 - Turbidity at RM 3.5 and Flow at RM 39.4, 2000-2005



Turbidity readings for 2006-2009 are not shown in Figure 13 because MCES switched to a different meter in March 2006 and started reporting turbidity in nephelometric turbidity ratio units (NTRU). Different types of meters yield different turbidity readings. For this reason and others, the MPCA is proposing to replace the existing turbidity standards with TSS standards during the triennial rule revisions, 2008-2011. In January 2010, a site-specific TSS standard was proposed for the Mississippi River at Lock and Dam Nos. 2 and 3: summer average TSS concentration of 32 mg/L met in at least half of the summers over a long-term record (MPCA, 2010). Over the recent 10-year period of 2000-2009, the average summer TSS concentration at MI 3.5 ranged from 37 mg/L in 2009 to 164 mg/L in 2004, with a 10-year average TSS concentration of 96 mg/L. The median of the 10 summer means was 86 mg/L—more than two and a half times greater than the proposed standard for Lock and Dam Nos. 2 and 3.

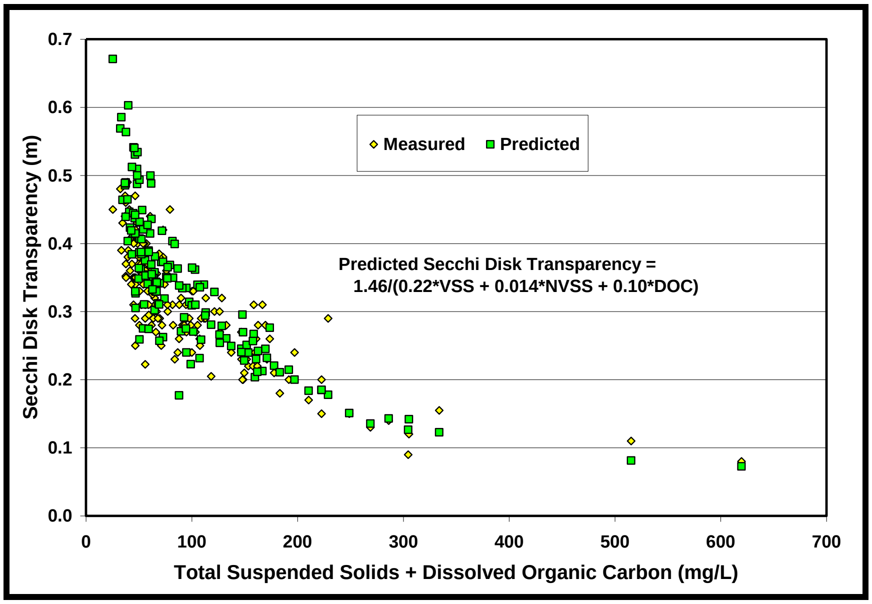
7.1 Suspended Solids as Predictors of Transparency and Turbidity

Light attenuation is an important factor in phytoplankton growth, and the CE-QUAL-W2 model represents this relationship by adjusting the maximum growth rate according to light, temperature, and nutrient availability (Cole and Wells, 2008). The Lake Pepin Phosphorus Study demonstrated the importance of light to phytoplankton growth in the Mississippi River downstream of its confluence with the Minnesota River (Larson et al., 2002). With excessive turbidity affecting light conditions in the lower Minnesota River, this relationship deserved further study.

Using data from the MCES monitoring program, Megard (2007) studied the relationship of suspended particles and dissolved material to transparency and turbidity in the lower Minnesota River. An equation for estimating the extinction coefficient for diffuse underwater light was developed in terms of three attenuators: volatile (organic) suspended solids (VSS), nonvolatile (inorganic) suspended solids (NVSS), and dissolved organic carbon (DOC). Figure 14 shows a similar relationship for another measure of underwater light: Secchi transparency. Secchi transparency is inversely related to the light attenuation coefficient (K, m^{-1}), yielding the equation

$$K = 0.22\ (VSS) + 0.014\ (NVSS) + 0.10\ (DOC)$$

Figure 14 - Transparency and Light Attenuators, Lower Minnesota River, 1996 & 2003-06



(Megard, 2007)

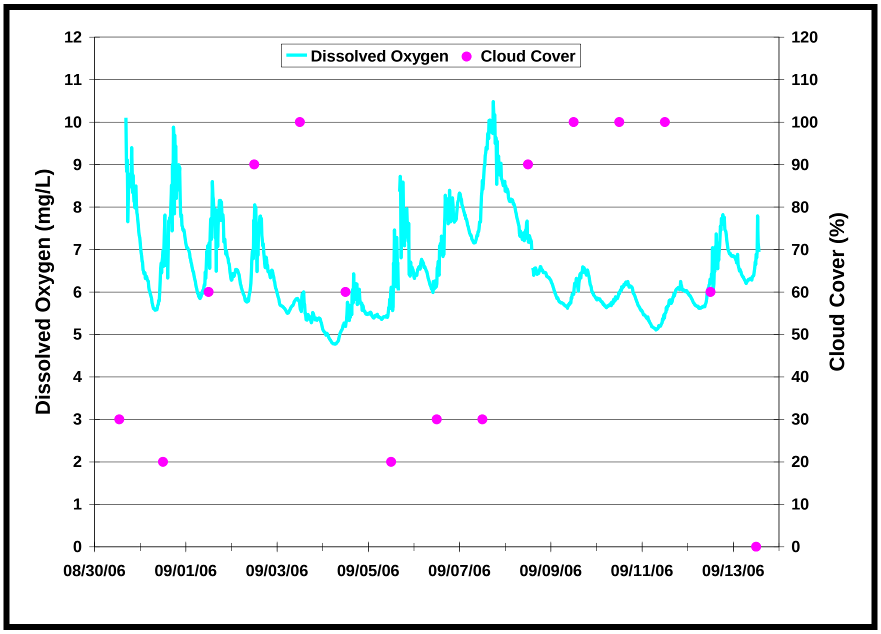
CE-QUAL-W2 contains three nearly parallel extinction coefficients for organic solids (EXORG; default, 0.2 m^{-1}), inorganic solids (EXINOR; default, 0.01 m^{-1}), and pure water (EXH2O; default, 0.45 m^{-1}). In the Lower Minnesota River Model, Megard's coefficients were applied as follows: VSS to EXORG, NVSS to EXINOR, and DOC to EXH20. The DOC coefficient was multiplied by the mean DOC concentration (5.8 mg/L). Megard (2007) showed that DOC concentrations were nearly constant in the lower Minnesota River.

While inorganic suspended solids dominate river concentrations, Megard (2007) revealed that organic suspended solids play an important role in attenuating light (note the larger VSS coefficients in the above equations). Organic solids scatter light more than inorganic solids. With turbidity measurements varying among different meters and protocols, suspended solids can provide a universal translator among turbidity methods. For example, Megard provided the following translation for MCES turbidity using the current analytical method:

$$\text{Turbidity (NTRU)} = 0.80\ (VSS) + 0.46\ (NVSS)$$

Both $HydrO_2$ (2007) and MPCA (2007c) noted that light was an important factor in DO metabolism in the lower Minnesota River. During two $HydrO_2$ assessments, the photic zone was restricted to 2.5-3.0 ft (0.76-0.91 m) or 20% and 31% of the water-column depth in July and September 2006, respectively. This was due to high turbidity from phytoplankton and suspended particles. MPCA (2007c) documented a decrease in diel DO fluctuation with greater cloud cover (Figure 15).

Figure 15 - DO Fluctuation and Cloud Cover at RM 1.2, September 2006



(MPCA, 2007c)

7.2 Model Results for Suspended Solids

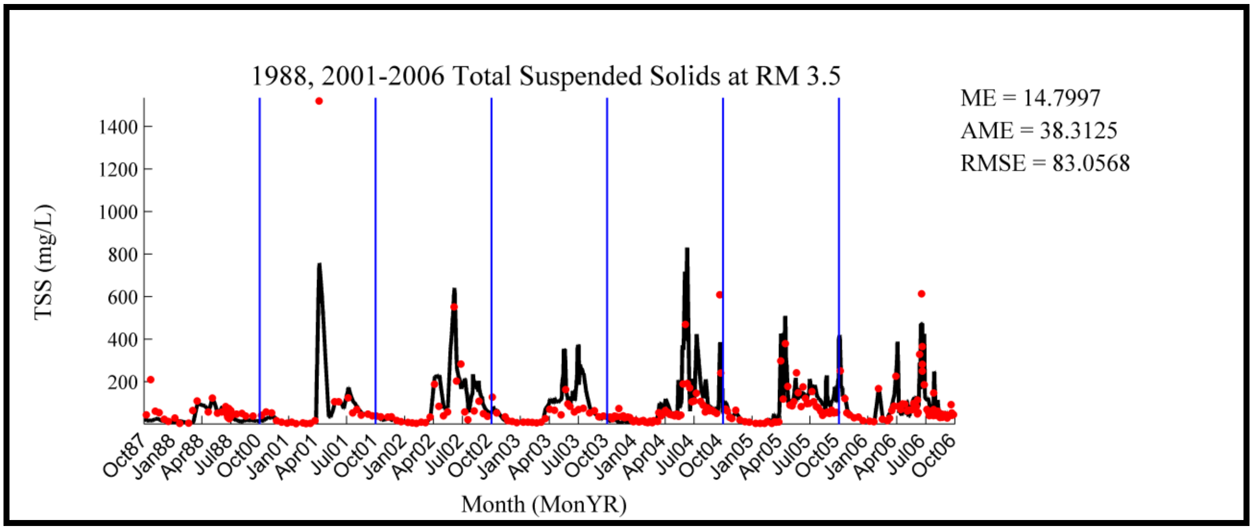
Figure 16 shows the calibration results for TSS at RM 3.5 for the seven years. The model tends to predict the overall trends well. The AME for TSS is 38 mg/L at RM 3.5, which is well below the target of 152 mg/L (10% of the range of measured data). TSS is a derived variable calculated from the following state variables: inorganic suspended solids (ISS = NVSS), nonliving organic matter (four groups), and phytoplankton biomass (three groups). The dominant form of suspended solids in the lower Minnesota River is inorganic. Any number of ISS groups with different characteristics such as settling rates can be defined in CE-QUAL-W2, but only one group is defined in the Lower Minnesota River Model. We did not have sufficient data (e.g., particle sizes) to define multiple ISS groups. Nonliving organic matter represents a wide range of dead organisms and degradation products generated in the river (e.g., bacteria, algae, and fish), on the landscape (e.g., leaves, grass, and crop residue), and at point sources (e.g., treated wastewater).

As described in the previous section, model settings for three light extinction coefficients were based on an analysis of transparency and suspended solids in the lower Minnesota River by Megard (2007). Model output includes the resulting light extinction coefficient for user-selected segments and times. Turbidity in NTU or NTRU can be estimated from model outputs for ISS and TSS using relationships described by Megard (2007) with VSS calculated as TSS minus ISS.

The CE-QUAL-W2 model, version 3.6, does not simulate sediment transport as in the HSPF or ECOMSED-RCA models so deposition and resuspension are not well represented in the Lower

Minnesota River Model. However, suspended solids are adequately modeled for understanding oxygen, nutrient, and phytoplankton dynamics in the lower Minnesota River.

Figure 16 - Model Results for Total Suspended Solids at RM 3.5, 1988 and 2001-2006



(Smith et al., 2010)

8 NUTRIENTS AND SUSPENDED SOLIDS

At this time, Minnesota has narrative but not numeric nutrient standards for rivers; however, numeric standards are currently being developed as part of the state's triennial water-quality rule revisions, 2008-2011. The state has long-standing toxicity-based standards for ammonia nitrogen and drinking-water standards for nitrate nitrogen (Minnesota State Rules, Chapter 7050). The state standard for un-ionized ammonia nitrogen that applies to the lower Minnesota River (Classes 2B and 2C) is 0.04 mg/L as a 30-day average concentration, which protects aquatic life against chronic toxicity. The un-ionized form of ammonia is toxic; the percent un-ionized is calculated from ambient temperature and pH. The ammonia standard must be met at least 50 percent of the days at which the river flow is equal to the lowest 30-day flow with a once in ten-year recurrence interval ($30Q_{10}$). The drinking-water standard for nitrate nitrogen is 10 mg/L, but this applies only to rivers designated for water supply and not to Class 2B and 2C waters. In the triennial rule revisions, the MPCA proposes to add nitrate standards based on aquatic-life toxicity.

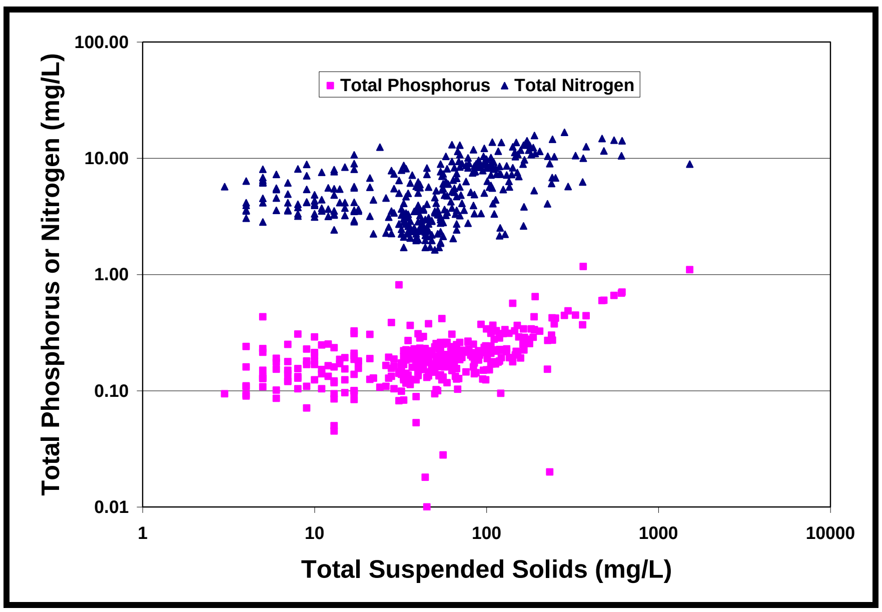
Nutrient concentrations are high in the lower Minnesota River. Table 5 shows the range and quartiles for nitrogen, phosphorus, and suspended-solids concentrations at RM 3.5 for the recent 10-year period of 2000-2009. Values reported as below the detection level were treated as at the detection level (e.g., "< 0.02" was treated as "0.02"). Total nitrogen was estimated by summing total Kjeldahl, nitrate, and nitrite nitrogen. The median TN concentration was 4.81 mg/L with NO3 representing the largest portion. The number of NO3 samples exceeding 10 mg/L was 26 of 351 at RM 3.5. NH4 concentrations were generally low with nearly 25% of the values reported as below the detection level of 0.02 mg/L. The median TP concentration was 0.19 mg/L with the median SRP concentration representing roughly a third (0.066 mg/L). Almost 10% of the SRP concentrations were reported as below the detection level of 0.005 mg/L. The 25th and 75th percentiles for TSS concentrations were 28 and 90 mg/L, respectively.

Table 5 – Nutrient and Suspended-Solids Concentrations (mg/L) at RM 3.5, 2000-2009

	TN	NO3	NH4	TP	SRP	TSS
Minimum	1.63	0.09	0.02	0.01	0.005	3
25th Percentile	3.22	1.72	0.02	0.14	0.027	28
Median	4.81	3.32	0.06	0.19	0.066	47
75th Percentile	7.93	6.56	0.14	0.24	0.101	90
Maximum	16.72	14.70	0.93	1.17	0.371	1520

Figure 17 provides a log-log plot of TP, TN, TSS concentrations at RM 3.5 over the same 10-year period. Phosphorus and nitrogen concentrations appear to increase with suspended solids at TSS concentrations greater than 35 mg/L. At lower TSS concentrations, TP and TN tend to hover around long-term median concentrations.

Figure 17 – Nutrient and Suspended-Solids Concentrations at RM 3.5, 2000-2009



The USACE-ERDC with help from MCES conducted research on nutrients and sediment in the lower Minnesota River (James, 2007). Analytical work was conducted at the Eau Galle Aquatic Ecology Laboratory in Spring Valley, Wisconsin. Research focused on the following tasks:

- Annual nutrient and sediment budgets. Researchers compiled loads for the river, tributaries, and point sources. They identified major loading sources and quantified the portion retained within the lower Minnesota River or exported to the Mississippi River.

- Biologically available phosphorus. The laboratory measured various forms of phosphorus in the river. Soluble reactive P is the form that fuels phytoplankton growth; however, some particulate P forms can readily recycle to biologically available P under certain conditions.

- Phosphorus dynamics. They studied processes that control how much phosphorus is in solution in the river, attached to suspended particles, or incorporated into phytoplankton. Physical, chemical, and biological processes can transform phosphorus.

- Sediment-bed nutrient fluxes. MCES collected sediment cores from the river bed, and the ERDC analyzed sediment characteristics and nutrients. They also measured nutrient release rates from the sediment bed under oxic and anoxic conditions.

Field work and laboratory analyses were conducted in 2005 and 2006. Annual budgets were compiled for water years 2004, 2005, and 2006 from loads estimated with the FLUX program (Walker 1996). The following sections summarize some key findings. For more information, see the full report on nutrient dynamics and budgetary analysis (James, 2007) or article on phosphorus dynamics (James and Larson, 2008).

8.1 Annual Nutrient and Sediment Budgets

The Minnesota River delivers high nutrient and sediment loads to the Mississippi River (LimnoTech, 2009; Kloiber, 2004; Larson et al., 2002). Table 6 lists annual loads at RM 3.5 for WY 2004-2006, which were typical of loads for years with normal flows over the past three decades (Kloiber, 2004). Mean annual flows ranked between the upper 50th and 15th percentiles among historical flows (Figure 3): 4080 cfs (116 m^3/s) in WY 2004, 5830 cfs (165 m^3/s) in WY 2005, and 7860 cfs (223 m^3/s) in WY 2006. During the three-year period, TSS loads from the Minnesota River averaged 740,000 metric tons per year (mt/yr). Much of the sediment is deposited downstream in lower Pool 2 and Lake Pepin (LimnoTech, 2009; Larson et al., 2002). Nitrogen loads averaged 55,000 mt/yr with NO3 as the dominant form, and phosphorus loads averaged over 1,400 mt/yr with nearly one-third as biologically available SRP. Nutrient loads from the Minnesota River contribute to eutrophication in the Mississippi River (LimnoTech, 2009; Larson et al., 2002).

Table 6 – Sediment and Nutrient Loads at RM 3.5, WY 2004-2006

	Load (metric tons per year)					
	TSS	TKN	NO3	NH4	TP	SRP
Water Year 2004						
Minnesota River at RM 3.5	710000	6900	33000	330	1400	410
Water Year 2005						
Minnesota River at RM 3.5	690000	7600	49000	370	1400	510
Water Year 2006						
Minnesota River at RM 3.5	830000	9700	60000	330	1500	450

(James, 2007)

Loads for the Minnesota River at Fort Snelling (RM 3.5) represented outputs in the annual budgets (Table 6). For inputs, the ERDC calculated loads for the Minnesota River at RM 39.4, 11 monitored tributaries (Table 1), and four point sources. The monitored tributaries represent 67% of the total watershed area of the Minnesota River downstream of Jordan. Loads were not estimated for unmonitored areas. Excellent effluent data for all variables were available for the Blue Lake and Seneca WWTPs. Only flow, TSS, and NH4 were frequently monitored at the airport stormwater outfalls during the three years. At the Black Dog GP, MCES collected only 15 sets of water-quality samples at the two outfalls during the summers of 2005 and 2006.

Table 7 shows the relative contribution by each major category (river, tributaries, and point sources) as a percentage of the combined load to the Minnesota River between Jordan and the mouth. James (2007) lists loads from individual tributaries and point sources. The upstream boundary dominated inputs to the nutrient and sediment budgets, with the Minnesota River at Jordan providing over 88% of the TSS, TN, and TP loads. While their annual load contributions were small compared to the Minnesota River at Jordan, the tributaries and point sources delivered some loads more or less out of proportion to their flow contributions. The tributaries as a group delivered higher percentages of NH4 and TP and lower percentages of NO3 than their flow percentages. This simply means that NH4 and TP concentrations were generally higher and NO3 concentrations were lower in the tributaries as a group than in the Minnesota River at Jordan. Nitrogen may have more time to convert from ammonium to nitrate in the river. Compared to other sources, point sources contributed loads higher in NH4 and SRP concentrations, and they delivered negligible TSS loads.

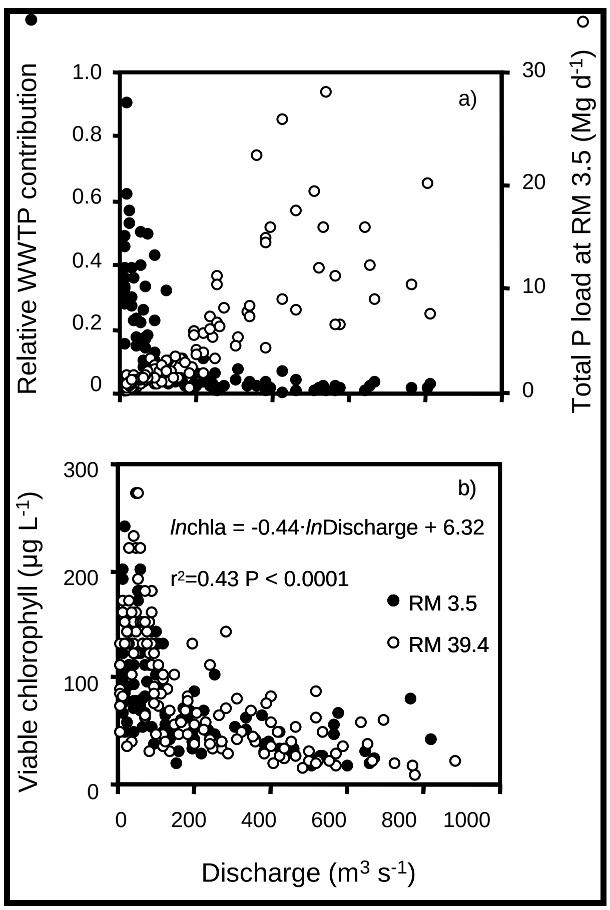
Table 7 – Sediment and Nutrient Inputs from Major Source Categories, WY 2004-2006

	Percent of Total Flow or Load to Lower 40 Miles						
	FLOW	TSS	TKN	NO3	NH4	TP	SRP
Water Year 2004							
Minnesota River at RM 39.4	92.0	95.9	90.8	95.4	84.1	92.2	89.9
Monitored Tributaries	4.2	4.0	6.8	2.0	7.1	5.0	4.2
Point Sources	3.8	0.0	2.5	2.6	8.8	2.7	5.9
Retention (-) or Export (+)	+0.8	-39.2	+4.6	+3.8	+50.1	-4.9	-4.9
Water Year 2005							
Minnesota River at RM 39.4	94.1	96.8	92.4	97.4	71.1	91.2	88.7
Monitored Tributaries	3.2	3.2	4.8	0.9	4.7	4.4	3.8
Point Sources	2.7	0.0	2.7	1.7	24.2	4.4	7.5
Retention (-) or Export (+)	+1.7	-21.8	-3.0	+4.5	+27.6	-8.8	-1.3
Water Year 2006							
Minnesota River at RM 39.4	95.0	91.6	92.9	97.3	89.1	88.4	82.9
Monitored Tributaries	3.6	8.4	5.5	1.2	6.7	7.4	4.4
Point Sources	1.5	0.0	1.6	1.5	4.2	4.2	12.7
Retention (-) or Export (+)	+1.1	-22.0	-3.6	-3.9	+43.2	-10.9	-12.7
July 15–Sept 30, 2006							
Minnesota River at RM 39.4	94.2	85.7	91.4	64.6	65.7	52.0	8.3
Monitored Tributaries	5.8	8.9	5.6	2.1	9.9	7.1	4.9
Sediment Bed Flux						2.1	5.8
WWTPs	5.1	0.1	6.1	45.6	34.2	31.3	74.8
Black Dog GP	-5.0	5.4	-3.0	-12.1	-9.7	7.7	6.3
Retention (-) or Export (+)	+7.0	-36.8	-12.4	+4.5	+365	-8.9	-33.8

(James, 2007)

A detailed budget of FLUX-estimated loads was also compiled for the low flow period of July 15 - September 30, 2006 (Table 7). Results differed from the annual budgets. With less river flow to dilute effluent flows, the relative portion of nutrient loads contributed by the two WWTPs increased greatly. During the 11-week period, the Blue Lake and Seneca WWTPs contributed 34.2, 45.6, and 74.8% of the NH4, NO3, and SRP loads, respectively, while the Minnesota River at RM 39.4 contributed 65.7, 64.6, and 8.3%, respectively. In addition to flows being low at Jordan, concentrations of inorganic N and especially SRP were lower during this period than at higher flows. Figure 18 shows the increasing portion of TP loads contributed by WWTPs as river flows decreased. During the low flow period, the Black Dog GP was a net sink for nitrogen but a net source for suspended solids and phosphorus.

Figure 18 – Flow Relationships with Phosphorus and Chlorophyll, April-Nov., 2004-06



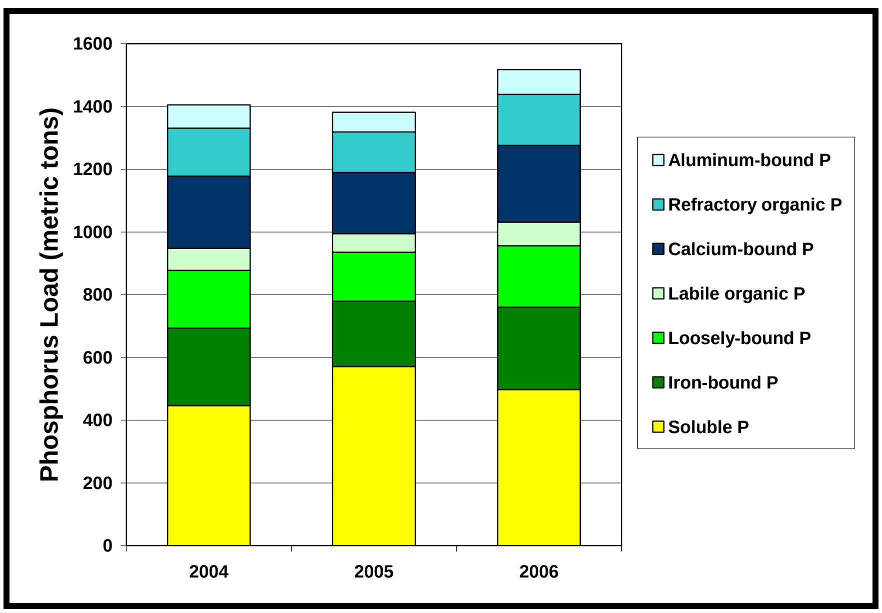
(James and Larson, 2008)

Table 7 also lists the net retention or export of nutrients and sediment over each time period. The percentages were calculated by subtracting the output at RM 3.5 from the combined inputs and dividing the result by the combined inputs. A high percentage of the suspended-solids load was retained and likely deposited in the channel or floodplain of the lower Minnesota River. The percentage ranged from 22% in 2005 and 2006 to 37-39% in 2004 and late summer 2006. A smaller percentage of phosphorus (5-11%) was retained either as sediment deposits or phytoplankton biomass. Most phosphorus was transported downstream to the Mississippi River. In contrast, more NH4 left the reach than entered by percentages ranging from 28 to 50% on an annual basis and by more than three-fold in the summer low-flow period. James (2007) pointed to decaying algal biomass as the likely source of exported NH4 and SRP during the low-flow period.

8.2 Biologically Available Phosphorus

Some particulate P forms (e.g., loosely bound, iron-bound, and labile organic) are more readily recycled to biologically available P, while others (e.g., aluminum-bound, calcium-bound, and refractory organic) are more persistent. The former are termed "biologically labile," while the latter are "biologically refractory." James (2007) demonstrated that during higher flows (i.e., greater than 200 $m^3 \cdot s^{-1}$) and higher TSS loads, approximately one-half of the particulate P load from the lower Minnesota River during WY 2004-2006 occurred as biologically labile forms (labile organic, loosely bound, and iron-bound P segments in Figure 19). Greater than 70% of the TSS load occurred when discharges exceeded 200 $m^3 \cdot s^{-1}$ during WY 2004-2006. These forms easily convert to biologically available SRP under certain conditions, such as iron-bound P transformations under anoxic conditions.

Figure 19 - Phosphorus Load by Fraction at RM 3.5, WY 2004-2006



(James, 2007)

The total P load from the lower Minnesota River in WY 2004-2006 was roughly split into thirds as soluble P, biologically labile P, and biologically refractory P. Because most phosphorus is transported downstream, high levels of SRP and biologically labile P in the lower Minnesota River have important implications for eutrophication in the Mississippi River and Lake Pepin. James and Larson (2008) estimated that the recycling potential of the P load via diffusive sediment P flux under anoxic conditions was approximately 17 $mg/m^2/d$.

8.3 Phosphorus Dynamics

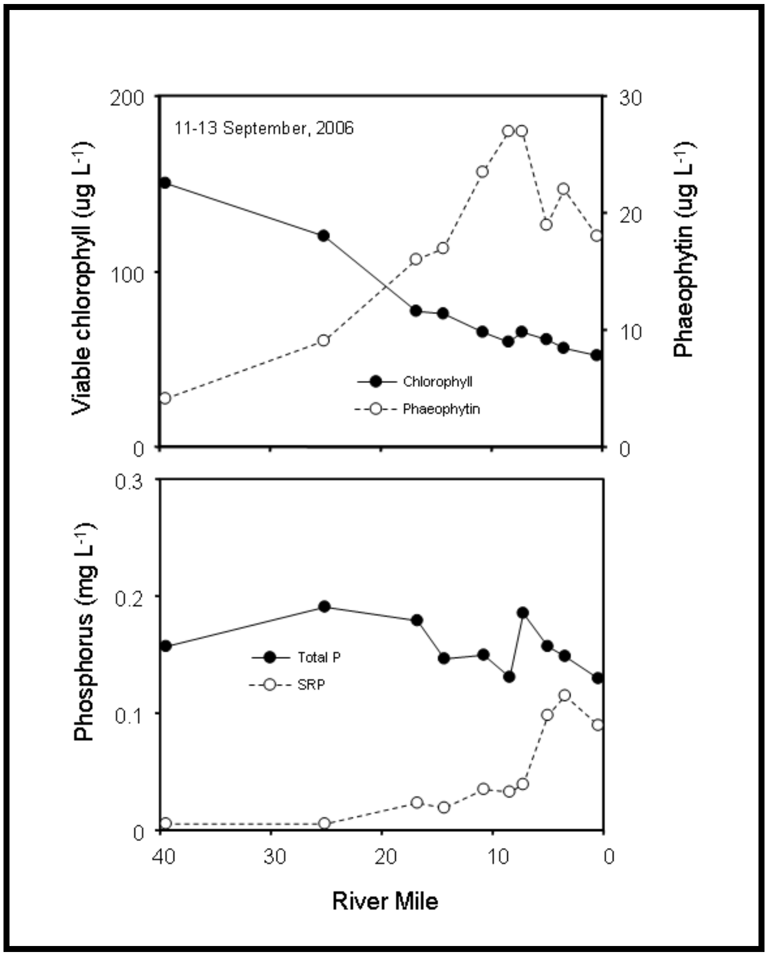
Phosphorus is not static in rivers; its form changes with physical, chemical, and biological processes. The ERDC studied phosphorus dynamics in the lower Minnesota River, especially processes that control transformations between soluble and particulate forms (James, 2007). They attempted to understand what controls SRP concentrations in the river and found different answers at different flows. Phosphorus demonstrated complex seasonal and flow patterns.

At higher flows, SRP concentrations in the river remained fairly stable and at equilibrium with P adsorbed (loosely bound) to suspended solids. Laboratory experiments were conducted using assays with known SRP concentrations to determine the crossover point when SRP started adsorbing to suspended solids collected from the Minnesota River. This point is called the equilibrium P concentration (EPC). Adsorption occurred above an EPC of 0.117 mg/L, while desorption occurred below this point. The EPC was similar to the mean SRP concentration in the river when the suspended solids were collected (0.116 mg/L). Samples were collected at higher flows by design. This suggested that suspended solids controlled SRP concentrations via phosphate buffering under these conditions. The researchers noted that the EPC for the lower Minnesota River was high relative to EPC values reported for other systems. They also noted that phosphate buffering can be an important P source for phytoplankton growth.

During summer low-flow conditions, biological controls became increasingly important in P dynamics. Figure 20 contains a longitudinal plot of CHLA and P concentrations from RM 40 to the mouth during September 11-13, 2006, when river flows were low (~800 cfs or ~23 m^3/s). During this period, concentrations of inorganic suspended solids were relatively low (40-50 mg/L). Between RM 40 and 25, SRP concentrations were very low while CHLA concentrations were very high. This suggested phytoplankton uptake of SRP and conversion to particulate organic P as P was incorporated into phytoplankton biomass.

Between RM 25 and the mouth, the algae increasingly died and decomposed, as evidenced by the decrease in viable chlorophyll *a* (an indicator of living algae) and the increase in phaeophytin *a* (a degradation product of chlorophyll *a*). The mineralization of organic P to inorganic P partially explains the increase in SRP concentrations near the mouth. SRP loads from the WWTPs provide another explanation, with sediment P flux playing a minor role.

Figure 20 - Chlorophyll and Phosphorus Concentrations, RM 40 to Mouth, 9/11-13/06



(James, 2007)

Abiotic controls over SRP concentrations at higher flows shifted to biotic controls at lower flows. Phosphate buffering by suspended solids maintained high SRP concentrations at higher flows, providing a P source for phytoplankton growth. While SRP concentrations became very low in the upper reach during low river flows, they were restored by WWTP effluent loads and algal die-off in the lower reach.

8.4 Sediment-Bed Nutrient Fluxes

The ERDC used information from the sediment-bed assessment to determine representative sampling locations for collecting cores to measure sediment characteristics and nutrient fluxes (James, 2007). Analyses were conducted at the Eau Galle Aquatic Ecology Laboratory. Total sediment P was positively related to sediment moisture, silt, and clay content, suggesting that more flocculent and finer sediments were associated with high P concentrations. The sediment contained mostly refractory P forms, but the most common labile form, iron-bound P, was positively correlated with rates of P release from the sediment. P release rates ranged from 0.7 to 6.5 $mg/m^2/d$ under oxic conditions; rates were 3.7 to 4.8 times higher under anoxic conditions.

Total and exchangeable nitrogen (N) in the sediment were also positively related to moisture, silt, and clay content. Bacterial nitrification occurred under oxic conditions, converting most ammonium to nitrate, so ammonium release occurred primarily under anoxic conditions. Anoxic NH4 release rates ranged from 2.2 to 41.8 mg/m^2/d. Unlike phosphorus, nitrogen release rates were not significantly correlated to sediment characteristics or exchangeable N concentrations.

Using the sediment-bed map and regression relationships among the sediment characteristics, the ERDC estimated that SRP release rates averaged 21.3 mg/m^2/day under anoxic conditions and 4.0 mg/m^2/day under oxic conditions in the lower 26 miles. Estimated SRP loads contributed by the sediment bed to the lower Minnesota River during the low flow period of July 15 – September 30, 2006, represented only 5.8% of the total SRP budget (Table 7). With increasing deposition of fine particles under lower flows, this portion might increase but is estimated to remain under 10% of the total P load at RM 3.5. At flows greater than 7000 cfs (200 m^3/s), the sediment bed contributes less than 1% to the P budget.

HydrO$_2$ (2007) conducted a pilot study in which they measured sediment nutrient fluxes in the field. While more difficult, field measurements involve less disturbance of the sediment and provide more natural conditions than laboratory measurements. On September 4, 2006, nutrient samples were collected from special chambers positioned on the river bed at RM 11.2 to measure sediment oxygen demand. The mean SRP release rate was 5.4 mg/m^2/d under oxic conditions, which is similar to the rate estimated by James (2007) for sand-silt sediments at RM 11.5 (4.8 mg/m^2/d). The mean NH4 release rate from the chambers was 48.1 mg/m^2/d. If representative of a larger area, this sediment release rate could represent a significant NH4 source to the river. Not enough data was collected to estimate the sediment-bed contribution to the NH4 budget.

8.5 Model Results for Phosphorus and Nitrogen

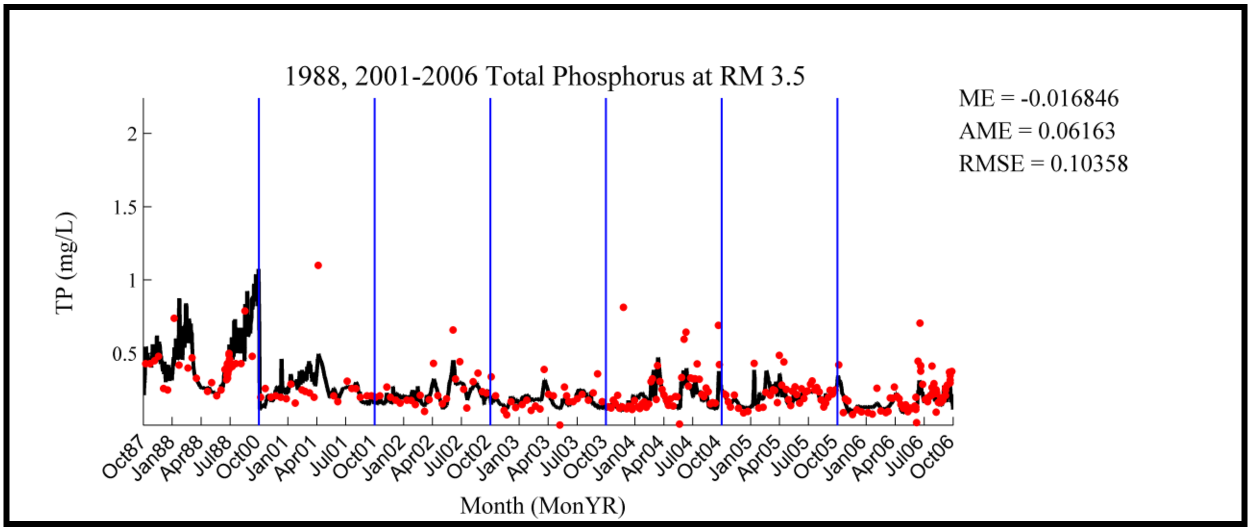
The CE-QUAL-W2 model, version 3.6, includes a state variable for SRP (PO4) and simulates various forms of particulate organic P, but it only partially supports particulate forms of inorganic P (Cole and Wells, 2008). CE-QUAL-W2 includes simple phosphorus sorption to particles and subsequent settling, but the model does not support desorption and more complex kinetics. For this reason, the phosphorus portioning coefficient (PARTP) was set to zero in the current application (i.e., particulate inorganic P was not modeled). The Lower Minnesota River Model includes the state variable PO4 and simulates particulate organic P associated with state variables for CBOD, nonliving organic matter, and phytoplankton.

Sediment phosphate and ammonium release can be represented in the model as a zero-order process, as a first-order process linked to organic matter settling and decay, or as a combination (Cole and Wells, 2008). In the Lower Minnesota River Model, it is modeled as a zero-order process that is linked to sediment oxygen demand (SOD) rates, temperature, and DO concentrations. Nutrient release rates are specified as a fraction of SOD rates: 0.010 for NH4 and 0.001 for PO4. The DO half-saturation constant (O2LIM) for aerobic processes was set to 0.1 mg/L.

Figure 21 and Figure 22 show the calibration results for TP and SRP (PO4) at RM 3.5. TP is a derived variable calculated from SRP and the particulate organic P forms mentioned above. The model tends to slightly under predict TP at all locations. However, the AME is 0.10 mg/L at RM

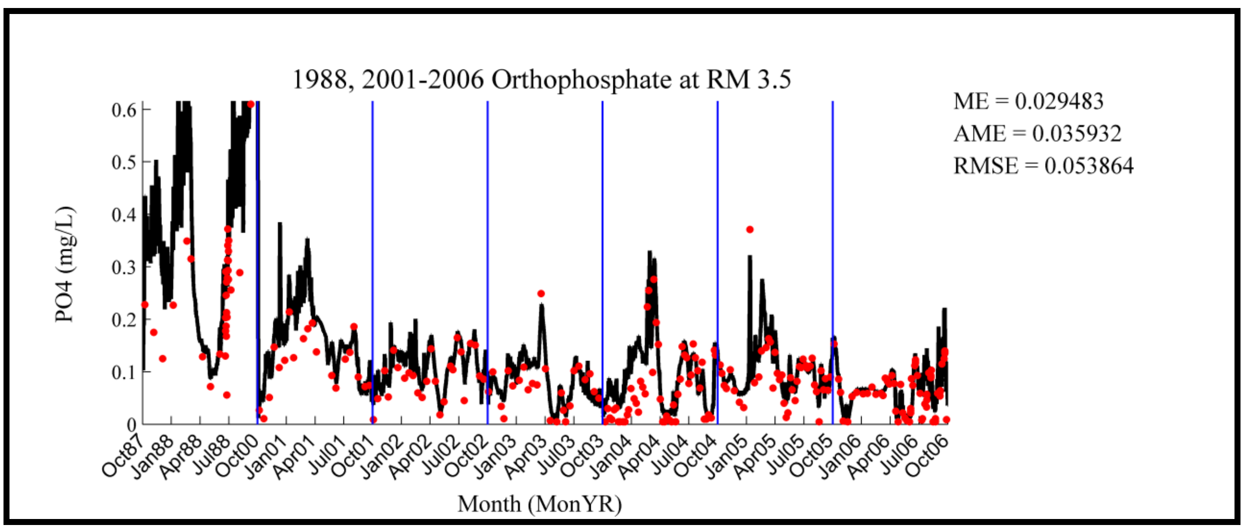
3.5, which is below the target of 0.11 mg/L (10% of the measured data range). The model does a good job with SRP predictions. At Fort Snelling, the AME is 0.04 mg/L, which is well below the target of 0.06 mg/L. The model tends to over predict SRP at the three downstream stations (RM 14.3, 8.5, and 3.5). This may be due to the SRP:TP ratios used to estimate SRP from effluent TP at the WWTPs when SRP was not available. Ratios of 0.90 and 0.81 were assigned to Blue Lake and Seneca, respectively, from linear regressions of SRP and TP (2004-2006). The equations over predict effluent SRP at low TP concentrations. The median effluent SRP:TP ratio during this period was 0.6 (see Section 11.1). River TP and SRP concentrations decrease between 1988 and 2001-2006, especially in late summer. This is partly due to lower river flows and less dilution in 1988, but it may also be partly due to reduced effluent loads.

Figure 21 - Model Results for Total Phosphorus at RM 3.5, 1988 and 2001-2006



(Smith et al., 2010)

Figure 22 - Model Results for Orthophosphate at RM 3.5, 1988 and 2001-2006



(Smith et al., 2010)

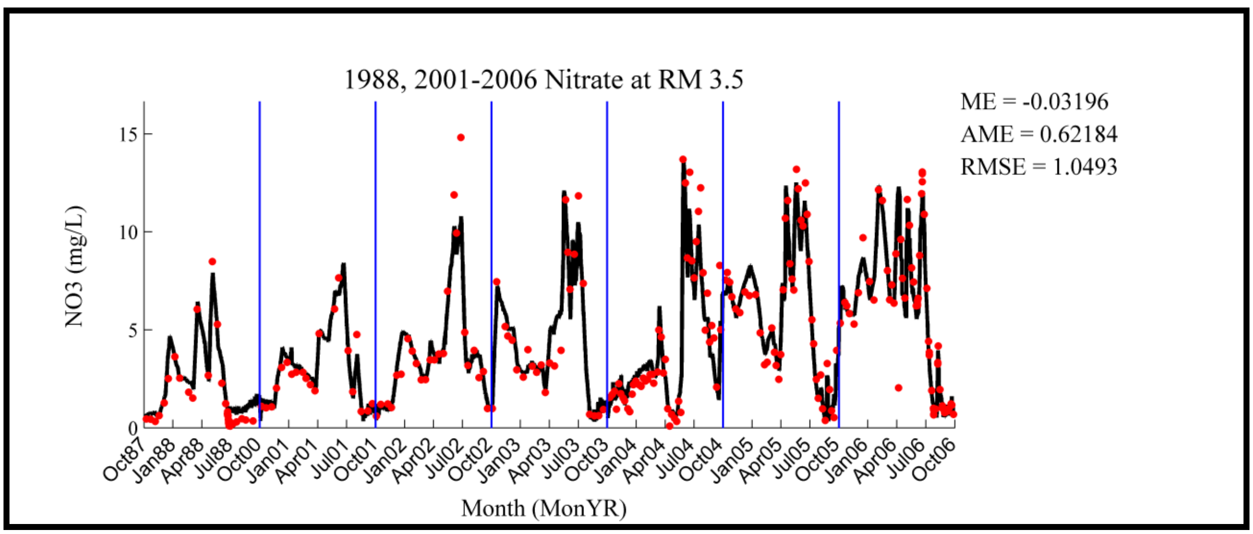
CE-QUAL-W2, version 3.6, allows simulation of inorganic N as nitrate-nitrite and ammonium and organic N associated with CBOD, nonliving organic matter, and phytoplankton. In the current model's inputs and outputs, "NO3" includes both nitrate and nitrite nitrogen. When NH4 was reported as below the detection limit (roughly a quarter of samples at RM 3.5 in 2000-2009), the value was set to the detection level of 0.02 mg/L. The below-detection values were initially set to zero, but the model produced better results when the values were set to 0.02 mg/L.

Figure 23 and Figure 24 show the calibration results for NO3 and NH4 at RM 3.5 for seven years. The user can also request model output for various derived N variables; for example, TKN was compared against measured data during the calibration. The model tends to do very well with NO3 predictions. At Fort Snelling, the AME is 0.62 mg/L, which is well below the calibration target of 1.46 mg/L (10% of measured data range).

As with most variables, the AME for NH4 increases as the river approaches the mouth; however, even at Fort Snelling, the AME is 0.12 mg/L, which is much less than the target of 0.25 mg/L. During the summer of 1988, the model under predicts NH4 beginning at RM 14.3, which was downstream from Blue Lake and two smaller wastewater treatment plants (later closed). Effluent NH4 loads were well defined in all years, so nitrification or another rate may have been different in the river in 1988. Model coefficients for all years are identical with the exception of faster CBOD decay rates for the WWTPs in 1988. The model also did very well with TKN predictions. At Fort Snelling, the AME is 0.32 mg/L, which is below the target of 0.47 mg/L.

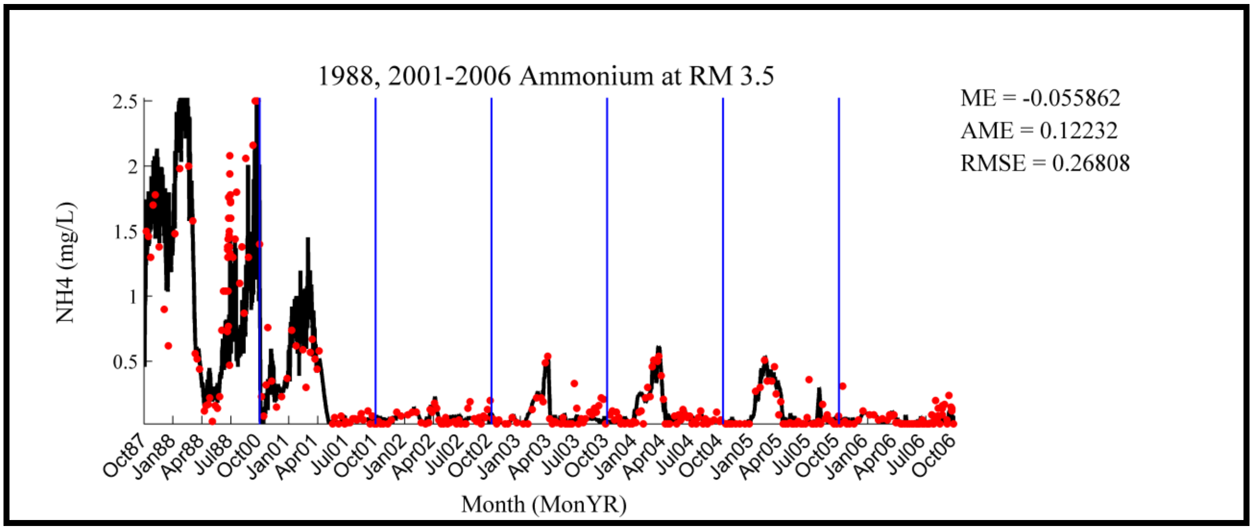
Note the differences in water quality between 1988 and the later years. The addition of nitrification at the WWTPs in the mid-1990s partially explains the decrease in NH4 concentrations and increase in NO3 concentrations in the river. For estimating the concentrations of un-ionized ammonia (toxic form), the user could combine model-estimated NH4 with continuously measured temperature and pH at RM 3.5. Temperature is simulated in the model and pH can be simulated, but adequate data (e.g., total inorganic carbon) were not available to support pH.

Figure 23 - Model Results for Nitrate Nitrogen at RM 3.5, 1988 and 2001-2006



(Smith et al., 2010)

Figure 24 - Model Results for Ammonium Nitrogen at RM 3.5, 1988 and 2001-2006



(Smith et al., 2010)

9 PHYTOPLANKTON

The State of Minnesota currently has numeric eutrophication standards for lakes but only narrative standards for rivers and other water bodies. As part of triennial rule revisions during 2008-2011, numeric water-quality standards for nutrients and their impacts to river and stream ecosystems are being developed, with promulgation expected in 2011. While the lower Minnesota River is not yet formally listed as impaired due to excess nutrients, several studies have documented the highly eutrophic conditions of this reach.

Following are three examples of the studies. In the second phase of the DO TMDL study of the lower Minnesota River, phosphorus load reductions were targeted because the study identified phosphorus as the cause of high levels of algae whose respiration and decomposition produced high BOD and low DO concentrations (MPCA, 2004). Sixty miles downstream of the confluence of the Minnesota and Mississippi Rivers is a 20-mile-long natural impoundment, Lake Pepin, which is listed as impaired for excess nutrients. By modeling the system, LimnoTech (2009) demonstrated that upstream loading of algal biomass from the Minnesota River was an important source of algal biomass to Lake Pepin, and upstream loading exceeded in-lake productivity even during low-flow summer periods. Finally, in a study of phosphorus and chlorophyll relationships in 116 temperate streams around the globe, Van Nieuwenhuyse and Jones (1996) found the highest summer average chlorophyll concentration in the Minnesota River at Jordan (total chlorophyll *a*, 170 µg/L, 1976-1992).

The current project did not include a special study focused strictly on phytoplankton dynamics; however, all studies touched on phytoplankton to some degree. The following sections examine results for two phytoplankton measures, chlorophyll *a* and biomass, and explore their seasonal and spatial patterns and their relationships to temperature, light, and nutrients.

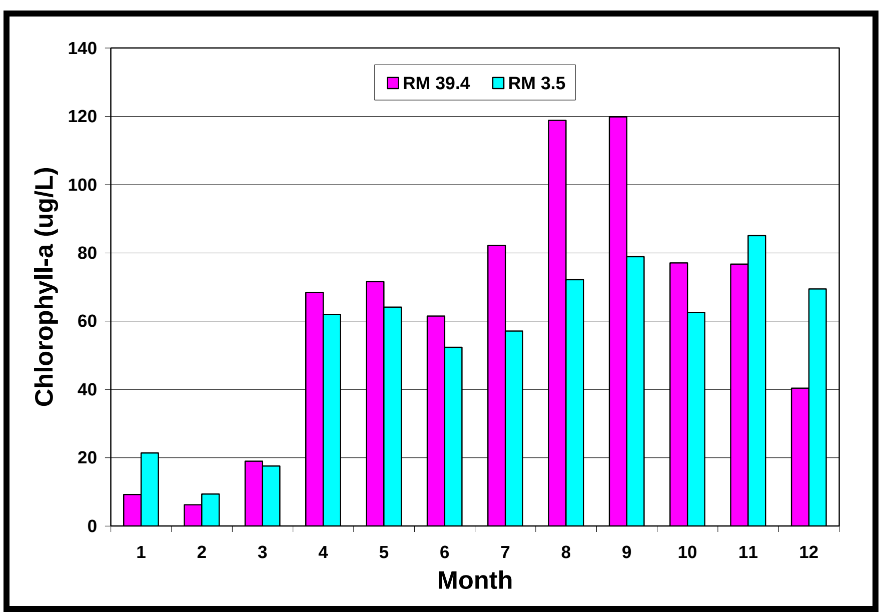
9.1 Chlorophyll *a* Patterns and Relationships

Figure 25 displays mean monthly chlorophyll *a* concentrations at RM 39.4 and RM 3.5 for the period 2003-2009. Unless noted, "chlorophyll *a*" (CHLA) means phaeophytin-corrected chlorophyll *a*, which is associated with living algae. This period was chosen because MCES used a consistent analytical method (modified monochromatic) to determine CHLA throughout the seven years. Algal levels naturally change from month to month in response to changes in water temperature, river flow, and solar energy.

At both monitoring stations, mean monthly concentrations were low during January through March and then peaked above 60 μg/L in April and May during 2003-2009. As references, the state standard for CHLA is less than 30 μg/L as a summer average in shallow lakes of southwestern Minnesota, and the CHLA goal in the DO TMDL study is 57 μg/L as a low-flow summer average at Jordan. The mean monthly concentration declined somewhat in June but then reached a second higher peak in August and September that exceeded 100 μg/L at Jordan. In a lake, this level would indicate hypereutrophic conditions. During the seven-year period, concentrations at RM 39.4 exceeded 200 μg/L on specific dates in April 2004, August 2006, and August 2008.

Starting in September, seasonal patterns at the two stations diverged. At RM 39.4, mean monthly CHLA concentrations dropped from August to December during 2003-2009. $HydrO_2$ (2007) found that solar energy in terms of visible light energy per day was more than twice as high during their survey in July 2006 than when they returned in September. They cited decreased solar energy as one of the factors restricting gross primary production in the September survey. Mean monthly CHLA concentrations at RM 3.5 were lower than at RM 39.4 during most of the year, especially July-September, but they rose to a third peak above 80 μg/L in November and exceeded mean concentrations at Jordan in November through February.

Figure 25 - Mean Monthly Chlorophyll *a* Concentrations at RM 39.4 and 3.5, 2003-2009



During 2003-2009, mean annual CHLA concentrations were 67 and 56 μg/L at RM 39.4 and 3.5, respectively, and summer means were 95 and 65 μg/L. For the period 1976-1996, Meyer and Schellhaass (2000) reported mean annual chlorophyll *a* concentrations of 60 and 53 μg/L at RM 39.4 and 3.5, respectively. They examined total chlorophyll *a* (trichromatic method, not corrected for phaeophytin *a*) and flow-weighted concentrations, but results for the recent seven-year period are similarly high. The earlier report observed that concentrations peaked in May and October at RM 3.5.

Algal levels also change in response to river flows. Figure 18 in Section 8.1 shows CHLA concentrations decreasing with increasing flows during April-November, 2004-2006. James (2007) attributed this response to flushing and cellular washout at higher flows. During the ice-free months, CHLA concentrations were generally highest during periods of lower flow.

At lower flows in late summer, the decrease in CHLA concentrations between RM 39.4 and 3.5 becomes more apparent. For example, Figure 20 in Section 8.3 shows that CHLA concentrations associated with living algae decreased from 150 μg/L at RM 39.4 to 22 μg/L at RM 3.5 during September 11-13, 2006, while phaeophytin *a* associated with dead algae increased from 2.7 to 39.3 μg/L. This represented a drop in the percent viable CHLA from 97% at Jordan to 72% at Fort Snelling. MPCA (2007c) reported no correlation between CHLA concentration and river mile in July 2006, but they observed CHLA decreasing and phaeophytin *a* increasing in a downstream direction at lower flows in September 2006.

The senescence and settling of phytoplankton contributes to oxygen demand in the water column and sediment in the lower reaches of the Minnesota River. Smith et al. (2010) demonstrated in the model that algal respiration was also an important component of the DO deficit in the navigation channel during July-September 1988 and July-September 2006 (see Section 10.6). Greater depths in the channel lead to slower velocities and increased settling of phytoplankton, especially the heavier diatoms. MPCA (2007c) noted decreasing DO and pH from upstream to downstream stations during the sonde surveys, especially in September 2006, which may be explained by decreasing algal activity.

During oxygen assessments in July and September 2006, HydrO_2 (2007) found the photic zone limited to roughly one meter or 20-31 percent of the water column, which limited the growth potential of the phytoplankton community. In the channel, phytoplankton can settle or be mixed below the photic zone and become light limited. James (2007) found an inverse relationship between flow and CHLA concentrations and attributed this pattern to the influence of flow and residence time on algal growth and washout.

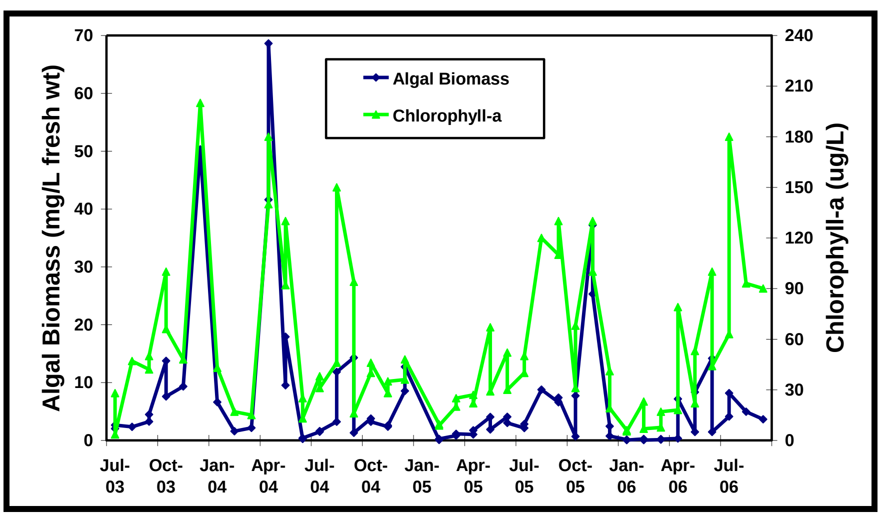
James (2007) examined relationships among phosphorus, suspended solids, and phytoplankton during 2004-2006 (see Section 8.3). At higher flows, equilibrium relationships between phosphorus and suspended solids appeared to maintain average SRP concentrations at 0.115 mg/L. At lower flows, phytoplankton dynamics became more important with growth and uptake reducing SRP concentrations to near zero at RM 39.4 as in late summer 2006 (Figure 20 in Section 8.3). On several occasions during 2004-2006, James (2007) found SRP declines from ~ 0.100 to < 0.010 mg/L on the falling limb of hydrograph peaks. On these occasions, the molar nitrogen-to-phosphorus ratios were greater than 50 suggesting potential P limitation in the upper reach between RM 39.4 and 25.1. However, effluent loads from the Blue Lake WWTP at RM 20.5 replenished SRP concentrations, which were probably assimilated by algae for growth downstream of the facility. Senescence exceeded growth in the lower reach in late summer 2006 as indicated by increasing phaeophytin *a* concentrations. Through senescence, particulate organic P in the form of algae at Jordan eventually converts to SRP downstream in the water column or sediment bed. Additional contributions from the Seneca WWTP, algal senescence, and sediment bed led to SRP concentrations greater than 0.100 mg/L downstream of RM 7.2 as seen in September 2006.

9.2 Phytoplankton Biomass and Groups

MCES collected phytoplankton samples at RM 3.5 from July 2003 through September 2006 and at RM 39.4 from February 2005 through September 2006. Dr. Jeffrey Janik, a consultant who has analyzed many samples from the Mississippi River and Lake Pepin, identified and counted the phytoplankton to the species level and measured biovolumes. Assuming a specific gravity of 1.0, biomass (µg/L fresh weight) is equivalent to the biovolume (mm^3/m^3). Species identifications and biomass estimates were also available from Dr. Janik for samples collected at RM 3.5 during January-September 1996.

Figure 26 plots phytoplankton biomass and CHLA at RM 3.5 over the three years. Biomass generally peaked in the cooler months of spring and fall. The three highest values that were greater than 30,000 µg/L occurred on 5/2/96, 12/5/03, 4/15/04, and 11/4/05, which corresponded to CHLA concentrations greater than 120 µg/L. During some summers, biomass grew higher than 10,000 µg/L as in 1996, 2004, and 2006. Biomass levels at RM 39.4 generally tracked closely with RM 3.5 during 2005 and 2006, but there were days in the late summer when biomass at Jordan exceeded biomass at Fort Snelling. This reinforces the observation that phytoplankton levels decline from RM 40 to the mouth of the Minnesota River under summer low-flow conditions.

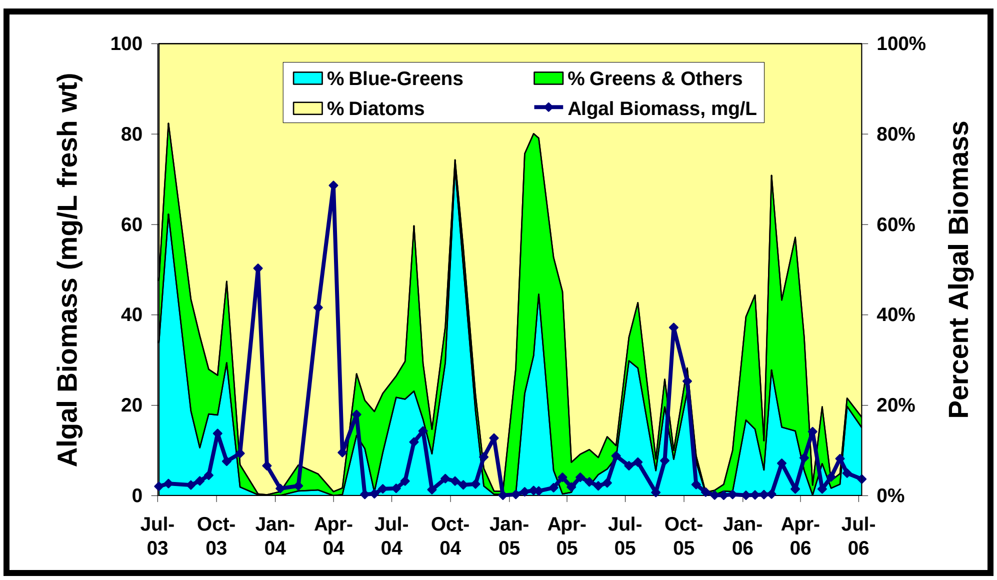
Figure 26 - Phytoplankton Biomass and Chlorophyll *a* at RM 3.5, 2003-2006



While CHLA generally tracked with phytoplankton biomass at RM 3.5 during 2003-2006, the ratio of biomass to CHLA varied greatly. This is expected because different phytoplankton species have different ratios. Physiological state also affects the stoichiometry. The median ratio of biomass (µg/L fresh weight) to CHLA (µg/L) in samples collected at RM 3.5 in 1996 and 2003-2006 was 79. The 25th and 75th percentiles for the ratio were 46 and 160, respectively. These are low compared to the average value of 500 reported for marine phytoplankton by Strickland (1966), who estimated dry weight as 20% of fresh weight. Smith et al. (2010) applied a ratio of 67.5 to convert CHLA (µg/L) to biomass (µg/L dry weight) in the CE-QUAL-W2 model after initially applying a ratio of 135 (default = 50). Version 3.6 of the model does not support variable stoichiometry.

Figure 27 displays the total phytoplankton biomass (blue line) along with the percent biomass represented by each major group (stacked areas) over time at RM 3.5. Diatoms were often the dominant phytoplankton group in the lower Minnesota River during 2003-2006, especially when total algal biomass was high. Moving water in a river favors these generally heavier species. Combining data from 1996 and 2003-2006, diatoms represented over 90% of the biomass at RM 3.5 in the cooler months of April, May, November and December. They represented over 75% of the biomass in January and summer (June-September) and over 60% in the transition months of February, March, and October. Blue-green algae represented 15-27% of the biomass in July, September, and October, while green algae and other groups represented 31-36% of the biomass in February and March. During these years, biomass was lowest in late winter (February).

Figure 27 - Biomass of Major Phytoplankton Groups at RM 3.5, 2003–2006



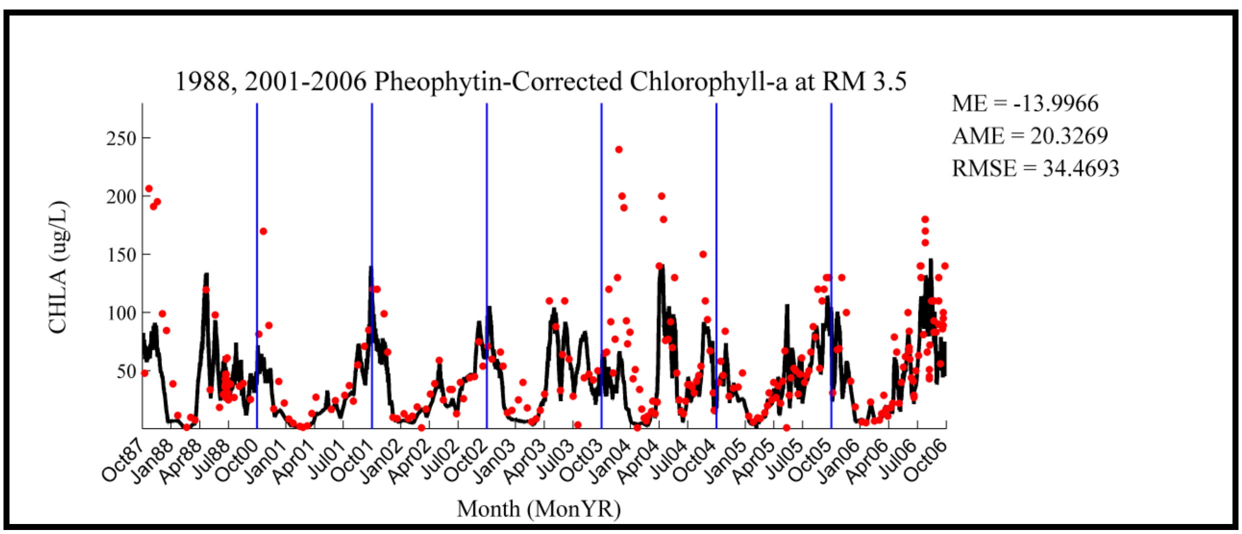
9.3 Model Results for Chlorophyll *a*

The CE-QUAL-W2 model allows any number of phytoplankton groups to be defined and simulated (Cole and Wells, 2008). Three groups were defined in the Lower Minnesota River Model: diatoms, blue-green algae, and others. Many of the algal coefficients were initially based on those applied in the Upper Mississippi River – Lake Pepin Model (LimnoTech, 2009), but some settings were adjusted during model calibration. The Lake Pepin model also contains three phytoplankton groups, but a portion of diatoms were moved to the "other" group to represent a summer assemblage of algae, leaving the original diatom group as a cooler weather assemblage.

In CE-QUAL-W2, phytoplankton are modeled as biomass (mg/L dry wt) and related to carbon, phosphorus, and nitrogen through stoichiometric ratios. The ratios are constant for each phytoplankton group, but they can vary among groups. At RM 39.4, biomass data were only available for February 2005 through September 2006, so a surrogate was needed to define phytoplankton biomass at the upstream boundary. CHLA concentrations served as the surrogate using a conversion factor of 0.0675 mg biomass/µg CHLA. The biomass was split into the three phytoplankton groups using biomass data for RM 39.4 when available. When not available, the splits were based on mean monthly splits at RM 3.5 from 1996 and 2004-2006. Therefore, the best calibration data set for phytoplankton was for the period February 2005 through September 2006 when biomass data were available for both RM 39.4 and 3.5.

Figure 28 show the final calibration results for CHLA concentrations at RM 3.5 during 1988 and 2001-2006. CHLA is a derived variable calculated from the combined biomass results for the three phytoplankton groups. The model generally follows the trends of peaks and valleys in CHLA concentrations. At Fort Snelling, the AME was 20 µg/L, which was less than the calibration target of 24 µg/L (10% of measured data range). At higher concentrations, the model tends to under predict. In the models of 2005 and 2006, the AMEs for the predicted biomasses of diatoms, blue-green algae and other algae met the calibration targets (Smith et al., 2010).

Figure 28 - Model Results for Chlorophyll *a* at RM 3.5, 1988 and 2001-2006



(Smith et al., 2010)

10 OXYGEN

In 1998 the lower reach of the Minnesota River from RM 22 to the mouth was added to the state's list of impaired waters for not meeting dissolved-oxygen standards intended to maintain a healthy fish community and protect aquatic life. In 2004 the U.S. Environmental Protection Agency approved the state's TMDL report that identified pollutant sources and quantified load reductions needed to meet the standard (MPCA, 2004). While the TMDL report was approved, the reach remains on the inventory of impaired waters for not meeting DO standards. The critical conditions for testing compliance with the standard are very low river flows during the summer, which have not occurred in recent years.

Between RM 40 and 21 of the Minnesota River, the DO standard is not less than 5 mg/L as a daily minimum (Minnesota State Rules, Chapter 7050). At RM 21, just upstream of the Blue Lake WWTP, the standard changes to not less than 5 mg/L as a daily average. The standard is applied year-round and at all but extremely low flows. The rule requires compliance with the DO standard 50 percent of the days at which the river flow is equal to the lowest weekly flow with a once in ten-year recurrence interval ($7Q_{10}$ flow). Seasonal $7Q_{10}$ flows were applied in the WLA study; for example, a $7Q_{10}$ flow of 282 cfs (8.0 m^3/s) was computed for RM 25.1 during the summer months of June through September (MPCA, 1985). At higher flows, compliance with the DO standard is expected to occur on successively higher percentages of days. Under extremely dry conditions when river flows fall below the $7Q_{10}$ statistic, the DO standard does not apply; however, effluent limitations must still be met.

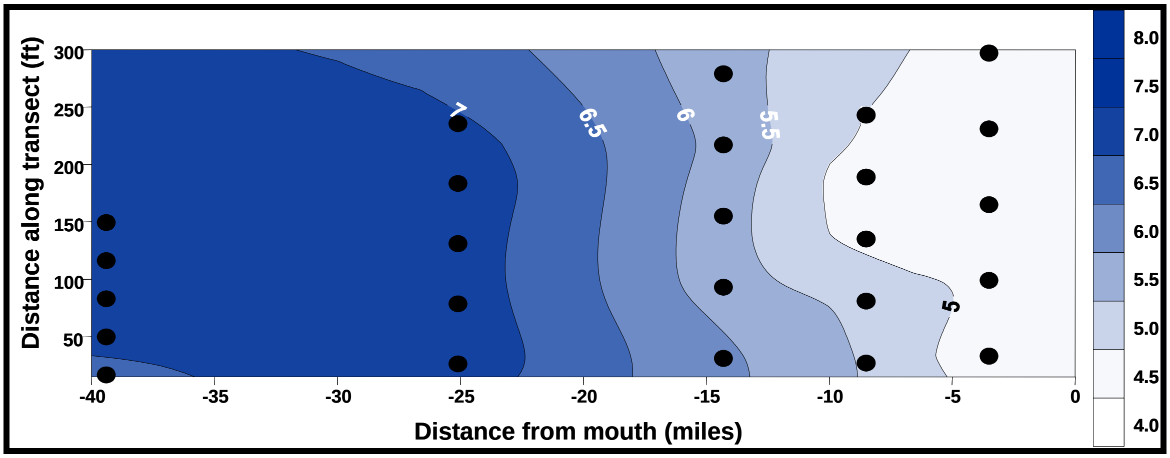
As in all northern temperate rivers, DO concentrations in the lower Minnesota River vary with the seasons because the solubility of oxygen increases as water temperature decreases. Over the past 10 years, 2000-2009, mean monthly DO concentrations at RM 3.5 ranged from 7.28 mg/L in July to 13.37 mg/L in December. Concentrations also vary with river flows. For example, when flows at Jordan fell below 2,000 cfs (56.6 m^3/sec) in late summer 2003, daily minimum DO concentrations at RM 3.5 decreased below 6 mg/L and daily DO fluctuation increased to more than 2 mg/L (Figure 4). Phytoplankton activity, pollutant loads, decomposition, ice, wind, and other factors also affect DO concentrations in the lower Minnesota River.

The most recent prolonged period when DO concentrations were frequently less than 5 mg/L at RM 3.5 occurred during June through August 1988. On June 1, 1988, mean daily flow at Jordan decreased below 2,000 cfs (56.6 m^3/sec), and in July and August, the flow averaged 330 cfs (8.5 m^3/s), which is near the summer $7Q_{10}$ flow. Since 1988 the Blue Lake and Seneca WWTPs were upgraded and river flows have been higher, resulting in generally higher DO concentrations. From field measurements collected weekly by MCES at RM 3.5 over the past ten years, DO concentrations decreased below 6 mg/L on 10 dates and below 5 mg/L on only two dates. Low DO concentrations occurred most often in late summer at lower flows.

On one of these occasions in August 2003, the USGS recorded low DO concentrations during a sampling run for the mixing study (Section 5.1). The ERDC used the Surfer© program and USGS data to estimate and plot DO concentrations over a wider area. Figure 29 shows the resulting plot with measured DO displayed as black circles and estimated DO shown in shades of blue. Near the water surface, a zone of DO concentration below 5 mg/L likely extended from

RM 10 to the mouth. The zone of low DO was more extensive near the river bed. On the next day as required by permit, operators at the Seneca WWTP started aerating the effluent to 16 mg/L (as measured at the treatment plant), which may have helped to improve DO conditions later in the summer.

Figure 29 – Surface Dissolved-Oxygen Concentrations (mg/L), RM 40 to Mouth, 8/21/2003



(ERDC)

While the DO standard has been met since 1988, the lower Minnesota River shows the potential for concentrations to fall below 5 mg/L under summer low-flow conditions. The question remains how the river will respond during a prolonged drought period with river flows near the $7Q_{10}$ statistic and point-source discharges near their permitted limits. The monitoring program and modeling project were designed to answer this question.

The WLA study of DO and BOD in the lower Minnesota River included an analysis of model sensitivity to different inputs (MPCA, 1985). The sensitivity analysis ranked the rates for reaeration, phytoplankton respiration, and sediment oxygen demand among the most important for simulating and understanding DO in the river. Learning from the previous effort, partners in the current project placed a priority on measuring these and other oxygen-related rates. Field measurements would help bracket appropriate settings for the Minnesota River from ranges in the literature. Also, by measuring rates concurrently, a snapshot of the various credits and debits to the oxygen budget could be developed during a short survey of the river.

$HydrO_2$ (2007) conducted two assessments of oxygen dynamics in the lower Minnesota River during the summer of 2006 when river flows at Jordan were less than 2,000 cfs (56.6 m^3/s). The work included measurements or estimates of the following processes:

- **Reaeration** or the transfer of oxygen between the atmosphere and river
- **Atmospheric diffusion** or the movement of oxygen molecules from high to low concentrations between the atmosphere and river
- **Water-column production and respiration** or oxygen gains and losses in the water column due largely to photosynthesis, respiration, and decay

- **Sediment oxygen demand** (SOD) or the loss of oxygen due to the decomposition of organic matter in the sediment bed
- **Community substrate oxygen demand** (CSOD) or the loss of oxygen due to biochemical processes across all substrates, including sediment (SOD), rocks, logs, and aquatic plants
- **Community oxygen metabolism** or oxygen gains and losses in the river from all sources and sinks in the air, water, and sediment

A full assessment was completed during July 17-24, 2006, at an average flow of 2,220 cfs (62.3 m^3/s), and a scaled-down assessment was completed during August 31-September 4, 2006 at an average flow of 966 cfs (27.4 m^3/s). Reaeration and diffusion were measured over the entire 40 miles, while the other rates were measured at six stations selected for their location in relation to point sources and the navigation channel. In tandem with the $HydrO_2$ assessments, the MPCA conducted synoptic water-quality surveys where they continuously monitored DO, pH, temperature, and conductivity at multiple stations with sonde-equipped buoys and collected grab samples for analytical tests (MPCA, 2007c).

After analyzing results from the two assessments, $HydrO_2$ concluded that all of the above processes are important to consider when evaluating current and future DO levels in the lower Minnesota River. The following sections summarize findings about individual components of the DO budget. For more information, see the reports by $HydrO_2$ (2007) and MPCA (2007c).

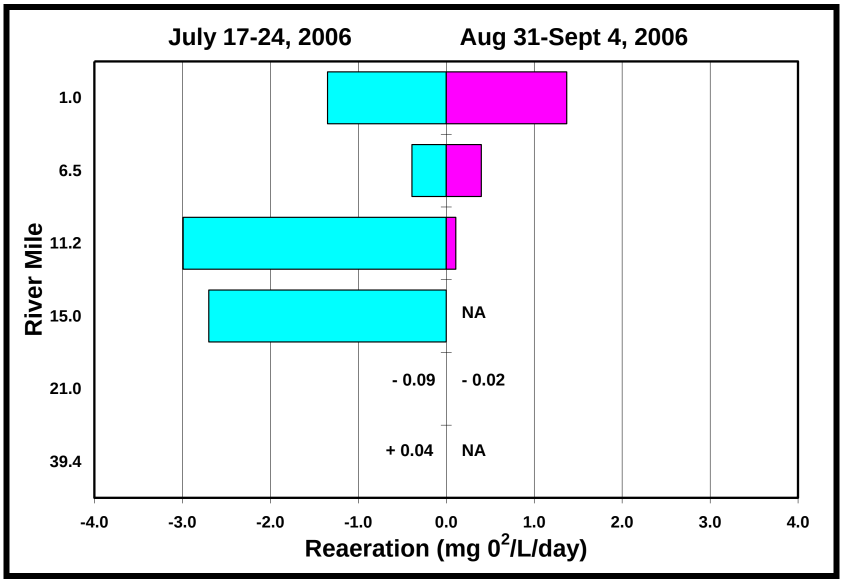
10.1 Reaeration

Reaeration rate coefficients measured in the lower Minnesota River were typical for deep, slow-moving waters with little turbulence ($HydrO_2$, 2007). Coefficients ranged from 0.4 to 1.2 per day in the July survey and from 0.2 to 1.7 per day in the September survey (base *e* at 20º C). In general, the coefficients were lower in September than in July. Stream reaeration is a function of turbulence, and lower flows in September yielded less turbulence. In deep, slow-moving rivers, reaeration generally declines with decreasing flow.

Different techniques were applied in the upper and lower reaches due to complicating factors in the lower 20 miles (e.g., discharges, withdrawals, and pooling). A gas tracer was used in the upper reach, while a diffusion dome was used in the lower reach. Concurrent measurements using both techniques between RM 20.0 and 18.7 yielded only a 12% difference, providing confidence in reaeration rate coefficients determined with the surrogate dome method compared to the recognized accuracy of the gas tracer method.

Figure 30 shows reaeration in terms of the net rate of oxygen change at specific river miles. This rate is computed by multiplying the reaeration rate coefficient by the water-column DO deficit. A positive rate ("gain") results from oxygen transferring from the atmosphere to the river. A negative rate ("loss") shows oxygen being purged from the river to the atmosphere.

Figure 30 - Oxygen Reaeration at Six Sites during Two Surveys in 2006



($HydrO_2$, 2007)

Reaeration is normally a major source of oxygen to rivers. This was seen in the September survey at RM 1.0 and 6.5 where reaeration contributed 0.4 and 1.4 mgO_2/L/day, respectively, to the river (Figure 30). However, in the July survey, oxygen produced by phytoplankton led to supersaturated conditions, which resulted in reaeration "off gassing" up to 3.0 mgO_2/L/day from the water column to the atmosphere. Based on $HydrO_2$'s extensive work on various water bodies, this was an unusual occurrence related to high algal productivity and resulting supersaturated DO conditions. Reaeration effects on the oxygen budget were greater within the navigation channel, which may be related to algal activity. Upstream of the channel at RM 21.0 and 39.4, the net effect of reaeration on DO concentrations was minor during the two assessments; that is, gains due to reaeration were balanced by losses at these two locations.

10.2 Water-Column Production and Respiration

Biological production and respiration in the water column were strong components of oxygen dynamics during two surveys under low flow conditions in summer 2006 ($HydrO_2$, 2007). Changes in oxygen levels due to these factors were measured with light and dark bottles. At six river stations in the July survey, gross primary production (GPP) ranged from 7 to 17 gmO_2/m^2/day, while respiration (R) ranged from 6 to 27 gmO_2/m^2/day using the enclosed bottle technique (Table 8). Advancing into the fall season with the accompanying drop in solar energy and light, production decreased below 7 gmO_2/m^2/day in the September survey, while respiration remained high at 6-17 gmO_2/m^2/day. Phytoplankton are likely the main source of GPP in the lower Minnesota River because high turbidity and low light inhibit periphyton and macrophyte growth. CHLA concentrations at the six stations averaged 125 μg/L during the July survey and 82 μg/L during the September survey. In addition to phytoplankton, bacteria and other organisms contribute to respiration rates in the water column.

Table 8 - Water-Column Production and Respiration, July and September, 2006

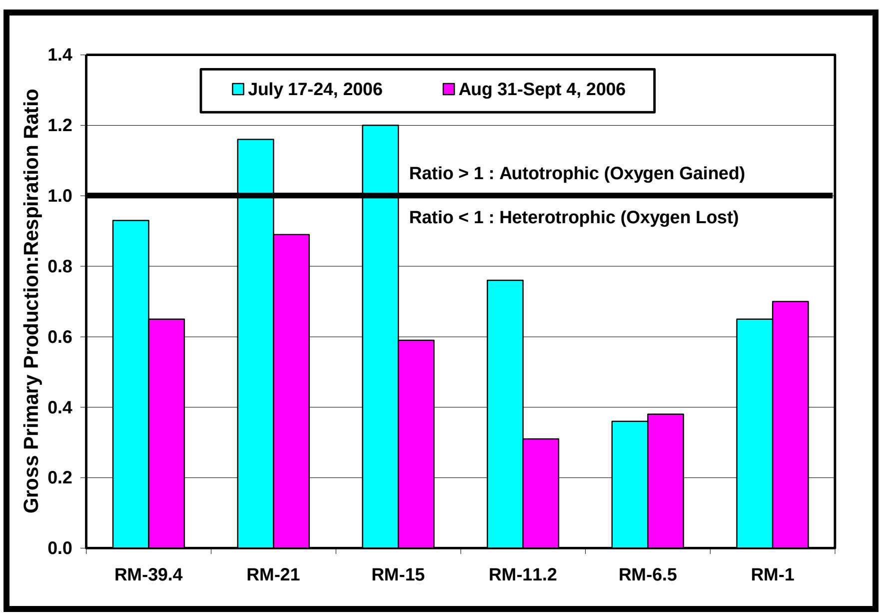
	Gross Primary Production (GPP), Respiration (R), and Phaeophytin-Corrected Chlorophyll *a* (CHLA) (GPP & R, $gmO_2/m^2/day$; CHLA, µg/L)					
	7/17/06-7/24/06			8/31/06-9/4/06		
Station	**GPP**	**R**	**CHLA**	**GPP**	**R**	**CHLA**
RM 39.4	9.29	9.97	130	4.40	6.78	110
RM 21	7.22	6.21	110	5.19	5.84	75
RM 15	16.77	13.99	113	4.50	7.69	70
RM 11.2	7.93	10.44	113	4.33	14.40	73
RM 6.5	9.74	26.91	138	6.33	16.80	94
RM 1	6.45	9.90	145	6.58	9.38	71

($HydrO_2$, 2007)

Combining the two rates into a single GPP:R ratio provides insights into the trophic state of the river and the net impact on the DO budget. A ratio of one or greater indicates an autotrophic state in which phytoplankton produce enough DO via photosynthesis to exceed respiration demands in the river. A ratio of less than one indicates a heterotrophic state in which phytoplankton produce insufficient DO to meet the demands of respiration.

The GPP:R ratios in Figure 31 show a metabolic progression at the three upstream sites from an autotrophic state in July 2006 to a hetereotrophic state in September 2006. This coincided with a halving of solar energy between the two surveys. Only in July at the three sites upstream of the navigation channel did the ratio approach or exceed one, indicating that phytoplankton were a net source of oxygen to the river. At all six stations in September and the three downstream stations in July, the ratio was less than one, indicating that phytoplankton were a net sink of oxygen from the river.

Figure 31 – Production-to-Respiration Ratios at Six Sites during Two Surveys in 2006

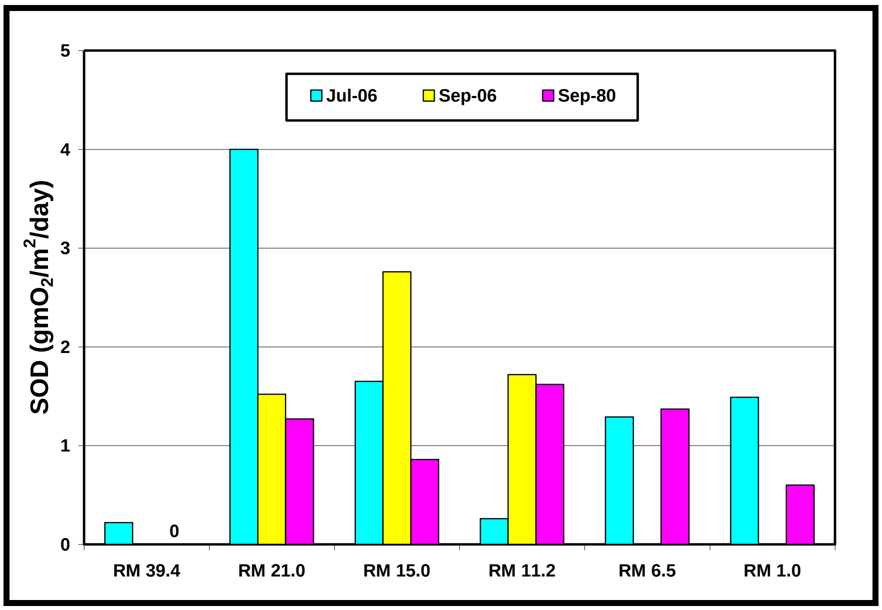


($HydrO_2$, 2007)

10.3 Sediment Oxygen Demand

$HydrO_2$ (2007) measured rates of sediment oxygen demand in the field with large chambers deployed by a diver to assure effective seating on the river bed. They applied information from the sediment-bed assessment when selecting sites to match the dominant substrates of silt and sand. Figure 32 displays the measured SOD rates, which ranged from 0.22 to 2.76 $gmO_2/m^2/day$ with one exception: 4.01 $gmO_2/m^2/day$ at RM 21 in the July survey. This maximum rate and the minimum rate of 0.22 $gmO_2/m^2/day$ at RM 39.4 in July were corroborated by CSOD calculations (see next section). Except for the one high value, SOD rates measured in the lower Minnesota River were low to moderate compared to other studies in an extensive data base of SOD measurements using a similar in situ chamber method.

Figure 32 - Sediment Oxygen Demand at Six Sites during Three Surveys in 1980 and 2006



($HydrO_2$, 2007)

In September SOD rates were measured a second time at three of the six stations to evaluate changes under lower river flows. Rates decreased greatly at RM 21 but increased at RM 15 and 11.2. The diver observed increased flocculent deposits on the sediment bed and chambers in September. Decreased flow and current velocity can lead to increased settling of fine organic matter, which can increase SOD rates at lower river flows.

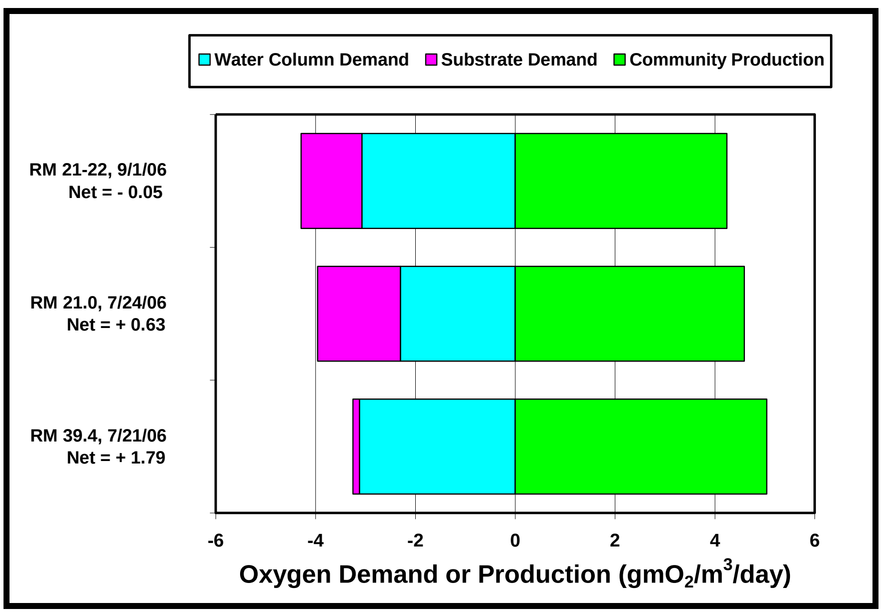
$HydrO_2$ compared SOD rates measured in 2006 with rates measured by the MPCA in 1980 at roughly the same locations ($HydrO_2$, 2007; MPCA, 1985). River flows in September 1980 ranged from 857 to 1,830 cfs. Figure 32 shows that rates measured in September 1980 near RM 7, 11, and 21 were similar to rates measured in either July or September 2006, providing some basis for the application of specific rate ranges within certain reaches of the study area.

10.4 Total Community Production and Demand

HydrO_2 (2007) had planned to calculate total community oxygen metabolism and community substrate oxygen demand by applying the diel curve method to continuous data collected during the sonde surveys (MPCA, 2007c). Unfortunately, data collected at the downstream stations (RM 15, 11.2, 6.5, and 1.0) were not suitable for this method due to complicating factors including equipment failure, supersaturated conditions, pooling effects, and navigation traffic. HydrO_2 was only able to compile complete oxygen budgets for RM 39.4 and 21 in the July survey and near RM 21-22 in the September survey. These sites are located upstream of major point-source discharges and the navigation channel, which could change the mix of factors affecting the oxygen budget.

Figure 33 shows the total community oxygen demand and production under the three settings. On the plus side of the oxygen budget, gross primary production in the water column likely dominated community production. CHLA concentrations and GPP rates were high at these sites (Table 8 in Section 10.2). With light limiting the growth of attached algae and submersed vegetation, phytoplankton were likely the major sources of oxygen under these circumstances. The net effect of diffusion over the air-water interface had a negligible effect on the oxygen balance at these upstream sites (Figure 30 in Section 10.1).

Figure 33 - Oxygen Demand and Production at Two Sites on Three Dates in 2006



(HydrO_2, 2007)

On the negative side of the oxygen budget, respiration and decomposition in the water column dominated total community oxygen demand in the river under the three settings (Figure 33). Phytoplankton respiration was likely a major sink, but the activities of bacteria and other organisms also contributed to water-column demand. Community substrate oxygen demand played a lesser role in the oxygen budget, although it represented a sizeable portion at RM 21 during the July survey. Of the various substrates (e.g., sediment, deadfall, and rocks), sediment was the largest source of CSOD on the three occasions, ranging from 54% at RM 39.4 to 90% at RM 21.0 in the July survey.

The net effect of community production and demand was positive in July with gains of 0.63 and 1.79 gmO_2/m^3/day at RM 21.0 and RM 39.4, respectively. It changed to slightly negative (-0.05 gmO_2/m^3/day) at RM 21-22 in September. From the previous sections, it is likely that water-column respiration contributed an even larger portion of oxygen demand at sites in the navigation channel, and reaeration played a larger role in the oxygen budget at downstream sites.

MPCA (2007c) noted spatial and temporal patterns in DO concentrations from the two sonde surveys in July and September 2006. DO concentrations decreased in a downstream direction from RM 39.4 to the mouth. Vertically, concentrations were generally well mixed at stations upstream of the navigation channel, but wide top-to-bottom differences occurred at downstream stations, especially during the day. In July, algal activity produced supersaturated DO conditions and wide diel DO fluctuation. DO flux between the minimum concentration near dawn and maximum concentration in late afternoon ranged from 4 mg/L at the upstream sites to 1.5 mg/L near the mouth.

10.5 Biochemical Oxygen Demand

Since the WLA study in 1985, the Blue Lake and Seneca WWTPs were upgraded to advanced secondary treatment with nitrification. As shown for the Seneca WWTP in Figure 2 (Section 2), mean annual effluent CBOD concentrations were generally greater than 12 mg/L before the upgrades. Since 1993, they have been consistently less than 6 mg/L as annual means and less than 5 mg/L as summer means. Many facilities that discharge to the Minnesota River upstream of Jordan have also upgraded to higher treatment levels. The improvements contributed to decreasing trends in BOD_5 concentrations at RM 39.4 that have been documented in two reports. MPCA (2002) showed a 34% decrease over the period 1977-2001, and Kloiber (2004) showed a 38% decrease over the period 1976-2002. The MPCA report showed a similar decline in BOD_5 concentrations at Fort Snelling.

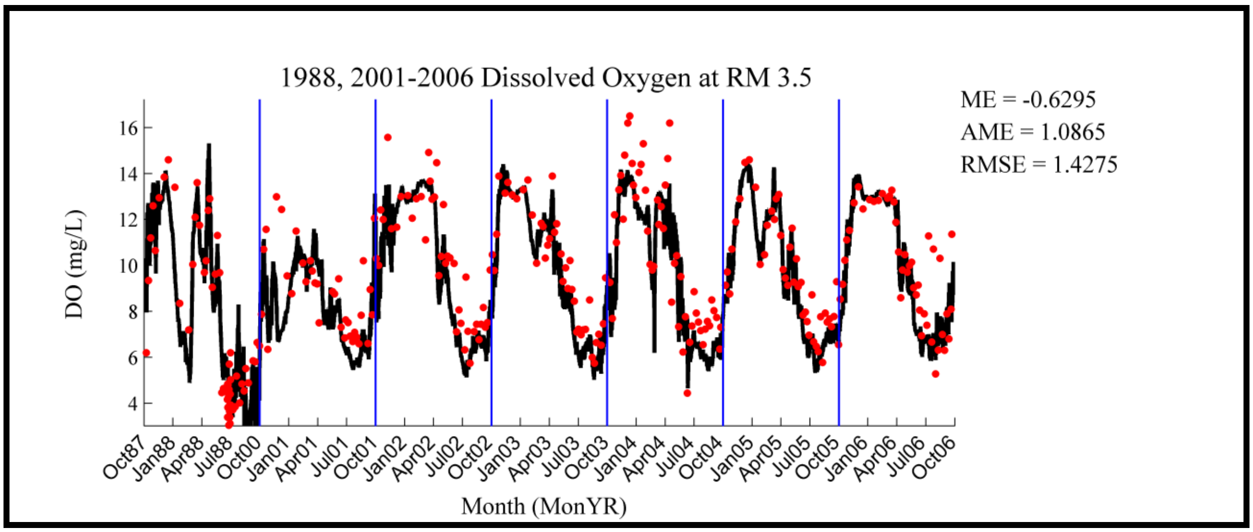
During the three-year monitoring program, 2004-2006, ultimate BOD and CBOD tests were run on unfiltered and filtered samples from the river, tributaries, WWTPs, and main airport outfall on roughly a seasonal basis. Ultimate tests are conducted for up to 70 days by MCES Laboratory Services. Appendix A in Smith et al. (2010) contains summary statistics for the ultimate-to-5-day (U:5) ratios and bottle decay rates. Test results will be valuable in a revised WLA study because U:5 ratios and CBOD decay rates are important settings in the model.

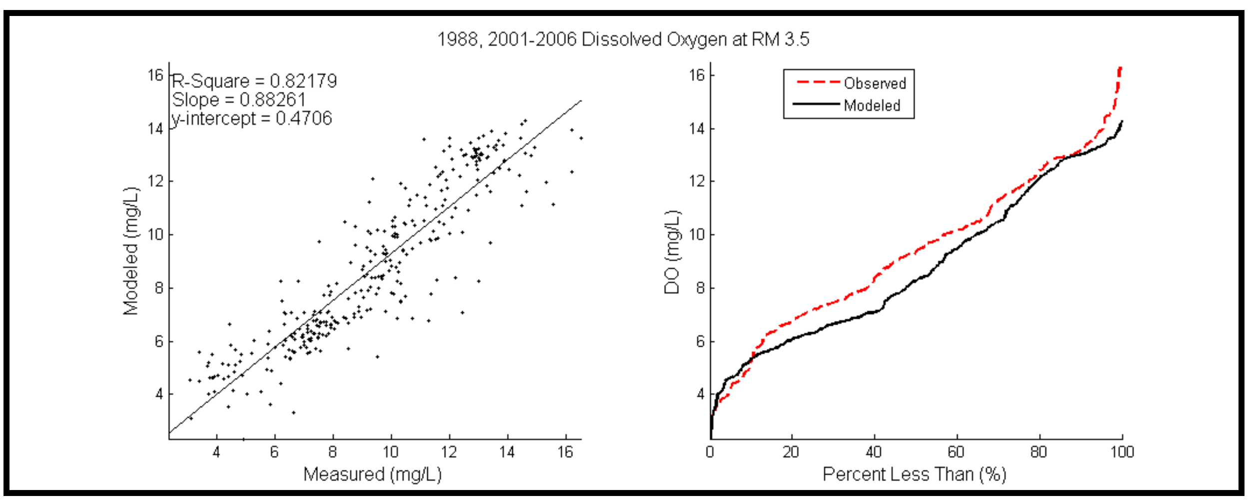
Effluent characteristics have changed considerably since the 1985 WLA study. For example, the mean U:5 ratio for unfiltered CBOD increased at Blue Lake from 3.9 in 1980 to 7.5 in 2004-06, and the ratio increased at Seneca from 3.2 in 1980 to 10.2 in 2004-06. This was accompanied by slower decay rates because advanced treatment has removed organic matter that is readily degraded (labile), leaving material that degrades more slowly (refractory). From the results of the 2004-2006 tests, the average bottle CBOD decay rate was 0.03/day in effluent samples from Blue Lake and Seneca compared to 0.06/day in samples from the Minnesota River at Jordan (base *e* at 20º C).

10.6 Model Results for Oxygen

CE-QUAL-W2 is an advanced water-quality model that simulates the complex DO relationships among the atmosphere, water column, sediment bed, organic matter, phytoplankton, nitrogen forms, and CBOD loads (Cole and Wells, 2008). Some rates measured in the HydrO_2 assessments were applied directly to the CE-QUAL-W2 model, such as sediment oxygen demand; others were applied indirectly to evaluate model settings, such as reaeration equations (Smith et al., 2010). In its entirety, the field work informed the modeling team about the importance of different factors to oxygen dynamics in the Minnesota River. Flow and temperature dominated oxygen dynamics at most flows and seasons, but phytoplankton, the sediment bed, and other factors became increasingly important at lower flows in the summer (Figure 34).

Figure 34 - Model Results for Dissolved Oxygen at RM 3.5, 1988 and 2001-2006





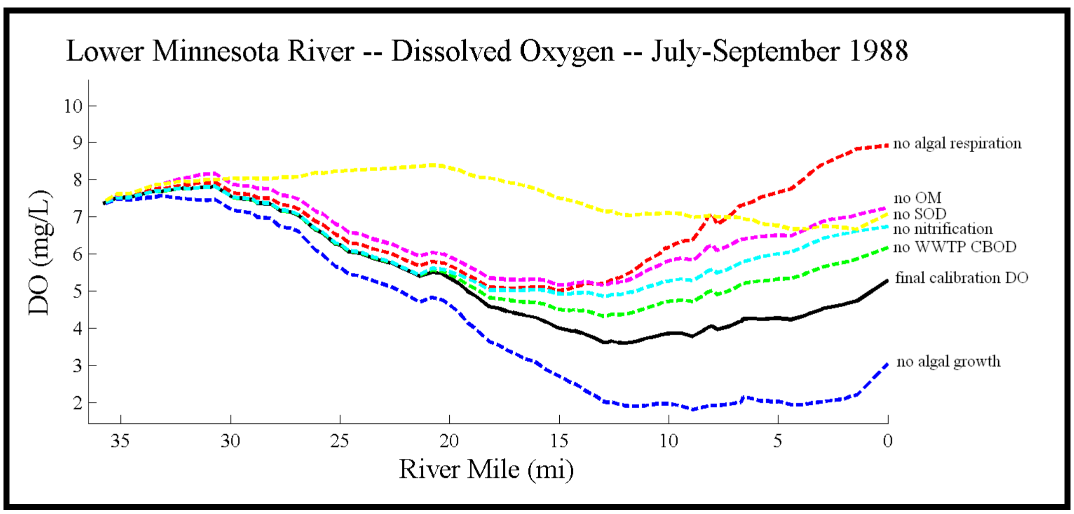
(Smith et al., 2010)

Figure 34 shows the final calibration results for DO concentrations at RM 3.5 in 1988 and 2001-2006. Note in particular the DO concentrations and model performance under summer low-flow conditions as in 1988, 2001, 2003, and 2006. The model does a good job of predicting DO concentrations, especially in the upper reach of the river; however, the seven-year mean error for DO indicates that the model slightly under predicts DO. This is especially prevalent during the summer periods in most water years. Figure 34 also includes one-to-one and cumulative distribution plots. At the lower 60% of measured values, the model tends to under predict the data by approximately 0.63 mg/L at RM 3.5. Although the model under predicts DO levels, the model is well within the standard accepted level of tolerance for DO, 1.00 mg/L, and is well within the target of 1.28 mg/L at RM 3.5 (10% of measured data range).

Smith et al. (2010) conducted a component analysis of various DO sources and sinks using the models of 1988 and 2006 from the final calibration. The 1988 model was selected because it represented very low summer flows in the river and pre-upgrade CBOD and NH4 loads from the WWTPs. The 2006 model represented a short low-flow period in late summer, post-upgrade effluent loads, and a robust calibration data set. In a component analysis, a single DO source or sink (or group of related inputs) is turned off in the model while other model inputs remain at their calibration settings. The difference in resulting DO concentrations between the component analysis and calibration reveals the importance of each component.

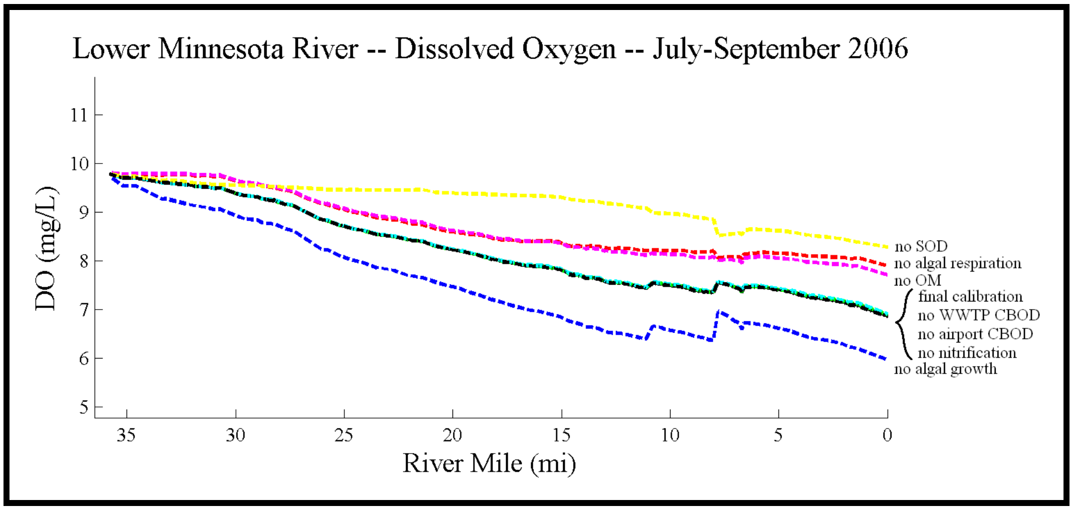
Figure 35 and Figure 36 display the results of the component analyses for 1988 and 2006 as average DO concentrations over the three-month period, July-September, on a longitudinal plot from RM 36 to the mouth. In both plots, the distance between lines widens as the river approaches the mouth, especially in the navigation channel in 1988. This shows that all components with the exception of SOD become more important in the lower reach. The lines are more widely spread in 1988 because river flows at Jordan were lower than in 2006; the mean flow during July-September was 304 cfs in 1988 and 1,850 cfs in 2006. Higher river flows in 2006 also afforded more dilution for effluent loads.

Figure 35 – Components of Modeled DO, RM 35 to Mouth, July-September 1988



(Smith et al., 2010)

Figure 36 - Components of Modeled DO, RM 35 to Mouth, July-September 2006



(Smith et al., 2010)

In both years, phytoplankton played a major role in DO dynamics, but respiration offset production to some degree, reducing the net effect. SOD was a major sink especially between RM 21.4 and 15.0 where the highest measured SOD rate (4.0 $gmO_2/m^2/day$) was applied. Nonliving organic matter (OM) from the upstream boundary and tributaries as well as OM from dead algae within the river represented a major source of oxygen demand in both summers. Effluent CBOD and NH4 loads from the WWTPs played a lesser role in 2006 compared to 1988 due partially to more river flow and dilution in 2006 and partially to treatment upgrades. Removing all effluent CBOD loads from the two WWTPs in the 2006 model resulted in little or no change in river DO concentrations. In the component analysis, NH4 loads were left as in the calibration inputs, but nitrification was turned off, preventing all NH4 from converting to NO3 and consuming oxygen. The two WWTPs were significant sources of NH4 in 1988. Oxygen demand due to nitrification was a larger component than effluent CBOD loads in the 1988 model, but both were negligible in the 2006 model. Only a subset of potential sources and sinks were tested in the component analysis; for example, reaeration was not tested.

11 MAJOR DISCHARGERS

Four major dischargers were defined in the Lower Minnesota River Model: Blue Lake WWTP, Seneca WWTP, Black Dog GP, and MSP airport. The dischargers are mentioned in the study area description (Section 2), monitoring program (Section 4), budgetary analysis (Section 8.1), and elsewhere. The following sections contain additional information.

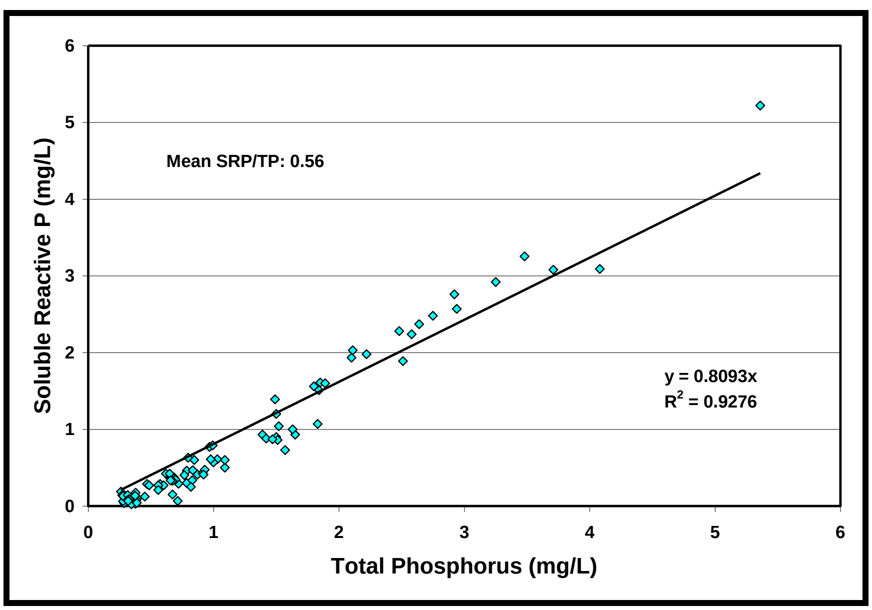
11.1 Blue Lake and Seneca Wastewater Treatment Plants

Effluent quality at the Blue Lake and Seneca WWTPs increased greatly as a result of improvements made in the mid-1990s. As shown for the Seneca WWTP in Figure 2 (Section 2), mean annual effluent $CBOD_5$ concentrations decreased from greater than 12 mg/L before 1992 to less than 6 mg/L after 1992. Currently, mean summer effluent $CBOD_5$ concentrations generally fall between 3 and 4 mg/L with many values reported as < 3 mg/L (e.g., 35% of reported values during 2000-2007). During 2000-2009, the mean summer $CBOD_5$ concentration of grab samples collected from the Minnesota River at RM 39.4 averaged 2.5 mg/L and ranged from 1.3 mg/L in 2001 to 3.7 mg/L in 2009.

With the facility improvements, effluent NH4 concentrations were also greatly reduced due to nitrification, but effluent NO3 concentrations increased as a consequence (Figure 2). Combined annual average TP loads from both WWTPs decreased nearly 80% from 1156 lb/day in 1991 to 244 lb/day in 2009. Not only concentrations and loads have changed; the characteristics of the effluent have changed as well. For example, organic matter in the discharge is more refractory (i.e., decomposes in months to years instead of days to weeks) as evidenced in slower CBOD decay rates and higher CBOD U:5 ratios (Section 10.5). A component analysis of the CE-QUAL-W2 model showed that eliminating effluent CBOD loads from both facilities during July-September 2006 resulted in little or no change to DO concentrations at RM 3.5 (Figure 36).

Phosphorus characteristics have changed, too. Effluent TP is assumed to contain a high portion of bioavailable P, measured as SRP. MCES has routinely monitored TP in effluent for over 25 years, but SRP is measured only by special request. Effluent SRP concentrations were measured at the Blue Lake and Seneca WWTPs before operations were optimized to biologically remove P in the mid-1990s. The average ratios of SRP to TP in the effluent were 0.84 at Blue Lake and 0.78 at Seneca during 1991-1992. SRP measurements were again requested for the current study. During WY 2004-2006, the average ratios decreased to 0.62 and 0.56 at Blue Lake and Seneca, respectively, with even smaller SRP fractions at effluent TP concentrations under 1.0 mg/L (Figure 37). Biological P removal at the WWTPs not only reduced the total P discharged to the receiving water; it also reduced the portion of bioavailable P. It is unknown how much of the remaining effluent P (TP minus SRP) is biologically labile (i.e., easily recycled to SRP under certain conditions) or refractory (i.e., tightly bound to particles or slow to degrade).

Figure 37 – Effluent TP Concentrations, Seneca WWTP, Water Years 2004-2006



Despite these improvements, the Blue Lake and Seneca WWTPs continue to enrich the Minnesota River with nutrients, especially under lower river flows when there is less dilution. In a nutrient budget of the lower Minnesota River during summer low-flow conditions in 2006, James (2007) showed that the two WWTPs contributed 34.2%, 45.6%, and 74.8 % of the NH4, NO3, and SRP loads, respectively, to the lower 40 miles while contributing only 5.1% of the flow (Section 8.1). The question is how this enrichment affects aquatic life and recreation. In further examination of the low flow budget in 2006, James (2007) thought it probable that SRP loads from the Blue Lake WWTP at RM 20.5 were assimilated by phytoplankton for growth after SRP supplies were nearly depleted at RM 39.4 (Section 8.3). In the lower reach, phytoplankton senescence occurred to a greater degree, so their decomposition and additional effluent loads contributed to high SRP concentrations (>0.100 mg/L) near the mouth. Nutrient loads from the Minnesota River may impact aquatic life and recreation at downstream locations in the Mississippi River. The effects of nutrient enrichment under other conditions can be further explored with the model.

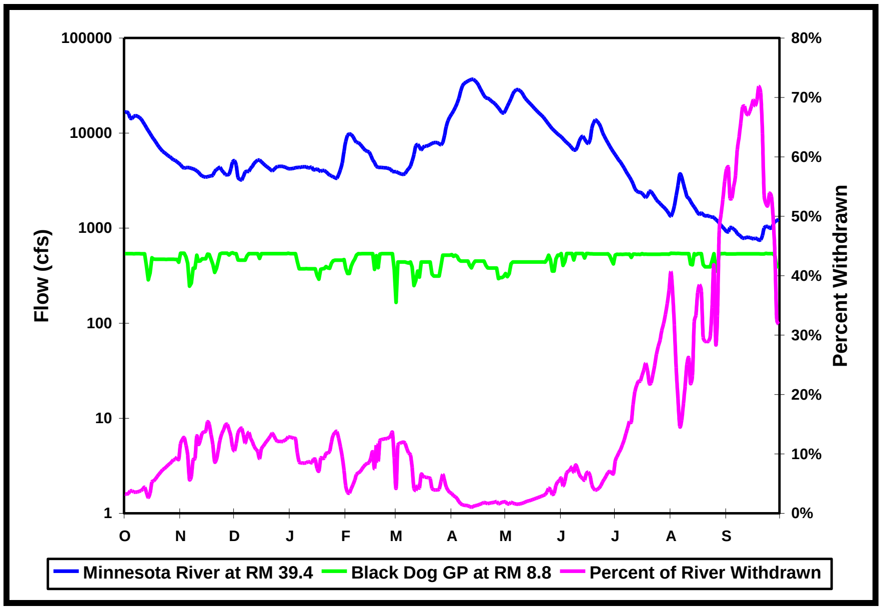
11.2 Black Dog Generating Plant

Modeling the lower Minnesota River identified at least one deficiency in the monitoring program: insufficient water-quality monitoring of the Black Dog GP outfalls. The water appropriation permit for the Black Dog GP requires monitoring of intake flows, and the NPDES permit requires monitoring of flow and temperature at the two cooling-lake outlets. However, while Xcel Energy has conducted environmental studies on the facility's effects on the river (e.g., Xcel Energy, 2007), water quality is not routinely monitored.

The Black Dog area presents a complex problem for monitoring and modeling. As seen in the aerial photograph on the title page, the area includes the 538-megawatt Black Dog GP facility and its intake, the three-mile-long Black Dog cooling lake with outfalls at each end, and the watershed draining into the lake. The facility, cooling lake, and watershed all potentially affect water quality. As seen in Figure 38, the withdrawal rate varies greatly with seasonal temperatures and energy demands for heating and cooling. During April through September 2006, the percent of river flow withdrawn by the Black Dog GP varied from 1% in early April to 72% in mid-September. Discharges at the two outfalls vary with withdrawal rates and operational controls to meet thermal requirements. Designing a water-quality monitoring program for the highly variable cooling-lake discharges would be challenging.

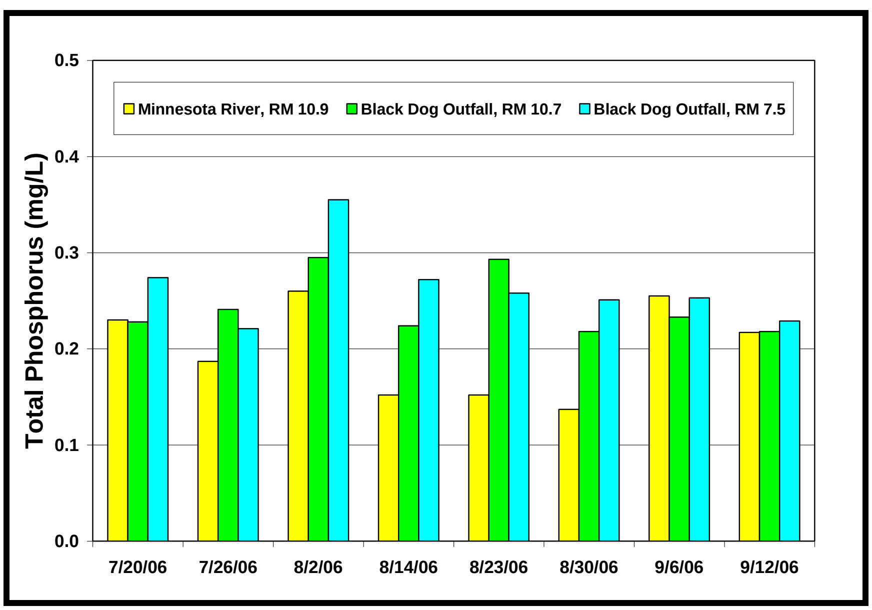
Fifteen 24-hour composited water-quality samples were collected concurrently from the two outfalls and an upstream river site during the summer in 2005 and 2006 when river flows were low. Under these conditions, the percent of river withdrawn by the facility typically increases as would, presumably, potential effects on water quality. Figure 39 shows the results for TP concentrations during the low flow period in July-September 2006 as an example. Results for TP and other variables varied from date to date: sometimes concentrations in one or both outlets were higher than those at the upstream river site, sometimes they were lower. The only conclusions drawn were the following: 1) the Black Dog complex shows the potential to change water quality, positively or negatively, 2) the effects can vary day to day if not hourly, and 3) more water-quality monitoring is needed to understand the effects.

Figure 38 - Black Dog GP Withdrawal Rate Compared to Flow at RM 39.4, WY 2006



James (2007) hinted at the potential effects in a budgetary analysis of FLUX-estimated loads during the summer low-flow period in 2006. Based on excellent flow data but limited water-quality data, the budget suggested that the Black Dog complex was a net sink for nitrate and net source of ammonium, phosphorus, and suspended solids under these conditions. Some processes that may occur in the lake include denitrification, settling, sediment resuspension by wind or rough fish, sediment-bed nutrient release, phytoplankton activity, reaeration by wind, and bio-chemical oxygen demand. James (2007) noted peak phaeophytin *a* concentrations between RM 10.8 and 7.2 during this period and thought it conceivable that flow through the Black Dog GP contributed to phytoplankton death. He concluded that more research is needed.

Figure 39 – TP Concentrations at Black Dog Outfalls and RM 10.9, July-September, 2006



With limited water-quality data for Black Dog Lake and the two outfalls, a simplified approach was applied in the CE-QUAL-W2 model (Smith et al., 2010). The intake and two outfalls were defined as one withdrawal and two loading points in the river model. Black Dog Lake was not modeled but could be added as a branch given sufficient data. Xcel Energy provided daily or hourly flow and temperature data for the seven modeled years. Discharge and intake flows frequently differed, presenting challenges for the model calibration and budgetary analysis.

For periods when water-quality samples were collected (low flow summer periods in 2005 and 2006), the data were used to define model inputs at the two outfalls. No water-quality data were available for 1988, 2001-2004, and most of 2005 and 2006. For these periods, model outputs for a segment directly upstream of the outfall at RM 10.7 were used as inputs for the two outfalls. These "reflected" input files required an extra model run and assumed that the Black Dog complex has no effect on water quality. This assumption may be valid under some conditions (e.g., high river flow and low withdrawal rate) but not others (e.g., low river flow and high withdrawal rate).

In a sensitivity analysis, Smith et al. (2010) removed the Black Dog intake and outfalls from the 1988, 2003, and 2006 models. On an annual basis, model results with and without Black Dog were similar, but the intake and outfalls were retained for completeness and for future applications when more data are available. Model performance under summer low-flow conditions would benefit from additional water-quality monitoring at the outlets under targeted conditions. In 2008 Xcel Energy proposed major modifications to the cooling water systems to better protect aquatic life, including large baffles running lengthwise down the middle of the lake to recirculate and more thoroughly cool water before it is discharged to the river.

11.3 Minneapolis-St. Paul International Airport

Information on the water quality of stormwater discharges from the MSP international airport presented a number of challenges for defining inputs to the CE-QUAL-W2 model:

- Few data were available on airport flows and loads in 1988, when drought conditions would have reduced stormwater. As a result, airport discharges were not defined in the 1988 model.

- The MAC was required to monitor only a subset of the modeled variables, and the required sampling frequencies for many variables decreased after October 2004. For water years 2001-2004, the MAC provided excellent data for flow, temperature, $CBOD_5$, NH4, and TSS and good data for DO, TKN, and TP. After October 2004, they provided daily data for flow and $CBOD_5$; weekly data for temperature, NH4, and TSS; and only occasional data for DO, TP, and TKN. For regulatory purposes, tracking $CBOD_5$ loads became the primary interest.

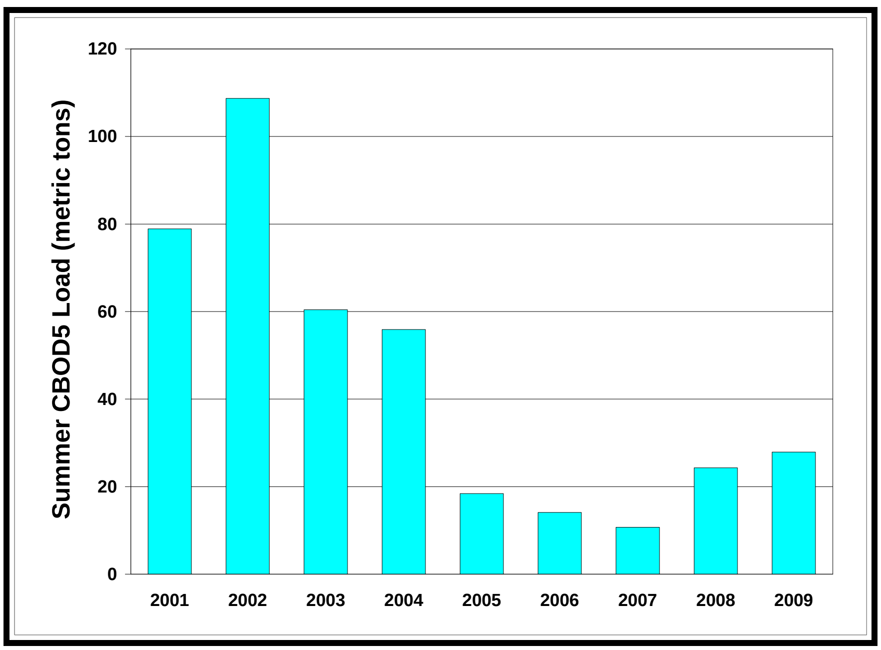
- Airfield, deicing, and stormwater improvements completed in 2005 resulted in some disruptions in sampling (e.g., as new ponds filled) and changes in water quality. Figure 40 shows decreased summer $CBOD_5$ loads from the stormwater discharges to the river since the improvements.

- Due to construction in 2004, MCES collected only 23 samples at the main outfall (old outfall 020, new outfall SD010) from February 2005 to September 2006. The full suite of modeling variables was analyzed in only 17 samples from the one outfall.

- Characteristics of the CBOD discharged from the MSP outfalls varied greatly. For example, ultimate CBOD (unfiltered) ranged from 7 to 2005 mg/L in 11 samples collected and analyzed by MCES. The CBOD U:5 ratio varied from 1.4 to 14.3. Ultimate CBOD results from the MAC for samples collected at three outfalls from 2001 to 2004 also varied greatly.

In the model, inputs were defined for the MSP outfalls, but different variables required different levels of estimation (Smith et al., 2010). While the airport outfalls were assigned their own CBOD group separate from the river and WWTPs, only a single common U:5 ratio and decay rate were applied to all airport outfalls at all times. The outcome at times was highly variable model results for the derived variables $CBOD_5$ and dissolved organic carbon at RM 3.5. This is an area for further evaluation in the model and possible improvement with additional monitoring.

Figure 40 - MSP Airport $CBOD_5$ Loads to the Minnesota River, June-Sept., 2001-2009



(Data: Outfalls SD006, SD010, and SD012 from the MPCA)

12 MODEL APPLICATION

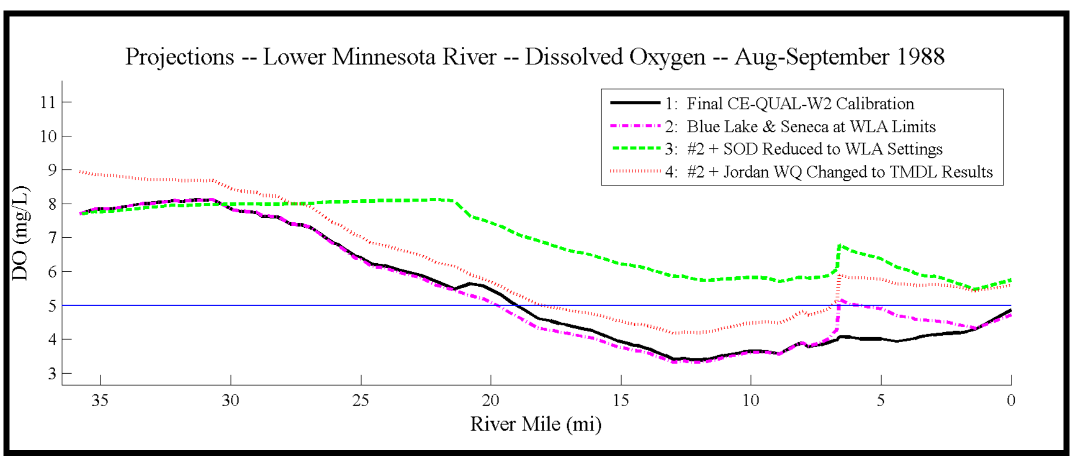
Once the model was fully tested and met performance targets, a number of loading scenarios were applied to demonstrate the model's capabilities (Smith et al., 2010). The scenarios were based on current permit limitations and completed load allocation studies. In one scenario, results from the Minnesota River Basin Model (HSPF framework) were translated and input to the Lower Minnesota River Model (CE-QUAL-W2 framework). The objectives were to show the following: 1) the model produces reasonable results even when loads are greatly increased or decreased, 2) the model can be linked to other models, and 3) the model is suitable for application in future load allocation studies and facility or watershed planning. Four loading scenarios were applied to the Lower Minnesota River Model:

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

Figure 41 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 in 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 in the river 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 utility in future load allocation studies and other applications. Scenario D demonstrated the ability to translate 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 41 - Scenario Results for DO, RM 35 to Mouth, August-September 1988



(ERDC)

13 SUMMARY

The Metropolitan Council led a cooperative effort of federal, state, and local agencies to develop a water-quality model of the lower 40 miles of the Minnesota River for use in facility and watershed planning. The water-quality issues in order of priority were dissolved oxygen, ammonia, nutrients, and sediment. The partners chose the CE-QUAL-W2 model framework and designed a three-year monitoring program to support it. Long-term monitoring programs for the river, tributaries, and discharges were enhanced to fulfill model data requirements. More intensive monitoring was added at low river flows during the summer to assess water quality under critical conditions for dissolved oxygen and eutrophication. Special field studies were conducted to guide decisions and support model inputs. These included extensive work on oxygen and phosphorus dynamics. Data from the customized monitoring program improved model performance and thereby decreased model uncertainty. The project demonstrated the benefits of designing a multi-year monitoring program to support a specific model framework. This section summarizes the results.

Hydrodynamics play a prominent role in the water quality of the lower Minnesota River, as it does in all rivers. However, this reach differs from most in being part of a navigation system to support commercial barge traffic. Major factors influencing the hydrodynamics are flows from the large upstream watershed, channel morphology including the navigation-system modifications, and pooling effects behind Lock and Dam No. 2 in the Mississippi River. Impacts of the withdrawal and discharges at the Black Dog Generating Plant vary with energy demand and river flow, with withdrawal rates ranging from 0% to over 100% of the river flow. In studies by the USGS, ground-water inflows to the river were shown to be minor.

From the modeling effort, Smith et al. (2010) found that river discharge was the main driver of water quality for the majority of the year, and at flows greater than approximately 50 m^3/s (1,800 cfs), transport dominated water quality. The hydrodynamics become more complex at lower river flows due to the increased influence of the navigation system, pooling, withdrawals, and discharges. At lower flows in summer, greater depths and slower velocities in the navigation channel increasingly affect sediment, light, phosphorus, phytoplankton, and oxygen dynamics.

The USGS studied mixing characteristics of the river at long-term monitoring stations under various flows during 2004-2006 (14 dates). The river was well mixed with respect to turbidity in over 90% of profiles taken upstream of the navigation channel, but vertical differences greater than 20% in turbidity occurred in nearly half of the profiles taken within the channel. Vertical differences greater than 0.5 mg/L in DO concentrations were observed only at the three downstream sites within the navigation channel at flows less than 3,000 cfs (85 m^3/s). At RM 14.3 (Savage), vertical DO differences greater than 0.5 mg/L were observed in only 2 of 27 profiles (7%); while at RM 3.5 (Fort Snelling), they occurred in 28 of 71 profiles (39%). Only one vertical temperature difference of greater than 2º C was measured in April 2004 at RM 8.5 (Black Dog). No vertical differences greater than 10% in conductivity or greater than 0.5 in pH were recorded.

During June through September, 2006, the USGS collected eight pairs of discrete and integrated water samples at RM 39.4 and 3.5 (Jordan and Fort Snelling). MCES performed laboratory and statistical analyses to compare water quality in the paired samples. Differences can indicate when and where the river was not well mixed for a particular variable. At both sites concentrations of TSS and CHLA were significantly different between sampling protocols. At RM 39.4, the average ratio between concentrations in the discrete and integrated samples was 0.93 for both constituents, indicating that the difference was small. At RM 3.5, the ratio was similar for TSS (0.92) but lower for CHLA (0.76). BOD_5 and nutrient concentrations were not significantly different between discrete and integrated samples. The findings have implications for assessments that require representative data for the entire channel cross-section.

MCES developed a map of surficial sediment-bed types in the lower 26 miles of the Minnesota River using seismic-profiling data collected by the USGS in September 2003. The dominant types were silt-sand (57%) and sand-gravel (30%) with no strong longitudinal patterns over the reach. In 2005 and 2006, MCES collected sediment cores at sites chosen with the map, and the ERDC analyzed sediment characteristics. Organic content was low, averaging only 2.5%. Moisture content and sediment density proved to be good predictors of sediment composition, with moisture content positively correlated with clay and silt and negatively correlated to sand. Sediment density showed strong relationships in directions opposing moisture content.

At low river flows in September 2006, MPCA (2007c) and $HydrO_2$ (2007) observed increased settling at RM 3.5 and 1.2 with fine material deposited on the surface of the river bed. High amounts of fine organic material may generate more SOD and result in higher SRP and NH4 release rates. Using chambers seated on the river bed, $HydrO_2$ (2007) measured SOD rates in July and September 2006 that ranged from 0.22 to 2.76 gmO_2/m^2/day with one higher exception. These rates are low to moderate compared to rates measured in other rivers, and they are similar to rates measured by the MPCA near the same locations in 1980. $HydrO_2$ (2007) and Smith et al. (2010) confirmed that SOD remains an important component of oxygen dynamics in the lower Minnesota River. In the laboratory, James (2007) measured oxic and anoxic rates of SRP and NH4 release from sediment cores but concluded that sediment release rates played minor roles in nutrient budgets for 2004-2006 even under summer low-flow conditions.

Despite efforts to reduce sediment loads to the Minnesota River, suspended-solids concentrations remain high. Over the recent 10-year period of 2000-2009, the median TSS concentration at RM 3.5 was 47 mg/L. Average summer TSS concentrations at this site ranged from 37 mg/L in 2009 to 164 mg/L in 2004. Megard (2007) showed how suspended solids in the river relate to light attenuation, transparency, and turbidity, which in turn impact phytoplankton and other aquatic life. Due to their scattering effect, organic solids play an important role in transparency and turbidity; however, inorganic solids are more prominent in the river. With turbidity measurements varying among different meters and protocols, suspended solids can provide a universal translator among turbidity methods. $HydrO_2$ (2007) and MPCA (2007c) noted that light through its effects on phytoplankton was an important factor in DO metabolism in the lower Minnesota River. During two surveys in July and September 2006, the photic zone was restricted to less than one meter from the water surface or less than one-third of the water-column depth.

Nutrient levels are also high in the river. During 2000-2009, the median TN concentration at RM 3.5 was 4.81 mg/L with NO3 representing the largest portion (3.32 mg/L). NH4 concentrations were generally low at this site with nearly 25% reported as below the detection level of 0.02 mg/L. The median TP concentration was 0.19 mg/L with biologically available SRP representing roughly a third (0.066 mg/L). However, almost 10% of SRP concentrations at RM 3.5 were reported as below the detection level of 0.005 mg/L.

James (2007) compiled nutrient and sediment budgets for the lower Minnesota River for water years 2004-2006. During the three-year period, TSS loads from the Minnesota River averaged 740,000 mt/yr. TN loads averaged 55,000 mt/yr with NO3 as the dominant form, and TP loads averaged over 1,400 mt/yr with nearly one-third as SRP. Approximately one-half of the particulate P is biologically labile or easily recycled to SRP under certain conditions. Sediment and nutrient loads from the Minnesota River have important downstream effects on sedimentation and eutrophication in the Mississippi River.

During 2004-2006, the Minnesota River at Jordan contributed over 88% of the TSS, TN, and TP loads to lower 40 miles of the river (James, 2007). At lower flows, the portions of TN and TP loads contributed by the Blue Lake and Seneca WWTPs increased. During an 11-week period in late summer 2006, the two WWTPs contributed 34.2, 45.6, and 74.8% of the NH4, NO3, and SRP loads, respectively. The lower Minnesota River was a deposition zone for suspended solids, with an annual retention of 22% of the total TSS load in 2005 and 2006 and 39% in 2004. The reach was also a sink for TP with 5-11% of the annual load retained in 2004-2006; however, most TP was transported downstream. The reach was a net source of NH4 with 27.6-50.1% more exported than received in the three years.

Phosphorus dynamics are complex in the lower Minnesota River. James (2007) studied the relationship of phosphorus to suspended solids. At higher flows, suspended solids maintained SRP concentrations around 0.115 mg/L through equilibrium processes. At lower flows, biotic factors became increasingly important. At times, SRP concentrations were nearly depleted at RM 39.4 through algal assimilation but increased downstream with inputs from WWTPs and other sources. The upper reach (RM 40-20) showed the most potential for P limitation of algal growth. Phytoplankton senescence and decomposition of organic N and P likely contributed to increased NH4 and SRP concentrations in the lower reach under summer low-flow conditions.

High levels of nutrients in the Minnesota River support high levels of phytoplankton. During 2003-2009, mean annual CHLA concentrations were 66 and 56 μg/L at RM 39.4 and 3.5, respectively, and summer means were 90 and 65 μg/L. Algal levels vary from month to month in response to changes in temperature, flow, and solar energy, with peak CHLA concentrations typically occurring in late summer at RM 39.4. Phytoplankton samples were collected at RM 3.5 during 1996 and 2003-2006. Biomass generally peaked in the cooler months of spring and fall. Diatoms dominated the phytoplankton community, representing over 60% of the biomass in all months and over 90% in April, May, November, and December. Blue-green algae thrived during low flows in late summer and early fall, when they represented 15-27% of the biomass.

CHLA concentrations at RM 39.4 exceeded concentrations at RM 3.5 during the ice-free season and especially in late summer. James (2007) and MPCA (2007c) showed viable chlorophyll *a* concentrations decreasing from RM 39.4 to the mouth in September 2006 as phaeophytin *a* concentrations increased, suggesting algal die-off in the lower reaches under summer low-flow conditions. Increased water-column depths and lower current velocities in the navigation channel may settle or mix phytoplankton out of the narrow photic zone, leading to senescence. In turn, this could contribute to oxygen demand in the water column and sediment bed.

Low DO concentrations occur most often in late summer at lower flows in the river. The most recent prolonged period when DO concentrations were frequently less than 5 mg/L at RM 3.5 occurred during June through August 1988. In July and August 1988, the flow averaged 330 cfs, which is near the summer $7Q_{10}$ flow. Since 1988 the Blue Lake and Seneca WWTPs have been upgraded and river flows have been higher, resulting in generally higher DO concentrations in the summer. From field measurements collected weekly by MCES at RM 3.5 over the past ten years, DO concentrations decreased below 6 mg/L on 10 dates and below 5 mg/L on only two dates. Despite the good record, the data show a potential for low DO concentrations, and the question remains how the river will respond under lower flows and higher BOD loads. The Lower Minnesota River Model provides a tool for answering this question.

Phytoplankton production and respiration are strong components of oxygen dynamics in the lower Minnesota River, especially during summer low-flow conditions. Oxygen dynamics are a complex mixture of physical and biochemical factors. Physical factors include temperature, flow, wind, ice, and light. In the model, good representation of the hydrodynamics generally translated to solid DO predictions at higher flows. At lower flows and warmer temperatures, biochemical factors become more important. These include phytoplankton activity, decomposition of nonliving organic matter, and sediment oxygen demand. As evidence of this activity, diel DO fluctuation greatly increased at river flows less than 2,000 cfs during the summer.

In July and September 2006, $HydrO_2$ (2007) measured reaeration, sediment oxygen demand, and phytoplankton activity in the lower Minnesota River. They concluded that all are important to oxygen dynamics under summer low-flow conditions. Reaeration rate coefficients were typical for deep, slow moving waters with little turbulence. Reaeration is normally a source of oxygen to rivers; however, supersaturated DO conditions due to heavy phytoplankton activity led to “off gassing” of oxygen via reaeration in July. With the exception of two upstream sites in July, phytoplankton respiration exceeded production, showing the river to be predominantly hetereotrophic during summer low-flow conditions as seen in 2006.

Using the Lower Minnesota River Model, Smith et al. (2010) conducted component analyses of DO in the models of 1988 and 2006, focusing on late summer, July-September. In both summers, phytoplankton played a major role in DO dynamics, but respiration offset production to some degree, reducing the net effect. SOD was a major sink especially between RM 21.4 and 15.0 where the highest measured SOD rate was applied. Nonliving organic matter from the upstream boundary (Jordan), tributaries, and in-stream algal production represented a major source of oxygen demand in both summers. Effluent CBOD and NH4 loads from the WWTPs played a smaller role in 2006 compared to 1988 due partly to more river flow and dilution in 2006 and partly to treatment upgrades. Removing all effluent CBOD loads from the two WWTPs in the 2006 model resulted in little or no change in river DO concentrations.

Effluent quality at the Blue Lake and Seneca WWTPs has improved greatly since the WLA study in 1985. CBOD, NH4, and TP concentrations and loads are consistently well below permit limitations. Effluent characteristics have changed as well, with organic matter becoming more refractory (that is, slower to degrade) and phosphorus becoming less biologically available. While current effluent CBOD and NH4 loads have little effect on river DO concentrations, the two WWTPs continue to enrich the river with phosphorus and nitrogen, contributing to eutrophication in the lower Minnesota River and at downstream locations on the Mississippi River.

The Black Dog GP cooling-lake outfalls and MSP airport stormwater outfalls were challenging to monitor and model. Additional work is needed to understand their impact on the water quality of the lower Minnesota River. Limited sampling from 2005 and 2006 indicated that the Black Dog complex (that is, facility, lake, and watershed) can affect river water quality—sometimes negatively, sometimes positively. With its very dynamic withdrawal and discharge, Black Dog's effects could vary on a daily if not hourly basis. From a budgetary analysis of July 15 – September 30, 2006, James (2007) noted that Black Dog may be a potential sink for nitrogen and source for sediment and phosphorus but recommended further study. As a result of airfield improvements in 2005, CBOD loads from the MSP stormwater outfalls have decreased, but characteristics such as the decay rate and ultimate-to-5-day ratio have not been adequately studied.

The Lower Minnesota River Model was calibrated against seven years: three years with enhanced monitoring, WY 2004-2006; a drought year, WY 1988; and three contiguous years with a variety of flows, WY 2001-2003 (Smith et al., 2010). The model simulates discharge, water elevation, temperature, total dissolved solids, inorganic suspended solids, DO, SRP, NH4, NO3, silica, three groups of BOD, three groups of phytoplankton, and four forms of organic matter. The two-dimensional model grid includes 90 longitudinal segments and up to 111 vertical layers. The calibration strategy focused on performance during summer low-flow conditions. A set of parameters was developed to meet performance targets in the model of 1988, and then this set was successfully applied to models of 2001-2006. The settings for model coefficients are identical in all years with the exception of faster CBOD decay rates for the WWTPs in 1988, which reflect measured changes in effluent characteristics before and after treatment upgrades.

The seven years provided a wide range of hydrologic variability (Smith et al., 2010). The fact that one calibration was developed that captured the trends in water quality over a range of flows 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 $7Q_{10}$ 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 (Smith et al., 2010). With rare exceptions, the statistical measures of model performance were excellent and met calibration targets. 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 days 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 miles). This is a conservative approach but results in more certainty about the model statistics.

Four loading scenarios were applied to the model to demonstrate its potential use in facility and watershed planning (Smith et al, 2010). In one scenario, output from the Minnesota River Basin

Model (HSPF) was translated and used as input to the CE-QUAL-W2 model, showing the Lower Minnesota River Model's ability to act as a bridge to other modeling efforts. The scenario results were reasonable, adding confidence in the model's performance and utility. The results of the calibration and application of the Lower Minnesota River Model show that it is an acceptable tool for studying dissolved oxygen, nutrients, phytoplankton, and turbidity under a variety of conditions.

14 RECOMMENDATIONS FOR FUTURE WORK

Information from monitoring and modeling projects can identify changes to the monitoring program that are needed to track the progress of implementation plans and prepare for future studies and management decisions. It can identify deficiencies, such as important but infrequently monitored pollutant sources, and it can identify excesses, such as frequent sampling of a minor source. Model sensitivity analyses can guide monitoring priorities. Results from modeling and monitoring can fine tune sampling elements (e.g., protocols, variables, locations, and frequencies) and maximize the efficiency of monitoring programs.

Similarly, modeling is an iterative process. This project benefitted from previous modeling of the lower Minnesota River and concurrent modeling of the Mississippi River. During the project, we discovered areas in the model framework that could be improved. A future effort will hopefully benefit from our observations and bring modeling to the next level.

14.1 Modeling

- Add sediment transport to the CE-QUAL-W2 model. Basic scour and deposition routines comparable to those in the HSPF model would have benefited this project and provided an improved link between the Minnesota River Basin and Mississippi River models.

- Add sediment diagenesis to the CE-QUAL-W2 model. While the Lower Minnesota River Model was generally insensitive to sediment nutrient fluxes, it played a role during summer low-flow conditions and may be important in systems with deeper impoundments. Sediment oxygen demand plays an important role in the river under summer low-flow conditions.

- Add inorganic particulate P as a state variable in the CE-QUAL-W2 model to capture equilibrium processes. These processes are important in the Lower Minnesota River.

- Add variable stoichiometry to phytoplankton kinetics in the CE-QUAL-W2 model. While less important in highly eutrophic systems like the Minnesota River, this feature becomes more important in systems with low nutrient levels and in model projections with reduced nutrient loads. River miles 40 to 20 of the lower Minnesota River show the potential for P limitation of algal growth. Adding this feature would make CE-QUAL-W2 comparable to the ECOMSED-RCA model.

– Support separate input files for water-quality variables measured at different frequencies in the CE-QUAL-W2 model (Smith et al., 2010).

– With additional data, improve modeling of the effects of the Black Dog GP, cooling lake, and watershed.

– With additional data, improve modeling of the stormwater discharges from the MSP airport.

14.2 Monitoring

– Conduct a field study on the effects of barge traffic on the water quality of the lower Minnesota River, and if important, build a navigation model linked to the CE-QUAL-W2 model.

– Conduct a study of the effectiveness of aeration at the Seneca WWTP. MCES conducted an aeration study of the river during September 21-25, 1998, but conditions were not ideal (e.g., water temperatures were cooling). In early September 2006, MPCA (2007c) measured DO concentrations ranging from 9 to 13 mg/L near the end of the effluent pipe when MCES reported effluent DO concentrations greater than 17 mg/L. With a probe inserted into the effluent pipe on September 17 and 25, 2009, MCES measured DO concentrations of 23.4 and 24.8 mg/L when the facility was aerating.

– Conduct water-quality monitoring of the Black Dog GP outfalls during the summer when the facility is withdrawing large portions of river flow. Long-term monitoring would inform future model development.

– Conduct ultimate CBOD tests on representative samples from stormwater discharges at the MSP airport to determine the CBOD characteristics after the airfield improvements.

– Many of the model enhancements listed in Section 14.1 would require additional monitoring of the lower Minnesota River to support these features.

14.3 Laboratory

– Develop analytical methodology for determining low levels (<10 mg/L) of particulate organic carbon (OC). Current technology utilizes combustion methods that are not precise enough for low levels. High precision at these levels is required to be able to distinguish between soluble and total OC fractions. The CE-QUAL-W2 user manual recommends monitoring total OC at a minimum and the dissolved and/or particulate fractions if resources allow. These data can improve the simulation of organic matter and phytoplankton.

– Develop analytical methodology for precise BOD_5 and $CBOD_5$ determinations below the current arbitrary limit of 2 mg/L. The limit is due to the requirement for a minimum DO depletion of 2 mg/L in the BOD bottle. This is important for modeling the effects of advanced facilities with effluent BOD concentrations on the order of ambient concentrations.

– Develop analytical methodology for reliably determining CBOD by inhibiting any nitrogenous demand present. Currently, only one product is available as an inhibitor, and it is exhibiting biodegradation in many samples. Using nitrate and nitrite measurements to track nitrification is a very costly and imprecise substitute for inhibition. CBOD decay and nitrification are tracked separately in the CE-QUAL-W2 and other water-quality models, so reliable CBOD measurements at a reasonable cost are needed.

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