Chippewa River, Minnesota
Technical Support for

Un-ionized Ammonia TMDL

Prepared for:

U.S. EPA Region 5
Minnesota Pollution Control Agency

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General information on TMDLs can be found on the Web at the following sites:
Minnesota Pollution Control Agency
www.pca.state.mn.us/water/tmdl/
www.pca.state.mn.us/water/basins/mnriver/index/
U.S. Environmental Protection Agency
www.epa.gov/owow/tmdl/

Executive Summary

The lower segment of the Chippewa River (segment 07020005-501, first listed in 1998 as segment 07020005-001 see note below), running from the confluence with the Minnesota River upstream 12.6 miles to the Watson Sag Diversion, has been listed by the State of Minnesota as not supporting designated uses due to elevated concentrations of un-ionized ammonia. Un-ionized ammonia is toxic to fish and other aquatic life, and Minnesota has established a water quality criterion of 0.04 mg/L or less of un-ionized ammonia instream to protect aquatic life.
Because the waterbody has been formally listed as impaired on Minnesota's Clean Water Act Section 303(d) list submitted to EPA, this segment of the Chippewa River has been scheduled for development of a Total Maximum Daily Load (TMDL), specifying how much ammonia can be loaded to the river without contravening water quality standards, and how that ammonia load should be allocated among point and non-point sources. This document, developed under the auspices of U.S. EPA Region 5, is intended to provide technical support to Minnesota for the estimation of the TMDL.

Monitoring data upon which this segment was listed as impaired are collected downstream of the Montevideo, MN permitted wastewater treatment plant (WWTP) discharge. This point source is the major contributor of ammonia to the reach under dry weather, low flow conditions. The waterbody was originally listed as impaired based on monitoring data collected during the late 1980s and early 1990s. In April 1993, the Minnesota Pollution Control Agency (MPCA) issued a NPDES permit that authorized construction of an upgraded facility with an expanded discharge, and, in 1994, MPCA assigned ammonia limits for the first time.

Monitoring in 2001 revealed exceedances of the un-ionized ammonia criterion. Examination of the data reveals that the exceedances of the standard noted instream during 2001 occurred during a time period in which the Montevideo WWTP was out of compliance with its discharge permit, and was loading more ammonia to the river than is allowed. This suggested that the waterbody segment might best be addressed through a compliance enforcement order, followed by a de-listing from the 303(d) list. Such a conclusion would be warranted, however, only if the limits established in the Montevideo permit are indeed fully protective of water quality standards in the presence of existing and permitted upstream loads of ammonia. This analysis concludes that the approach used by MPCA to develop the permit limits is generally sound. Modifications are suggested to explicitly account for the margin of safety in the permit limits and to promote additional monitoring to better assess ammonia loads in the watershed.

To evaluate these issues, two water quality models were created. The first is an application of the HSPF watershed model to the entire Chippewa River watershed. The HSPF model was adapted from an existing modeling effort in the Minnesota River basin to assess the relative impact of point and nonpoint ammonia loads in the Chippewa River watershed. The second model is a spreadsheet-based dilution analysis of the potential impacts of Montevideo permitted ammonia discharges across historically observed, seasonal instream low flows. **These analyses revealed that nonpoint and point source loads other than the Montevideo WWTP contribute a small portion of the total ammonia load in the watershed. As a result, the TMDL analysis primarily focused on the wasteload allocation for the Montevideo WWTP.**

The MPCA calculates mass limits using the permitted concentration limits coupled with the average wet weather (AWW) design flow for the WWTP, which in the case of Montevideo was 2.47 mgd. This wet weather hydraulic design accommodates the collection system infiltration and inflow that is expected under wet weather conditions. When the treatment plant is experiencing maximum wet weather flow rates, it is highly unlikely that the river will be at a low flow, drought condition. Therefore, the critical load discharged by the WWTP under wet weather conditions should be coupled with a river condition somewhat greater than the dry weather low flow condition. Establishment of a representative wet weather design flow for the river can be accomplished by using the dry weather low flow (the ${}_{30}Q_{10}$) multiplied by a factor representing the ratio of WWTP wet weather to dry weather design flows; i.e., the ratio AWW/ADW. This approach is consistent with past recommended procedures (Winslow, MPCA 1985).

Results of the analysis provide the following TMDL seasonal wasteload allocations (WLA), load allocations (LA), and explicit margin of safety (MOS) under dry weather and wet weather conditions:

Table ES-1. Dry Weather Seasonal TMDL, WLA, LA, and MOS

Season	TMDL (kg/day)	WLA (kg/day)	LA (kg/day)	MOS (kg/day)	% MOS
Spring	197.5	86.5	30.0	81.1	41.1%
Summer	13.4	7.0	3.2	3.3	24.5%
Fall	40.4	24.0	3.8	12.6	31.3%
Winter	61.3	36.0	2.4	22.9	37.3%

Table ES-2. Wet Weather Seasonal TMDL, WLA, LA, and MOS

Season	TMDL (kg/day)	WLA (kg/day)	LA (kg/day)	MOS (kg/day)	% MOS
Spring	659.9	289.7	100.0	270.1	40.9%
Summer	44.8	23.4	10.5	10.9	24.4%
Fall	135.1	80.4	12.7	42.0	31.1%
Winter	204.9	120.6	8.1	76.3	37.2%

An additional implicit margin of safety was accounted for in the analysis by using conservative assumptions when determining the in-stream low flow discharge rates.

Note: The Chippewa River as listed in 1998 was divided into two listings and the length of the listed segment changed. The length of the listed segment is 12.6 miles. The assessment unit ID has been changed to 07020005-501. The original reach for the Chippewa River assessed in the 1998 305b report is 07020005-001, Chippewa River, Dry Weather Creek to Minnesota River, 11.20 miles. The reach length was based on Reach File 1. For the 2002 305b report, the reach was split into two reaches because of the Watson Diversion and the reach length was based on the National Hydrography Dataset (NHD).
07020005-501, Chippewa River, Watson Sag Diversion to Minnesota River, 12.6 mile
07020005-502, Chippewa River, Dry Weather Creek to Watson Sag Diversion, 3.2 miles

1 Introduction and Description of the Watershed

An 12.6-mile segment of the Chippewa River (HUC-segment: 07020005-501) appears on Minnesota's 2004 303(d) list as not supporting designated uses due to elevated concentrations of un-ionized ammonia. This report describes the estimation of a Total Maximum Daily Load (TMDL) to achieve the water quality standard for un-ionized ammonia in the Chippewa River.

The Chippewa River starts in northeast Douglas County and flows about 130 miles southwest to Montevideo, Minnesota, at the confluence with the Minnesota River in Chippewa County, Minnesota. The Chippewa River Watershed is one of the largest watersheds in Minnesota and covers a 2,085 square mile area that includes portions of Chippewa, Kandiyohi, Swift, Stearns, Pope, Stevens, Douglas, Grant, and Otter Tail Counties The southernmost 12.6-mile segment that is on Minnesota's 2004 303(d) list flows from the Watson Sag Diversion to the Minnesota River. The Chippewa River watershed and the location of the listed segment are shown in Figure 1-1.

1.1 SETTING

The Chippewa River watershed covers three ecoregions: the North Central Hardwood Forest, the Northern Glaciated Plains, and the Western Corn Belt Plains. The North Central Hardwood Forest includes the Belgrade-Glenwood Outwash Plain along the east-central edge of the basin and the Alexandria Moraine Complex along the remainder of the eastern half of the watershed. The Northern Glaciated Plains include the Big Stone Moraine on the far western edge, the Appleton-Clontarf Outwash Plain along the lower Chippewa River, and the Benson Lacustrine Plain within the south-central section of the watershed. A description of each ecoregion is included in Table 1-1 (MPCA 2002) and EPA (2002).

Table 1-1. Chippewa River Watershed Ecoregions

Geomorphic Setting	Soil Description	Slopes (percent)
North Central Hardwood Forest Ecoregion		
Belgrade-Glenwood Outwash Plain	Well-drained, sandy, and loamy	2-6
Alexandria Moraine Complex	Well-drained, loamy, silty, sandy, and mucky	6-45
Northern Glaciated Plains		
Big Stone Moraine	Well-drained, silty, and loamy	6-12
Appleton-Clontarf Outwash Plain	Poorly-drained and extensively tiled	2-6
Benson Lacustrine Plain	Poorly-drained, extensively tiled, silty clay to silt loam	0-2
Western Corn Belt Plains	Glaciated till plains, hilly loess plain	--

1.2 Climate

The climate in the Chippewa River Watershed is continental, with cold dry winters and warm wet summers. According to historical data (MPCA 2002), average temperatures range from 10.1 °F in January to 72.5 °F in July, and average annual precipitation ranges from 25 to 28 inches. About 18 inches of precipitation fall from May through September.

1.3 Hydrology

One active USGS stream gaging station is located in the watershed. The station is located on the Chippewa River near Milan, Minnesota, at Highway 40. Flow has been monitored at this station since 1937 and data are available to present. Six additional gaging stations were installed throughout the

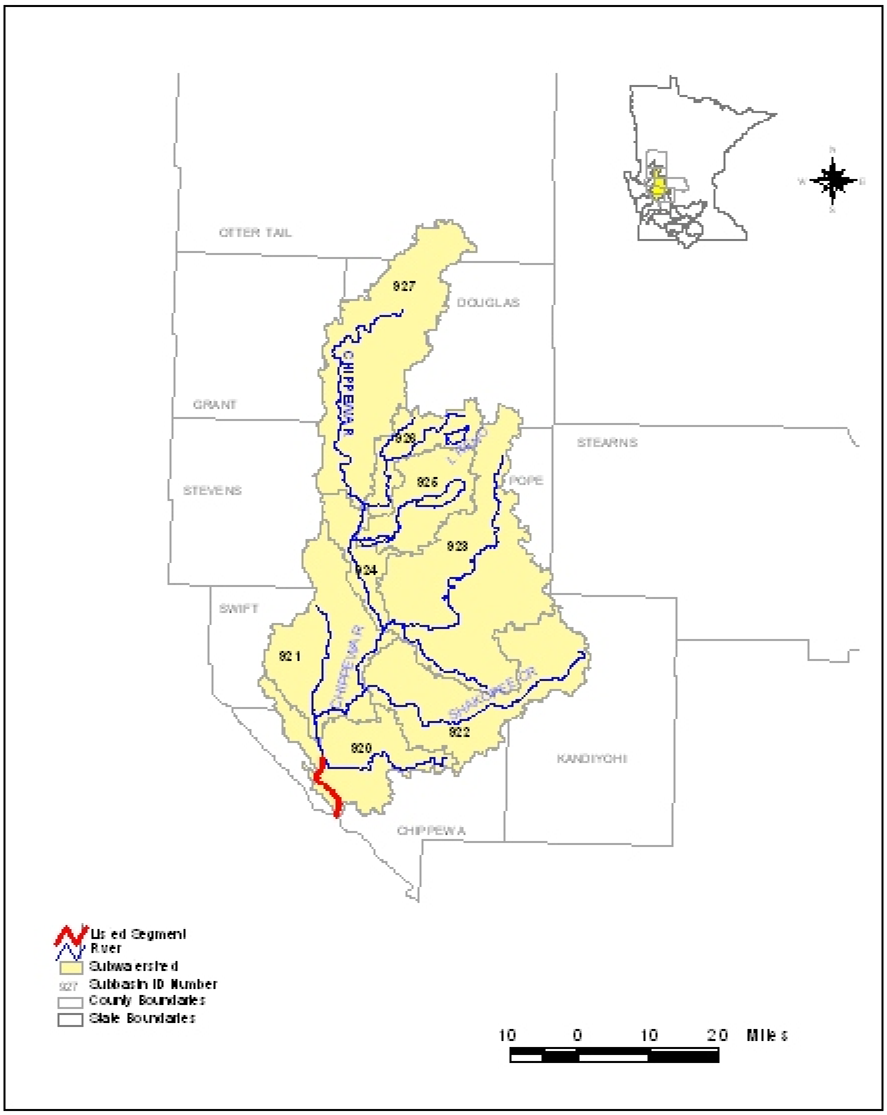

Figure 1-1 Chippewa River Watershed and Listed Segment 07020005-501

watershed as part of a diagnostic study performed by the Chippewa River Watershed Project (CRWP). Daily flow measurements were recorded from April through October in 1998 and 1999 at each CRWP automatic sampling station. According to CRWP data collected in 1998 and 1999, flows in the Chippewa River vary from 66.5 cubic feet per second (cfs) at Cyrus (near the headwaters) to 159 cfs at Milan during low flow conditions and from 579 to 2,910 cfs during high flow conditions (Olson and Churchill, 2000).

There is a U.S. Army Corps of Engineers flood control project on the Chippewa upstream of Montevideo, Minnesota. This project provides a flow diversion from the Chippewa River into the Lac qui Parle Reservoir on the Minnesota River. The flow diversion is designed to provide flood protection to downstream areas including Montevideo. The project was authorized by Congress in 1936 and construction was completed in 1951 (http://www.mvp.usace.army.mil/flood_control/LacQuiParle/; accessed 9/4/2002). The diversion consists of two control structures, the Chippewa diversion dam and the Watson Sag Weir. Flood waters are diverted through the weir into the Watson Sag which flows into the Lac qui Parle Reservoir upstream of the Chippewa River confluence with the Minnesota River, effectively bypassing the Chippewa River. The structures work in concert to store and release flood waters while maintaining some flow in both the Chippewa River and the Watson Sag.

During non-winter months, the inflow is split approximately equally between the Watson Sag and the Chippewa River below the diversion. However, during low flow conditions, the U.S. Army Corps of Engineers sometimes lets more water go through the diversion dam and allows only 10 to 20 percent of the flow to go over the Sag under the theory that the Chippewa River has more critical in-stream flow needs (Kenton Spading, U.S. ACOE, via telefax). As winter approaches, the gate at the Chippewa diversion dam is closed and subsequently becomes frozen shut. During icing periods, a low-flow outlet diverts about 10 percent of inflow into the Watson Sag (in order to maintain its aquatic habitat) while the remaining 90 percent continues down the Chippewa River (http://www.crh.noag.gov/ncrtc/forecast_groups/min/ wtsm5/wtsm5_new.html; accessed 9/4/02).

1.4 Monitoring Data

Water Quality monitoring data for un-ionized ammonia were available from USGS and MPCA from a number of stations in the Chippewa River Watershed (Table 1-2). Although none of the stations have a continuous period of record during 1971 to 1994, there were a total of eight exceedances of the un-ionized ammonia criterion out of 182 samples. Seven exceedances occurred downstream of the Montevideo WWTP: six at station CH-0.5 and one at USGS 05305400. One occurred elsewhere, downstream from a WWTP in Glenwood, Minnesota with a concentration of 41 µg/l. Locations, dates, and concentrations of the violations at Montevideo are summarized in Table 1-3. A time series of un-ionized ammonia from 1971 to 1994 at CH-0.5, immediately downstream of the Montevideo WWTP discharge location, is shown in Figure 1-2.

Table 1-2. Water Quality Monitoring Stations in the Chippewa River Watershed with Un-ionized Ammonia Data During 1971 to 1994.

Station Number	Station Description	Sample Count
CH-0.5	CHIPPEWA RIVER AT BRIDGE ON MN-7 AT MONTEVIDEO	105
CH-20	CHIPPEWA RIVER SH-40 E OF MILAN	39
CH-77.6	CHIPPEWA R AT CSAH-18 4 MI SE OF CYRUS	2
CH-79	CHIPPEWA R AT ROAD BTN S34/3 3.5 MI SE OF CYRUS	2
CH-80.8	CHIPPEWA R AT ROAD BTN S27/34 2.5 MI SE OF CYRUS	2
CH-81.9	CHIPPEWA R IN T125NR40WS28NWQSEQ EAST OF CYRUS	2
CH-84.2	CHIPPEWA R AT MN-28 AT CYRUS	3
CH-153.4	CHIPPEWA R BR AT CSAH-7 0.5 MI S OF MILLERVILLE	1
CH-154.5	CHIPPEWA R BR AT CR-60 0.5 MI E OF MILLERVILLE	1
CHL-0.8	L CHIPPEWA R AT CR-73 3.5 MI SE OF CYRUS	2
OUT-8.6	OUTLET CK AT CSAH-14 5 MI SW OF STARBUCK	2
OUT-11	OUTLET CK RD IN SEQ OF S8 4.5 MI SW OF STARBUCK	2
OUT-12	OUTLET CK AT ROAD BTN S5/8 4 MI SW OF STARBUCK	2
OUT-14.2	OUTLET CK AT RD BTN S33/34 2.5 MI SW OF STARBUCK	2
OUT-16.7	OUTLET CK DIRECTLY ABOVE STARBUCK WWTP DISCHARGE	1
OUT-17.1	OUTLET CK AT CR-29 AT STARBUCK	1
PRK-0.02	PERKINS CK AT CSAH-54 AT GLENWOOD	1
PRK-0.4	PERKINS CK AT 6TH ST T125N/R38W/S12/NWQ/NEQ/SEQ	1
PRK-0.5T1	PERKINS CK TRIB IN T125N/R38W/S12/NEQ/NWQ/SEQ	1
PRK-0.5T2	PERKINS CK TRIB IN T125N/R38W/S12/NEQ/NWQ/SWQ	1
PRK-0.9	PERKINS CK AT 8TH AVE T125N/R38W/S12/NEQ/NWQ/NEQ	1
SIG-0.4	SIGNALNESS C RD BTN S15/16 4.5 MI SW OF STARBUCK	2
WKY-INLET	DI TO WHISKEY L IN T129N/R39W/S22/SWQ NR BRANDON	1
USGS 05305400	CHIPPEWA RIVER AT MONTEVIDEO, MN	5

Table 1-3 Exceedances of Un-ionized Ammonia Criterion in Segment 07020005-501, 1971 to 1994.

Station	Date	Un-ionized Ammonia Concentration (µg/l)
CH-0.5	5/18/1981	81
CH-0.5	7/11/1985	41
CH-0.5	7/6/1988	248
CH-0.5	8/11/1988	259
CH-0.5	9/6/1988	252
USGS 05305400	8/15/1989	296
CH-0.5	8/31/1994	53

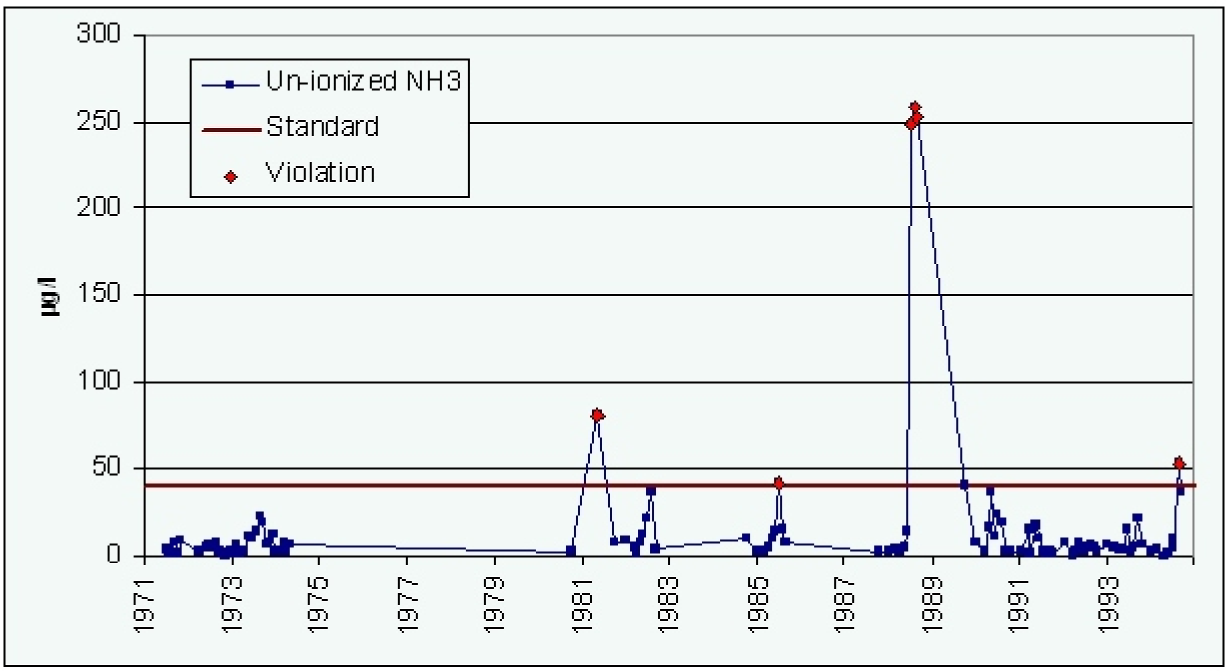

Figure 1-2 Un-ionized Ammonia at Station CH-0.5 1971-1994

Regular ambient monitoring of the Chippewa River at Montevideo by MPCA ceased in 1994. Ammonia problems at this site were believed to be resolved by an upgrade and new permit limits at the Montevideo WWTP. Additional monitoring by MPCA was conducted in 2001 and revealed continuing exceedances of the un-ionized ammonia criterion. Six measurements were taken during 2001 at CH-0.5, two of which (33 percent) were in excess of the un-ionized ammonia standard during 2001 (186 µg/l on 8/28 and 52 µg/l on 9/19). As discussed in Section 3.1.1, the Montevideo WWTP was discharging ammonia in excess of its permit effluent limits during this period.

2 Water Quality Standards and Numeric Water Quality Targets

2.1 USE CLASSIFICATION AND WATER QUALITY STANDARD FOR UN-IONIZED AMMONIA

Chippewa River segment 07020005-501 is assigned use classifications of 2B, 3B, 4A, 4B, 5, and 6 under Minnesota Pollution Control Agency Rules. These classifications include aquatic life and recreation, industrial consumption, agriculture and wildlife, aesthetic enjoyment and navigation, and other uses. Class 2B signifies waters that must support aquatic life, including cool and warm water sport or commercial fish, and recreation, including swimming. These use classifications do not include protection as a source of drinking water. For surface waters not protected for drinking, most of the applicable standards are associated with Class 2. In general, if the Class 2 standards are met, other uses such as industrial and agricultural uses are also protected.

The Class 2B standard for un-ionized ammonia is 0.04 mg/L expressed as Nitrogen (Minnesota Rule 7050.0222 Subp. 4). It is a chronic standard, which is defined as “the highest water concentration of a toxicant to which organisms can be exposed indefinitely without causing chronic toxicity.” The basis of the standard is for the protection of the aquatic community from toxicological effects.

2.2 TMDL ENDPOINTS

Ammonia is a natural by-product of biological activity resulting from the decomposition of nitrogen-containing compounds and from the hydrolysis of urea. Sources of ammonia include 1) excretion, and 2) decomposition of food and fecal matter. Sources of ammonia can be natural or man-caused such as natural organic plant and animal matter, municipal and industrial wastewaters, and runoff from animal feedlots. Ammonia dissolves easily in water where it is found in two forms, NH3 (un-ionized) and NH4+ (ionized). The total amounts of these two forms is the "Total Ammonia Nitrogen," simply referred to as “ammonia” in this document. The un-ionized form (NH3) is toxic to fish and other aquatic life, while the ionized form (NH4+) is not. The percentages of un-ionized to ionized ammonia is a factor of water temperature and pH, as discussed below. As a result, the TMDL endpoint is expressed as the total seasonal ammonia concentrations allowable in the Chippewa River to achieve the water quality standard for un-ionized ammonia under critical flow conditions.

The un-ionized ammonia Class 2B water quality standard of 0.04 mg/L is the indicator for whether this reach of the Chippewa River supports uses. MPCA’s stated recurrence interval for achieving the standard calls for exceedances no more than once in three years. While Minnesota regulations do not provide an explicit statement of the averaging period or acceptable frequency of exceedances for chronic criteria for the protection of aquatic life, MPCA uses internal guidance documents that detail technical methodology and guidelines for a) the development of water quality criteria for toxic substances[1], and b) development of water quality-based effluent limitations for toxic substances[2]. For interpretation of the standard as a water quality target this TMDL therefore relies on MPCA guidance which are generally consistent with U.S. EPA guidance.

[1] *Guidelines for the Development of Surface Water Quality Standards for the Protection of Aquatic Life, including Human Health and Wildlife.* MPCA Guidance Document, Jan. 1990, Rev. July 1993, Rev. Aug. 2000 (draft).

[2] *Methodology for the Development of Water Quality-Based Effluent Limitations for Toxic Substances.* MPCA Guidance Guidance Document, Jan. 1995.

The Minnesota guidance addresses both the magnitude (i.e., how much of a pollutant (or pollutant parameter such as toxicity), expressed as a concentration, is allowable) and the duration (the period of time (averaging period) over which the instream concentration is averaged for comparison with criteria concentrations) for compliance with the water quality standard.

Magnitude

The magnitude component of the target is the chronic criterion of 40 µg/L un-ionized ammonia, as expressed in the Minnesota Rule.

Duration: Averaging Periods

For protection of aquatic life, U.S. EPA derives acute criteria from 48- to 96-hour tests of lethality or immobilization. U.S. EPA derives chronic criteria from longer-term (often greater than 28-day) tests that measure survival, growth, reproduction, or in some cases, bioconcentration.

For chronic criteria, U.S. EPA recommends an averaging period of 4 days (EPA 1985), implying that the 4-day average exposure should not exceed the continuous (chronic) concentration criterion. U.S. EPA selected the 4-day averaging period based on the shortest duration in which chronic effects are sometimes observed for certain species and toxicants.

Under MPCA guidance, the duration of compliance for ammonia can be averaged over a 30-day period as long as the effluent and ambient variability of ammonia concentration is low. In practice, ambient ammonia concentrations averaged over four days should not exceed the chronic standard by more than a factor of two during a low flow period. Because these conditions are met in the Chippewa River, a 30-day duration period was used to assess compliance with the water quality standard.

In addition, compliance with the standard is assessed at critical low flow conditions, defined as the 30Q10 instream dilution flow. Here "30Q10" refers to the 30-day average low flow that occurs, on average, once every 10 years. Note that use of a design flow method also recognizes that occasional rare exceedances of the criterion are likely to occur during periods in which flow is less than the 30Q10.

The rule further specifies that allowable un-ionized ammonia concentrations should be calculated from pH, water temperature, and total ammonia according to a formula provided by Emerson et al. (1975). The percent of total ammonia that is un-ionized is given by

$$f = \frac{1}{10^{(pK_a - pH)} + 1} \times 100$$

where $pK_a = 0.09 + \frac{2730}{T}$ and T is the water temperature in degrees Kelvin (°C + 273.17).

Based on site-specific monitoring data for temperature and pH collected at eleven MPCA monitoring sites located in the Upper Minnesota River Basin (Carol Sinden, MPCA, personal communication), the seasonal total ammonia concentrations to be achieved to maintain compliance with the un-ionized ammonia water quality standard are presented in Table 2-1.

Table 2-1. Seasonal Temperature, pH, and Allowable Total Ammonia Concentrations to Achieve Compliance with the Un-ionized Ammonia Water Quality Standard

Season	Temperature ©)	pH	Total Ammonia (mg/L)
Spring (Apr - May)	14.0	8.2	1.04
Summer (Jun - Sep)	23.5	8.2	0.53
Fall (Oct - Nov)	10.5	8.2	1.35
Winter (Dec - Mar)	0.0	7.8	7.73

3 Source Assessment

3.1 PERMITTED POINT SOURCES

3.1.1 Montevideo WWTP

The Montevideo WWTP is located 0.5 miles upstream from the confluence of the Chippewa River with the Minnesota River, and has been operating under a permit since 1974. MPCA issued the plant's current operating permit in April 1993, which authorized construction of an upgraded facility with an expanded discharge. The plant is designed to treat an average dry weather flow of 0.74 million gallons per day (MGD), an average wet weather flow of 3.00 MGD, and a peak hourly flow of 4.20 MGD.

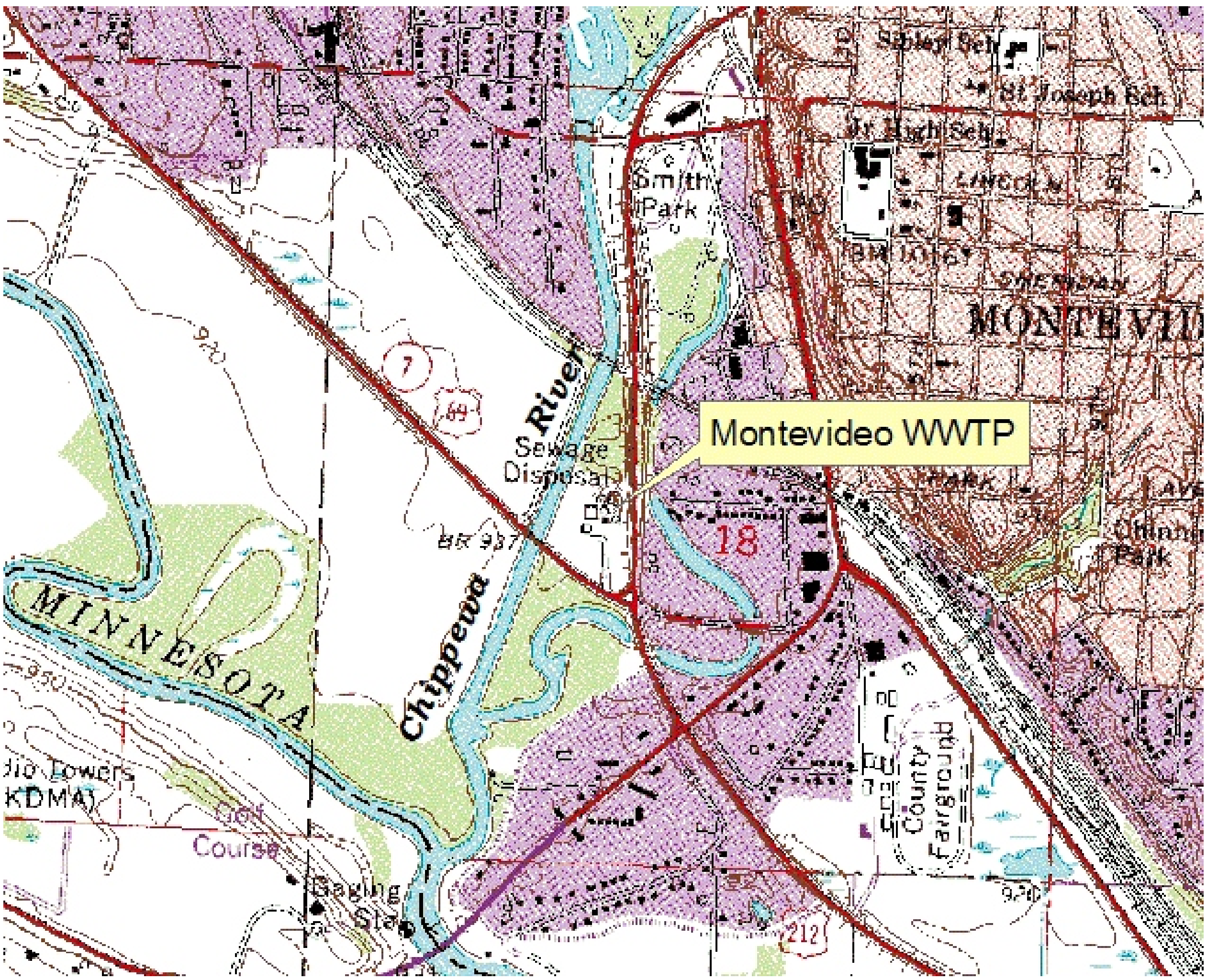

Figure 3-1 Location of the Montevideo wastewater treatment plant

When the treatment facility expanded in 1994, MPCA assigned ammonia limits for the first time and calculated them based on toxicity, using the Chippewa River 7Q10 low flow and the WWTP Average Dry Weather (ADW) design flow of 0.740 MGD. Because the treatment facility was expanding to an Average Wet Weather (AWW) design flow of 3.0 MGD and the discharge was a short distance upstream of an outstanding resource value water reach of the Minnesota River, MPCA froze mass loadings for $CBOD_5$ and TSS limits at the existing AWW design flow of 2.47 MGD. Mass limits for ammonia were also calculated using the AWW design flow of 2.47 MGD.

The permit includes specific concentration and mass limitations on discharges of 5-day CBOD, TSS, Ammonia N, Fecal Coliform Bacteria, pH, and Total Residual Chlorine. There are no limits related to temperature of the effluent. The pH must be maintained between 6.0 and 9.0 at all times. The actual average discharge during 1999-2001 was 0.899 MGD. The average concentration and load of Ammonia N from January 1998 to May 2002 were 1.32 mg/l and 5.25 kg/day, respectively. Figure 3-2 shows the permitted and actual concentrations and loads of Ammonia N during this period. Violations of the limits are shown in red. During the summer and fall of 2001, the Montevideo plant exceeded both the concentration and mass limits for ammonia and the concentration limit was exceeded in October of 1998 and June of 1999. The maximum reported monthly average concentration in the Discharge Monitoring Reports (DMRs) was 13.1 mg/l in October 1998 and the maximum reported monthly average load was 34.65 kg/day in August 2001. The Permit Compliance System (PCS) generated automatic notices of non-compliance for June to September 2001. In October through December the plant returned to compliance and the PCS system automatically generated a “back in compliance” notice. No enforcement action was pursued by MPCA, and no correspondence relating to the non-compliance period is in MPCA’s file (Enrique Gentzch, MPCA, personal communication, 8/13/2002).

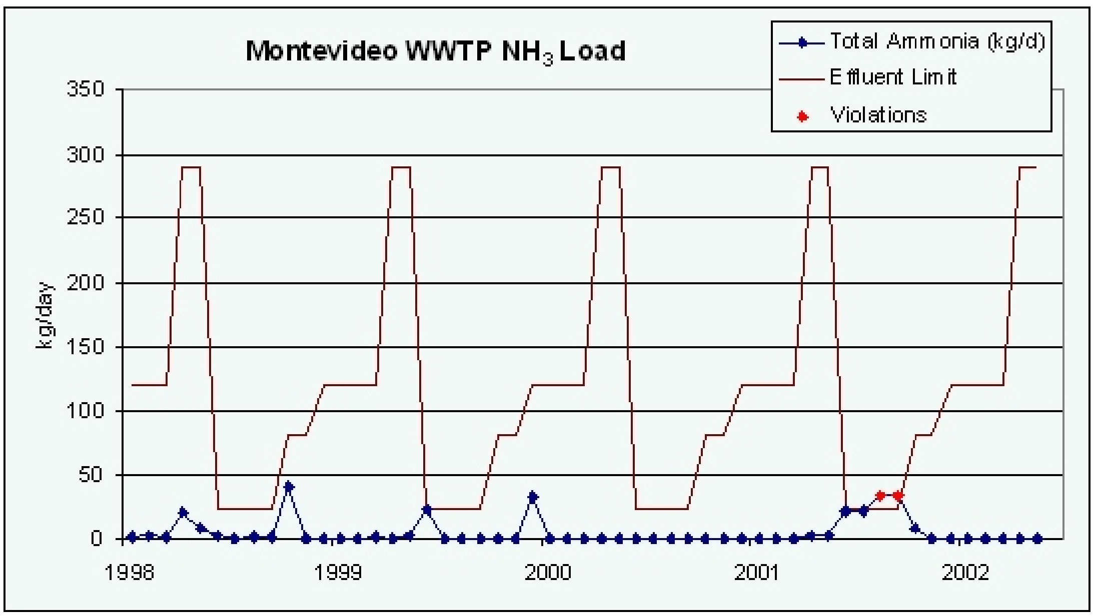

Figure 3-2 Ammonia Load in Montevideo WWTP Effluent 1/1998 - 5/2002

3.1.2 Other WWTPs and Stabilization Pond Systems

There are seven other WWTPs in the Chippewa River Watershed (Table 3-1 and Figure 3-3). None are close to segment 07020005-501 with the exception of the Watson WWTP, which also discharges to segment 07020005-501 and is about eight miles upstream from the confluence of the Chippewa River with the Minnesota River. According to data provided by MPCA, Watson’s average discharge during 1999-2001 (0.025 MGD) is very small relative to Montevideo’s average discharge during the same period.

Table 3-1 Other WWTPs with direct discharges to the Chippewa River Watershed

Plant	NPDES Permit Number	Average Discharge 1999-2001(MGD)	Average Discharge 1986-1992(MGD)
Benson WWTP	MN0020036	0.354	0.311
Hancock	MN0023582	0.138	0.124
Kensington	MN0021598	0.047	0.035
Kerkhoven	MN0020583	0.11	0.113
Starbuck	MN0021415	0.174	0.172
Watson	MN0022144	0.025	0.002

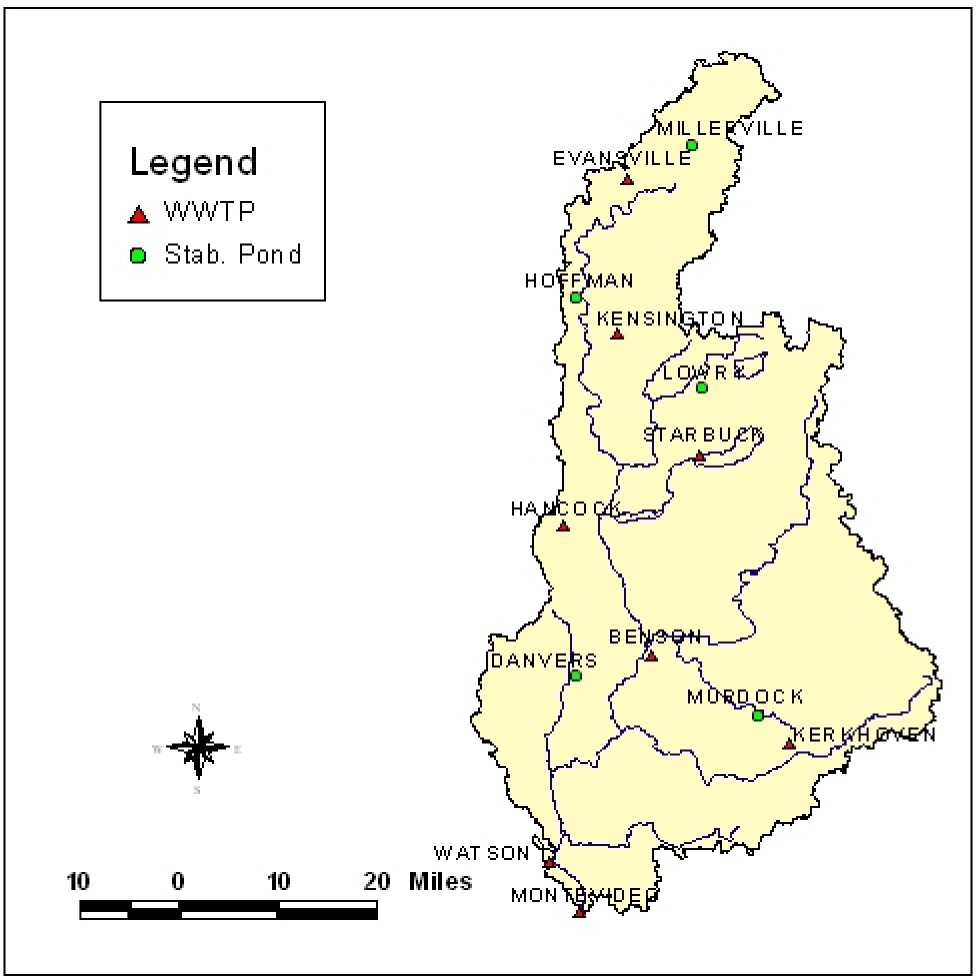

Figure 3-3 WWTPs and Stabilization Ponds in the Chippewa River Basin

Corrections: Evansville has a stabilization pond; delete Millerville.

The Benson WWTP is the only other plant in the watershed with permitted limits for ammonia (Table 3-2). It is about 48 river miles upstream of the confluence of the Chippewa River with the Minnesota River. During 1999-2001 it had an average discharge of 0.354 MGD and an average ammonia load of 0.54 kg/day (1.2 lb/day). Other WWTPs without ammonia limits are assumed to have an average NH_3 -N concentration of 7.3 mg/l, based on information provided by MPCA.

Table 3-2 Effluent Limits for Ammonia-N at Benson WWTP

Month	Monthly Average Daily Concentration	Monthly Average Daily Load
June - Sept	3 mg/l	8.87 kg/day (19.6 lb/d)
Oct - Nov	10 mg/l	29.56 kg/day (65.2 lb/d)
Dec - Mar	NA	NA
Apr - May	4 mg/l	11.82 kg/day (26.1 lb/d)

There are five stabilization pond systems in the Chippewa River Watershed of sufficient size to have a permit for discharging waste water (Table 3-3 and Figure 3-3). None discharge to segment 07020005-501. Stabilization ponds are permitted to discharge during two time periods, April 1 - June 15 and September 15 - December 15. Typically the operators perform two discharges of 7 to 10 days each per period, and the discharge events occur near the middle of the periods. However, at any one time not all ponds are discharging. For the purposes of the modeling analysis (Section 4), the spring and fall discharges from all ponds to a given reach were represented by a statistical, semi-circular distribution apportioned over the permitted discharge window. Annual discharge was assumed to be approximately 75 percent of inflow volume based on DMR data of the ratio of inflow to discharge for several representative stabilization ponds. The discharge volume was divided equally between the spring and fall discharge periods. Specific concentrations of pollutants in effluent from stabilization ponds are generally not monitored. For organic P, ortho P, nitrate, ammonia, organic N, and fecal coliform bacteria, typical spring and fall concentrations reported in an MPCA study were used (Helgen, 1992), and transformed into daily loads. Ultimate BOD was estimated from BOD5 data reported from DMRs for several representative ponds, using a conversion factor of 2.28.

Table 3-3 Stabilization Ponds in the Chippewa River Watershed

Pond	NPDES Number	1990 Discharge (MGD)	2000 Discharge (MGD)
Danvers	MN0025593	0.0036	0.0169
Evansville WWTP	MN0023329	0.073	0.060
Hoffman	MN0021199	0.0486	0.0643
Lowry	MN0024007	0.0299	0.0304
Murdock	MN0054305	0.0531	0.0501

3.1.3 Community Septic Systems and Home Septic Systems

The Minnesota River basin contains a significant number of households that are served by wastewater disposal systems that involve direct or semi-direct discharge to a tile drain line. In most cases, the effluent first passes through a septic tank, where settling and partial anaerobic digestion occurs. Some systems are believed to have direct discharges without a septic tank, but MPCA staff indicate that the majority of such systems do include a septic tank (personal communication with Jim Klang, 10/22/2001). In the Minnesota River basin, systems of this type include both community systems ("Unsewered Communities") and individual household wastewater disposal systems ("ISTS").

To determine the distribution of direct-discharge ISTS within model sub-watersheds, data were utilized from the Minnesota River Assessment Project (MRAP) Report (MPCA, 1994), Volume IV. A sub-report contained within Volume IV of MRAP, SWCD Methodology of Land Use Assessment (Mueller and Wehrenberg, 1993), provides numeric land use data associated with 32 original minor watersheds in a data matrix in Addendum H.

Data were extracted from the matrix for the total acreage, the total cropland acreage, and number of ISTS draining to tiles for each of the 32 minor watersheds. Under the assumption that most, if not all, of the ISTS are in rural farming areas, the density of ISTS was analyzed as a function of total cropland area within each minor watershed. Excluding watersheds with no ISTS draining to tiles listed, the number of direct-to-tile ISTS per 10,000 acres of cropland ranged from 1.55 in minor watershed #26082 within the Chippewa River watershed to 38.15 in minor watershed #30056 within the Blue Earth River watershed. The density of ISTS in the region of the Chippewa River Watershed is relatively small at 13 home septic systems per 10,000 acres of cropland.

3.2 Nonpoint Sources

3.2.1 Land Use

Land use for the model was initially set up to correspond to the period of available more intensive monitoring data (1986-92). Land use data for 1989 were obtained from the Minnesota Land Management Information System at the University of Minnesota (Figure 3-4). Further processing of the land use data was necessary to reclassify the data into land uses used by the HSPF model (see Section 4).

The model was calibrated to1986-92 monitoring data, using the 1989 land use. Simulation of current conditions required a land use update. Current land use data for the Chippewa River Watershed were not readily available as a spatial coverage. The data were updated to year 2000 estimates to reflect significant changes in land use and agricultural practices as follows:

1. Significant amounts of agricultural land have been retired or converted to native grass, wetlands, or woods under three programs: the USDA Conservation Reserve Program (CRP), the USDA Conservation Reserve Enhancement Program (CREP), and the Minnesota Reinvest in Minnesota (RIM) program. Current county-level totals for each program were obtained and spatially averaged in the GIS to estimate shifts in agricultural land use to other classes from 1989 to 2000.
2. Use of conservation tillage has increased greatly over the last decade. Year 2000 residue transects provided by audits at the local level were used to adjust the fraction of cropland classified as conservation tillage (greater than 30 percent residue).
3. Numbers of farm animals in the Chippewa River watershed were used to estimate the amount of manured lands in 1989. Since the same data were not available in 2000, MPCA estimates of manured lands in 2000 were used instead.
4. No changes were made in the urban land coverage.

Land use for 1989 and 2000 is summarized in Table 3-4.

Table 3-4 Chippewa River Watershed Land Use in 1989 and 2000

Land Use	1989	2000
Forest	5.69 %	6.36 %
Marsh/Wetlands	2.94 %	5.50 %
High Till Cropland	63.49 %	32.92 %
Low Till Cropland	9.07 %	32.89 %
Pasture/Grass	13.19 %	16.48 %
Manure Application	3.70 %	3.92 %
Urban	1.93 %	1.93 %

3.2.2 Animal Operations

Animal operations are a significant portion of agricultural activity in the Chippewa River Watershed and potential source of ammonia, most notably from manure application. The numbers of animals by type are shown in Table 3-5, as estimated from 1991 data from the Minnesota Agricultural Statistics Services. Livestock data from 1991 were used in the model because they provided a good fit with the 1986 to 1992 monitoring data used to calibrate the model. Also, the statistics for that year were more informative than more recent year's data. The *Beef* and the *Dairy* categories are a subset of the *Cattle and Calves* category. For 2000, complete data were available for *Cattle and Calves* and *Hogs and Pigs* only. MPCA estimated that total manure generation in the Chippewa River watershed increased by about 5.4 percent from 1991 to 2000 (personal communication from Nick Gervino, MPCA, to Klaus Albertin, Tetra Tech, 7/2/02).

Table 3-5 Livestock in the Chippewa River Watershed in 1991

Type	Number of Animals (1991)
Cattle and Calves	76000
Beef	10000
Dairy	18000
Hogs and Pigs	110000
Sheep and Lamb	6900
Layers and Broilers	129000

Calculation of the amount of manure generated on the land area receiving manure applications is important to the watershed modeling because manured lands receive significantly higher estimates of nutrient build-up rates (including ammonia) in the watershed model. For modeling purposes, the influence of animal manure is input as an equivalent area for manure application (see Tetra Tech, 2002). For the calibration period (1986-92), the land area receiving manure application in the Chippewa River basin was 46,732 acres; for 2000, the estimate is 49,508 acres.

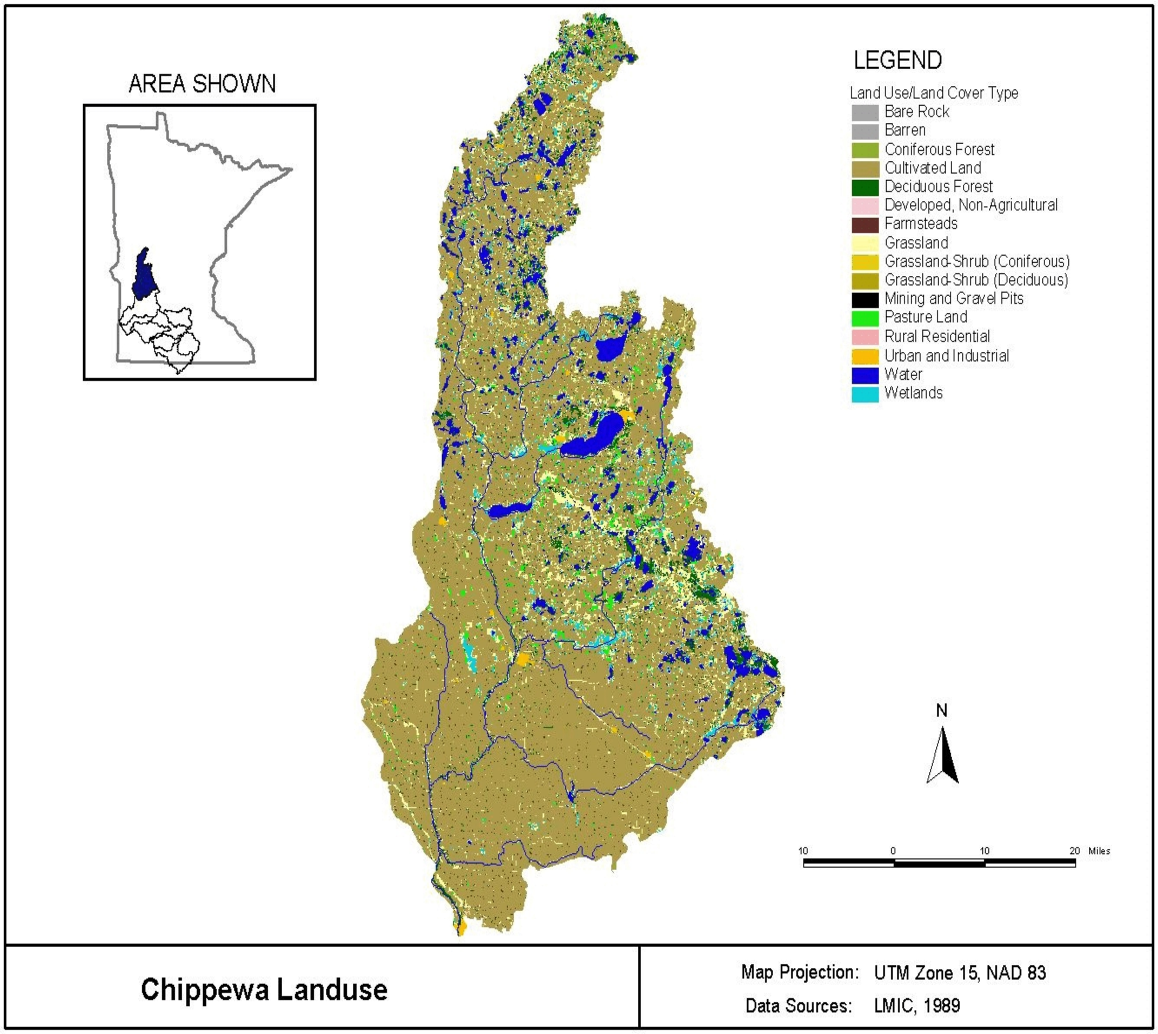

Figure 3-4 1989 Land use in the Chippewa River Basin

4 Technical Approach

4.1 Assessment of Relative Impact of Point and Nonpoint Source Loads

A linkage analysis was used to assess the relative impact of ammonia load from point and nonpoint sources in the Chippewa River watershed. The linkage analysis guided the selection of an appropriate TMDL modeling approach to determine the allowable pollutant loads in the watershed to achieve the un-ionized ammonia water quality standard.

The data analysis described in Section 3 suggests that exceedances of the un-ionized ammonia standard are primarily associated with discharges from the Montevideo WWTP, located a short distance upstream of the MPCA monitoring station at Chippewa River mile 0.5. There are other point and nonpoint sources of ammonia loading in the watershed. To assess the relative impact of point and nonpoint source loads in the watershed, a linkage analysis was developed for the TMDL to address all sources of ammonia loading and evaluate their impact over the range of flow and meteorological conditions that are present in the watershed. To complete the analysis, a dynamic watershed model was adapted from a previous analysis of the Minnesota River drainage basin to simulate point and nonpoint loading from the entire upstream Chippewa River watershed across the full range of potential conditions. Calculation of un-ionized ammonia concentrations also required estimates of temperature and pH. Finally, instream transformations, including the decay of organic material and growth cycles of algae are potentially important factors for the instream ammonia concentrations.

4.2 Linkage Analysis Model Selection

The model selected for the linkage analysis was the Hydrologic Simulation Program-FORTRAN or HSPF (Bicknell et al., 1996). This EPA-supported watershed model is widely used in TMDL applications, and provides a simulation of the complete hydrological cycle and the watershed and instream nutrient and algal cycles. The model is also capable of simulating water temperature and pH. In the Chippewa application, water temperature is simulated and calibrated; however, pH has not been simulated due to lack of sufficient data to calibrate the carbonate cycle. Instead, the un-ionized ammonia analysis was conducted using an upper-percentile estimate of observed pH.

As described below, the watershed model determined that the primary cause of instream water quality impairment during low flow conditions in the listed reach is loading from the Montevideo WWTP. In addition, the calibration of the model for ammonia did not provide a particularly good fit to individual observations at the monitoring station below Montevideo - due largely to the lack of loading data from the WWTP itself during the calibration period. Accordingly, a second, spreadsheet-based modeling tool was created to evaluate the dilution of Montevideo discharges and associated un-ionized ammonia concentrations using seasonal conditions. The spreadsheet model is described in Section 5 and forms the basis for the TMDL allocations.

The following discussion describes the linkage analysis of relative point and nonpoint source loads to provide the basis for the spreadsheet model used to develop the TMDL allocations.

4.3 MODEL DESCRIPTION

The Hydrological Simulation Program – FORTRAN (HSPF) is a comprehensive package developed by EPA and USGS for simulating water quantity and quality for a wide range of organic and inorganic pollutants from complex watersheds. HSPF includes components to address urban and agricultural watershed hydrology, surface water quality analysis, and pollutant decay and transformation on the land surface and in the water column. It is a continuous simulation model that typically operates on an hourly time step.

The Chippewa watershed model presented in this report was derived from a previous model of the watershed developed as part of a larger suite of linked HSPF models of the Minnesota River basin created by Tetra Tech for MPCA. Full details are provided in Tetra Tech (2002).

The model divides the watershed into eight sub-watersheds, numbered 920 through 927 (Figure 4-1). Land use in each of these subwatersheds is allocated to seven categories (plus open water), as described in Section 3. Montevideo is located near the outlet of sub-basin 920, at the downstream end of the watershed.

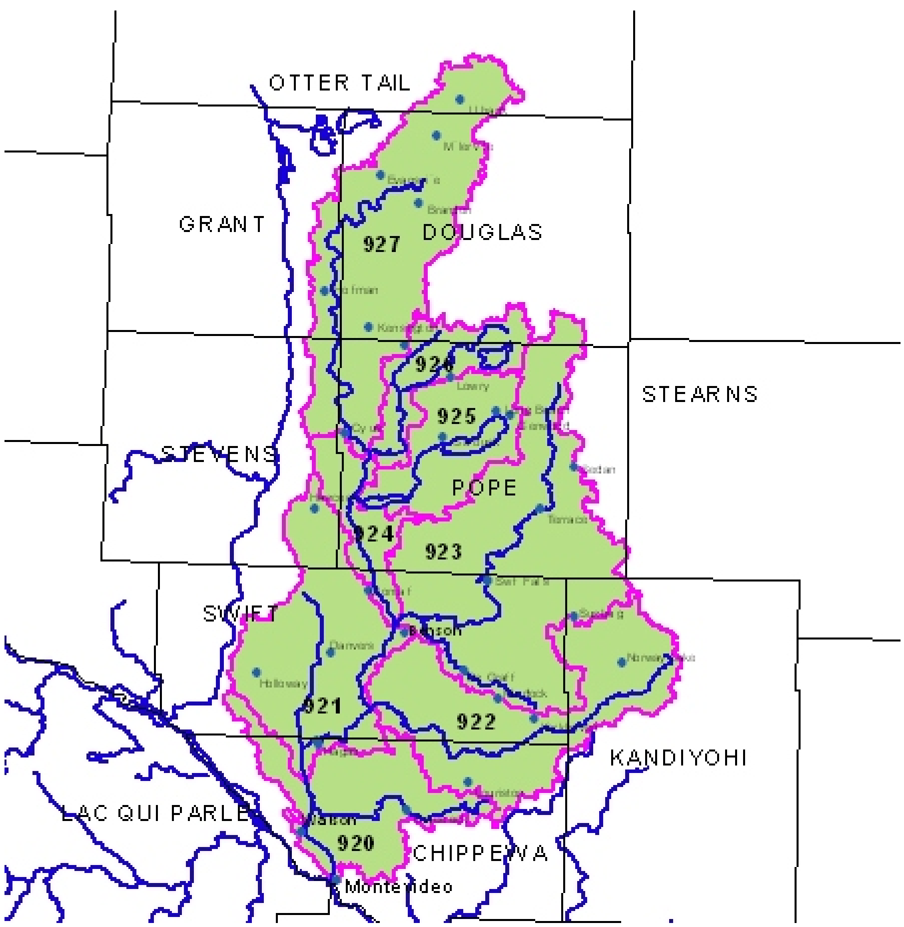

Figure 4-1. Sub-watersheds for Chippewa River HSPF Model

Tile Drainage

An important enhancement was made to the model to represent tile drainage. Land use in the Minnesota River basin is primarily agriculture (typically in corn/soybean rotation). A dominant characteristic of the basin is the presence of extensive tile drainage, as many of the soils are naturally poorly drained. Installation of tile drainage has converted what were predominantly glacial plain outwash depressional wetlands into productive farmland. The presence of tile drains, which include both surface and subsurface inlets, has radically altered the natural hydrology of the area. Surface inlet tile drains, in particular, may also play a significant role in the transport of sediment and pollutants from agricultural land to the river.

It is not feasible to simulate individual tile drain systems at the large basin scale. Further, neither the location nor the total density of tile drainage is known throughout the basin; in most areas, only the public tile drains and ditches are documented in spatial coverages, and the extent of private tile drains is known only for limited areas.

The HSPF model does not contain any routines for the explicit representation of tile drains. In the model, incoming moisture (precipitation, snow melt) cascades through a series of stores, including interception storage (storage above the soil surface), surface detention, upper zone soil storage, lower zone soil storage, and active groundwater. A significant portion of the total incoming moisture is re-emitted to the atmosphere as evaporation and plant transpiration ("evapotranspiration"). Excess moisture at the soil surface becomes surface runoff. Moisture that enters the soil profile is partitioned between interflow and storages that may result in ground water discharge (or loss to deep ground water).

In typical applications of HSPF, surface runoff represents the quick flow storm response, interflow an intermediate time-scale hydrologic response, and groundwater discharge the base flow hydrologic response. In such applications, interflow represents lateral movement of water through the shallow soil profile.

A large number of parameters describe the movement between the various moisture stores in the model. Key rate parameters include the infiltration rate, which controls movement of water from upper to lower soil zone storage, and the interflow inflow rate, which controls movement of water into interflow. These rates, combined with the available capacity in each of the stores, determine the disposition of water and the resulting shape of the outflow hydrograph.

At a gross or basin scale, the net effect of tile drainage is to move water relatively rapidly out of surface storage without direct surface drainage. Accordingly, it is to be expected that tile drainage is best represented in HSPF as an interflow component, with a response time that is somewhat slower than direct surface runoff, but quicker than groundwater discharge, represented by a relatively fast recession coefficient. In fact, tile drainage constitutes a range of different hydrologic response times. The fraction of the net discharge from tile drains that comes from surface inlets is a rapid-response component. However, tile drain outflow also contains a slower component that consists of subsurface flow that has percolated through the upper soil layers and into the drains through lateral soil flow. As a result, the net tile drainage can, in HSPF, be expected to be represented as a combination of interflow and ground water discharge.

There are some limitations to the representation of tile drainage as interflow in HSPF. Within HSPF, the rate of interflow inflow is determined in relation to infiltration. First, the infiltration capacity, IBAR (in/hr), is determined as

$$IBAR = (INFILT / (LZS/LZSN)^{INFEXP}) * INFFAC$$

where INFILT is the infiltration rate parameter (in/hr), LZS is the current lower soil zone storage (in) and LZSN is the nominal lower soil zone storage. The sum of interflow plus infiltration (IIBAR) is then determined via a ratio to IBAR:

$$IIBAR = IBAR * INTFW * 2.0^{(LZS/LZSN)}$$

where INTFW is the interflow inflow rate parameter (dimensionless). Thus, in HSPF the rate of interflow discharge depends on the extent to which the lower soil zone capacity is filled. In reality, tile drain discharge will depend on the capacity of the tiling and the hydraulic head at the tile outlet. Actual tile drainage has an upper limit determined by pipe capacity, regardless of the extent to which infiltration has filled the lower soil zone capacity. As a result, HSPF simulations that obtain a good general representation of tile drainage discharge under normal conditions are likely to over-estimate interflow discharge during large precipitation events with dry antecedent conditions. This effect is indeed evident in HSPF simulations of the Minnesota River basin.

Available data do not allow direct determination of the interflow inflow parameter to represent tile drainage; instead, this parameter must be determined through calibration. The MnRAP studies of selected sub-watersheds suggest that there is generally a decreasing trend of intensity of tile drain density from the southeast (LeSueur, Blue Earth) to the northwest portions (Yellow Medicine, Chippewa) of the Minnesota River basin (Figure 4-2). Basins to the southeast have tile densities on the order of 0.03 km/ac, while those to the northwest have tile densities less than 0.01 km/ac. While the sample of basins in MnRAP is small, this trend coincides with changes in soil type and other anecdotal information. Accordingly, the interflow inflow parameter is expected to be higher in the southeast, and lower in the northwest portions of the Minnesota River basin.

The HSPF representation of interflow differs sufficiently from the actual physics of tile drainage that the INTFW parameter cannot be estimated from first principles or soil properties. Instead, it must be determined through calibration. As noted above, the interflow portion of the hydrograph depends on INTFW, INFILT, and LZSN simultaneously. The latter two parameters are determined from soil properties (see below), leaving INTFW as a calibration parameter.

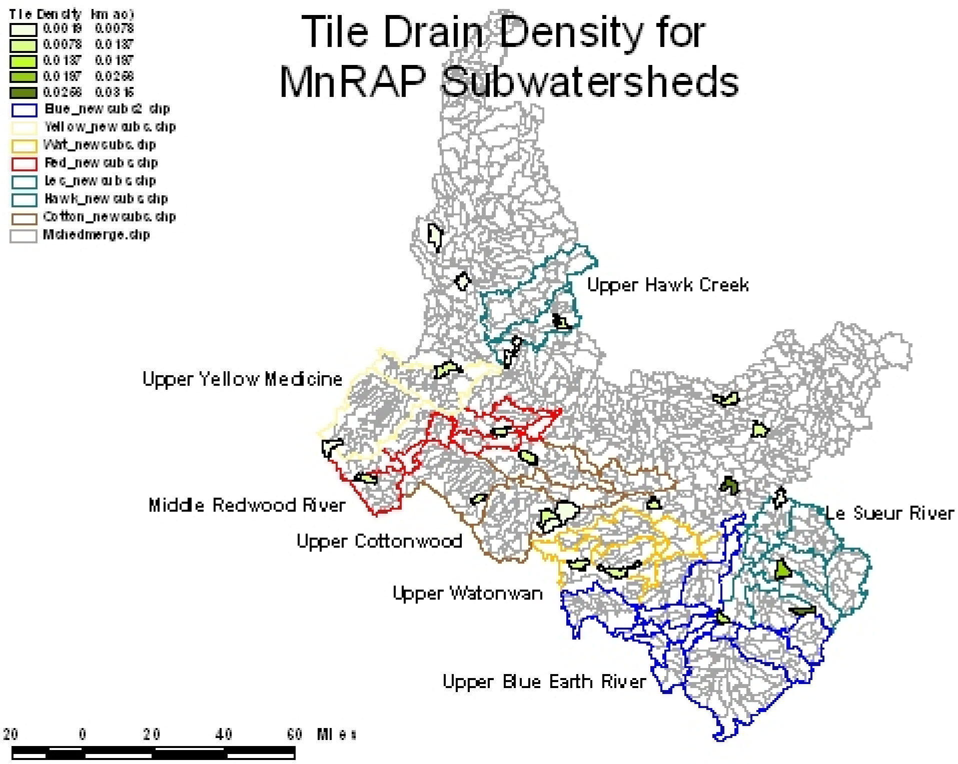

Figure 4-2. Tile Drain Density Spatial Distribution in the Minnesota River Basin

USGS HSPF modeling of the Heron Lake basin (just south of the Minnesota River drainage and heavily tiled) used a value of INTFW of 3.4 to obtain a good fit to flows at multiple gages (Jones and Winterstein, 2000). Calibration efforts in the Minnesota River basin reveal that this value of INTFW is also appropriate for the areas near the Heron Lake basin, such as Watonwan River. We found, however, that slightly lower interflow inflow (INTFW=3) provided a better fit in the months of January through March, when ice and snow may impede inflow to tile drains. In the LeSueur River basin, which has, on average, the highest clay content of soils and low infiltration rates, a higher value of INTFW is appropriate, with a value of 4 during the spring and summer providing a better fit. Lower values of INTFW (down to a low of about 1.0 in the Chippewa River basin) are specified for the basins with less dense tile drainage.

Community and Individual Septic Systems

Community and direct-discharge ISTS sources should be included within the watershed model. Unsewered communities are specified explicitly to the model, as their locations and population served are known to MPCA. The large number of direct-discharge ISTS must be approached on an aggregate basis. The approach used consists of a statistical estimation of the number of direct-discharge ISTS by model reach, combined with an estimate of pollutant concentrations and loads in typical septic system effluent.

It is important to note that individual onsite sewage treatment systems that do not have direct-discharge connections (e.g., septic systems with leach fields) are not explicitly included within the model. Rather, these systems contribute to observed background concentrations in ground water discharge.

Unsewered communities are specified as point sources in the model. Direct-to-tile ISTS are represented on an aggregate basis by model reach/sub-watershed. Both ISTS and unsewered communities are represented on a population basis. The general steps for including these sources are conceptually simple:

1. Calculate the population discharging to a given reach.
2. Associate the loading rate per capita.
3. Add this load to the appropriate reach.

For ISTS, a household size of 2.1 was assumed (email from Jim Klang, MPCA, to Sabrina Cook, 2/14/01). Onsite wastewater disposal with drain fields (no direct discharge to tile lines) is not explicitly included in the model, but forms part of the ground water loading background.

Loading rates per capita should be derived based on data after initial settling in a septic tank. Table 4-1 gives estimated loading rates and pass through percentages.

Table 4-1. Loading Representation of ISTS and Unsewered Communities

Parameter	Loading rate or concentration	Assumed pass-through fraction	Notes
Water	50 gal/person/day	1.0	Value for rural areas supplied by Jim Klang, MPCA
Total-N	53 mg/L	0.72	Mid-point of range supplied by Jim Klang, MPCA; see also USEPA, 1980 and 1993 for pass-through.
Ammonia-N	50 % of Total N		Assumed that much of the organic N will be broken down in the septic tank, while a portion of the ammonia N will be oxidized in the drainage system before reaching a stream.
Nitrate-N	50 % of Total N		
Phosphate	20 mg/L	0.43	Midpoint of range supplied by Jim Klang, MPCA, consistent with Caraco et al., 1998; USEPA, 1980; USEPA, 1993
CBODu	175 mg/L BOD-5, with ratio CBODu/ BOD5 of 1.2.	1.0	Concentration from Jim Klang, MPCA. Ratio for untreated waste: Thomann and Mueller, 1987.
Fecal Coliform	1.0E6 #/100 ml	0.0135	Concentration from Jim Klang, MPCA. Resulting loads consistent with Metcalf & Eddy, 1991; USEPA, 1999. Pass through based on approximately 4-day time of travel to major reaches under low flow (determined in calibration).

4.4 LINKAGE ANALYSIS MODEL CALIBRATION

Calibration of the Minnesota River HSPF models is described in detail in Tetra Tech (2002). In that work, models were developed for nine different sub-basins, which enabled a cross-sectional approach to calibration. This approach seeks one family of parameters that works across multiple watersheds. A cross-sectional approach provides greater robustness and believability to the calibration, as it demonstrates that the apparent fit is not simply a curve-fitting artifact. The calibration process for HSPF is sequential, beginning with the calibration of flow, followed by calibration of water quality. Sediment and dissolved pollutant loading depend on the representation of flow, while sorbed pollutant loading depends on the simulation of sediment.

4.4.1 HSPF Hydrologic Calibration

Flow was calibrated to USGS gaging data for 1986 through 1992. This period includes a wide range of hydrologic conditions, encompassing both very dry and wet years. The calibration was accomplished within the HSPF Expert System, HSPEXP (Lumb et al., 1994). The calibration is cross-sectional in nature, in that the same parameters are used for a given land use in all watersheds, except where justified by known differences in soils and topography.

The gage selected for calibration is the USGS gage on the Chippewa River near Milan. This gage is located upstream of the Watson Sag diversion, and thus is not affected by the unmonitored diversions that reduce flow in the lower Chippewa near Montevideo.

The primary external forcing of the model is provided by time series of precipitation, temperature, and potential evapotranspiration. As the model is run on an hourly time step, hourly data are required for these meteorological variables. There are not, however, any hourly weather stations in or near the Chippewa basin. For precipitation, daily totals were obtained from eight Cooperative Summary of the Day reporting stations (Figure 4-3). These daily totals were then disaggregated into surrogate hourly time series using the WDMUtil companion program to HSPF and assigned to sub-basins as weighted totals based on a Thiessen polygon analysis. Daily maximum and minimum temperature data were available from Benson and Glenwood, while potential evapotranspiration was estimated using the Penman method and employing wind and solar radiation data from Sioux Falls, South Dakota, the nearest first-order weather station.

A number of the parameters used in HSPF reflect properties of the soils. These parameters can be derived from, or related to, reported soil characteristics. This approach has two important advantages: First, it reduces the number of unconstrained or “free” parameters that must be addressed in calibration. Second, it helps to ensure that variability in parameter values between basins is systematic and based on physical evidence.

For the simulation of hydrology, parameters for infiltration rate (INFILT; in./hr.) and lower zone soil storage (LZSN; in.) can be related to soil parameters. Infiltration estimates in soils coverages are based on ring infiltrometers under dry conditions, and do not reflect actual infiltration rates during storm events, when surface sealing may occur. Based on calibration, effective infiltration rates during events appear to be on the order of 1/40 to 1/50 of reported infiltrometer infiltration rates vertically averaged over the soil profile. The 1/50 ratio appears to be applicable to basins in which the clay fraction of surface soils is on the order of 31 percent (e.g., LeSueur), while the 1/40 ratio is applicable at a clay fraction of about 25 percent. As the clay fraction in Chippewa soils averages about 25 percent, the 1/40 ratio was used, yielding an estimated value of INFILT of 0.075 in/hr.

Soil moisture storages are estimated from data on available water capacity, multiplied times an assumed rooting depth. The final estimate obtained in the Chippewa calibration was 4.3 in., consistent with a mean rooting depth of 82 cm. A complete input file for the Chippewa River model is included as an appendix to this report.

Data limitations (particularly the lack of hourly precipitation data at most stations) suggest that the model will often not predict the shape of individual storm hydrographs. Therefore, the emphasis in calibration is on achieving long-term consistency between model simulations and observed gage data, rather than evaluating fit on a point-by-point basis. To do this, the HSPEXP system (Lumb et al., 1994) was employed, and the default calibration targets provided in HSPEXP became the objectives for the Minnesota River hydrologic calibration. These targets:

- Predict total flow volume over the period of simulation within 10 percent.
- Predict the volume of the 10 percent highest flows within 10 percent.
- Predict the volume of the 50 percent lowest flows within 15 percent.
- Obtain a seasonal volume error (winter versus summer components of flow) of less than 10 percent.

Simulated and observed monthly flows for the Chippewa River near Milan are shown in Figure 4-5. While the model reproduces the general trend, there are obvious discrepancies in individual months. These discrepancies are largely due to a combination of sparse/unrepresentative meteorological data and difficulties in simulation of spring snow melt.

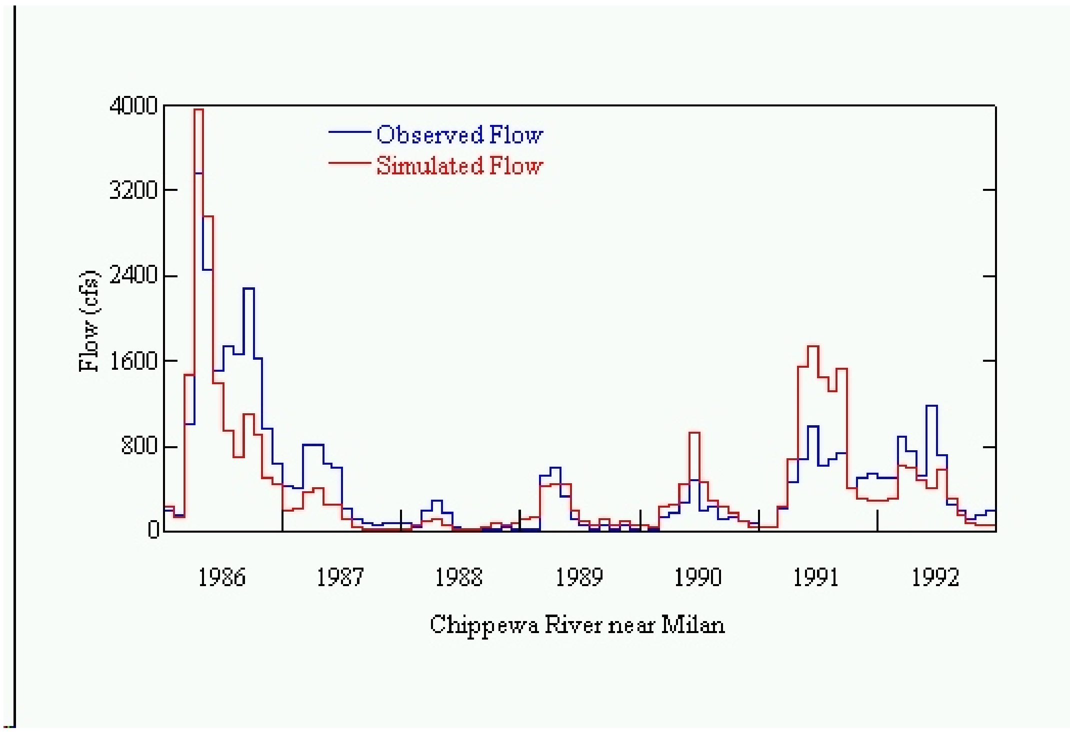

Figure 4-3. Observed and Simulated Flows at the USGS Gage, Chippewa River near Milan, MN

Calibration statistics for the Chippewa River basin (calibrated at Milan above the Watson Sag flow diversion) are shown in Table 4-2. The shaded cell for seasonal volume error connotes that the statistic did not meet the calibration objectives.

Of these targets, the most difficult to meet for the Minnesota River basin is the seasonal volume error, specifically the winter (December-February) component. This is because a significant portion of runoff in the basin occurs with snowmelt, and small discrepancies in the representation of temperature can move spring flow peaks before or after the end-of-February cutoff specified in HSPEXP for calculation of the winter portion of seasonal volume error.

Table 4-2. HSPF Model Hydrologic Calibration Statistics

Statistic	Result	Target
Total Flow Volume	-2 %	10 %
50% Lowest Flows	2.2 %	10%
10% Highest Flows	7.2 %	15%
Seasonal Volume Error	29.5 %	10%

For extension of the model downstream to Montevideo, the approximate behavior of the Watson Sag Diversion is simulated by routing one half of the water present near Milan at low flow into the downstream reach.

4.4.2 HSPF Water Quality Calibration

As noted above, the water quality calibration is based on a methodology of finding a common set of parameters that may be applied across all major watersheds, with variation among watersheds based on external evidence. The data available for calibration consist primarily of point-in-time grab samples collected at or near the mouths of major watersheds. Observations at these points represent the combined net impact of all upstream loads, as well as interactions in and between the water column and sediment. Few samples are available from lower order headwater streams. Continuous or event-mean observations are not available, so it is not possible to clearly distinguish whether a lack of fit to individual observations is due to bias in the model parameters, mistiming of temporal events, or random variability in sampling results.

The complete water quality calibration of the Minnesota River basin models is described in Tetra Tech (2002); only the ammonia component is summarized here. Ammonia nitrogen is a minor constituent of the total nitrogen loading for most land uses. It is included as a separate constituent in the model primarily because of the potential for elevated ammonia loading from manure application areas.

In addition to low rates of loading from the land surface, most of the ammonia observed in monitored watersheds of the Minnesota River basin is attributable to wastewater treatment plants and ISTS. Internal generation from the breakdown of organic matter in the stream can also be important. Because of these issue, it is not reasonable to attempt to calibrate nonpoint ammonia loads to instream observations.

Given these considerations, the literature-based parameter values for ammonia developed for the original Minnesota River HSPF models were accepted for use in the revised model. Surface accumulation rates for ammonia are expected to be very low, because of the rapid oxidation of ammonia on the surface, and are essentially nominal values, except for manured land.

Subsurface concentrations of ammonia are more important to model simulations than surface loading. Here, the values developed in the original HSPF models for LeSueur, Redwood, and Watonwan major watersheds appeared to be reasonable, and were applied to the other basins as well, including the Chippewa. It should be noted that subsurface concentrations specified to the model for a reactive parameter like ammonia are *not* equivalent to concentrations observed in ground water. This is because the model simulates only higher order streams, and significant nitrification of ammonia is expected to occur in the lower order feeder streams that are not included in the model. Therefore, the values that should be specified to the model are not actual ground water concentrations but rather the *exerted* concentration present when flow reaches a simulated reach. In any case, as noted above, model results are not very sensitive to the nonpoint component of ammonia loading.

The only extensive water quality data available for calibration are those collected by MPCA and USGS in the Chippewa River below the Montevideo discharge. Application of the model against these data shows a fairly weak fit (Figure 4-4), in which the model appears to reproduce some of the general trend in ammonia concentrations, but misses a number of higher-concentration observations. The fit appears to be better for the period after 1990. The median values from pairs of observed and simulated values are 0.13 and 0.05, for observed and simulated values respectively, over the entire calibration period. For 1990-92 the medians are much closer (0.08 versus 0.06). Statistical tests conclude that the observed and simulated values are not equal. At the same time, the ammonia parameters appear to yield reasonable results for other Minnesota River subwatersheds (Tetra Tech, 2002).

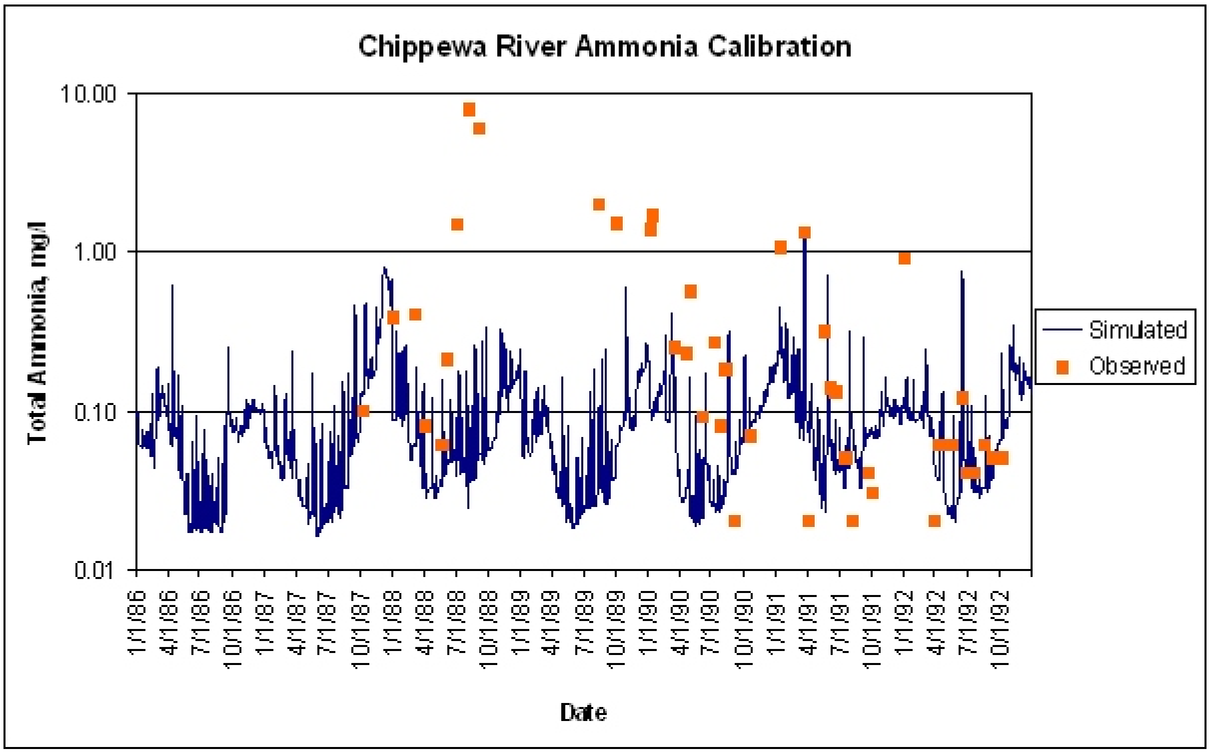

Figure 4-4. Results of HSPF Model Calibration for Ammonia, Chippewa River below Montevideo

The match between observed and predicted ammonia concentrations at the lower Chippewa station is thus weak, and does not prove the adequacy of the HSPF model application. However, the only monitoring station in the watershed for which adequate calibration data are available is the station located just downstream of the Montevideo discharge. Thus, any inaccuracies in the representation of the Montevideo discharge are likely to have a large impact on model predictions. It appears that the uncertainty in specifying this discharge is the major source of uncertainty in model predictions; therefore, data do not exist with which to judge the model performance in representing the net effects of upstream watershed loads.

For the calibration period (1986-92), the data relative to the Montevideo discharge are extremely limited. According to MPCA, ammonia concentrations in the Montevideo effluent were not monitored during the calibration period. Instead, ammonia loads from the plant are simulated based on a constant typical ammonia concentration for secondary wastewater treatment plants in Minnesota, provided by MPCA, applied to reported monthly average discharges. No seasonal variations were imposed, although it is well known that nitrification varies strongly with temperature. In addition, all of Minnesota's DMR data prior to 1989 were lost during a storage warehouse fire. Thus, for the period prior to 1989 flow also is unknown and is represented by replicating the 1989 monthly pattern.

Given these data shortcomings, it is not surprising that the model does a poor job of replicating individual concentration measurements below Montevideo, particularly during low flow periods when the plant discharge may dominate the ammonia balance. High concentrations seen in monitoring data, but not predicted by the model, are likely due to higher-than-estimated ammonia loads from the Montevideo WWTP. The ammonia parameters used in the watershed model generally perform well across other major watersheds in the Minnesota River basin (Tetra Tech, 2002). On this basis, the model is deemed to provide a reasonable (but uncalibrated) representation of the upstream concentration likely to be present at Montevideo. However, the lack of a satisfactory calibration result suggests the need to employ a second linkage tool in the TMDL allocation process, which is provided by the spreadsheet dilution model.

5 TMDL Allocation Analysis

A TMDL is the total amount of a pollutant that can be assimilated by the receiving water while still achieving water quality standards. TMDLs can be expressed in terms of mass per time or by other appropriate measures. TMDLs are composed of the sum of individual wasteload allocations (WLAs) for point sources, load allocations (LAs) for nonpoint sources, and natural background levels. In addition, the TMDL must include a margin of safety (MOS), either implicitly or explicitly, that accounts for the uncertainty in the relationship between pollutant loads and the quality of the receiving waterbody. Conceptually, this definition is denoted by the equation:

$$\text{TMDL} = \sum \text{WLAs} + \sum \text{LAs} + \text{MOS}$$

To develop the un-ionized ammonia TMDL for the listed segment of the Chippewa River, the following approach was taken:

- Simulate baseline conditions
- Assess source loading alternatives
- Determine the TMDL and source allocations

The primary wasteload allocation component of the TMDL is presented in terms of mass per time in this report. The load allocation to upstream background sources is reported first on a concentration basis, which is converted to a flow-dependent mass allocation.

5.1 TMDLs and Source Allocations

A spreadsheet model was developed to assess allowable loads at seasonal, steady-state, dry weather, and wet weather flow conditions. As discussed above, current Minnesota guidance calls for the use of a seasonal 30-day in-stream low flow period to assess allowable discharges at design flow conditions. However, because the Montevideo permit was originally developed using 7-day low flow conditions, anti-backsliding provisions of Minnesota's regulations prohibit the use of a less stringent design condition to develop new effluent limits. As a result, the WLA for this TMDL is based upon the currently permitted seasonal effluent concentrations.

The MPCA calculates mass limits using the permitted concentration limits coupled with the average wet weather (AWW) design flow for the WWTP, which in the case of Montevideo was 2.47 mgd. This wet weather hydraulic design accommodates the collection system infiltration and inflow that is expected under wet weather conditions. When the treatment plant is experiencing maximum wet weather flow rates, it is highly unlikely that the river will be at a low flow, drought condition. Therefore, the critical load discharged by the WWTP under wet weather conditions should be coupled with a river condition somewhat greater than the dry weather low flow condition. Establishment of a representative wet weather design flow for the river can be accomplished by using the dry weather low flow (the ${}_{30}Q_{10}$) multiplied by a factor representing the ratio of WWTF wet weather to dry weather design flows; i.e., the ratio AWW/ADW. This approach is consistent with past recommended procedures (Winslow, MPCA 1985).

Applying this approach, the following relationship is derived with example calculations for summer conditions:

1. Dry Weather TMDL

TMDL = WLA + LA + MOS, or

(Qmix)(Cstd)K = (Qadw)(Ce)K + (Qr)(Cr)K + MOS

Where:

TMDL = total allowable ammonia load, kg/day
Qr = river design flow at 30Q10, cfs
Qadw = effluent design flow at Average Dry Weather (ADW), cfs
Qmix = Qr + Qadw
Cstd = water quality standard. The total ammonia concentration at ambient pH and temperature that meets the 0.04 mg/L un-ionized ammonia standard, mg/L
Cr = headwater ammonia concentration, mg/L
Ce = current permit limit for ammonia, mg/L
K = units conversion factor (2.4466)
MOS = explicit margin of safety and growth, kg/day

Example: solve for MOS (summer season)

MOS = TMDL – WLA – LA, or
MOS = (Qmix)(Cstd)K – (Qadw)(Ce)K – (Qr)(Cr)K

Where:
Qr = summer 30Q10 = 9.20 cfs
Qadw = 0.74 mgd = 1.14 cfs
Qmix = 10.34 cfs
Cstd = 0.53 mg/L total ammonia
Ce = 2.5 mg/L total ammonia effluent limit
Cr = 0.14 mg/L headwater total ammonia
K = 2.4466

Then:

MOS = (10.34 cfs)(0.53 mg/L)(2.4466) – (1.14 cfs)(2.5 mg/L)(2.4466) – (9.20 cfs)(0.14mg/L)(2.4466)
MOS = 13.41 kg/day – 6.97 kg/day – 3.15 kg/day
MOS = 3.29 kg/day (or 24.5% of the total TMDL)

2. Wet Weather TMDL

TMDL = WLA + LA + MOS, or

[(Qr)(AWW/ADW) + Qaww] (Cstd)K = (Qaww)(Ce)K + (Qr)(AWW/ADW)(Cr)K + MOS

Where:

AWW/ADW = ratio of average wet weather design to average dry weather design of WWTF (2.47 mgd / 0.74 mgd = 3.34)

Qaww = WWTF average wet weather design flow. Note: the term (Qaww)(Ce)K represents the currently permitted mass load.

Example: solve for MOS (summer season)

MOS = TMDL – WLA – LA, or
MOS = [(Qr)(AWW/ADW) + Qaww] (Cstd)K – (Qaww)(Ce)K – (Qr)(AWW/ADW)(Cr)K

Where:
Qr = summer 30Q10 = 9.20 cfs
Qaww = 2.47 mgd = 3.82 cfs
AWW/ADW = 3.34
Cstd = 0.53 mg/L total ammonia
Ce = 2.5 mg/L total ammonia effluent limit
Cr = 0.14 mg/L headwater total ammonia
K = 2.4466

Then:

MOS = [(9.20 cfs)(3.34) + 3.82 cfs] (0.53 mg/L)(2.4466) – (3.82 cfs)(2.5 mg/L)(2.4466) – (9.20cfs)(3.34)(0.14mg/L)(2.4466)
MOS = 44.8 kg/day – 23.3 kg/day – 10.5 kg/day
MOS = 11.0 kg/day (or 24.5% of the total TMDL)

In the following sections, this relationship is applied under seasonal design conditions. As depicted in Table 5-1, Qr, Qadw, and Qaww values are specified based on field gage data and permitted design conditions. Table 5-2 presents the permitted ammonia concentrations (Ce), upgradient ammonia concentrations (Cr), and the allowable ammonia concentrations below Montevideo (Cstd) that meet the water quality standard for un-ionized ammonia. The total ammonia concentration that meets the water quality standard for un-ionized ammonia was developed from ambient pH and temperature conditions as reported in Section 2 (see Table 2-1).

Table 5-1 Design Condition Flow Values

Season	Qr (cfs)	Qadw (cfs)	Qaww (cfs)
Spring (Apr - May)	76.5	1.14	3.82
Summer (Jun - Sep)	9.2	1.14	3.82
Fall (Oct - Nov)	11.1	1.14	3.82
Winter (Dec - Mar)	2.1	1.14	3.82

Table 5-2 Design Condition Ammonia Concentrations

Season	Ce (mg/L)	Cr (mg/L)	Cstd (mg/L)
Spring (Apr - May)	31.0	0.16	1.04
Summer (Jun - Sep)	2.5	0.14	0.53
Fall (Oct - Nov)	8.6	0.14	1.35
Winter (Dec - Mar)	12.9	0.47	7.73

Seasonal 30-day low flow data were estimated based on surrogate flow series at Montevideo for 1938-2000 based on the USGS gaging on the Chippewa River near Milan (05304500). According to USGS, this station has a drainage area of 1,880 square miles. The flow data were adjusted to account for additional drainage area below this gage, including Dry Weather Creek and Spring Creek, for a total area of approximately 2,083 square miles. Flow from the ungaged areas was calculated by drainage area ratio to the Milan gage. This estimate is based on the Minnesota DNR Minor Watershed File - 1995 version. Flow at the diversion may thus be estimated as 1.08989 times the flow at the gage. U.S. Army Corps of Engineers operations at Watson Sag divert approximately 50 percent of the low flow during non-icing conditions, and 10 percent of the low flow during icing conditions. Flow in the Chippewa past the diversion is therefore estimated as 0.5 times flow above the diversion, although these proportions may vary during low flow conditions.

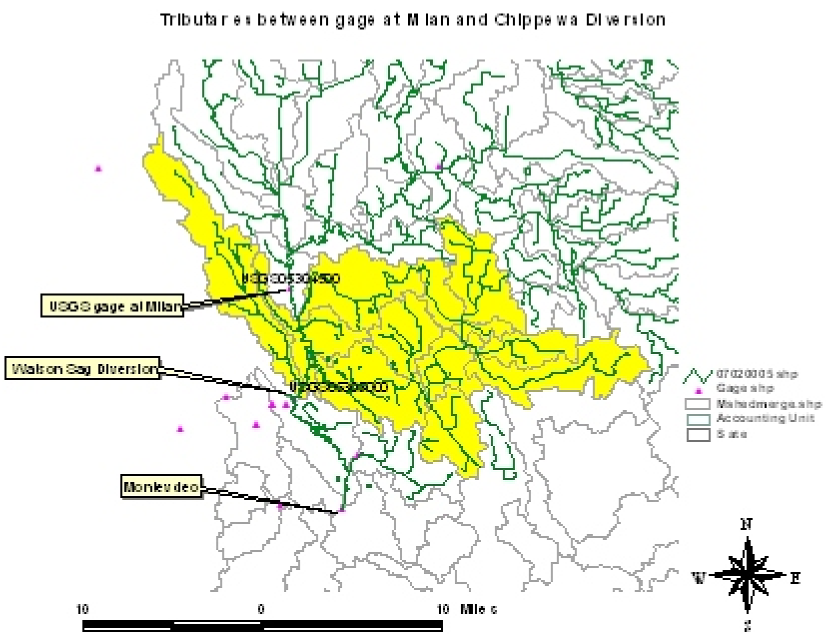

Figure 5-1. Minor Watersheds in the Chippewa River Basin between Milan and Watson Sag

5.2 Wasteload Allocations (WLAs)

To achieve water quality standards it is necessary to manage the WLA for the major contributor of ammonia in this reach, the Montevideo WWTP. This wasteload allocation was calculated using the spreadsheet dilution model with the following assumptions:

- The analysis is conducted over gaged flows from 1938 through 1991.
- The un-ionized ammonia concentration is calculated using the seasonally median water temperature and pH.
- The effluent limits are specified for four seasons (rather than monthly), consistent with existing MPCA practice.
- The most restrictive effluent limit will apply to the June-September period. Therefore, the rare

allowed exceedances (one in three years) are likely to be concentrated in this time period to avoid imposing an unattainably low effluent limit.

Based on these assumptions, the spreadsheet model derives seasonal dry weather and wet weather WLA values as presented in Table 5-3.

Table 5-3. Seasonal Montevideo WWTP WLAs

Season	Ce (mg/L)	Dry Weather WLA (kg/day)	Wet Weather WLA (kg/day
Spring (Apr - Mar)	31.0	86.5	289.7
Summer (Jun - Sep)	2.5	7.0	23.4
Fall (Oct - Nov)	8.6	24.0	80.4
Winter (Dec - Mar)	12.9	36.0	120.6

No revisions to existing wasteload allocations for other point sources are proposed as part of this TMDL. In terms of the ammonia load exerted at Montevideo, the influence of these sources is small to insignificant, and it is appropriate to treat them as part of the general background for the impaired reach. However, any significant expansions of upstream point sources would require additional analysis to ensure that water quality standards in the listed reach continue to be met.

5.3 Load Allocations (LAs)

Load allocations are assigned to nonpoint sources. Based on the HSPF modeling analysis in Section 4, the simulated background loads of ammonia (that is, all sources other than the Montevideo WWTP) constitute between 7 and 12 percent of the predicted assimilative capacity of the river under low flow conditions. **As a result, no single source other than the Montevideo WWTP constitutes a significant fraction of the load expressed at Montevideo.** A gross load allocation to all upstream sources is implied in the TMDL analysis by the assignment of constant background concentrations (by season) in the dilution analysis that serves as the basis for the wasteload allocation. These upstream concentrations, as shown in Table 5-2, are based on MPCA sampling data. Table 5-4 shows daily load based on seasonal ammonia concentrations and 30-day dry and wet weather low flows.

Table 5-4. Seasonal Ammonia Load Allocations

Season	**30-day low flow (cfs)**	**Dry Weather LA (kg/day)**	**Representative Wet Weather Flow (cfs)**	**Wet Weather LA (kg/day)**
Spring (Apr - May)	76.5	30.0	255.5	100.0
Summer (Jun - Sep)	9.2	3.2	30.7	10.5
Fall (Oct - Nov)	11.1	3.8	37.1	12.7
Winter (Dec - Mar)	2.1	2.4	7.0	8.1

These load allocation values were compared to the ammonia concentrations developed using the HSPF linkage analysis model (see Table 5-5). The model used the calibration period meteorology (1986-1992) and year 2000 land use and point source discharges. This meteorology represents the range of flow conditions, as it includes both very dry drought years (1987-1988) and wet years (1991-1992). The simulation predicts ammonia concentrations and water temperature; un-ionized ammonia concentrations were then estimated (using the equation specified in Minnesota regulations) under the assumption that pH = 8.4 (90th percentile of observed pH). The values were developed using four-day low flow conditions.

Table 5-5. Modeled Low Flow Load Allocations for Exerted Ammonia Load in the Chippewa River at Montevideo

			Modeled Predictions		
Season	**Month**	**Basin Monitoring (mg/L)**	**HSPF (mg/L)**	**4-day low flow (cfs)**	**Modeled LA (kg/day)**
Spring	April	0.16	0.07	25.38	4.19
	May		0.04	41.45	4.21
Summer	June	0.14	0.04	8.46	0.75
	July		0.03	3.38	0.24
	August		0.04	1.18	0.11
	September		0.06	1.40	0.20
Fall	October	0.14	0.09	1.60	0.36
	November		0.15	2.75	1.03
Winter	December	0.47	0.23	3.52	2.02
	January		0.16	2.40	0.92
	February		0.13	2.40	0.75
	March		0.12	2.68	0.81

As demonstrated, there is general agreement between the field measured ammonia concentrations above the Montevideo WWTP and those predicted using the HSPF model. Neither the field data nor the model predictions result in ammonia concentrations above Montevideo that would result in water quality standard exceedances. While no reduction in load allocations is proposed at this time, any major changes in land use that could lead to significantly increased ammonia loading would require additional analyses to ensure that water quality standards would continue to be met.

5.4 Margin of Safety

The Margin of Safety (MOS) is a required component of the TMDL. There are two acceptable methods for incorporating a Margin of Safety into a TMDL: 1) by implicitly incorporating the MOS through use of conservative modeling assumptions in the development of allocations, or 2) by explicitly specifying a portion of the TMDL as the MOS based on an analysis of uncertainty in modeling results. This TMDL applies both approaches. In addition, note that the margin of safety reserve is on the "exerted" ammonia load present in the Chippewa River at Montevideo, and translates to greater amounts at upstream sources due to loss of ammonia during transit.

A margin of safety is explicitly calculated using the approach described under Section 5.1 by subtracting the WLA and LA from the TMDL: MOS = TMDL-WLA-LA. The values are presented in Table 5-6 for dry weather conditions and Table 5-7 for wet weather conditions.

Table 5-6. Dry Weather Seasonal TMDL and MOS

Season	TMDL (kg/day)	WLA (kg/day)	LA (kg/day)	MOS(kg/day)	% MOS
Spring	197.5	86.5	30.0	81.1	41.1%
Summer	13.4	7.0	3.2	3.3	24.5%
Fall	40.4	24.0	3.8	12.6	31.3%
Winter	61.3	36.0	2.4	22.9	37.3%

Table 5-7. Wet Weather Seasonal TMDL and MOS

Season	TMDL (kg/day)	WLA (kg/day)	LA (kg/day)	MOS(kg/day)	% MOS
Spring	659.9	289.7	100.0	270.1	40.9%
Summer	44.8	23.4	10.5	10.9	24.4%
Fall	135.1	80.4	12.7	42.0	31.1%
Winter	204.9	120.6	8.1	76.3	37.2%

The implicit margin of safety for the Chippewa River un-ionized ammonia TMDL is accounted for through the use of conservative low flow estimates. In addition, field measurements to represent actual seasonal variation in pH, temperature, and in-stream ammonia levels upgradient of the Montevideo discharge were used.

A portion of the flow in the Chippewa River is diverted into the Lac qui Parle Reservoir on the Minnesota River to provide flood control. Under low flow conditions, the diversion is reduced to account for critical in-stream needs.[1] During average flow conditions, approximately 50 percent of the flow in the river is diverted to the reservoir. However, during low flow conditions, only 10 to 20 percent of the flow is diverted.[2] To provide a more conservative estimate of stream flow during low flow conditions, the 50 percent diversion was used in the flow estimates with a small adjustment to account for the larger watershed drainage area between the gage and the Montevideo discharge. The larger diversion results in lower flow in the stream and reduced dilution capacity resulting in lower allowable effluent concentrations and loads. The low flow discharges were calculated as follows:

1 Kenton Spading, U.S. Army Corps of Engineers ,via telefax (need date)

2 Ibid.

Table 5-8. Chippewa River 30Q10 Discharge at Montevideo WWTP under Low Flow Conditions (cfs)

Season	Gaged Discharge above Watson Diversion	50 Percent Diversion to Reservoir	Adjustment for 34 mi^2 Additional Drainage Area	Estimated Discharge
Climatic Year (Apr - Mar)	4.2	2.1	0.07	2.2
Spring (Apr - May	148	74	2.4	76.4
Summer (Jun - Sep)	17.7	8.9	0.3	9.2
Fall (Oct - Nov)	21.5	10.7	0.4	11.1
Winter (Dec - Mar)	4.0	2.0	0.07	2.1

For example, if a 20 percent diversion were used during the low flow summer period, the estimate stream flow would increase to 9.94 cfs and the resulting allowable ammonia load would increase.

Furthermore, the current wasteload allocation in the Montevideo WWTP NPDES permit was calculated at 7Q10 flow conditions, as specified in the Minnesota regulations in effect at that time. In a subsequent revision to the Minnesota rules, the water quality-based effluent limit that serves as the wasteload allocation for the Montevideo plant would now be calculated using 30Q10 flow conditions (see Methodology for the Development of Water Quality-based Effluent Limitations for Toxic Substances in Chapter 7050, Revised July 1993, p. 35) However, because of Minnesota's anti-backsliding provisions, the less stringent limit should not be applied. As a result, setting the WLA at the current permitted effluent limits accounts for anti-backsliding, builds in an additional reduction in allowable ammonia loads.

5.5 Seasonal Variation

A TMDL must consider seasonal variation in the derivation of allocations. This TMDL addresses seasonal variation by assessing compliance with the water quality standard under seasonal flow, ammonia load, and in-stream pH and temperature conditions. Thus, the analysis explicitly gives full consideration to seasonal variation, and the proposed WLAs for Montevideo are developed as seasonal allocations that ensure that the water quality target will be met throughout the year.

5.6 Future Growth

This TMDL does not include a specific allocation for future growth. Facility expansions or new facilities will be addressed through the NPDES permitting process. Such future growth will also be addressed by revisiting the TMDL allocations.

Reducing uncertainty in the prediction of low flow ammonia concentrations is one key to defining more precise wasteload allocations, reducing the MOS, and potentially gaining an additional reserve for future growth. Key data uncertainties that could be resolved are:

Monitoring of Upstream Point Source Loads

At present, of the permitted point sources in the watershed, only Montevideo and Benson have permit limits and monitor for ammonia. The contribution of other point sources to ammonia concentrations in the lower Chippewa is evidently small, but not well defined. Permitting of additional discharge capacity in the watershed may require the specification of ammonia effluent limits on existing dischargers other than Montevideo and Benson. In addition, ambient ammonia monitoring could be made a condition of the permits.

Flow Gaging

A key source of uncertainty for the TMDL is the absence of flow gaging in the lower Chippewa. Neither flow at Montevideo nor the portion of the upper Chippewa gaged flow that is diverted through Watson Sag is regularly and reliably monitored. Without such gaging, it is difficult to precisely assess the dilution capacity available at Montevideo. If such additional data are obtained it may be advisable to refine the TMDL with further modeling.

Finally, a new facility could be permitted in the watershed, without contravening the TMDL, provided that effluent limitations are based on the achievement of water quality standards. Because ammonia is not a conservative pollutant in the natural environment, it does not accumulate but is biochemically transformed and/or taken up by the aquatic biota.

Compliance Monitoring

At present, MPCA does not conduct regular ambient monitoring below Montevideo WWTP. The Chippewa River ambient monitoring station at River Mile 0.5 should be reactivated and sampled on a regular basis to ensure that the TMDL results in the attainment of water quality standards.

6 Public Participation

Dialog between the MPCA, the city of Montevideo, and the Chippewa River Watershed Project is ongoing to collect the necessary water quality data to demonstrate that the Chippewa River is meeting the ammonia water quality standard. Citizens attending monthly Chippewa River Watershed Project meetings are updated on the progress of the TMDL throughout the year. A conference call with MPCA, City and Montevideo WWTP staff took place on June 23, 2004 to share the results of the draft TMDL report and to discuss the monitoring needed. A public meeting was held 7-9 p.m. Thursday, July 22, 2004 in the Montevideo Community Center. The draft TMDL report is available to the public via the MPCA web site at http://www.pca.state.mn.us/water/tmdl.html. A public notice of the draft TMDL was posted in the State Register on July 6, 2004 and public comments could be sent to MPCA until August 6, 2004. No public comments were received during the public notice period.

7 Monitoring Plan

An information protocol has been developed by MPCA staff and shared with the city of Montevideo and the Chippewa River Watershed Project that describes the monitoring needed to determine if the river is meeting water quality standards for ammonia under critical conditions. Critical conditions for this impairment occur when the river is at low flow. Water quality samples will be collected by Montevideo Wastewater Treatment Plant (WWTP) staff and the Chippewa River Watershed Project staff, and some additional sampling will be done by MPCA staff. The sampling will be done this summer and in the following years if needed.

Monitoring under critical conditions is to be done weekly, at flows less than 250 cubic feet per second (cfs) between July 1 and October 31. While violations have occurred after the WWTP upgrade in 1994, the MPCA believes these are related to process control problems at the Montevideo WWTP. Data is needed to show that the water quality standard is being met. To remove this reach from the impaired waters list three years of monitoring is required with no more than one exceedance during the critical time period. Requirements for monitoring data regarding removal of water bodies from the Impaired Waters list are contained in the Guidance Manual for Assessing the Quality of Minnesota Surface Waters for the Determination of Impairment. 305(b) Report and 303(d) List. January 2004. Section X. This guidance document is available on the MPCA web site at http://www.pca.state.mn.us/publications/manuals/tmdl-guidancemanual04.pdf. The data collected will be used to assess whether the Chippewa River is meeting the un-ionized ammonia water quality standard, and if so a recommendation will be made to remove this portion of the river from the 303d (TMDL) list for ammonia impairment.

Water quality samples were collected by MPCA staff at two sites on the Chippewa River upstream and downstream of the Montevideo WWTP discharge point. The samples were collected as part of the MPCA's Milestone Monitoring Program in June and July 2004 and will be collected in August and September 2004. One of the sites (CH-0.5) is at the Highway 7 bridge in Montevideo just downstream of the WWTP outfall. The other site (CH-0.75) is 0.25 miles upstream of the outfall at the railroad bridge.

The samples were sent to the Minnesota Department of Health for analyses of total ammonia and fecal coliform among other parameters. Field parameters such as dissolved oxygen, pH, and stream temperature were measured at the sampling site. Un-ionized ammonia is calculated from total ammonia, pH and temperature values. The total ammonia nitrogen was non-detectable at both sites in the June 28, 2004 sample. The flows in the Chippewa River during the early summer of 2004 were generally high due to an unusually rainy June.

Data was also collected by MPCA staff during the summer of 2001 at both sites. There were some violations of the ammonia standard at the downstream site. These violations were attributed to an upset at the WWTP that was resolved.

This data will be used in future water quality assessments along with monitoring data to be collected by the WWTP at low flow to determine if the Chippewa River is currently meeting water quality standards for ammonia as a result of the upgrades at the WWTP. Once it is determined by monitoring data collected under critical conditions that the river is meeting un-ionized ammonia standards, the reach can be delisted from Minnesota’s 303(d) Impaired Waters List.

8 Implementation

Implementation of the Chippewa River Ammonia TMDL began in 1994 when the Montevideo WWTP upgraded its facility and an ammonia limit was placed in their permit. According to the findings of the draft Chippewa River Ammonia TMDL report, the Montevideo WWTP is the major contributor of ammonia to the impaired reach. The WWTP was upgraded in 1994 and seasonal ammonia limits were included in the NPDES permit. Dialog between the MPCA, the WWTP, and the Chippewa River Watershed Project is ongoing to collect the necessary water quality data to demonstrate that the Chippewa River is meeting the ammonia water quality standard. Monitoring is required to verify that the implementation is working. Upon monitoring verification the reach can be removed from the impaired waters list.

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