M.L. 2016 Project Abstract
For the Period Ending June 30, 2019

PROJECT TITLE: Tracking and Preventing Harmful Algal Blooms
PROJECT MANAGER: Daniel R. Engstrom
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FUNDING SOURCE: Environment and Natural Resources Trust Fund
LEGAL CITATION: M.L. 2016, Chp. 186, Sec. 2, Subd. 04a and M.L. 2015, Chp. 76, Sec. 2, Subd. 10

APPROPRIATION AMOUNT: $593,000
AMOUNT SPENT: $593,000
AMOUNT REMAINING: $0

Sound bite of Project Outcomes and Results
This project provides comprehensive data on the prevalence and toxicity of Harmful Algal Blooms (HABs) in Minnesota lakes today and in the past. By combining these data with updated modeling techniques, we provide a framework for predicting the timing and composition of HABs that can be tailored to individual lakes.

Overall Project Outcome and Results
Lakes are one of Minnesota's most precious resources and harmful algal blooms (HABs) threaten them both from an ecological and economic standpoint. This provides a survey of the current prevalence and toxicity of harmful algal blooms (HABs) in a subset of Minnesota lakes, determines if these blooms are increasing in frequency, and develops and refines modeling techniques that could be used to predict HABs in lakes across Minnesota. To this end, we intensively monitored five lakes in southwest and central Minnesota over 2 years for all major water chemistry parameters, algal biomass, and four cyanotoxins. In these lakes, and five additional lakes in northern Minnesota, we collected and dated sediment cores where fossil cyanobacterial pigments could be measured to track the occurrence of Cyanobacteria over the last 150 years. Finally, we chose one of the intensively monitored lake as a pilot study where we developed a watershed model (SWAT) and an in-lake hydrodynamic model (CE-QUAL-W2) to predict annual cyanobacterial bloom patterns. As a result of this project, we determined that in lakes which are already eutrophic, internal loading dynamics will play a key role in determining the size and toxicity of the bloom. Importantly, we found that even in shallow lakes (less than 16 ft maximum depth), temperature and oxygen dynamics are critical in terms of bloom timing and toxicity. Cyanobacteria pigment data from our sediment cores showed increasing HABs in some lakes over the 20th Century, but also demonstrate that conditions may have been even worse in the early to mid- 20th Century before the passage of the Clean Water Act. Our modeling results provide a framework for resource managers to predict seasonal bloom formation and persistence in lakes across the state using publicly available and widely used modeling techniques.

Project Results Use and Dissemination
Throughout this project we have provided numerous public updates on progress via the Science Museum of Minnesota's website and the St. Croix Watershed Research Station's blog, "Field Notes" including:

- "Featured Research Project" on SCWRS website: https://www.smm.org/scwrs/research/hab
- "Watching When, Where and Why Harmful Algae Happen in Minnesota Lakes" describing the beginning of the project: https://www.smm.org/scwrs/fieldnotes/watching-when-where-and-why-harmful-algae-happen-minnesota-lakes

- A primer on the "5 super powers of Cyanobacteria": https://www.smm.org/scwrs/fieldnotes/five-super-powers-cyanobacteria

We provided our expertise in major statewide news coverage of HABs over the course of this project, including:

- "Dogs as sentinels: Blue-green algae brings toxic mystery to Minn. Waters": https://www.mprnews.org/story/2016/05/24/water-toxic-algae-dogs-climate-change
- Two evening news spots on FOX21 Duluth in June of 2017 where reporters accompanied us in the field
- "Researchers search for clues to toxic algae blooms": https://www.mprnews.org/story/2017/08/17/researchers-search-for-clues-to-toxic-algae-blooms
- Participated in public call-in show on MPR for Minnesotans with questions about their lakes: https://www.mprnews.org/story/2018/04/03/water-month-state-of-minnesotas-lakes
- Participated in MPRs Climate Cast on the topic of HABs: https://www.mprnews.org/episode/2019/07/19/conditions-ripe-for-a-record-number-of-algae-blooms?fbclid=IwAR19XRU6hUPGjIt9-9d0Pj8H5pRfPLEYWSr-SpdE-g7yTlAANwQmZU7IaqQ

We co-organized two public workshops on HABs in cooperation with the University of Minnesota St. Anthony Falls Laboratory that were held in March of 2017 and 2018 which were each attended by ~70 people, including state agency personnel, local water district managers, academic researchers, private environmental consultants, and interested members of the public.

Major research results from this project were also presented at two separate meetings of the Association for the Sciences of Limnology and Oceanography in June of 2018 and February of 2019 using in-kind funding provided by the Science Museum of Minnesota. This is the largest meeting dedicated to aquatic science in the world and is held once a year. A PDF of the scientific poster presented in 2018 and the powerpoint presented in 2019 are included as a supplemental attachment to this report.

Additional Attachments include fact sheets created by SCWRS for HABs on Pearl Lake and the Madison Lake SWAT model and a report on the CE-QUAL-W2 model produced by USGS.

Environment and Natural Resources Trust Fund (ENRTF)
M.L. 2016 Work Plan Final Report

Date of Report: August 16, 2019

Final Report

Date of Work Plan Approval: June 7, 2016

Project Completion Date: June 30, 2019

PROJECT TITLE:	**Tracking and Preventing Harmful Algal Blooms**
Project Manager:	Daniel R. Engstrom
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Location: Statewide

Total ENRTF Project Budget:	**M.L. 2015, Chp. 76, Sec. 2, Subd. 10 Emerging Issues Account $**	**M.L. 2016, Chp. 186, Sec. 2, Subd. 04a Work Plan $**
ENRTF Appropriation:	**$93,000**	**$500,000**
Amount Spent:	**$93,000**	**$500,000**
Balance:	**$0**	**$0**

Legal Citation: M.L. 2016, Chp. 186, Sec. 2, Subd. 04a
M.L. 2015, Chp. 76, Sec. 2, Subd. 10

Appropriation Language :
M.L. 2016, Chp. 186, Sec. 2, Subd. 04a
$500,000 the second year is from the trust fund to the Science Museum of Minnesota for the St. Croix Watershed Research Station to identify species composition and timing of harmful algal blooms, understand the causes of bloom development in individual lakes, and determine how nutrients and climate interact to increase harmful algae outbreaks. This work must be done in cooperation with the University of Minnesota and the Minnesota Pollution Control Agency. This appropriation is available until June 30, 2019, by which time the project must be completed and final products delivered.

M.L. 2015, Chp. 76, Sec. 2, Subd. 10
$1,000,000 the first year is from the trust fund to an emerging issues account authorized in Minnesota Statutes, section 116P.08, subdivision 4, paragraph (d).

I. PROJECT TITLE: Tracking and Preventing Harmful Algal Blooms

II. PROJECT STATEMENT:

Harmful algal blooms (HABs), especially those caused by toxin-producing blue-green algae (Cyanobacteria), significantly reduce the recreational and ecological value of Minnesota lakes. They negatively impact water quality, degrade fisheries, and are a health concern for humans and domesticated animals. The duration, frequency, and extent of harmful algal blooms are increasing worldwide. New evidence points to similar changes in some Minnesota lakes, yet little information is available on historical trends in blooms or the present-day composition of algae associated with bloom formation and toxin production. Harmful algal blooms occur as discrete events and are known to relate to phosphorus concentration. However, the seasonality, water-quality conditions, and sediment-water interactions that drive these events are not well understood. A better understanding of the lake characteristics and nutrient-climate interactions that stimulate harmful algal blooms would facilitate new corrective measures and better allocation of management resources.

This project will address three key questions regarding the occurrence, composition, and causes of HABs: (1) when do they occur, in which type of lakes, and which species and toxins are present; (2) are they increasing in Minnesota, and if so, in which lakes; and (3) what are the main environmental factors causing bloom formation and toxin production? To answer the first question, we will intensively monitor a set of five lakes on a bimonthly (twice per month) basis for composition and abundance of algae, their associated toxins, and key water-quality variables including nutrients (N and P), temperature, dissolved oxygen levels, chlorophyll a, and phycocyanin. We will also deploy temperature and oxygen recorders to continuously monitor key variables that affect in-lake nutrient cycling, along with sediment traps to track seasonal changes in algal composition and abundance. This sampling will be done in collaboration with the DNR and PCA as part of their long-term Sentinel Lakes monitoring program. The second question will be addressed by analysis of dated sediment cores from 10 lakes for fossil algal pigments to assess historical changes in the abundance of cyanobacteria and other algae. This will allow us to determine where and when HABs have increased and link them to possible drivers such as land-use change and temperature increases. These lakes will also be selected from among those in the Minnesota Sentinel Lakes program and will cover a range of lake types (trophic status, lake depth, and size) and ecoregions of the state. The third question will focus on a single Sentinel Lake with known bloom problems. Here we will pair results from the monitoring and sediment cores with watershed and in-lake models of phosphorus loading to determine the critical factors leading to bloom development including watershed inputs, recycling from sediments, and changing lake temperatures.

This work will be done in collaboration with the University of Minnesota in their complementary project (038-B), “Increasing Harmful Algal Blooms in Minnesota Lakes”. The two research teams will coordinate monitoring effort on an overlapping set of lakes to extend the reach of this work from intensive laboratory studies to a broad range of observable field conditions in Minnesota lakes. Both research teams will regularly share data and results and coordinate the collection of samples when practical, and both groups will work jointly with the Minnesota Interagency Workgroup on Blue-Green Algae (MPCA, MDNR, MDH, MVMA) to update the agencies on our latest findings, coordinate research, response, and outreach efforts, and evaluate any emerging issues.

III. OVERALL PROJECT STATUS UPDATES:

Project Status as of July 1, 2016:

We have finalized the list of five Sentinel Lakes in Minnesota after receiving input from MPCA, MNDNR, and the St. Anthony Falls Laboratory (SAFL) Research group (who are working on a complementary LCCMR-funded HABs study). These lakes (Madison, St. James, Pearl, South Center, and Shaokatan) will undergo two years of monitoring for water quality, HABs, and cyanotoxins. We have coordinated with the MPCA to ensure all five lakes will be visited twice a month throughout the ice-free seasons of 2016 and 2017 (once each by both SCWRS and MPCA personnel).

In preparation for the first (2016) sampling season, we have acquired a YSI EXO2 multi-parameter sonde, capable of measuring vertical lake profiles for temperature, oxygen, conductivity, pH, turbidity, total algae, and Cyanobacteria. This instrument will be key for determining how HABs develop throughout the water column over the year, and what physio-chemical conditions accompany these blooms. Additionally, we have purchased and constructed five buoy arrays to be deployed in each of the study lakes. These arrays include both temperature and oxygen sensors and will record lake conditions throughout the ice-free season at 30-minute intervals. These data will help bridge the gap between bi-monthly sampling events and will record any major events that may lead to HABs fueled by internal loading of nutrients (i.e., wind-induced mixing of nutrient-rich anoxic bottom waters). Additionally, we have purchased all necessary miscellaneous supplies for the upcoming field season (i.e., bottles, chemistry reagents, calibration solutions).

Finally, thanks to the availability of the Emerging Issues funding, we were able to jump-start our sampling season shortly after ice-out in May, sampling all 5 lakes and installing the buoy arrays. We returned to sample all the lakes in June. In total, 251 water samples were collected from all 5 lakes including all major chemical, biological, and physical measures of interest (e.g., TP/TN, NOx/NH4, SRP, DIC, DOC, Chlorophyll *a*, phytoplankton, and four unique cyanotoxins [microcystin, cylindrospermopsin, anatoxin-a, saxitoxin]). An equivalent number of samples were also collected on alternate weeks over this time period by MPCA personnel, including phytoplankton and cyanotoxin samples to be later transferred to and analyzed by SCWRS.

Project Status as of January 1, 2017:

We have completed the first season of field sampling on the five selected sentinel lakes. Each lake was visited at least twice a month from May through October by either SCWRS or MPCA personnel. All analytes were collected for each sampling events and no events or samples were canceled or lost. In total, 524 water samples were collected across all lakes by SCWRS with a comparable number collected by MPCA in alternating weeks. We also successfully retrieved the buoy arrays from all 5 lakes in October and were able to upload all of the data that had been recorded over the previous 6 months (110,336 data points from South Center alone).

Water chemistry analysis has nearly been completed by the SCWRS Chemistry Laboratory. All analyses other than dissolved nitrogen species (NH4 and NOx) have been completed prior to January 1. Phytoplankton samples have all been preserved, settled, and condensed and are ready to be enumerated. We have purchased and installed an ABRAXIS Microplate reader for the purpose of analyzing our cyanotoxin samples via the ELISA plate method. This work will begin and proceed rapidly following a staff training and certification for the procedure by ABRAXIS scheduled in March 2017.

Project Status as of July 1 2017:

This project is currently right on track with the completion of one field season and the analysis of all water quality samples and the collection and processing of sediment cores from 4 of 10 lakes in this study. The remainder of the sediment cores will be collected in the Fall/Winter of 2017-2018.

We are currently in the middle of our second field season. Each of the five study lakes has been sampled bi-monthly, in cooperation with MPCA, since May of 2017. During our first sampling event, we also re-deployed temperature and oxygen sensors, as well as sediment traps in each of the 5 lakes, which will monitor the status of the lakes every 30 minutes throughout the entire open-water season.

Cyanotoxin samples analyzed by our newly purchased and installed ABRAXIS ELISA plate reader have provided the first systematic measurements of the cyanotoxins anatoxin-a, cylindrospermopsin, and saxitoxin in the state of Minnesota. All toxins were detected in at least one lake in 2016, coinciding with the peak bloom periods of July-September.

In March of 2017, we co-hosted a HABs workshop in collaborating with the St. Anthony Falls Laboratory, University of Minnesota Extension, and Minnesota Sea Grant. This day-long workshop focused on the issues of research, monitoring, and outreach as they pertain to HABs. This workshop focused on bringing together scientists, agency personnel, consulting companies and private citizens who were interested in these issues and

included administrators of state and federal agencies (MPCA, NOAA Sea Grant, MN DNR). The results of this workshop helped to foster further collaboration among HABs researchers in the State, as well as pinpoint future directions of HABs research that will be important to the citizens of Minnesota. We also invited participants to participate in coordinating a larger workshop in the future where the results from our study, and others in the State, can be presented to the general public.

Project Status as of January 1 2018:

This project continues to be on track with all activities. In November of 2017, we completed our second and final monitoring season for the five Sentinel Lakes selected for this project in Activity 2 (St. James Lake, Madison Lake, South Center Lake, Lake Shaokatan, and Pearl Lake). This included collecting water chemistry, cyanotoxin, and algae community samples twice a month from each lake in cooperation with the MPCA. All of the 2016 water quality samples have been analyzed and all water chemistry from 2017 has been analyzed in the SCWRS lab.

For Activity 3, dating has been completed on sediment cores collected from Madison Lake, Pearl Lake, Trout Lake, and Lake Shaokatan. Cores have been collected and are in the process of being analyzed for St. James Lake, Elk Lake, and Portage Lake. The remaining 3 cores (Carlos, Cedar, and South Center) will be collected before the end of the 2018 ice season (~mid-March) and are the top priority for analysis in the SCWRS laboratory. Algal pigment samples for the completed cores have also been shipped to Dr. Peter Leavitt at the University of Regina and are in the process of being analyzed.

For Activity 4, our partners at the USGS Water Science Center have completed their CE-QUAL2 models of both Madison and Pearl Lake which will be integrated with watershed models to be completed by SCWRS staff.

Project Status as of July 1 2018:

This project continues to be on track with all activities. Activity 1 has been completed and all of the sample collection and nearly all of the internal laboratory analyses have been completed for Activities 2 and 3.

For Activity 2, the final phase of analysis includes the phytoplankton samples which will be analyzed via microscopy. We have completed an initial assessment of the dominant taxa in these samples are in the process of doing our more quantitative biovolume assessments. We have begun to synthesize the field data collected from 2016 and 2017, which includes all of the water chemistry, cyanotoxins, and the buoy data (temperature and oxygen profiles).

For Activity 3, we have completed collection and analysis of sediment cores from all the lakes in this study. This includes all geochemical measurements (TP, BSi) as well as 210-Pb dating models. We have also completed pigment analysis on three of the cores (Pearl, Madison, Shaokatan) and the remaining core samples are in the queue for analysis at the University of Regina.

For Activity 4, we continue to partner with USGS to refine their predictive model for HABs in Madison Lake and they are currently working with the data provided by us from our monitoring efforts and buoys deployed in 2016. This model will be paired with a newly produced SWAT model that will be produced by SCWRS staff in the final year of this project.

Project Status as of January 1, 2019:

This project continues to move along with final analyses and synthesis in Activities 2 through 4 (Activity 1 has been completed).

For Activity 2, we have analyzed and synthesized all water chemistry data collected in both field seasons (2016 & 2017). We have completed all cyanotoxin analyses that were budgeted for, but have stored 131 additional toxin samples collected by SCWRS and MPCA in 2016 and 2017 for later analysis pending our amendment request (below). We have developed our phytoplankton enumeration and measurement technique using the CHARM laboratory inverted microscope and imaging software, and completing these samples will be a primary focus of this activity for the remainder of the project.

For Activity 3, all algae pigment analysis at the University of Regina have either been completed or are in the final stages of analysis. Diatom samples for creating diatom inferred-total phosphorus reconstructions is ongoing. We have completed diatom counts on Madison, Trout, St. James, and Pearl. Data from Pearl were not usable due to very poor diatom preservation, likely due to silica dissolution in highly alkaline waters. Comparison of finalized diatom and pigment data with the completed geochemical profiles will be the primary focus of the final synthesis for this activity.

For Activity 4, USGS has compiled stream and lake data for Madison Lake from agency sources as well as data collected as part of Activity 2 of this project. They have begun to use these data to calibrate a preliminary in-lake nutrient/algae model with an expanded taxonomic resolution for Cyanobacterial groups. This model will be refined and incorporated with SWAT modeling of Madison lake which is to be completed in the final period of this project.

Amendment Request (January 31, 2019):
We request a change in the budget for Activity 2 to increase the supplies budget from $5,000 to $10,500. This will allow us to purchase additional ELISA kits for analysis of 131 additional cyanotoxin samples that were collected in 2016 and 2017, but were not analyzed due to budgetary constraints. These samples were frozen after collection and are still viable for our ELISA toxin method. To pay for these analyses, we would like to shift remaining funds originally budgeted for Travel in Activity 2 ($4,000) and 3 ($1,500). This would increase the Supplies budget for Activity 2 to a total of $10,500 (up from $5,000). We had considerable savings in travel costs by combining trips for these two activities whenever possible, and by keeping our field team on site for multiple days rather than returning to SCWRS daily.

Amendment Approved by LCCMR 2/13/2019.

Overall Project Outcomes and Results:

Lakes are one of Minnesota's most precious resources and harmful algal blooms (HABs) threaten them both from an ecological and economic standpoint. This provides a survey of the current prevalence and toxicity of harmful algal blooms (HABs) in a subset of Minnesota lakes, determines if these blooms are increasing in frequency, and develops and refines modeling techniques that could be used to predict HABs in lakes across Minnesota. To this end, we intensively monitored five lakes in southwest and central Minnesota over 2 years for all major water chemistry parameters, algal biomass, and four cyanotoxins. In these lakes, and five additional lakes in northern Minnesota, we collected and dated sediment cores where fossil cyanobacterial pigments could be measured to track the occurrence of Cyanobacteria over the last 150 years. Finally, we chose one of the intensively monitored lake as a pilot study where we developed a watershed model (SWAT) and an in-lake hydrodynamic model (CE-QUAL-W2) to predict annual cyanobacterial bloom patterns. As a result of this project, we determined that in lakes which are already eutrophic, internal loading dynamics will play a key role in determining the size and toxicity of the bloom. Importantly, we found that even in shallow lakes (less than 16 ft maximum depth), temperature and oxygen dynamics are critical in terms of bloom timing and toxicity. Cyanobacteria pigment data from our sediment cores showed increasing HABs in some lakes over the 20th Century, but also demonstrate that conditions may have been even worse in the early to mid- 20th Century before the passage of the Clean Water Act. Our modeling results provide a framework for resource managers to

predict seasonal bloom formation and persistence in lakes across the state using publicly available and widely used modeling techniques.

IV. PROJECT ACTIVITIES AND OUTCOMES:

ACTIVITY 1: Jump-start lake monitoring program
Description: We will begin monitoring of algal blooms and associated limnological conditions at the onset of open water conditions in the first year of the project (April 2016). HABs typically appear in mid- to late-summer, but the conditions leading up to bloom formation cannot be understood without spring and early summer monitoring. Monitoring will be conducted at five Sentinel Lakes on a twice-monthly basis as described under Activity 2. In addition, we will instrument the five lakes with recording temperature and oxygen probes to continuously monitor chemical and physical lake conditions and sediment traps to track seasonal changes in algal composition and abundance.

Summary Budget Information for Activity 1:

ENRTF Budget: $ 93,000
Amount Spent: $ 93,000
Balance: $ 0

Outcome	Completion Date
1. *Accelerate the lake monitoring program by beginning in year-1 of the project (April, 2016) and extend the monitoring period to 7 months (April-October) for each of two years*	December 2016
2. *Instrument five lake with recording oxygen and temperature probes and sediment traps to continuously track changes in water column condition and algal composition and abundance*	December 2016

Activity Status as of July 1, 2016:

Sampling sites were finalized in coordination with MPCA and the SAFL research group. We selected sites that were 1) Sentinel Lakes and 2) had a documented history of HABs according to past MPCA/MNDNR Sentinel Lakes monitoring records. We also selected lakes over a range of depths to include both polymictic (mix continuously) and dimicitic (mix only in Spring and Fall) systems. The final list of lakes for monitoring is Madison Lake (SW MN; dimicitic), St. James Lake (SW MN; polymictic), Lake Shaokatan (W Minnesota; polymictic), South Center Lake (Central MN; dimictic), and Pearl Lake (NC MN; polymictic). We coordinated with the MPCA Citizen Lake Monitoring Program to distribute a letter to their volunteers on each of these lakes to make them aware of the work that was ongoing and also the presence of the submerged buoys over the next 2 years. We hoped by targeting citizen stakeholders around the lake, we can both encourage them with the knowledge of work being done on "their lake" and help to dissuade any concerns if our field crew or buoys are spotted by residents or visitors to the lake.

All necessary equipment for the field season was purchased. We purchased a YSI EXO2 Sonde for taking vertical profiles of physio-chemical variables, total algae, and total cyanobacteria. We also purchased temperature, dissolved oxygen (DO), and pressure sensors to instrument the submerged buoy arrays for each of the study lakes. These were set up to measure temperature every 0.5-1 meter and DO at the bottom and surface of each lake. In extremely shallow systems (i.e., Shaokatan and St. James), only the bottom DO sensor was deployed to prevent accidental damage of vandalism to sensors placed too close to the lake surface. Sediment traps were constructed at SCWRS and deployed in lakes deep enough to contain them (South Center, Madison, and Pearl) to test their effectiveness at integrating Cyanobacteria and total algae production over the season. In addition, we purchased all other necessary supplies (sample bottles, reagents, calibration standards) to cover the first two months (May and June) of sampling prior to the initiation of Activity 2 on July 1, 2016.

Two complete sampling events were carried out in May and June of 2016 on all five study lakes. In the first event, we also deployed the submerged buoy arrays in each lake and programmed them to record temperature and oxygen concentrations every 30 minutes for the entire field season (May through October). All buoys were successfully deployed and real-time corrected GPS coordinates were taken for each of their locations. Water samples were collected for TP/TN, NOx/NH4, SRP, DIC, DOC, Chlorophyll *a*, phytoplankton and cyanotoxins. In total, 251 unique samples were collected and transferred back to SCWRS for later analysis in our chemistry or CHARM (Center for Harmful Algal Research in Minnesota) laboratories. Additionally, vertical profiles were collected at each visit using the YSI EXO2 sonde for water column physio-chemical and biological variables (conductivity, temperature, DO, pH, turbidity, total algae, and total Cyanobacteria).

Activity Status as of January 1, 2017:

No activity during this reporting period

Activity Status as of July 1 2017:

This portion of the project has been completed. All necessary equipment was purchased and all samples from the early portion have been collected and analyzed.

Activity Status as of January 1 2018:

No activity during this reporting period

Activity Status as of July 1 2018:

No activity during this reporting period

Activity Status as of January 1, 2019:

No activity during this reporting period

Final Report Summary:

The prevalence and severity of harmful algal blooms (HABs) in Minnesota was one of the most pressing issues identified by both State agencies and the general public in Minnesota at the onset of this project. Because of this, the LCCMR decided to use Emerging Issues funding to jump-start this project a season ahead of when it would have normally started. Thanks to this timely support, we were able to formalize our sampling design, purchase all necessary equipment, complete construction of temperature and oxygen buoys, and begin our water quality sampling in May of 2016. In effect, this gave us an additional full field season for this project, doubling the amount of data that could be collected and analyzed over the course of this project. Importantly, this additional sampling year also gave us some insight into inter-annual variability of HABs in the lakes of this study and the range of timing and severity of HABs from year to year.

ACTIVITY 2: Identify species composition and timing of harmful algal blooms
Description: We will assess the relationship between algal communities and water quality in a representative group of Minnesota lakes to determine the distribution, abundance, and seasonality of bloom-forming species. Current water quality monitoring of five Sentinel Lakes (carried out by the MN DNR and MPCA) will be amended to include a twice-monthly algae sampling during the ice-free period over two years that will be analyzed by the St. Croix Watershed Research Station's CHARM Laboratory (Center for Harmful Algal Research in Minnesota), established with prior ENRTF support. Cyanobacteria that are detected will be quantified in terms of biomass

(bloom vs. non-bloom), danger to public health (toxin producing vs. non-toxin producing), and provenance (invasive vs. historically occurring).

Sampling will be done on an alternating 2-week basis for those lakes scheduled for routine monthly monitoring by DNR/MPCA field staff. Each site visit will include the collection of algal samples from near-surface and thermocline depths along with samples (epilimnion and hypolimnion) for water chemistry – total-N, total-P, nitrate/ammonia, soluble reactive P, dissolved organic and inorganic carbon – and algal toxins. Depth profiles of temperature, conductivity, pH, dissolved oxygen, chlorophyll *a*, and phycocyanin (a photosynthetic pigment specific to cyanobacteria) will be made with a YSI water-quality sonde specifically acquired for the project.

Soft algae (including cyanobacteria) will be identified and enumerated on a specialized "inverted" microscope recently acquired through ENRTF funding of our ongoing project, "Watershed-Scale Monitoring of Long-Term Best-Management Practice Effectiveness". Algal toxins, including microcystin, anatoxin-a, and cylindrospermopsin, will be analyzed by ELISA microplate reader.

Summary Budget Information for Activity 2:

ENRTF Budget: $ 165,800
Amount Spent: $ 165,800
Balance: $ 0

Outcome	Completion Date
1. *A quantification of the seasonality of harmful algal blooms across a representative sampling of Minnesota lakes*	June 2018
2. *The identification of bloom-forming species, the associated risk for toxin production, and the occurrence of invasive blue-green algae*	June 2018

Activity Status as of January 1, 2017:

We completed the field season, begun in May, in November. All 5 study lakes were visited twice monthly between May and October of 2016 (once by SCWRS and once by MPCA). We collected all necessary water samples and YSI profiles for each visit (detailed in Activity 1). We added one additional sampling trip to South Center Lake in November of 2016, due to the lake still being stratified during our October visit. Unfortunately, the lake was still stratified at this time, though we observed a considerable amount of hyoplimnetic erosion in our final sampling trip (full turnover likely occurred within days of our visit). Samples collected by MPCA that were to be analyzed by SCWRS (phytoplankton and cyanotoxins) were transferred to us after the season was completed. Water chemistry data and YSI profiles collected by MPCA will be shared with SCWRS once they have been completed.

YSI profiles from the year show clear connections between lake physics and the occurrence of Cyanobacteria blooms. Madison Lake provided one of the clearest examples of this (Fig. 1). Madison Lake was still mixed following Spring turnover when we visited in May and the entire water column was well oxygenated with low algae abundance and high clarity. On our subsequent visit in June, the lake had stratified and already driven oxygen down near 0 mg/L in the hypolimnetic bottom waters (Fig. 1). We observed a rapid accumulation of sediment P in the hypolimnion during this anoxic period, rising to 490 ug/L as compared to 84 ug/L in the surface waters. This stratification and anoxic hypolimnetic state was maintained through August. In August, stratification was rapidly broken, likely due to a series of storm events with high winds in this month, and the lake mixed once again releasing that P into the upper water column. A large Cyanobacteria bloom followed this mixing event and persisted through October (Figure 2). South Center Lake's hypolimnion also went hypoxic during the Summer, however the lake never mixed until late November leading to less abundant Cyanobacteria and better water clarity throughout the ice-free season. St. James and Pearl both were polymictic and had Cyanobacteria blooms from mid-Summer through October. Shaokatan remained in a clear-water, macrophyte-dominated, state throughout the season and never produced a Cyanobacterial bloom.

Water chemistry samples collected by SCWRS from the previous field season are being analyzed in the chemistry laboratory and all but the dissolved nitrogen species (NOx and NH4) have been completed. An ABRAXIS micro-plate reader has been purchased and installed for conducting cyanotoxin enzyme assays using the ELISA method. These samples are currently preserved via freezing, and will begin to be analyzed following a scheduled staff training and certification workshop scheduled in March of 2017.

Finally, all submerged buoy arrays were recovered from the study lakes in October and the data from them were recovered successfully. These data provided us 30-minute resolution on the condition of each of the lakes over the entire field season. In our dimictic systems (i.e., South Center), these data showed the onset and duration of stratification throughout the year and the accompanying oxygen depletion in the bottom waters (Fig. 3). In others, these results were more surprising. Pearl Lake, one of our polymicitic systems, actually demonstrated sustained periods of anoxia (~1 month) in the late summer (Fig. 4).

Activity Status as of July 1 2017:

We have nearly completed the analysis of toxin and water chemistry collected from the five intensively monitored lakes in 2016. These data are shown in Figure 5 over the entire sampling period. These data represent only the samples collected by SCWRS, as the MPCA samples are still being processed and formatted; those additional data will be incorporated into our analysis once they become available. These results provide the first systematic survey of all four major cyanotoxins in the state of Minnesota (Figure 5) and are plotted with a red-dashed reference line that corresponds to the minimum drinking standard as set by the state of Minnesota (microcystin, anatoxin-a), US EPA (cylindrospermopsin), or as recommended by the peer-reviewed literature (saxitoxin). We can now confirm the detectable presence of all four toxins in the study lakes as well as concentrations in excess of safe drinking standards for humans (and pets) for both microcystin and anatoxin-a. These toxins show strong seasonal trends, with the highest concentrations in late summer and fall, and also seem to persist for weeks to months within the water column. These peaks appear to be correlated to declining N:P ratios and algal biomass.

We began our second field season in May of 2017. This included the re-deployment of temperature and oxygen sensors in each of the five intensively studied Sentinel Lakes (Madison, St. James, Pearl, South Center, and Shaokatan). We also deployed sediment traps for the collection of silt and algae in the deeper lakes (Madison, Pearl, and South Center). These sensors will collect data on the temperature and oxygen stratification of the lakes every 30 minutes from May through October.

As laid out in our workplan, we have visited each of these five lakes bi-monthly, in cooperation with MPCA, in May and June, and will continue to monitor them through October of 2017. For 2017 we have added an additional hypolimnetic sample in Pearl lake (only surface water was collected in 2016) based on the observation of short periods of stratification and hypoxia in 2016. This lake was thought to remain mixed throughout the open-water season, and only through the installation of our sensors in 2016, were we able to determine it might be susceptible to anoxic nutrient release from the sediments.

Activity Status as of January 1 2018:

We have successfully completed both proposed monitoring seasons for the five lakes selected in this study. All lakes were visited twice monthly by SCWRS staff for both the 2016 and 2017 open water seasons (May – October).

We have now completed analyses on all water quality and toxin samples collected in the 2016 season and have finished analyzing all water chemistry samples from the 2017 season. Additionally, we successfully retrieved all

five oxygen and temperature buoys and sediment traps that were deployed in the lakes and have processed those data into isopleth charts. Notably, we extended the field season on South Center into November of 2017 so that we could capture the fall-mixing event that occurred too late in the season in 2016 (Figure 6).

In total, we have collected and analyzed 1,371 water quality samples in the 2016 and 2017 field seasons and recorded over 80,000 vertical profile measurements of temperature and oxygen from our five buoy arrays. These data will provide one of the most comprehensive syntheses of the environmental conditions that lead up to and follow HABs in Minnesota lakes.

Activity Status as of July 1 2018:

We have begun to synthesize the monitoring data collected in 2016 and 2017 and look at the major drivers for bloom formation and toxin production. Major findings thus far include:

- *South Center (dimictic):* gradual increase in TP and chl-*a*, both decreasing into the fall (Fig. 7A, C).
    - Saxitoxin and anatoxin-a increased, peaked, and subsided in parallel with *Dolichospermum* and A*phanizomenon flos-aquae* biomass (Fig. 7G,H and 8A).
    - Cylindrospermopsin peaked in fall with low abundance of *A. flos-aquae* and *Cylindrospermopsis raciborskii* (Fig. 7F).
- *Madison Lake (polymictic):* rapid increase in TP and chl-*a* following summer mixing events (Fig. 7A,C).
    - First mixing event led to a spike in saxitoxin and anatoxin-a from *A. flos-aquae* (Fig. 7G,H).
    - Second successive mixing events produced anatoxin-a and cylindrospermopsin and more complex blooms (*A. flos-aquae*, *C. raciborskii, Dolichospermum sp*.; Fig. 7F,H and 8C).
- *Pearl Lake (polymictic):* largest increase in TP and chl-*a* following late summer stratification and anoxia (Fig. 7A,C).
    - Mixing of anoxic waters in late summer led to fall blooms of *Microcystis* and *Dolichospermum* and peak concentrations of total microcystins (Fig. 7E and 8B).
- *St. James Lake (polymictic):* never stratified and chl-*a* increased with TP, peaking in mid-to-late Summer (Fig. 7A,C).
    - Total microcystin concentrations paralleled *Microcystis* biomass and decreased into the fall (Fig. 7E).
    - Decreasing TN (Fig. 7B) and increased light limitation from DOC (Fig. 7D) may have contributed to the decline in the toxic bloom in the fall, despite elevated TP (Fig. 7A).
- *Lake Shaokatan (polymictic):* lowest TP throughout the summer, but spiked to the highest TP concentrations in the fall. Despite this spike, chl-*a* never increased to levels seen in the other lakes and no bloom was observed (Fig. 7A, C).
    - No significant trends in cyanotoxins or bloom formation were observed throughout the year.
    - Despite a history of CyanoHABS, Shaokatan flipped to a macrophyte dominated state. The fall peak in TP corresponded to senescence of the dominant macrophytes, *Ceratophyllum demersum* and *Myriophyllum*; both are efficient users of phosphorus.

In summary, the lakes in this study which had the highest variability in stratification showed marked changes (Madison, Pearl) in both the cyanobacterial community and toxin production following mixing events. This included blooms into the late fall and temporal asynchrony of saxitoxin and anatoxin-a, likely produced by multiple different genera. Lakes that never mixed, or never stratified for long enough to produce anaerobic conditions, did produce CyanoHABs, but they followed more predictable seasonal patterns (unimodal peaks in

mid-summer). These results indicate the importance of understanding inter-annual variability in lake physical structure to predict the duration and toxicity of HABs.

Activity Status as of January 1, 2019:

All water chemistry data from 2016 and 2017 have now been compiled and we are in the process of final analysis and synthesis of these data. Trends for both water chemistry and cyanotoxins for both years are shown in Figure 10. We note that whereas certain toxins showed a consistent seasonal pattern across years, other (i.e., saxitoxin and cylindrospermopsin) were quite different. We hope to better understand this contrast through analysis of additional archived cyanotoxin samples as well as completion of the phytoplankton samples.

Final Report Summary:

This project represents the most intensive sampling of Cyanobacteria and their toxins conducted in the State of Minnesota to date. In cooperation with the MPCA, five lakes in southwest and central Minnesota were sampled every 2 weeks for water quality parameters. Additionally, each lake was instrumented with temperature and oxygen buoys which collected information on the physical stratification of the lake at 30-minute intervals from May through at least October (South Center Lake was monitored through November due to the longer stratification period and its proximity to SCWRS).

The role of temperature and oxygen on producing HABs

Although the connection between nutrients (primarily Phosphorus [P]) and HABs has long been known, there is still considerable uncertainty about how the timing of release may trigger more severe cyanobacterial blooms or the production of toxins. The use of buoys to measure thermal and oxygen structure in a lake represented a new technique for understanding what may be driving the most toxic Cyanobacteria blooms in Minnesota lakes. These buoys give us high-frequency (every 30 minutes) measurements of the lake's thermal structure and are necessary for understanding the potential role of internal loading of P from the sediments.

Internal loading of P to lakes from their sediments has been shown to increase by up to 40 times in the absence of oxygen (anoxia). Therefore, understanding if and when it occurs in lakes is critical when measuring the potential for blooms fueled by this sediment-bound P. It is often assumed that these internal loads are only important in weakly stratified lakes which are of intermediate depth. This is because shallow systems were thought to always be oxygenated due to continuous wind-mixing, and deep lakes are assumed to only mix twice a year when the water is cold and the photoperiod is still relatively short. To test these assumptions, we selected lakes along a depth gradient that ranged from 3 to 18 meters to monitor the presence of anoxia, and how it affected nutrient concentrations in the lake, as well as cyanobacterial abundance and toxicity.

The shallowest lakes in this study, St. James and Shoakatan, both behaved as expected with bottom waters super-saturated with oxygen for the majority of the year due to high rates of primary production by algae (St. James) and aquatic plants (Shaokatan) (Figure 11). The deepest lake, South Center, also behaved as assumed, with stratification beginning in the spring (May) and lasting well until the fall (November; Figure 3 and 6). In both cases, these results demonstrate that in extremely shallow systems (3m or less) and deep lakes (>15m), internal loading of nutrients is probably minimal. In these lakes fluxes of nutrients from sediments are either extremely slow due to oxygenated conditions (shallow lakes), or, timed during the part of the year where conditions are unfavorable for blooms due to cold water and short photoperiods (deep lakes). This indicates that bloom prediction and management strategies should focus on the timing and intensity of external loads from the watershed.

As expected, our lake of intermediate depth, Madison Lake (~10m), proved very susceptible to internal loading due to early mixing in August and then repeated mixing until the fall of 2016 (Figure 12). This prolonged anoxia

allowed for the accumulation of sediment P in the hypolimnion and the early mixing event in August caused that P to be mixed throughout the water column where it would be available to Cyanobacteria. This likely provided a second pulse of nutrients to the system in addition to the traditional external load associated with snow melt and elevated precipitation in the Spring (freshet). Because of this substantial potential for internal loading of P, the timing of this early mixing event (or whether it occurs at all) between years will be a key factor in determining the size of HABs and how long they persist into the fall.

Finally, perhaps the most surprising result from the buoy deployment was found in Pearl Lake. This lake is relatively shallow (5m) and was not expected to stratify due to its depth and round shape (high fetch relative to surface area). However, we found that not only did this lake stratify for a considerable amount of time in 2016 (>2 weeks), it rapidly developed anoxic conditions over this time period (Figure 4). This points to a previously unconsidered source of nutrients to lakes of similar depth (5-10m) that is important in predicting the timing and persistence of blooms between years (See "What's causing HABs in Pearl Lake?" fact sheet attached in Supplementary Material for more information).

The results from the temperature and oxygen buoys deployed for this study have revealed additional factors that may play a role in producing HABs in lakes of intermediate depths. As expected, internal loading of P is a major player in lakes of intermediate depth (10-15m), however it must also be considered in relatively shallow lakes that were previously though to mix continuously (5-10m).

Water Quality, Phytoplankton, and Toxin Monitoring

In order to understand the mechanisms that underlie HAB formation, intensive monitoring of a wide array of environmental parameters is necessary. In addition to characterizing the physical structure of a lake (i.e., temperature and oxygen stratification outlined above), we collected chemical, biological, and toxicity data on each of the five lakes at two-week intervals over 2 years (2016, 2017). Importantly, these data represent one of the first systematic survey of all four major cyanotoxins (microcystin, anatoxin-a, saxitoxin, cylindrospermopsin) in Minnesota. Figures 13 through 17 show a summary of all water chemistry data collected for the five lakes in this study over the 2 years.

Total phosphorus concentrations increased linearly throughout the spring and early summer in St. James which receives run-off from a largely agricultural watershed as well as the adjacent city of St. James, MN (population: ~4,600). No secondary pulses of nutrients were detected, consistent with the absence of internal loading described above. The peak cyanobacteria bloom (measured as chlorophyll-a) occurred in late-July and early August of both years and coincided with the highest TP concentrations. Both TP and algal biomass declined into the Fall. Microcystin concentrations also peaked in St. James, following the same pattern as TP and algal biomass (Figure 18). Microcystin concentrations exceeded the EPA's safe contact standard (8 ug/L) in St. James in both 2016 and 2017, during which period the genus *Microcystis* was the dominant algae found in the lake. Detectable concentrations of both anatoxin-a (Figure 19) in the Spring and Fall, and cylindrospermopsin (Figure 20) in the mid-Summer and Fall were also found in St. James. Both anatoxin-a and cylindrospermopsin were found in samples where the genus *Dolichospermum* was the most abundant Cyanobacteria.

South Center also had linear increases in TP and total algae from spring until mid-summer which tapered off into the Fall. Unlike St. James, South Center did have a secondary pules of internally derived nutrients in the late fall that was likely due to internal loading as the lake began to mix. However, because this event occurred late in the Fall as waters had already cooled, no secondary bloom of Cyanobacteria occurred in either year. All water chemistry data for South Center Lake are summarized in Figure 18. All four toxins were detected in South Center Lake over the course of the 2 years of sampling, however they were asynchronous and likely related to turnover in the dominant species of Cyanobacteria. Microcystin peaked in the late summer about a month after peak Cyanobacterial biomass and during a period where both TP and algal biomass was declining (Figure 18). This indicates that this toxin was likely being released from cells as they senesced during the cooler fall

temperatures. The genera *Dolichospermum* and *Aphanizomenon* were the most abundant during peak microcystin concentrations. Both genera were also likely responsible for the production of saxitoxin (Figure 21) and anatoxin-a (Figure 19) in both years, although the production of these toxins were asynchronous with saxitoxin peaking first, followed by anatoxin-a. Anatoxin-a concentrations exceeded drinking water standards in 2016 and saxitoxin concentrations exceeded standards in 2017 although both years had relatively similar bloom concentrations and timing (Figure 18). Cylindrospermopsin was detected in the later summer and fall (Figure 20) and was correlated to the appearance of the exotic species *Cylindrospermopsis raciborskii* in phytoplankton samples.

Shaokatan had very low concentrations of TP and total algae throughout most of the season in both years. Shaokatan was unique in that it was completely covered in aquatic plants (macrophytes) in both years. The MPCA had documented the flip of Shaokatan from a turbid, algae-dominated state to a clear-water, macrophyte dominated state prior to this study beginning. In 2016, after the macrophytes had died off we measured TP concentrations in Shaokatan ~150 µg/L, the highest measured in any lake during this study (Figure 15). The lack of an associated HAB demonstrates the effectiveness of macrophytes in locking up P in this system and reinforces the importance of protecting and reintroducing aquatic plants to shallow lakes in order to manage for HABs. As would be expected, toxin concentrations were very low in Shaokatan, although all 4 toxins were detected on at least one occasion over the 2 years (Figures 18-21). Very few Cyanobacteria were seen in any phytoplankton samples collected from Shaokatan, with only *Dolichospermum* being present in very low concentrations. Because Shaokatan is very shallow, it is possible that some of these toxins were being produced by benthic or epiphytic Cyanobacteria which would not have been collected in our water column samples.

Pearl TP and algal concentrations increased similar to St. James for the first part of 2016, but following a multi-week anoxic period (described in the previous section) TP and algae both increased at a more rapid pace (Figure 16). This secondary bloom was likely triggered by internally loaded sediment P that became unbound when the bottom waters become anoxic. This triggered both the largest biomass of Cyanobacteria and the highest concentrations of toxins, including microcystin concentrations exceeding the safe contact standard of 8 µg/L (Figure 18). This bloom was composed of primarily *Microcystis*, however, *Dolichospermum* was also abundant.

Madison TP and algal concentrations increased gradually in the first part of both years but jumped up in the late Summer due to early mixing events. Measured concentrations of TP in the hypolimnion exceeded 600 µg/l prior to the mixing event, almost 10x higher than was measured in the surface water (68 µg/l). Once this huge pool of nutrients was mixed in August of 2016 and 2017 the Cyanobacterial bloom peaked. In Madison, this bloom was primarily composed of the genus *Aphanizomenon* which produced a peak of anatoxin-a (Figure 19) coincident with the bloom and released microcystin (Figure 18) as the bloom senesced. Both toxins were observed in concentrations exceeding Minnesota Department of Health drinking water standards.

ACTIVITY 3: Reconstruct frequency of algal blooms relative to natural conditions

Description: We will determine where and when bloom-forming algae have increased in Minnesota lakes over the last century to better understand the causes and susceptibility of individual lakes to bloom development. We will use sediment paleolimnological methods to reconstruct the frequency and severity of cyanobacterial blooms in 10 lakes selected from the Sentinel Lakes monitoring program. Sediment cores will be collected from each lake and dated using radioisotopes at the St. Croix Watershed Research Station to establish a continuous history of lake condition over the last 150 years. Dated sections will be analyzed for fossil algal pigments, including those unique to blue-green algae, to determine presence, abundance, and frequency of harmful algal blooms in a historical context.

To obtain the sediment chronology, cores will be radiometrically dated by ^{210}Pb methods, supplemented as needed by identifying the 1963 ^{137}Cs peak that is remnant from the atmospheric testing of nuclear bombs. Based on typical sediment accumulation rates in Minnesota lakes, it should be possible to obtain reliable dates back to the mid to early 1800s in all lakes. Dating resolution will be roughly decadal overall, but more detailed (approximately 5-year) for the most recent 2-3 decades.

The sediment cores will be analyzed for a suite of components to assess changes in algal abundance and composition as well as nutrient levels that contribute to the development of HABs. Fossil pigments specific to cyanobacteria along with those produced by other algal groups will be the primary tool for reconstructing changes in HABs and overall lake productivity. In concert with pigment analyses, lake-water phosphorus content over time will be estimated by analysis of the remains of diatoms, a group of algae with certain species that are diagnostic of phosphorus content in the water in which they live. General algal productivity will be assessed by the accumulation of biogenic silica, which is largely composed of the glass cell walls of these diatoms. The phosphorus content (both total and extractable fractions) of the sediment will determine apparent loads of this essential nutrient.

Ultimately, core reconstructions will be compared with local land-use history, nitrogen deposition trends, and meteorological records (temperature, wind speed, precipitation) to determine whether any of these potential drivers of limnological change are correlated with shifts in lake productivity and, in particular, the abundance of HAB-forming algae.

Summary Budget Information for Activity 3:

ENRTF Budget: $ 171,900
Amount Spent: $ 171,900
Balance: $ 0

Outcome	Completion Date
1. *A comparison of historical changes in harmful algae among a large suite of Sentinel lakes to determine the geographic extent and timing of the problem*	January 2018
2. *An assessment of the likely drivers of increasing harmful algae by comparison of trends in lake sediment cores with changes in landscape, land-use, and climate over the period of record*	December 2018

Activity Status as of January 1, 2017:

We have selected five additional Sentinel Lakes with more infrequent or no history of Cyanobacterial blooms (Trout, Elk, Portage, Carlos, and Cedar) to complement the study lakes described in Activity 1 and 2. These lakes were selected to fill out the range of lake-types present in Minnesota and to give a clear picture of the historical occurrence and persistence of HABs across the state. Field work for coring these lakes is scheduled to begin after January 1, 2017, once ice conditions allow us to access these lakes.

Activity Status as of July 1 2017:

In February of 2017, we collected sediment cores from St. James, Madison, and Shaokatan through the ice. We have successfully dated the cores from Shaokatan and Madison. Due to the known high sedimentation rate of Madison, based on earlier work, over 3 meters of sediment were extracted from this lake! Our St. James core was found to contain glacial clays (dating back to ~11,000 years ago) after less than 50 cm down in the core. This could possibly be due to a hiatus in sedimentation at our coring site. We will recore St. James in a different area during the Fall of 2017. Loss-on-ignition profiles for Madison, Pearl and Shaokatan sediment cores have all been completed.

Activity Status as of January 1 2018:

We have completed dating and LOI for Madison, Pearl, and Shaokatan. We have also collected and completed dating on Trout Lake. Cores from Elk and Portage lakes have been collected and are in the process of being analyzed in the SCWRS laboratory. St. James Lake was re-cored in September of 2017 and this new sediment is currently being dated. South Center Lake will be cored in February of 2018 in conjunction with a Winter Ecology

undergraduate course which is being hosted by SCWRS. Carlos and Cedar lake cores will be collected before the end of the ice season in 2018 (early March).

Pigment samples have been preserved and shipped to Dr. Peter Leavitt at the University of Regina for all cores which have completed dating models (Madison, Pearl, Shaokatan, and Trout). We expect the results from these analyses no later than March of 2018 and we will continue to select and ship samples from the remaining cores as the dating models are completed.

Activity Status as of July 1 2018:

All ten lakes in this project have been cored and those cores have been dated and analyzed for all major geochemical elements at SCWRS. We have received completed pigment analyses from University of Regina on 3 of the cores and the other 7 remaining are currently being processed. Preliminary pigment analyses from Madison, Pearl, and Shaokatan are shown in Figure 9. These pigment data show a fairly diverse history of blooms across the three lakes. Madison may have had more intense, but less toxic, blooms in the past. Blooms in Shaokatan had gotten progressively worse up until the major shift to macrophyte dominance as a result of lake restoration in the last ~5 years. The concentration of Cyano pigments in Pearl Lake has remained relatively constant through time, with a small peak in the 1950s prior to the Clean Water Act.

Activity Status as of January 1, 2019:

We have received completed pigment profiles for 6 of the 10 lakes and the final 4 lakes are nearing completion at the University of Regina.

We have completed diatom counts on the sediment cores from Madison, St. James, and Trout. Diatom counting from Pearl Lake was halted due to very poor preservation of fossil diatoms in this core, likely due to silica dissolution. Pearl is a Chara-dominated lake with high alkalinity, which probably led to these preservation issues. We have prepared diatom slides for the final six lakes and counting is underway on those cores. Once completed, these data will be used to produce the diatom-inferred phosphorus model which will serve as an indicator of past water quality conditions. Pairing these data with our nearly completed pigment profiles will provide the best historical context for the prevalence and causes of HABs in these lakes.

Final Report Summary:

One of the biggest questions surrounding HABs is if they are actually increasing in frequency or if the public and researchers are just becoming more aware of them. In the absence of good long-term monitoring records, ^{210}Pb-dated sediment cores provide a means of measuring changes in a lake over the last 150 years. In the case of HABs, this can be done using fossil pigments left behind by algae that are unique to Cyanobacteria. Depending on the concentration and type of pigments present, we can tell reconstruct the abundance and types of Cyanobacteria that were in the lake historically. Additionally, other algae whose remains are preserved, in this case diatoms, can tell us about the conditions of the lake over that same time period based on what we know about the water quality preferences of those species in contemporary lake samples.

Cyanobacterial pigments

The concentration of fossil pigments from Cyanobacteria were measured in 15 samples from each of the ten MN DNR Sentinel Lakes selected as part of this study. These lakes ranged from shallow hypereutrophic lakes in SW Minnesota to deep oligotrophic lakes in NE Minnesota. As was expected given the wide range in lake type and geography, pigment results showed differing patterns across the state (Figure 22). It is important to note that pigment preservation in sediment cores is lake-specific, so patterns should be contextualized based on the current condition of that specific lake. For example, if a currently eutrophic lake shows no change in pigment

concentrations throughout the core, that would indicate that it was likely historically eutrophic (no change), likewise, if an oligotrophic lake showed the same pattern it would mean that it was historically oligotrophic (also no change).

Three of the five intensively monitored lakes, all of which were known to be eutrophic to hypereutrophic, had similar patterns of increasing Cyanobacteria through time. Both Shaokatan and South Center showed increasing Echinone (a pigment present in all Cyanobacteria) through time. St. James ad a similar pattern, however, due to this lake being dredged in the 1970s, we only were able to collect sediment from that time forward. This pattern reflects the gradual accumulation of nutrients from non-point source pollution over the 20th Century. Pearl shows fairly constant Echinone throughout the last 150 years, indicating that this lake was likely fairly naturally productive. In contrast, Madison actually shows a fairly steep reduction in Cyanobacterial pigments with peaks in the early to mid- 20th Century. Because this lake is adjacent to the city of Madison Lake, this is likely due to the history of this lake being used for point-source waste-water disposal prior to the passage of the Clean Water Act in 1972. Unfortunately, it appears that Cyanobacterial pigments have begun to increase again in Madison since ~1990, indicating the possibility of increasing problems in this lake.

The remaining five lakes are all meso- to oligotrophic lakes in north and northeastern MN. Trout, which acts as our control, is a nearly pristine lake on the edge of the Boundary Waters. It showed no significant change in Cyanobacterial pigments, as we would expect. Portage, which is located just north of Park Rapids, MN is a meso to eutrophic lake which shows little change in total Cyanobacterial abundance, but an increase in Canthaxanthin which is associated with the benthic Cyanobacteria *Nostoc*. Elk, which is located in Itasca State Park, shows the disappearance of Aphanizophyll, associated with N-fixing Cyanobacteria, after 1900 and very small increases in total Cyanobacteria (Echinone) as well as colonial Cyanobacteria (Myxoxanthophyll). Carlos, which is located near Alexandria, MN, shows gradual increases in Cyano pigments over the 20th Century, associated with the establishment of resorts and cottages in this popular recreational area. Interestingly, these pigments decline recently which could reflect the establishment of zebra mussels in this lake (first detected in 2009). Cedar Lake (in Morrison County, MN) shows little change in Cyanobacteria over time besides an increasing occurrence of the pigment Aphanizophyll in the late 20th Century which could indicate a switch in the lake to being N-limited.

In general, the Cyanobacterial pigments support data that HABs are becoming more frequent in our most disturbed lakes in Southern and Central Minnesota. There is evidence, however, that some lakes which may have received point-source pollution prior to the passage of the Clean Water Act have actually significantly improved (i.e., Madison) though they still remain impaired today. The less disturbed lakes in northern MN in this study show little change in the abundance of Cyanobacteria, but there is evidence of changes to the types of Cyanobacteria present in these systems.

Diatom reconstructions of past conditions

Due to the lack of long-term monitoring data available for nearly all lakes in Minnesota, we instead rely on using sediment cores to reconstruct the history of these systems. One of the most effective tools for this are diatoms, which are very sensitive to environmental conditions and have known optima and tolerance ranges for nutrients, especially phosphorus. Below are the results of this analysis from Madison, Shaokatan, and St. James.

The stratigraphic diagram for Madison showed the predominant diatoms that were driving the shifts in the community assemblages, as well as the results of the constrained cluster analysis, and the percentage of plankton throughout the core (Figure 23). The samples from the 1800s were dominated by *Aulacoseira ambigua* and *A. granulata*; these species are characteristic of nutrient-rich and turbid, wind-swept conditions. In the early 1900s, through the 1930s, the assemblage shifted to predominantly small *Stephanodiscus* species (*S. minutulus*, *S. hantzschii*, and *S. parvus*), which are also indicative of nutrient enrichment. From the 1940s to the 1960s there was another shift, characterized again by the small *Stephanodiscus* species (although in lower abundance than the early 1900s) and an increase in *Fragilaria mesolepta*, a planktonic mesotrophic indicator

which can be indicative of nitrogen enrichment. The assemblage from the 1970s to the core top was again dominated by *Aulacoseira ambigua* and *A. granulata*, with the addition of *Fragilaria crotonensis*, another eutrophic indicator. According to the constrained cluster analysis (Figure 23), the two largest breaks in the samples occurred between 1962 and 1973, and between 1883 and 1905.

Passive plotting of the Madison Lake core on the MN calibration set showed that there has been change in the diatom community assemblage in the core relative to the diatom communities in the calibration set samples (Figure 24). However, in the MN calibration set, TP is correlated with axis 1, and the major shifts in the diatom community occurred along axis 2, which is correlated with lake color. This suggests that TP is not the main driver of these shifts. The low λ_r/λ_p value of 0.12 also supports the conclusion that TP has not been the predominant driver of change in Madison Lake during the period of study.

In contrast to the more abrupt shifts observed in the diatom community in the Madison core (Figure 23), the diatom assemblage in Shaokatan showed much more subtle changes over time (Figure 25). The presence of *Aulacoseira ambigua*, *A. granulata*, and *Stephanodiscus minutulus* throughout the core indicate nutrient enrichment throughout the period of study, although all three of these species are very low in the topmost sample, which could indicate the start of a decline in nutrient levels in the lake and is consistent with the recent flip to a macrophyte-dominated clear state. The samples from the late 1970s through 2015 also showed a rise in two benthic species, *Cocconeis placentula* var. *lineata* and *Gomphonema sarcophagus*, which also suggests a rise in water clarity. The samples at the core top (2009 and 2015) showed an increase in *Fragilaria mesolepta*, a planktonic mesotrophic indicator which can be indicative of nitrogen enrichment.

As with Madison Lake, projection of the Shaokatan core on the MN calibration set showed change that is orthogonal to the TP axis; again, suggesting that TP was not the predominant driver of diatom community turnover in the lake and that the change was more aligned with lake color (Figure 26). The λ_r/λ_p value of 0.40 also suggests that although TP may have played a role in diatom community change, it was not the most important driver.

With the exception of the uppermost sample (2012), the diatom community in St. James Lake was dominated by planktonic eutrophic indicators such as *Aulacoseira ambigua*, *A. granulata*, and *Stephanodiscus minutulus* (Figure 27). The tychoplanktonic species *Staurosira construens* and *Staurosirella pinnata* were also abundant; these species are primarily benthic, but are often swept-up and suspended into the water column; many of these species are adapted to live on fine-grained sediments, such as those found in shallow lakes. The sample from 2012 showed a shift in the community, and was dominated by the eutrophic indicator *Fragilaria crotonensis* and by the planktonic diatom *Lindavia ocellata*. There is also a slight rise in *Asterionella formosa* and *Fragilaria crotonensis* in this sample, which could indicate nitrogen enrichment. However, these differences between the uppermost sample and the rest of the core could be the result of diagenetic processes. Again, none of the shifts in the diatom community were significant when compared to a broken stick model.

As with Madison and Shaokatan Lakes, projection onto the MN calibration set showed the change in St. James Lake was more closely aligned with axis 2, again suggesting drivers other than TP, such as lake color (Figure 28). However, the shift in the topmost sample (2012) shows some movement along axis 1, and this is reflected in a higher λ_r/λ_p value of 0.73.

ACTIVITY 4: Determine how nutrients and climate interact to favor harmful algae

Description: We will quantify phosphorus inputs and cycling in an intensively monitored sentinel lake to determine the critical factors leading to bloom development including watershed inputs, recycling from sediments, and changing lake temperatures. The study lake will be among those sampled in Activities 1 and 2, thus allowing us to pair monitoring of harmful algae blooms with mechanistic models that describe watershed inputs and internal recycling of nutrients. We will measure potential for in-lake recycling of legacy nutrients (phosphorus) by determining the fraction of labile phosphorus in the lake sediment. We will monitor bloom

formation by quantifying harmful algae in sediment traps and water column samples, along with potential environmental controls, including water chemistry, lake temperatures, and oxygen depletion of bottom waters.

These results will be paired with watershed and in-lake models to better understand the factors contributing to bloom formation. Specifically, we will use the Soil and Water Assessment Tool (SWAT), a watershed modeling program developed by the Agricultural Research Service (ARS) of the U.S. Department of Agriculture (USDA) to estimate watershed nutrient loads, present-day and in the past. Model construction requires inputs of hydrography, topography, soils, land cover, and agricultural management practices. For the study watershed, a SWAT model will be calibrated to current (2000-2010 average) land-use and climate conditions. In particular, the model will be constrained to match the sediment and phosphorus loads inferred from the sediment core data for this recent time period. Then, the model will be run to simulate sediment and phosphorus loads for selected periods in the past and tested against the sediment core data for these past periods. These model runs will build the mechanistic relationship between the erosional and fertilization history of the terrestrial watershed and how this history is recorded in the lake sediments.

To complete the analysis and infer the impact of watershed land use on lake-water quality, a coupled hydrodynamic – nutrient cycling model, CE-QUAL-W2, will be used to simulate algal and nutrient dynamics within the lake as well as the water-column physical parameters (temperature, dissolved oxygen) that govern them. A primary goal of in-lake modeling will be to partition the loading of phosphorus between external (watershed) and internal (sediment) sources. A secondary goal will be to use SWAT-model inferred nutrient loading for a selected past time period and see if a calibrated CE-QUAL-W2 model can predict the algal community as determined in the sediment core.

Summary Budget Information for Activity 4:

ENRTF Budget: $ 162,300
Amount Spent: $ 162,300
Balance: $

Outcome	Completion Date
1. *A mechanistic understanding of drivers of harmful algal blooms based on intensive monitoring of algae phenology and in-lake processes*	June 2019
2. *A determination of the relative importance of external loading vs. the internal recycling of phosphorus in terms of driving harmful algal blooms across lakes*	June 2019
3. *A predictive framework linking internal and external nutrient loads to the occurrence of harmful algal blooms in Minnesota lakes*	June 2019

Activity Status as of January 1, 2017:

Based on our first year of data, and in collaboration with the USGS Water Science Center personnel, we have decided on Madison Lake as the study lake for Activity 4. We picked this lake because of its sensitivity to changes in thermal stratification as well as the noxious Cyanobacteria blooms that were observed there in 2016. Additionally, Madison Lake was the site of the SAFL HABs Research Group's buoy, which provides an additional source of high-frequency data that could be used in our modeling efforts.

Activity Status as of July 1 2017:

No activity during this reporting period

Activity Status as of January 1 2018:

Our USGS partners have completed their CE-QUAL2 model for Madison Lake which will be integrated with the watershed modeling still to be completed by SCWRS. The completed report is located online at

https://pubs.usgs.gov/sir/2017/5056/sir20175056.pdf. The USGS' model is to be further refined based on Cyanobacterial response experiments to be conducted by USGS in the coming year.

Activity Status as of July 1 2018:

No activity during this reporting period

Activity Status as of January 1, 2019:

USGS compiled the available data for Madison Lake from the 2016 open-water season. Sentinel Lakes program water quality data for Madison Lake were available for nine dates in 2016. Available data included nutrients, algal biomass estimates, phytoplankton group biomass distribution, and temperature and dissolved oxygen profile data. In addition to lake water quality data, inflow water quality data were available for up to five dates for the two tributaries to Madison Lake.

Data were used to develop a draft calibration of the Madison Lake CE-QUAL W2 model for the ice-free season of 2016. The new version of the model increased the number of algal groups from four to five by distributing cyanophyte biomass into two sub-groups: non-nitrogen fixing morpho-types and heterocyst-forming filamentous morpho-types. Each algal group had its own specific nutrient-dependent growth physiology incorporated into the model calibration.

The resulting draft water quality model for 2016 had water balance and heat budget calibrations within acceptable ranges as defined in a previous publication of the model (linked in Jan. 1, 2018 update). Predicted component temperature and dissolved oxygen profiles from the model compared favorably to actual field data from the nine sampled dates in 2016. Predicted total algal biomass estimates from the model, based on the cumulative output from the five modeled algal groups, were also in general agreement with the chlorophyll *a* data from the lake water quality dataset. Nutrient calibrations were acceptable for phosphorus and some species of nitrogen, but the draft model currently underestimates the observed total nitrogen in the lake. This may result from the failure of the model to account for nitrogen fixation in the water column.

Calibration efforts will continue into the next quarter with a final model result, with estimates of internal versus external loads, by the end of the project.

Final Report Summary:

This activity produced predictive models for external inputs (SWAT) which could then be fed into a mechanistic in-lake model (CE-QUAL W2) to provide a predictive framework for HABs in Minnesota lakes. Madison Lake was chosen as the pilot lake for this project due to data availability and the dynamic nature of its stratification (i.e., early mixing).

Watershed Modeling

A computer model of the Madison Lake watershed can help identify sources and transport of nonpoint-source (NP-S) pollutants (sediment, phosphorus, and nitrogen), thus informing management decisions on how to clean up these pollutants and reduce noxious algal blooms in Madison Lake. These NP-S pollutants commonly come from row-cropped fields of corn and soybeans because of fertilizer applications and tillage practices that leave fields without living cover for much of the year. The modeling program applied to the Madison Lake watershed is called the Soil and Water Assessment Tool, or SWAT for short. SWAT was developed by the U.S. Department of Agriculture (USDA) Agricultural Research Service (ARS) to help understand and predict loads of NP-S pollutants from large river basins over long periods of time.

Input to the SWAT model relies on readily-available data from government-agency web sites. Topography was taken from LiDAR digital elevation models (DEMs) made available by the Minnesota Department of Natural Resources (MDNR) at a 3-m horizontal resolution. The DEM was hydro-modified to include drainage features (e.g., culverts) that correct for the false water impoundment by roads and other embankments. Soils data were taken from the SSURGO database made available by the USDA, which is the most spatially detailed soil data available. Land cover and crop types were taken from the USDA's crop data layer (CDL) datasets for 2014-18. This 5-year sequence of crops on the ground, at 30-m spatial resolution, provided an objective method for inferring typical crop areas, rotations, and locations in the watershed. Corn and soybeans accounted for nearly all the crop acreage, with minor areas of alfalfa and small grains (Figure 29). Representative amounts of inorganic fertilizer were added to all crops at the time of planting. Conservation tillage was assumed for all cropland, consisting of fall chisel plowing followed by spring disking or field cultivation. Weather data (daily precipitation and temperature) were taken from six weather stations (Amboy, Le Sueur, Mankato, St. Peter, Waseca, and Faribault) and averaged for the watershed centroid by simple inverse-distance weighting. The model was calibrated against measured daily outflow from Madison Lake for 2015-16, with Nash-Sutcliffe coefficient of efficiencies of 0.65 and 0.76 for 2015 and 2016, respectively, indicating very good model fits.

The resulting SWAT model identified which of the 197 modeled subbasins of the Madison Lake watershed were hot-spots of high sediment and nutrient yields (Figure 30). A *yield* is a mass per unit area of a selected land unit, per unit time, for example, tons per hectare per year (t/ha/yr) or kilograms per hectare per year (kg/ha/yr). In the Madison Lake watershed, high sediment, phosphorus, and nitrogen yields are consistent with each other and are driven primarily by sources, namely, the location of corn and soybean fields. Wetland, forest, and grasslands produce minimal yields of these NP-S pollutants. The cropland hot-spots of high yields are areas to target for remediation by alternative farming practices that reduce soil erosion and nutrient loss. The next steps would be to simulate possible remediation scenarios to see which ones will most efficiently reduce these pollutants while increasing landscape biodiversity and habitat, without placing undue burden on the farmers who are stewards of the land.

A fact-sheet, further detailing the SWAT portion of this Activity is included as a Supplemental File to this Report ("Construction and Calibration of a Computer Model of the Madison Lake Watershed").

In-lake modeling

A previously developed CE-QUAL-W2 model for Madison Lake, Minnesota, simulated the algal community dynamics, water quality, and fish habitat suitability of Madison Lake under recent (2014) meteorological conditions. Additionally, this earlier model simulated the complex interplay between external nutrient loading, internal nutrient loading from sediment release of phosphorus, and the organic matter decomposition of the algal biomass. However, the partitioning of cyanobacteria within the modeling framework was simplified to one group and did not account for how different cyanobacteria populations are affected by light conditions, the usage of nitrogen, temperature growth ranges, and differences in settling rates. To get a better handle on the proliferation of cyanobacteria in Madison Lake, the model required updates to at least partition the cyanobacteria into a group that fixed nitrogen and a second, more buoyant cyanobacteria group, that did not independently fix nitrogen.

To address the shortcomings of simulating cyanobacteria in the earlier model, the U.S. Geological Survey (USGS), in cooperation with the St. Croix Watershed Research Station (Science Museum of Minnesota), updated the Madison Lake CE-QUAL-W2 model to better characterize cyanobacteria into two groups. In addition to updating the cyanobacteria group differentiation, the entire portion of the model that handles the simulation of algal community dynamics was updated while preserving the model's predictive capabilities for nutrients, water temperature, and dissolved oxygen. The calibration and validation of the model was done under recent meteorological conditions with large and persistent cyanobacteria blooms (2014 and 2016). Overall, the model simulations predicted the persistently large total phosphorus concentrations in Madison Lake's hypolimnion, key

differences in nutrient concentrations between the two years, and cyanobacteria bloom persistence. As a product of this Activity, USGS produced a report which is currently being finalized. The most current draft of this report is attached as a supplemental file (Smith and Kiesling, 2019, Updates to the Madison Lake (Minnesota) CE-QUAL-W2 Water Quality Model for Assessing Algal Community Dynamics).

V. DISSEMINATION:

Description: We will collaborate with the Minnesota Interagency Workgroup on Blue-Green Algae (MPCA, MDNR, MDH, MVMA) to update the agencies on our latest findings, coordinate research, response, and outreach efforts, and evaluate any emerging issues. The Workgroup currently meets twice each year.

In addition, we will distill results from this study into compelling, accessible, and readable stories that will be widely distributed through electronic communications channels. This will include feature articles focusing on specific research results; regular short blog posts about the methods, activities, and people behind the study, and a quarterly email newsletter, "*Field Notes*", with links to the articles and blog posts.

A final project report will document all findings for reference by state personnel, presentations at regional meetings will apprise stakeholders of our methods and results, and publications in peer-reviewed journals will inform the wider academic research community.

Status as of July 1, 2016:

No activity during this reporting period

Status as of January 1, 2017:

No activity during this reporting period

Status as of July 1 2017:

We co-hosted a workshop at the St. Anthony Falls Laboratory's St. Paul campus entitled, "Freshwater Cyanobacteria - Harmful Algal Blooms (HABs) in Minnesota: Past, Present, and Future", which invited researchers, agency personnel, and private consultants working on HABs in Minnesota. This all-day workshop was held on March 28th with representation from the University of MN-Twin Cities, University of MN-Duluth, University of MN Extension, MN DNR, MPCA, MN Department of Health, Minnesota Sea Grant, Minnesota Natural Resources Research Institute, USGS Minnesota Water Science Center, the Science Museum of Minnesota, and others.

We have produced two full-length "Field Notes" articles that have been disseminated via our website and social media and have received thousands of page views since being published. The first article, "Watching When, Where and Why Harmful Algae Happen in Minnesota Lakes," introduced the public to our project and highlighted the need for this research in Minnesota. The second article, "Five super powers of Cyanobacteria", described the evolutionary backdrop that allows Cyanobacteria to produce harmful blooms. These articles continue to be available online at: https://www.smm.org/scwrs/fieldnotes/watching-when-where-and-why-harmful-algae-happen-minnesota-lakes and https://www.smm.org/scwrs/fieldnotes/five-super-powers-cyanobacteria.

Finally, we continue to use social media, traditional media, and outreach events to reach the broadest possible audiences with our message about awareness, understanding, detection, and prevention of HABs. In the last year our work has attracted the attention of both radio and television outlets. A Minnesota Public Radio story included radio play and a web video:
http://www.mprnews.org/story/2016/05/24/water-toxic-algae-dogs-climate-change

and a field interview with Duluth Fox 21 News resulted in two evening news special reports:
http://www.fox21online.com/news/local-news/water-worries-investigating-toxic-water-on-minnesota-lakes/40394034
http://www.fox21online.com/news/local-news/Water-Worries-Researchers-Look-For-Answers/40413736
Public outreach has included a St. Croix Watershed Research Station "Friends" event in July 2017 where project leader Dr. Adam Heathcote and graduate student David Burge teamed to present a public talk on "Harmful Algal Blooms: When Good Algae Go Bad") and show attendees what good algae and HABs look like under the research microscope. A "Members Behind the Scenes" event held at the Science Museum of Minnesota gave over 500 museum guests and families a chance to learn about the superpowers of cyanobacteria and see them under the microscope.

Status as of January 1 2018:

We have continued to share our research highlights with the public via social media, including coverage of our coring trips to Trout Lake and Itasca State Park (Elk and Portage Lakes). Additionally, we have been featured on an MPR story regarding HABs in August of 2017 (https://www.mprnews.org/story/2017/08/17/researchers-search-for-clues-to-toxic-algae-blooms).

We are also co-sponsors and co-organizers of the upcoming second HABs workshop, to be held at the SAFL auditorium on March 29th. This workshop is open to the general public and hopes to attract researchers, agency personnel, and interested citizens to come learn about and share their own knowledge on HABs in Minnesota.

Status as of July 1 2018:

We continue to publicize the importance of this ENRTF-supported work through our social media channels. We co-organized and participated in the 2nd Minnesota HABs workshop on March 29th, 2018. Adam Heathcote served as the program moderator and also participated as part of an expert panel which took questions from the ~70 registrants who attended the meeting. On April 3rd, 2018, Adam Heathcote was one of two experts asked to spend an hour on MPR taking questions on Minnesota lakes, many of which revolved around the problems with HABs (https://www.mprnews.org/story/2018/04/03/water-month-state-of-minnesotas-lakes).

Status as of January 1, 2019:

We participated in an open forum for questions surrounding HABs in Minnesota at the 2018 Minnesota Water Resources Conference. Together with our partners at USGS (Richard Kiesling), and our collaborators at St. Anthony Falls Laboratory (Shahram Missaghi), we provided a brief background on the issues of HABs in Minnesota, highlighted the outreach efforts by the Science Museum and University of Minnesota Extension, and fielded questions from the audience. We hope to parlay this effort into a special session on HABs at the 2019 MWR Conference (application pending).

Final Report Summary:

In the process of completing this project we have created the Minnesota HABs Working Group, which is a collaboration between universities, agencies, and consulting firms working on HABs in Minnesota. With this group we hosted workshops in 2017 and 2018 that were open to the public and drew near capacity registrants in both years. We also participated in a Q&A panel on HABs at the Minnesota Water Resources Conference, which included Dr. Adam Heathcote (SCWRS), Dr. Sharahm Missaghi (University of Minnesota Extension), and Dr. Richard Kiesling (USGS). From these collaborations, we have developed an entire special session devoted to HABs which will be held at the 2019 Minnesota Water Resources Conference.

The research produced from this project has been featured prominently on traditional media including multiple interviews with MPR and local Minnesota TV affiliates. Additionally, we have produced a number of articles meant for the general public about our work on HABs which were featured on the Science Museum of Minnesota's website and have given public presentations intended for the general public. Major research results from this project were also presented at two separate meetings of the Association for the Sciences of Limnology and Oceanography in June of 2018 and February of 2019 using in-kind funding provided by the Science Museum of Minnesota. This is the largest meeting dedicated to aquatic science in the world and is held once a year.

VI. PROJECT BUDGET SUMMARY:

A. ENRTF Budget Overview:

Budget Category	$ Amount	Overview Explanation
Personnel:	$ 384,100	1 sediment geochemist at 8% FTE for 2 years ($21,600); 1 algal and diatom analyst at 50% FTE for 2.5 years ($122,400); 1 algal toxin specialist and data analyst at 50% FTE for 3.5 years ($128,400); 1 hydrologist/watershed modeler at 35% FTE for 2 years ($70,100); 1 field technician at 75% FTE for 2 years ($41,600)
Professional/Technical/Service Contracts:	$ 68,800	USGS CE-QUAL modeling of in-lake process and field monitoring of discharge over 3 years ($50,000); Fossil pigment analysis by specialized external lab ($18,800)
Equipment/Tools/Supplies:	$ 35,500	Dissolved oxygen and temperature recording probes ($24,000); Field supplies including sediment traps, sample bottles, vials & reagents ($11,500)
Capital Expenditures over $5,000:	$ 30,000	YSI Water-quality sonde ($20,000); ELISA microplate reader ($10,000);
Travel Expenses in MN:	$ 14,100	Field work: sediment core collection and twice-monthly lake sampling
Other: Analytical Services	$ 60,500	Lab analysis of water samples (N, P, DOC, DIC) and sediment cores: radiometric dating (Lead-210, Cesium-137); biogenic silica; loss-on-ignition, sediment phosphorus and metals
TOTAL ENRTF BUDGET:	**$593,000**	

Explanation of Use of Classified Staff: N/A

Explanation of Capital Expenditures Greater Than $5,000: A dedicated water-quality sonde with sensors for chlorophyll a and phycocyanin is required for the intensive (twice-monthly) lake monitoring as outlined under Activity 1. An ELISA microplate reader for analysis of cyanobacterial toxins as described under Activity 2.

Number of Full-time Equivalents (FTE) Directly Funded with this ENRTF Appropriation: 5.36

Number of Full-time Equivalents (FTE) Estimated to Be Funded through Contracts with this ENRTF Appropriation: N/A

B. Other Funds:

Source of Funds	$ Amount Proposed	$ Amount Spent	Use of Other Funds
Non-state			
Science Museum of Minnesota	$ 215,000	$ 215,000 as of Jul 1, 2019	Unrecovered support services (lab & equipment maintenance, infrastructure, project administration), 43% of direct costs
State			
DNR & MPCA (in-kind)	$ 105,000	$ 105,000 as of Jul 1, 2019	Support in collecting water and phytoplankton samples from Sentinel Lakes
TOTAL OTHER FUNDS:	**$ 320,000**	**$ 320,000**	

VII. PROJECT STRATEGY:

A. Project Partners: DNR/MPCA (Sentinel lakes monitoring)
U.S. Geological Survey (CE-QUAL-W2 modeling)

B. Project Impact and Long-term Strategy:

This project will provide a statewide assessment of whether the threat of HABs is increasing in Minnesota and, if so, it will help identify the factors most likely contributing to that change. There has been only limited, short-term monitoring of HABs in Minnesota lakes, and evidence for changes in bloom frequency and severity is largely anecdotal. The reconstruction of past algal abundance from sediment cores, as outlined in this study, will provide a solid historical context for the present-day condition of Minnesota lakes. While excess nutrients, particularly phosphorus have long been known to stimulate algal growth, there are other factors that may play an equally important role; these include changes in the thermal structure of lakes (duration and stability of stratification), surface water temperatures and length of the growing season, atmospheric deposition of reactive nitrogen, invasive species such as the common carp, and internal feedback from the growth and senescence of the algal blooms themselves.

This study will improve our ability to predict when HABs occur, when they produce toxins, and how long those toxins persist. Again, monitoring of algal blooms and their toxins has been largely discontinuous and non-systematic so that we have only limited information about the seasonality, abundance, and composition of HABs or their associated toxins in our lakes. Because algal blooms and toxin production are relatively short-term events, high-frequency, systematic monitoring and modeling of both algae and associated physico-chemical conditions is needed to understand the risk posed by HABs and the factors contributing to their development.

This project integrates an extensive package of watershed monitoring data, sediment analytical results, watershed modeling, and in-lake modeling in a way that will engender mechanistic understanding of how and why harmful algal blooms occur. A key benefit of the project is the transferability of the results. Models are inherently flexible in their application, and the lessons learned in calibrating the models to our study site can be passed along in fitting the models to other sites. Furthermore, the calibrated models can be run with possible future land use or climate data, thus giving tremendous predictive power to infer potential impacts on our lakes.

Finally, as a long-term strategy, this study will establish infrastructure and capacity to identify harmful algae and toxins within the state of Minnesota; our state agencies currently outsource much of this work. The research staff who will carry out this project already possess expertise in algal identification and ecology. The work carried out here will help hone those skills, particularly with cyanobacteria and other soft algae, which are taxonomically difficult and environmentally complex. We anticipate that this study will raise additional questions about HABS and that solutions to the problem will involve long-term research investment beyond that outlined here.

C. Funding History:

Funding Source and Use of Funds	Funding Timeframe	$ Amount
MPCA (Lake of the Woods nutrient mass-balance study)	January 2012 -- July 2016	$ 300,000
ENRTF (M.L. 2014, Chp. 226, Sec. 2, Subd. 03g; "Watershed-Scale Monitoring of Long-Term Best-Management Practice Effectiveness") to establish Center for Harmful Algae Research in Minnesota (CHARM lab)	July 2014 -- June 2017	$ 900,000
ENRTF (M.L. 2009, Chap 143, Sect 2, Subd 05c "Cooperative Habitat Research in Deep Lakes") MN DNR subcontract to SMM	July 2010 -- June 2013	$ 90,000

VIII. FEE TITLE ACQUISITION/CONSERVATION EASEMENT/RESTORATION REQUIREMENTS: N/A

IX. VISUAL COMPONENT or MAP(S): See attached figures

X. RESEARCH ADDENDUM: See attached Research Addendum

XI. REPORTING REQUIREMENTS:

Periodic work plan status update reports will be submitted no later than the end of the months of July 2016, January 2017, July 2017, January 2018, July 2018, and January 2019. A final report and associated products will be submitted between June 30 and August 16, 2019.

Figures

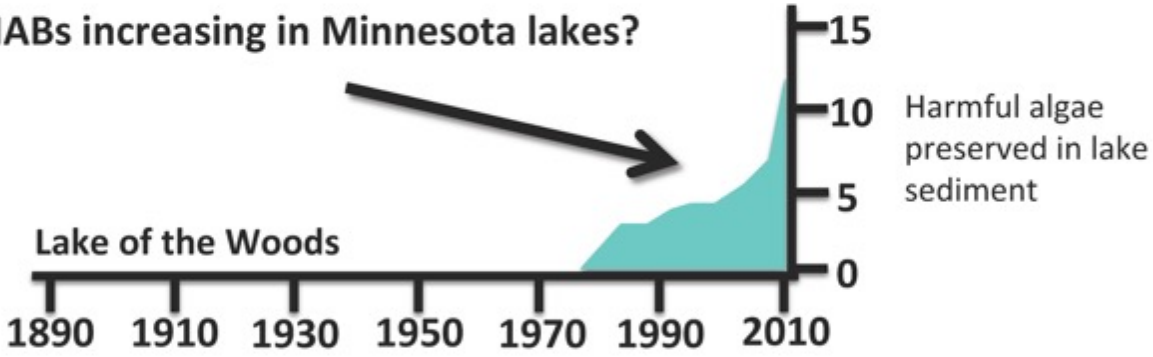
Environment and Natural Resources Trust Fund (ENRTF)
2016 Work Plan
Project Title: *Tracking and Preventing Harmful Algal Blooms*

What's going on with Harmful Algal Blooms (HABs) in Minnesota lakes?

- **What algae are present, when do they bloom, and are they harmful?**

HABs to the public:
a soupy green mess

- **HABs increasing in Minnesota lakes?**



- **Excess phosphorus causes HABs, but which is the bigger problem?**
  - ? **Watershed inputs**
  - ? **In-lake recycling**

The Ghost of Phosphorus Past

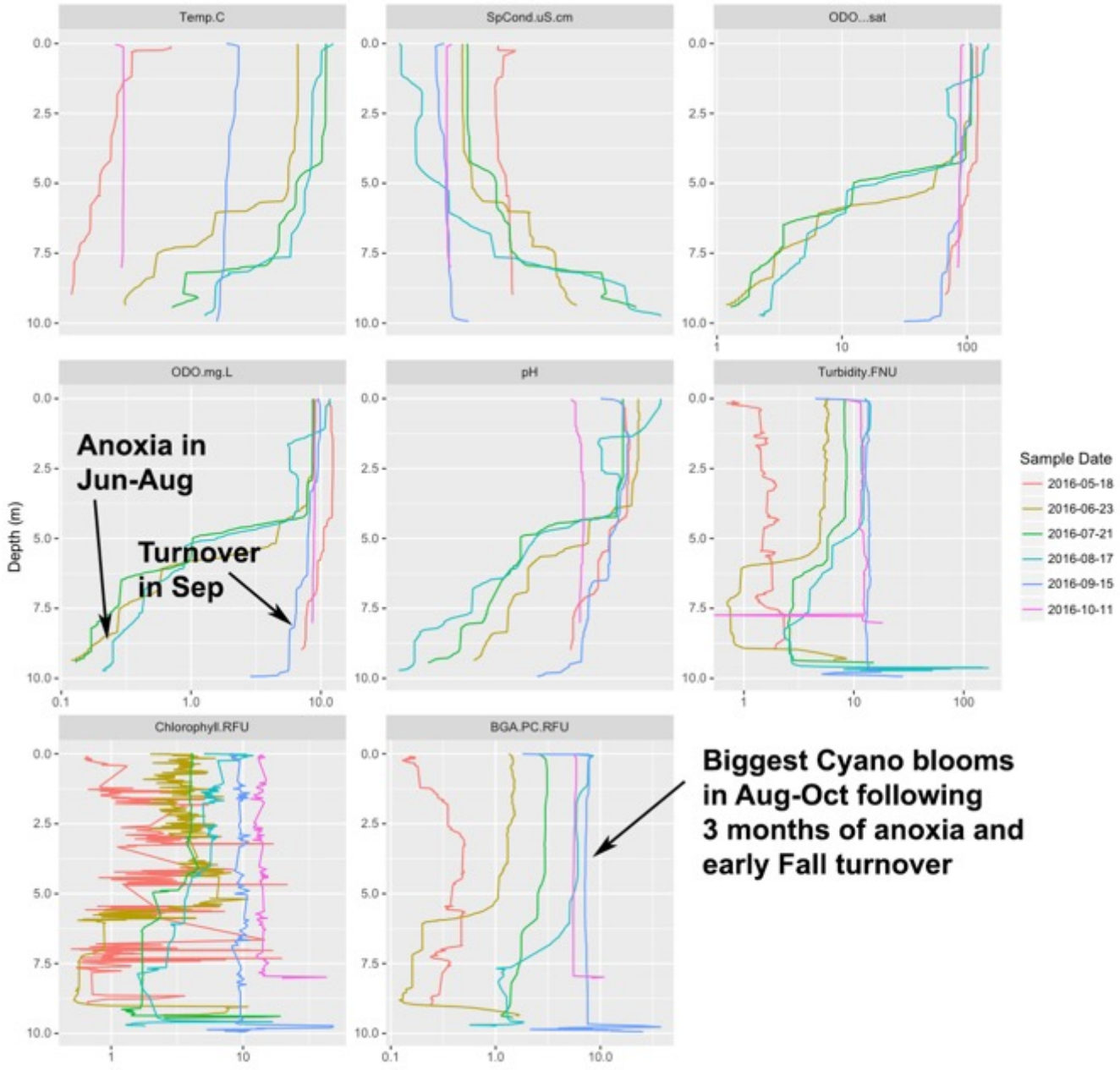


Figure 1. Example of YSI profiles collected from Madison Lake over the 2016 field season. Oxygen concentration profiles (ODO) show rapid anoxia occurring after stratification begins in June and persists until the lake turned over in September. Phycocyanin profiles (BGA.PC) show peak Cyanobacteria abundance following the turnover event, likely due to the release of sediment nutrients.

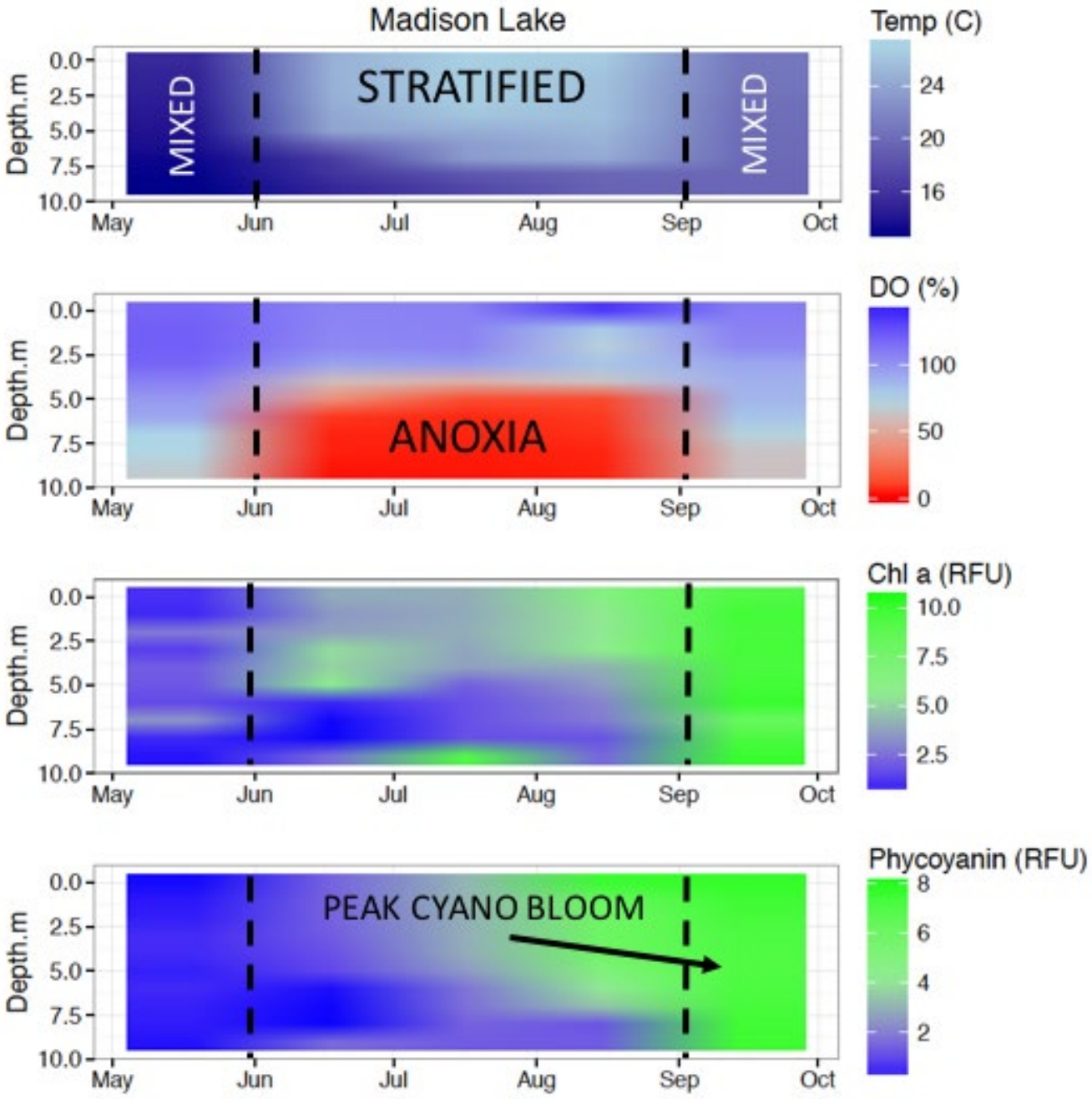


Figure 2. Interpolation of YSI profiles over the entire field season show the relationship between temperature stratification, oxygen and Cyanobacteria blooms. The largest blooms of the year in Madison Lake occurred followed a period of anoxic hypolimnetic conditions that allowed sediment P to accumulate in the bottom waters. This P was then released to the entire water column following a rapid turnover event in September.

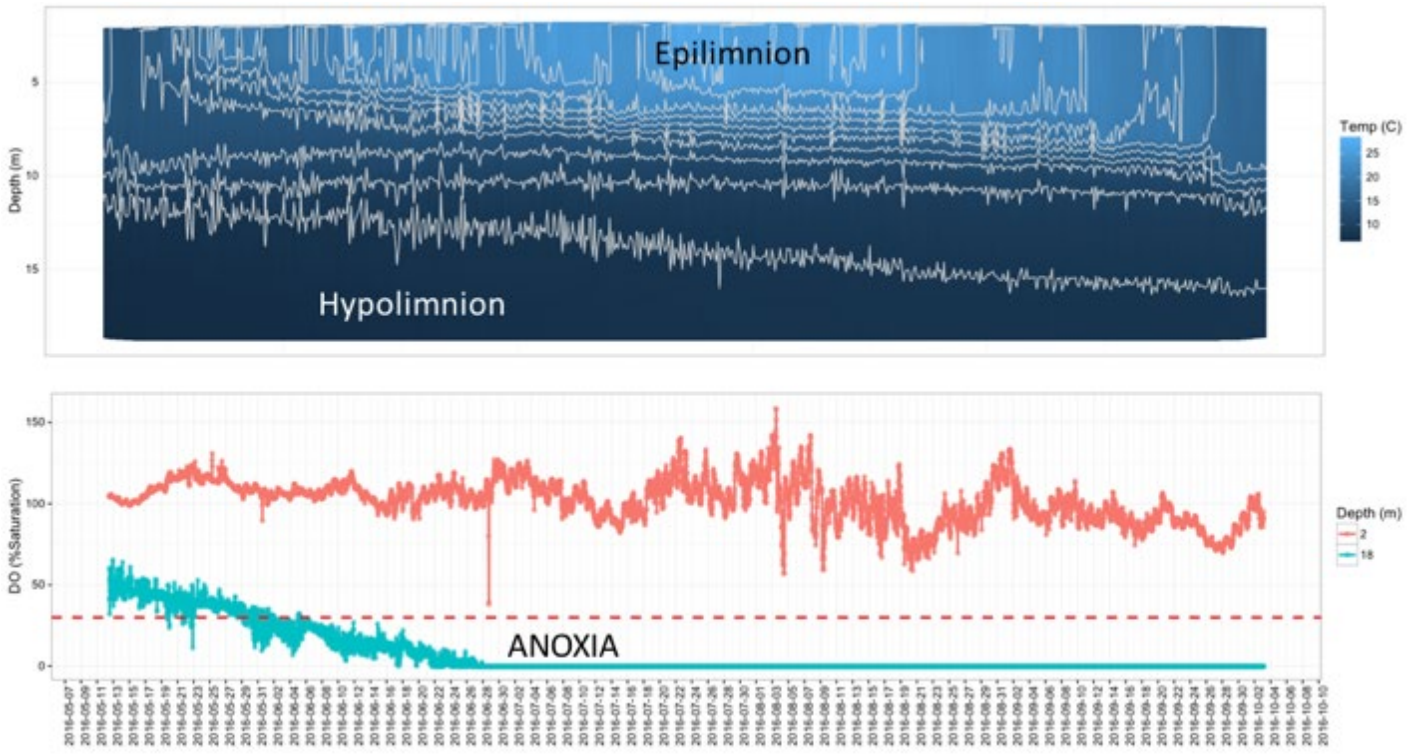


Figure 3. Temperature and dissolved oxygen data from the submerged buoy array at South Center Lake. These data show a typical dimictic lake that remained stratified throughout the entire ice-free season. Complete oxygen depletion (anoxia) occurred by the end of June and was maintained through November.

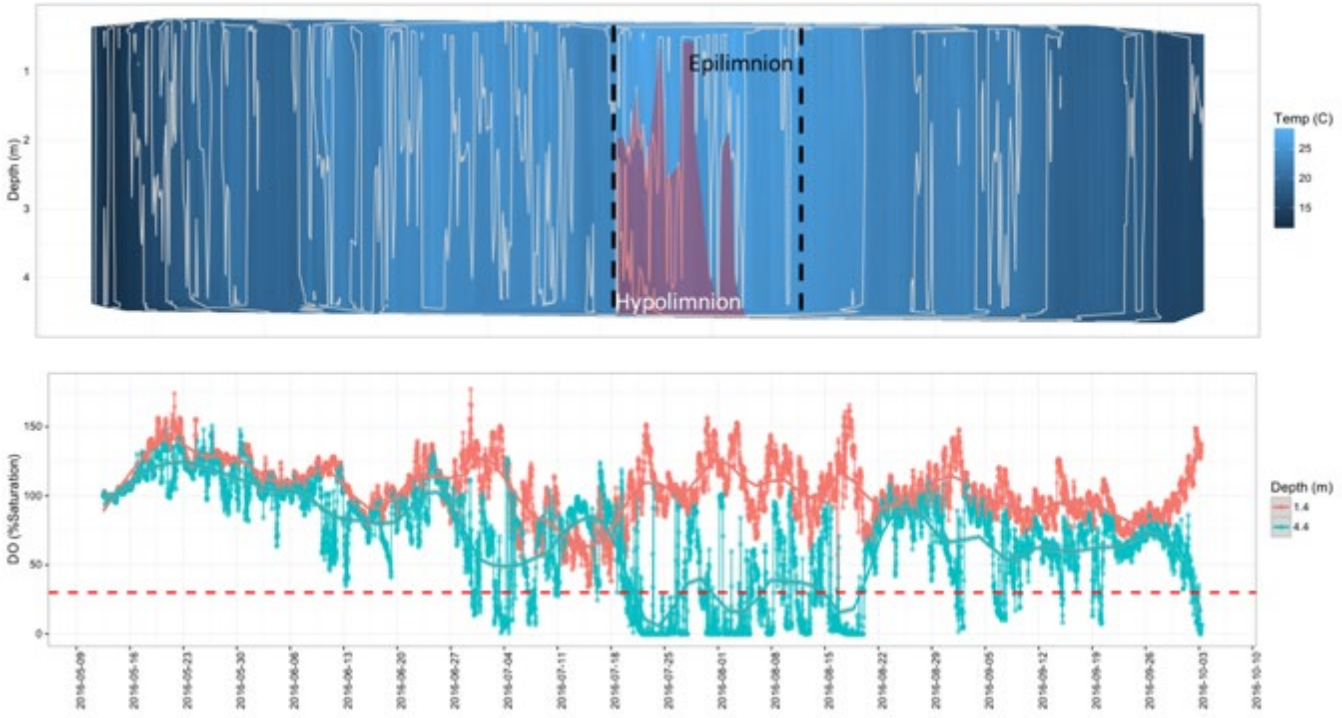


Figure 4. Temperature and dissolved oxygen data from Pearl Lake that show a surprisingly persistent stratification event occurring from mid-July through mid-August (red-shaded region in top panel). This event led to anoxic conditions and the likely release of sediment P into the water column following complete mixing at the end of August.

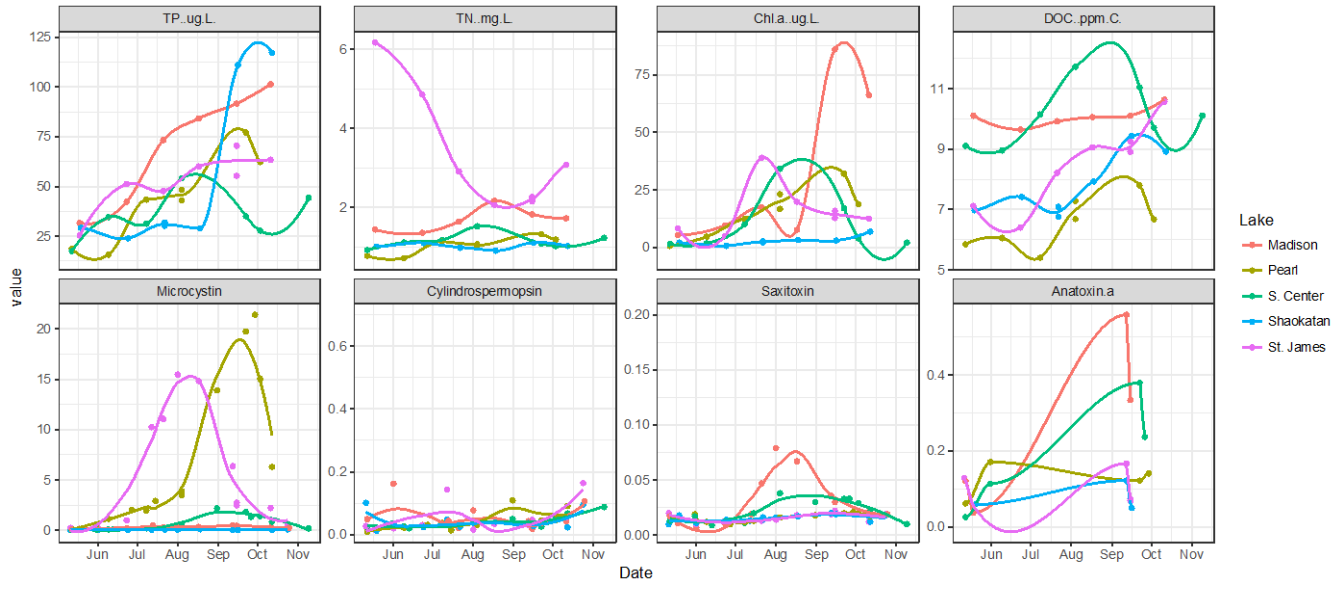


Figure 5. Water chemistry and cyanotoxin concentrations measured in the five intensively studied Minnesota lakes in 2016. Line colors correspond to each of the lakes (see legend) and dashed red line corresponds to the minimum safe drinking water concentration.

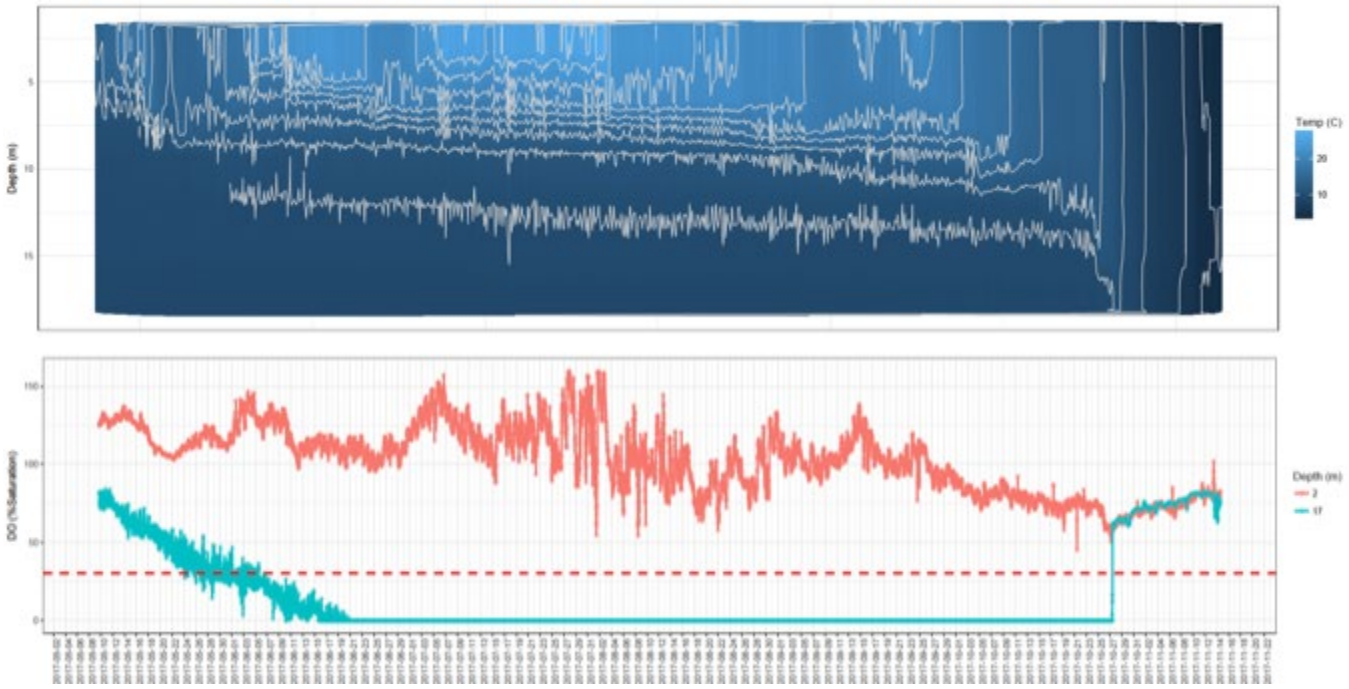


Figure 6. Isopleths of temperature over the 2017 field season in South Center Lake (top) and dissolved oxygen concentrations at the surface (orange) and bottom (teal) of the lake over the same time period. These data include measurements of temperature and oxygen collected every 30 minutes from the beginning of May until mid-November. We hope these data will illustrate the importance of stable stratification on reducing the impact of internal loading from sediment-bound nutrients. South Center Lake will serve as a strong contrast to the other polymictic lakes selected for this study which may be mixing sediment nutrients constantly, or periodically, throughout the open-water season.

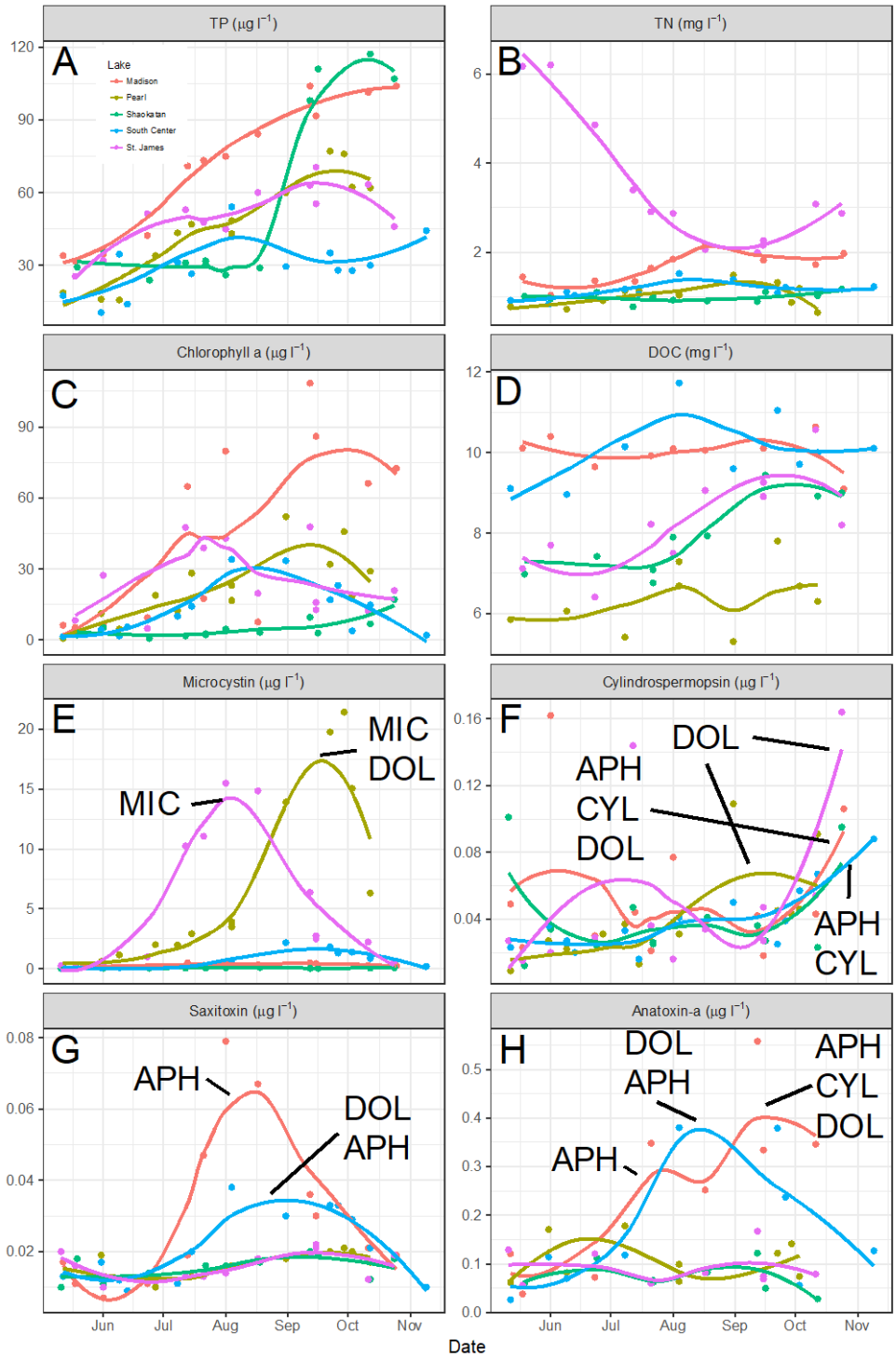


Figure 7. Water quality (A-D) and cyanotoxin (E-H) concentrations from the five lakes in this study. Colored lines correspond to different lakes (see legend). Three letter abbreviations correspond to the dominant Cyanobacteria genera in the sample (MIC: Microcystis, DOL: *Dolichospermum*, APH: *Aphanizomenon*, CYL: *Cylindrospermopsis)*

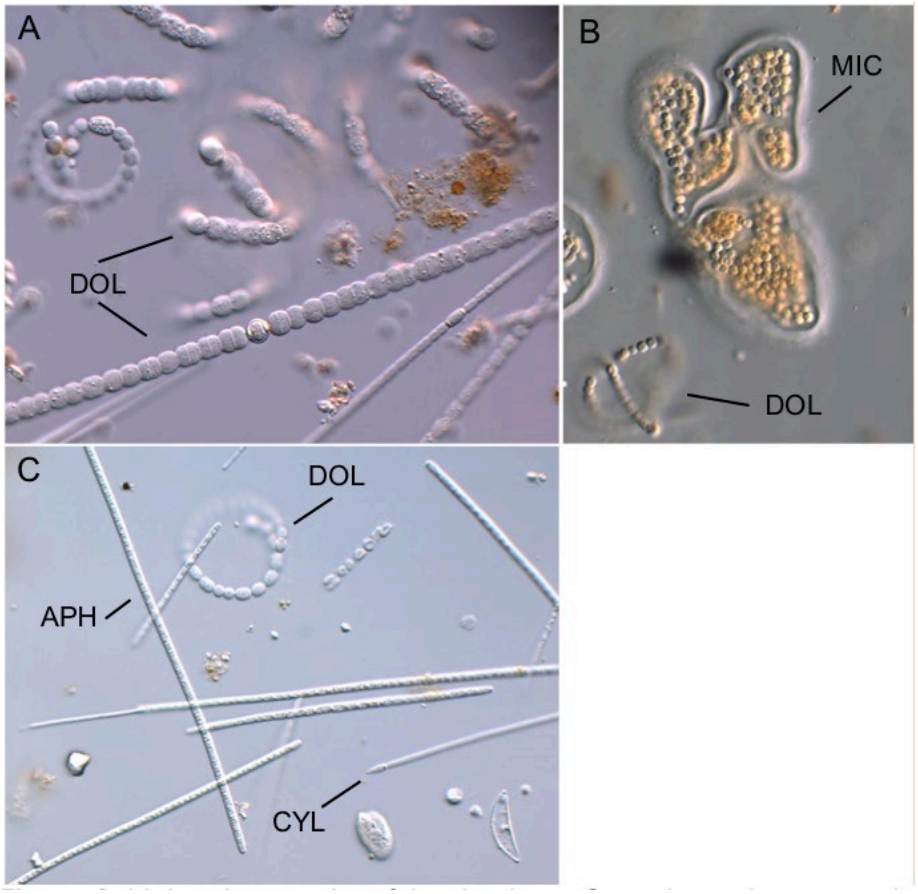


Figure 8. Light micrographs of the dominant Cyanobacteria community in selected samples. A) *Dolichospermum* and *Aphanizomenon*, South Center (8/4/2016), B) *Microcystis*, Pearl (9/29/16), C) Aphanizomenon, *Cylindrospermopsis*, *Dolichospermum*, Madison (9/21/16). MIC: *Microcystis*, DOL: *Dolichospermum*, APH: *Aphanizomenon*, CYL: *Cylindrospermopsis*

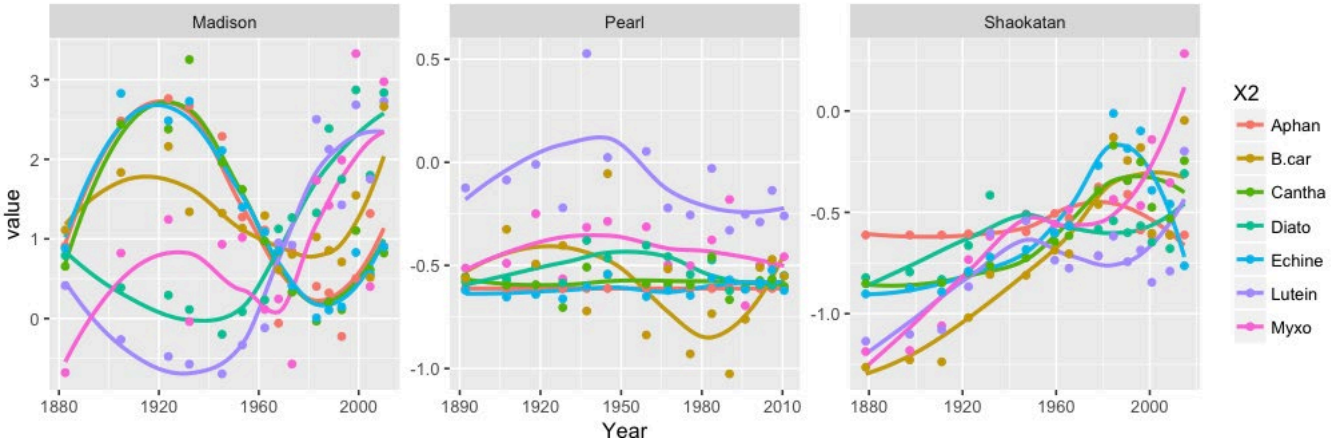


Figure 9. Algal pigment concentrations through time for three of the lakes in this study. Pigment values have been scaled to unit variance so they can be shown on the same graph. Madison and Shaokatan both show major increases in the pigment associated with colony-forming toxic Cyanobacteria (Myxo). Madison shows peak concentrations of many of the pigments associated with Cyanos (Aphan, Cantha, Echine) may have actually peaked in the 1920s and declined until recently (with the exception of Myxo). Total algae production (B.car) also follows this pattern. This may indicate that whereas algae blooms were present, and perhaps more intense, in the past; these blooms are currently made up of more toxic forms. Shaokatan shows increases in total algal production and most of the Cyanobacteria pigments that is consistent with an increasingly eutrophic system. Pearl shows less of a clear pattern in pigment concentrations, with most pigments remaining relatively flat over the entire time period or even decreasing.

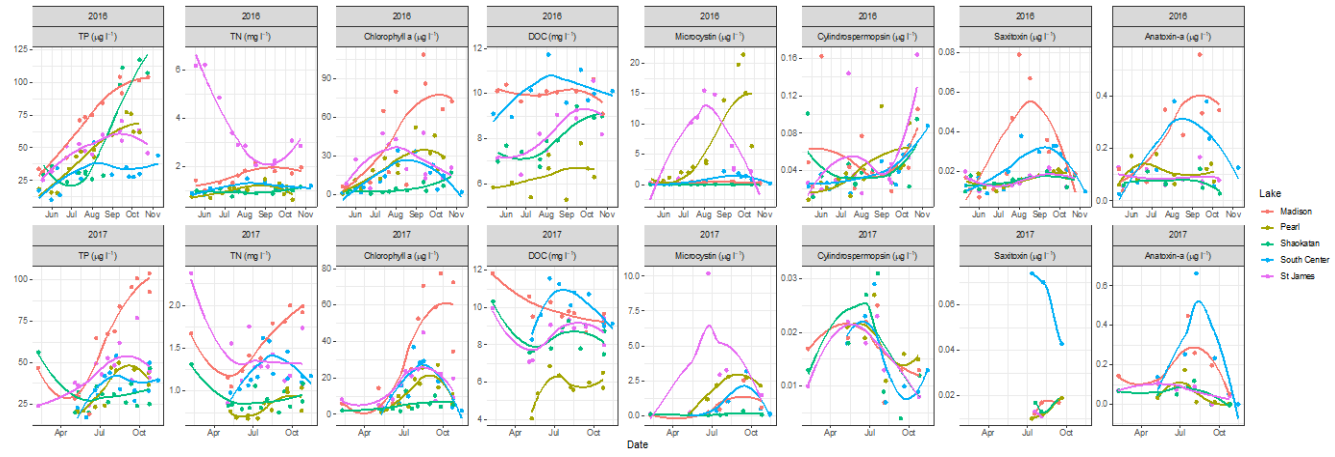


Figure 10. Major water chemistry and cyanotoxin concentrations for all five lakes in this study in 2016 and 2017. Algae density (as Chlorophyll *a*), microcystin, and anatoxin-a concentrations seemed to follow similar patterns across both years in the lake of this study, however, cylindrospermopsin and saxitoxin results were very different.

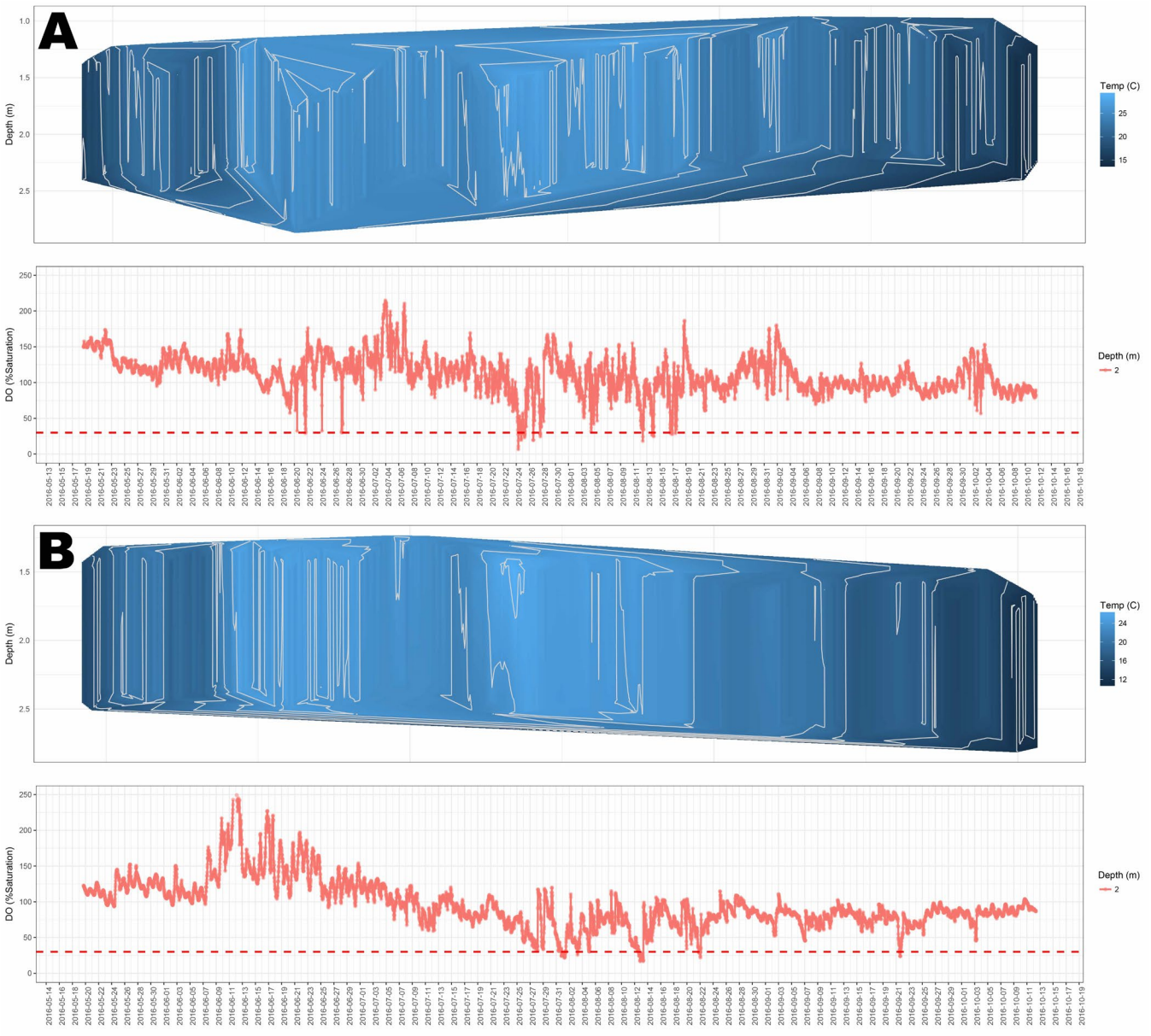


Figure 11. Temperature isopleths and bottom dissolved oxygen (DO) concentrations from buoy data collected in 2016 for Lake St. James (A) and Lake Shaokatan (B). Both lakes were mixed for almost the entire open-water season, with only very brief periods (~1-2 days) of oxygen depletion. Red dashed line represented hypoxic concentrations.

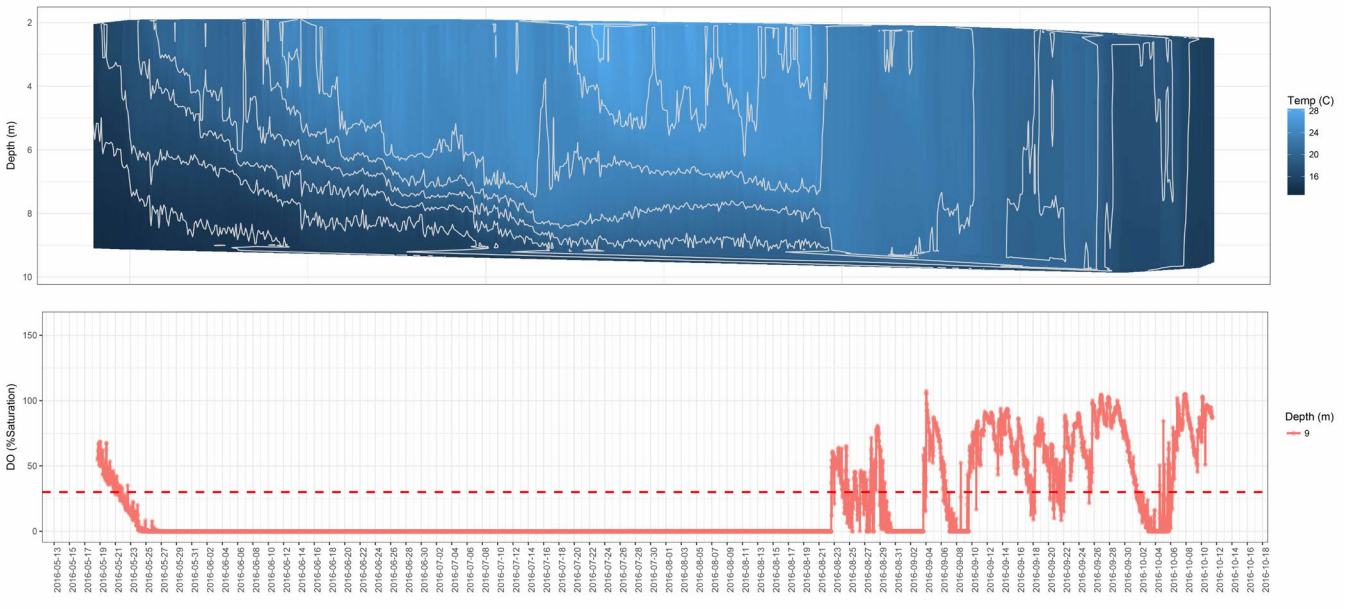


Figure 12. Temperature ispoleths and bottom dissolved oxygen concentrations for Madison Lake in 2016. Data show early stratification in May followed by prolonged anoxia in the hypolimnion until a mixing event in late August. The lake then alternated between stratified and mixed for the rest of the ice-free season. Red dashed line represents the hypoxic zone.

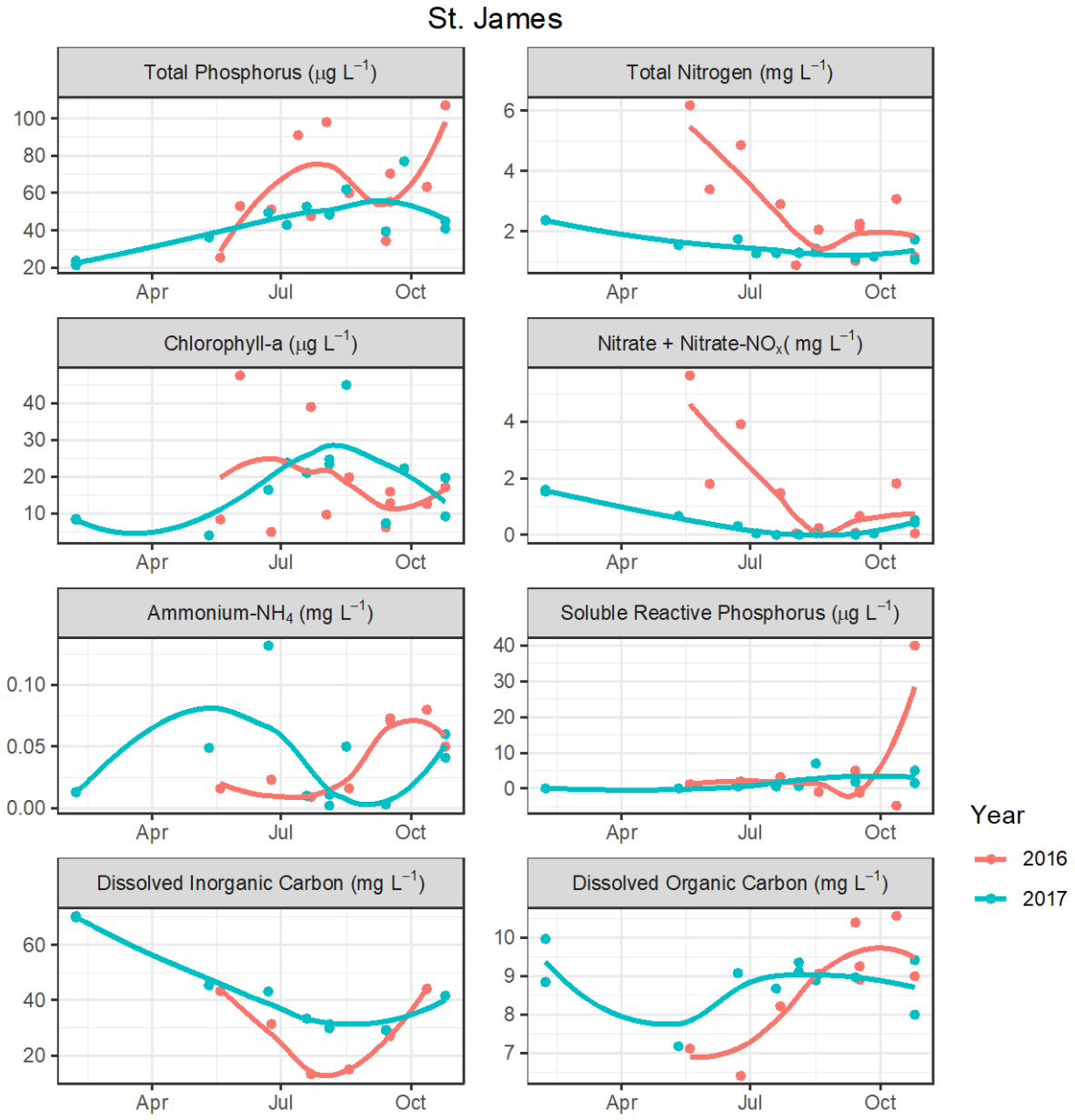


Figure 13. Water chemistry summary for all analytes collected on St. James Lake in 2016 (orange) and 2017 (turquoise). Lines represent locally weighted polynomial regression (LOWESS) smoothers of the data to show general trends.

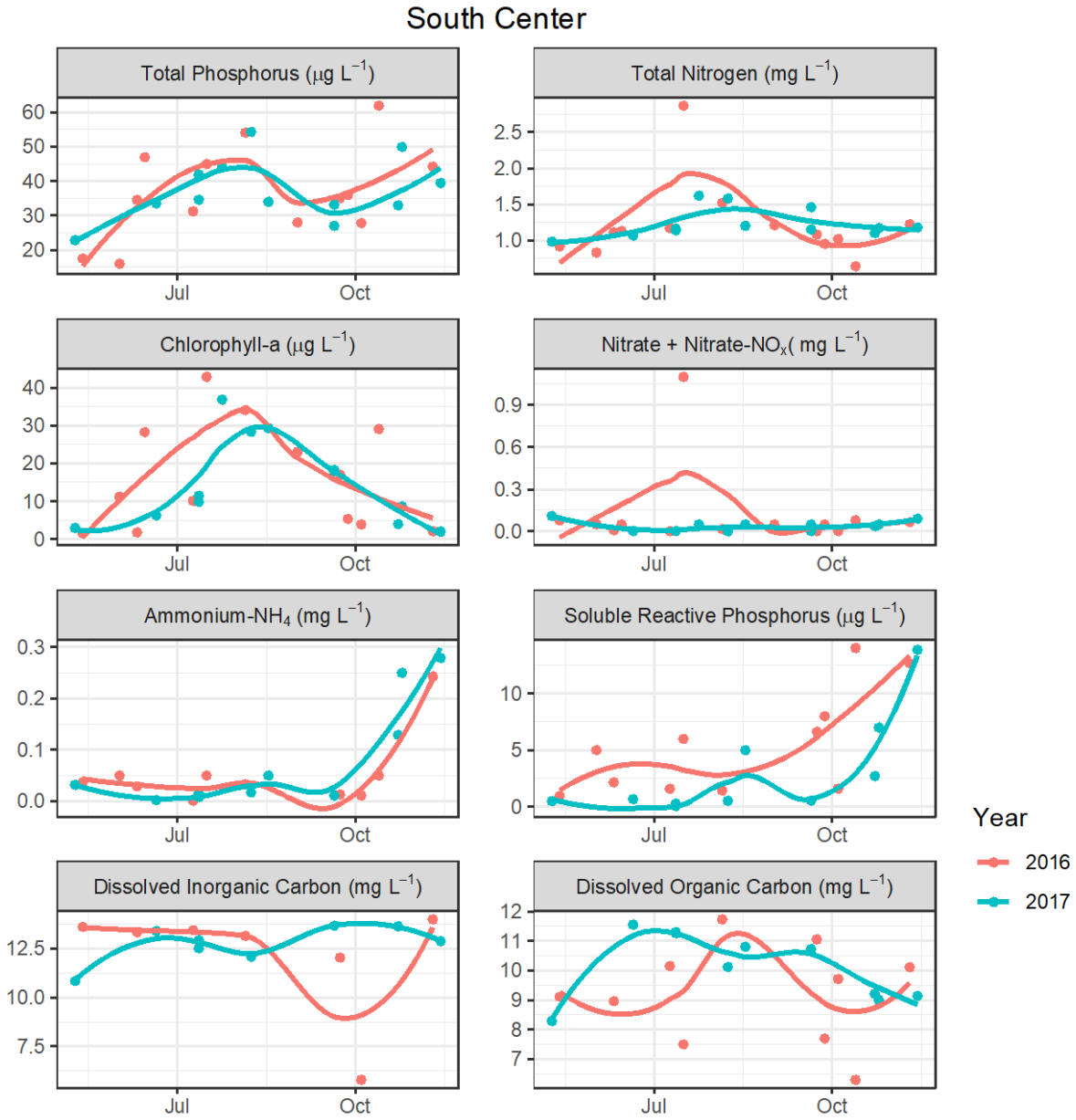


Figure 14. Water chemistry summary for all analytes collected on South Center Lake in 2016 (orange) and 2017 (turquoise). Lines represent locally weighted polynomial regression (LOWESS) smoothers of the data to show general trends.

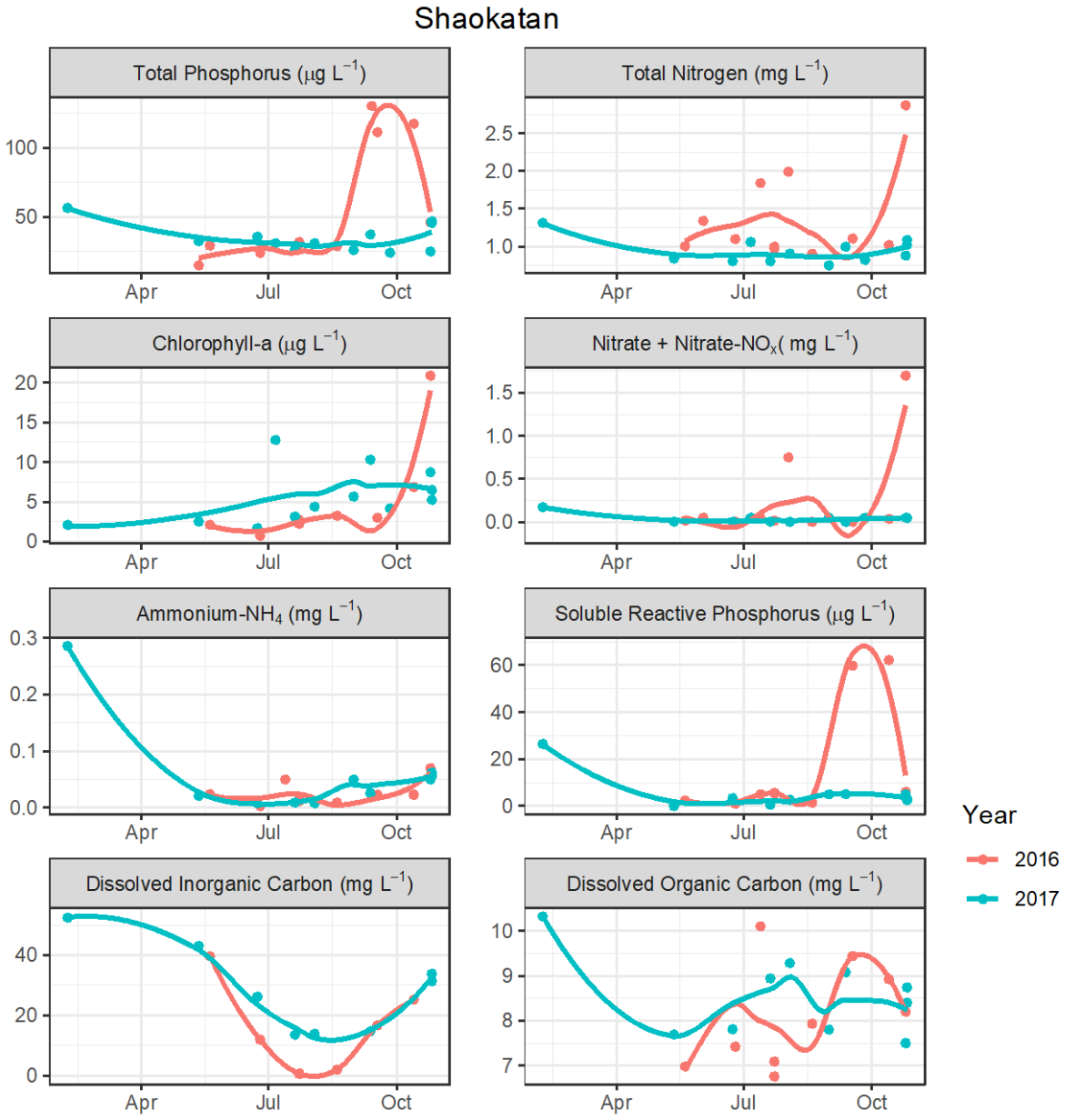


Figure 15. Water chemistry summary for all analytes collected on Lake Shaokatan in 2016 (orange) and 2017 (turquoise). Lines represent locally weighted polynomial regression (LOWESS) smoothers of the data to show general trends.

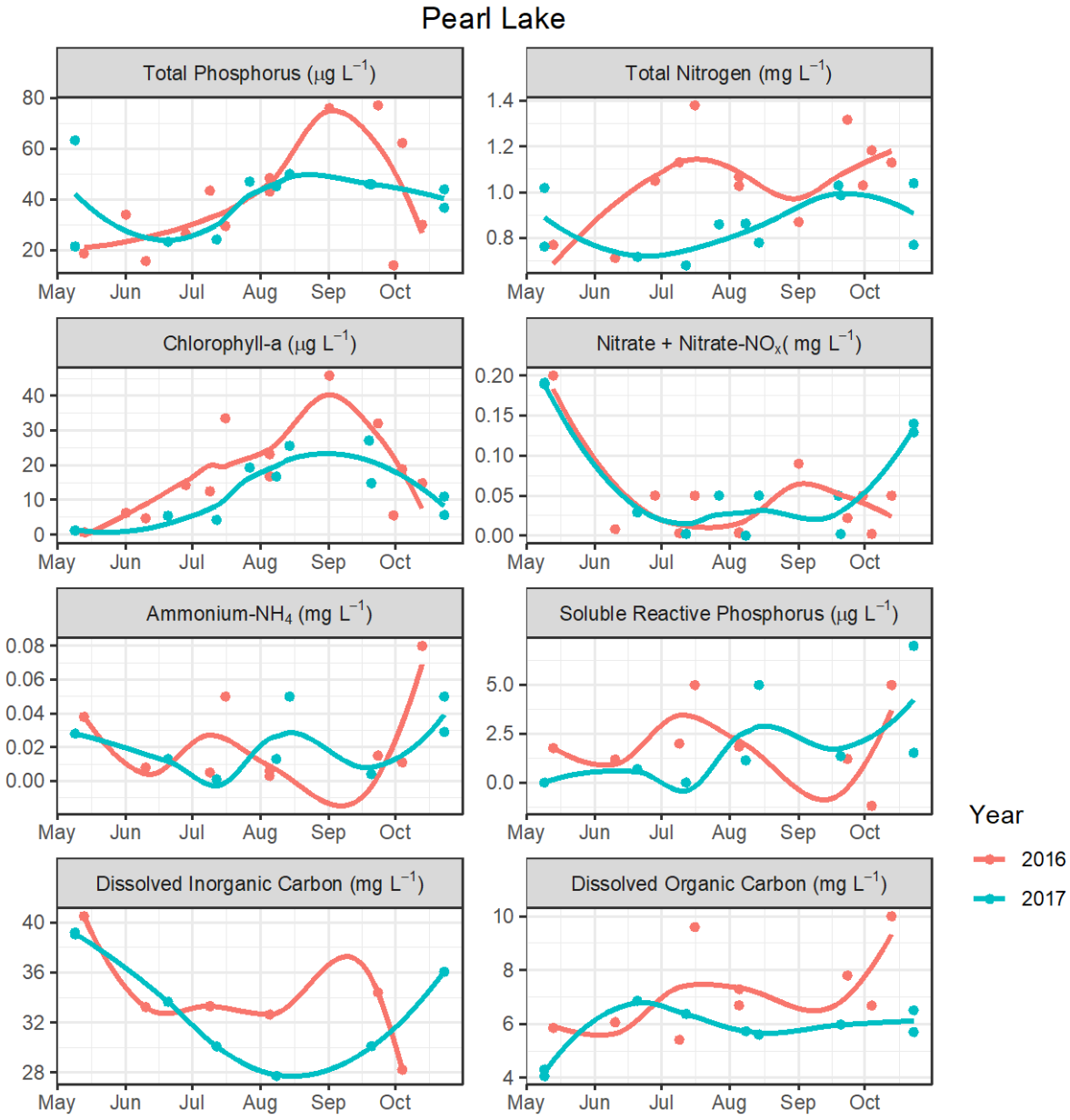


Figure 16. Water chemistry summary for all analytes collected on Pearl Lake in 2016 (orange) and 2017 (turquoise). Lines represent locally weighted polynomial regression (LOWESS) smoothers of the data to show general trends.

Figure 17. Water chemistry summary for all analytes collected on Madison Lake in 2016 (orange) and 2017 (turquoise). Lines represent locally weighted polynomial regression (LOWESS) smoothers of the data to show general trends.

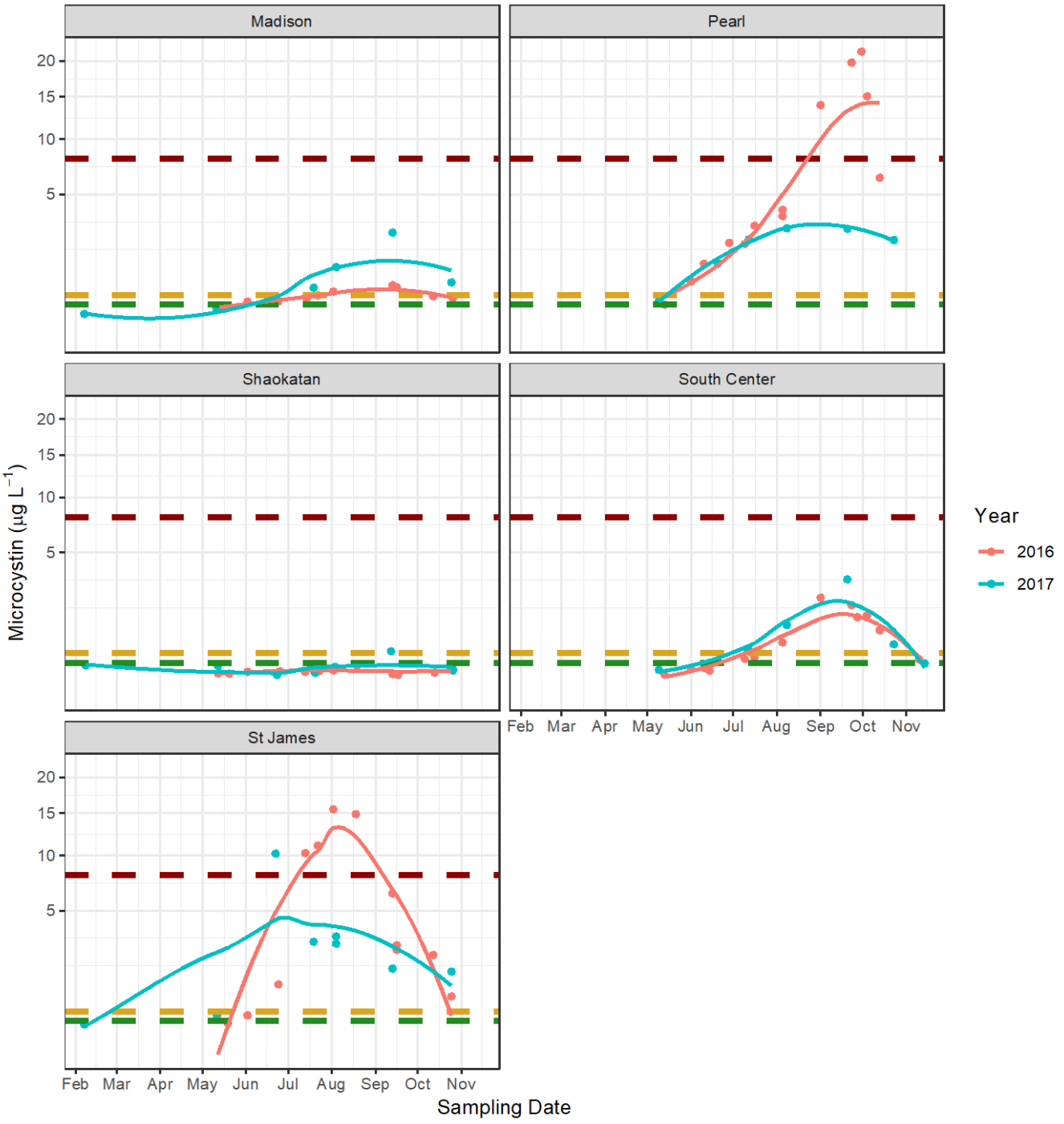


Figure 18. Microcystin concentrations in all five lakes in this study in 2016 (orange) and 2017 (turquoise). Green dashed line represents instrument detection limits (0.15 µg/L), yellow dashed line represents minimum safe drinking water standard (0.3 µg/L; EPA, MDH), and red dashed line represents recreational contact standard (8 µg/L; EPA).

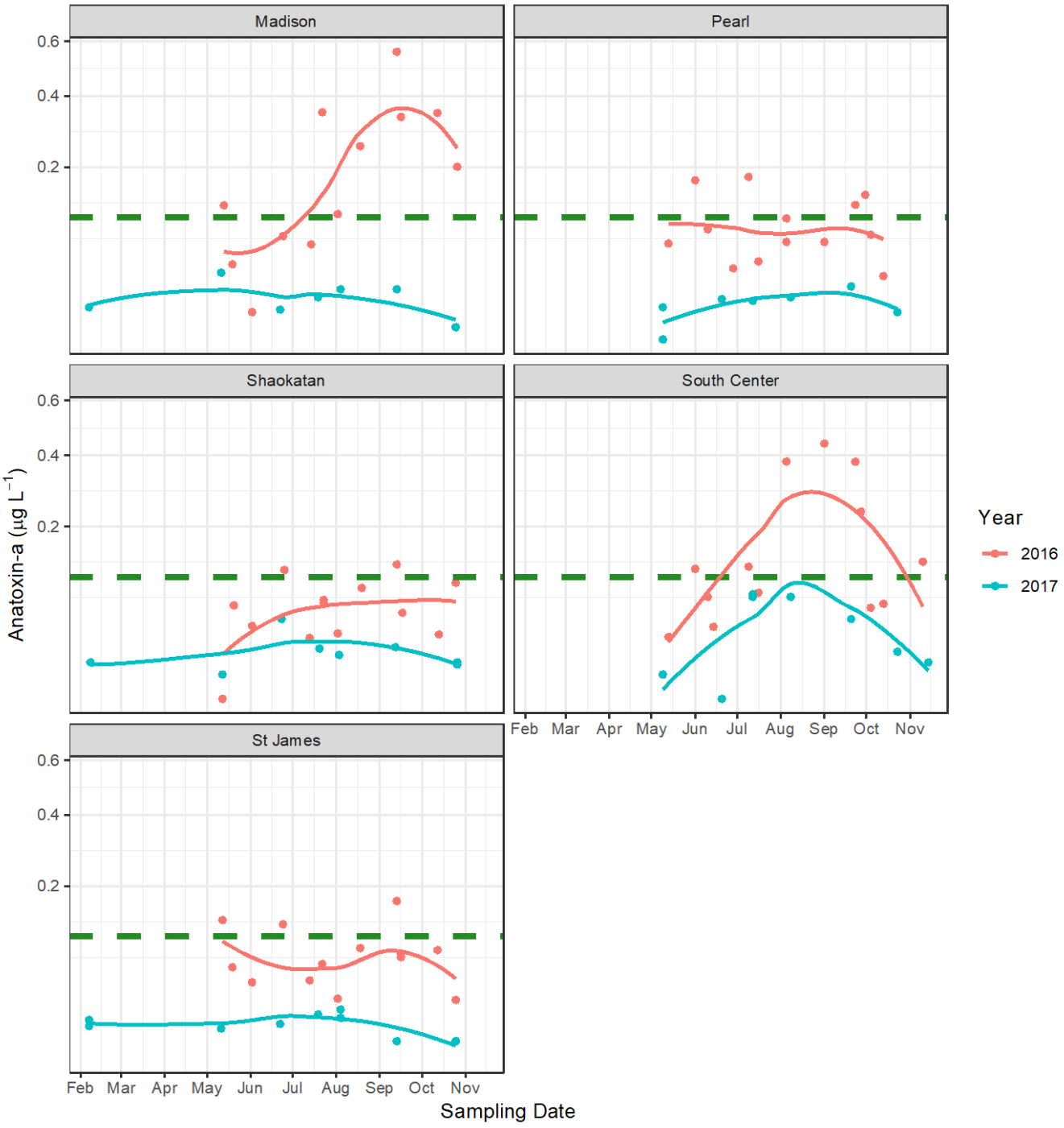


Figure 19. Anatoxin-a concentrations in all five lakes in this study in 2016 (orange) and 2017 (turquoise). Green dashed line represents instrument detection limit, which is the same as the MDH minimum drinking standard (0.1 µg/L).

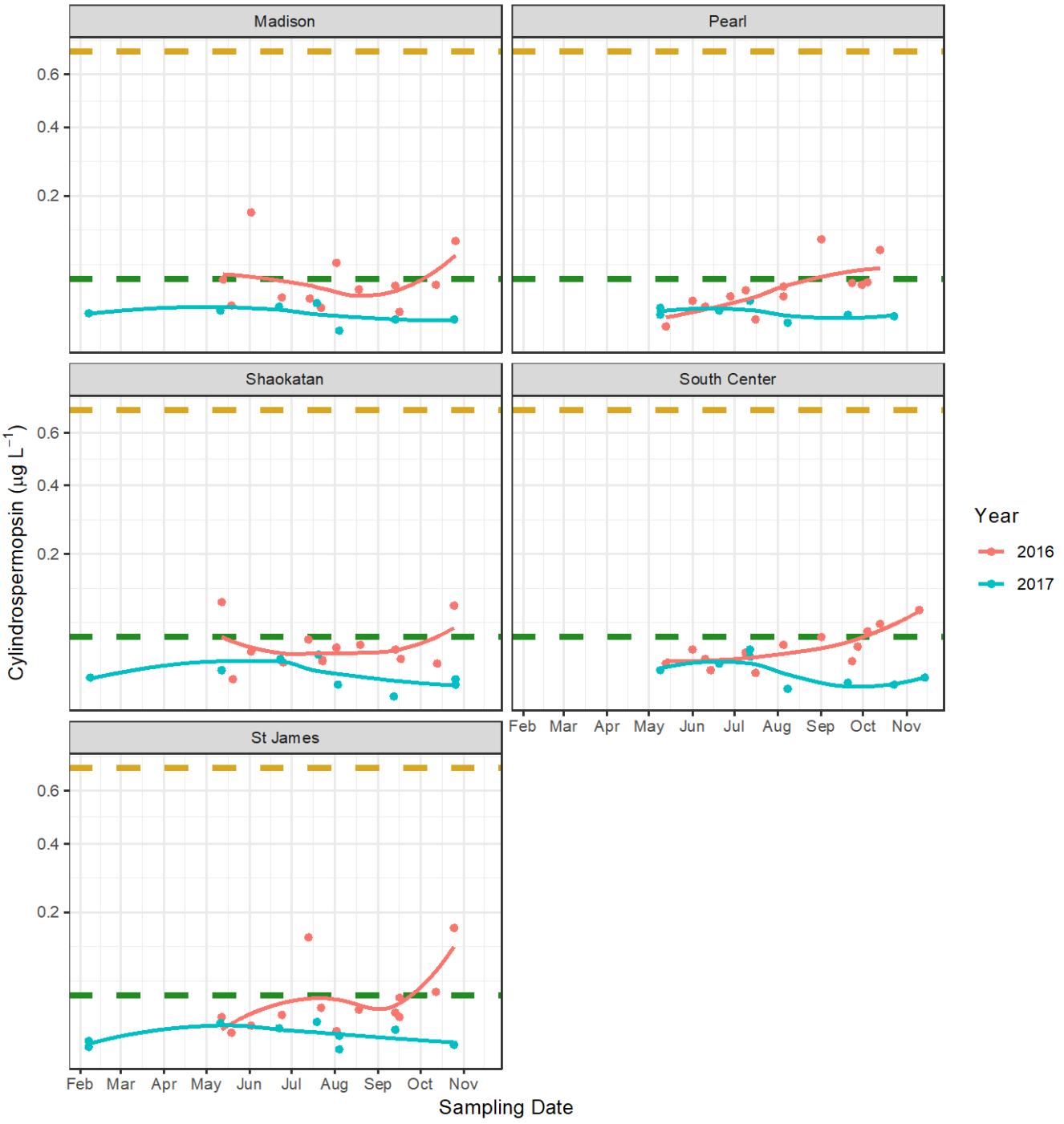


Figure 20. Cylindrospermopsin concentrations in all five lakes in this study in 2016 (orange) and 2017 (turquoise). Green dashed line represents instrument detection limits (0.05 µg/L) and yellow dashed line represents minimum safe drinking water standard (0.7 µg/L; EPA).

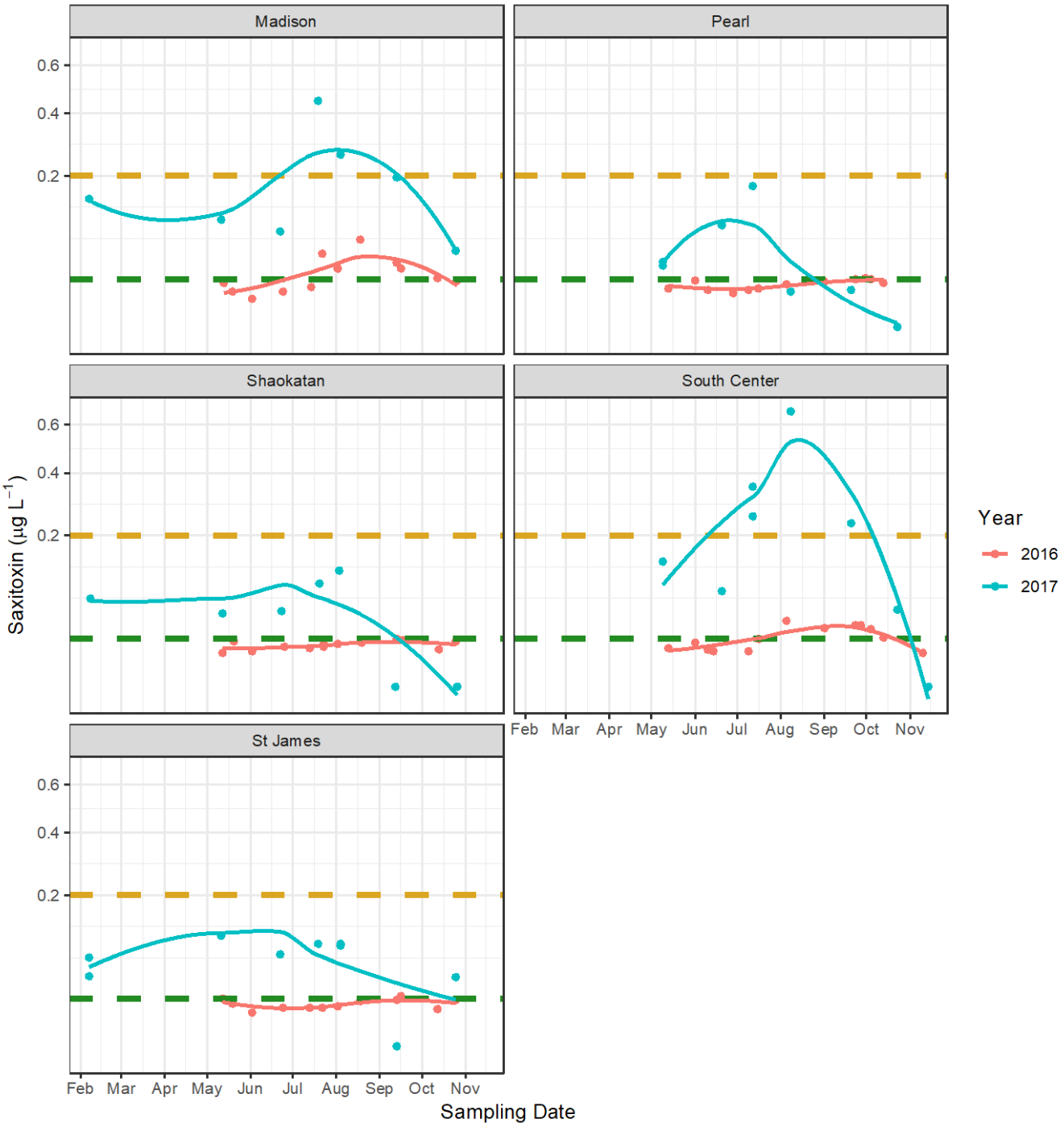


Figure 21. Saxitoxin concentrations in all five lakes in this study in 2016 (orange) and 2017 (turquoise). Green dashed line represents instrument detection limits (0.02 µg/L) and yellow dashed line represents minimum safe drinking water standard from the Ohio Department of Health (0.2 µg/l) due to a lack of any state or **federal** standards in Minnesota.

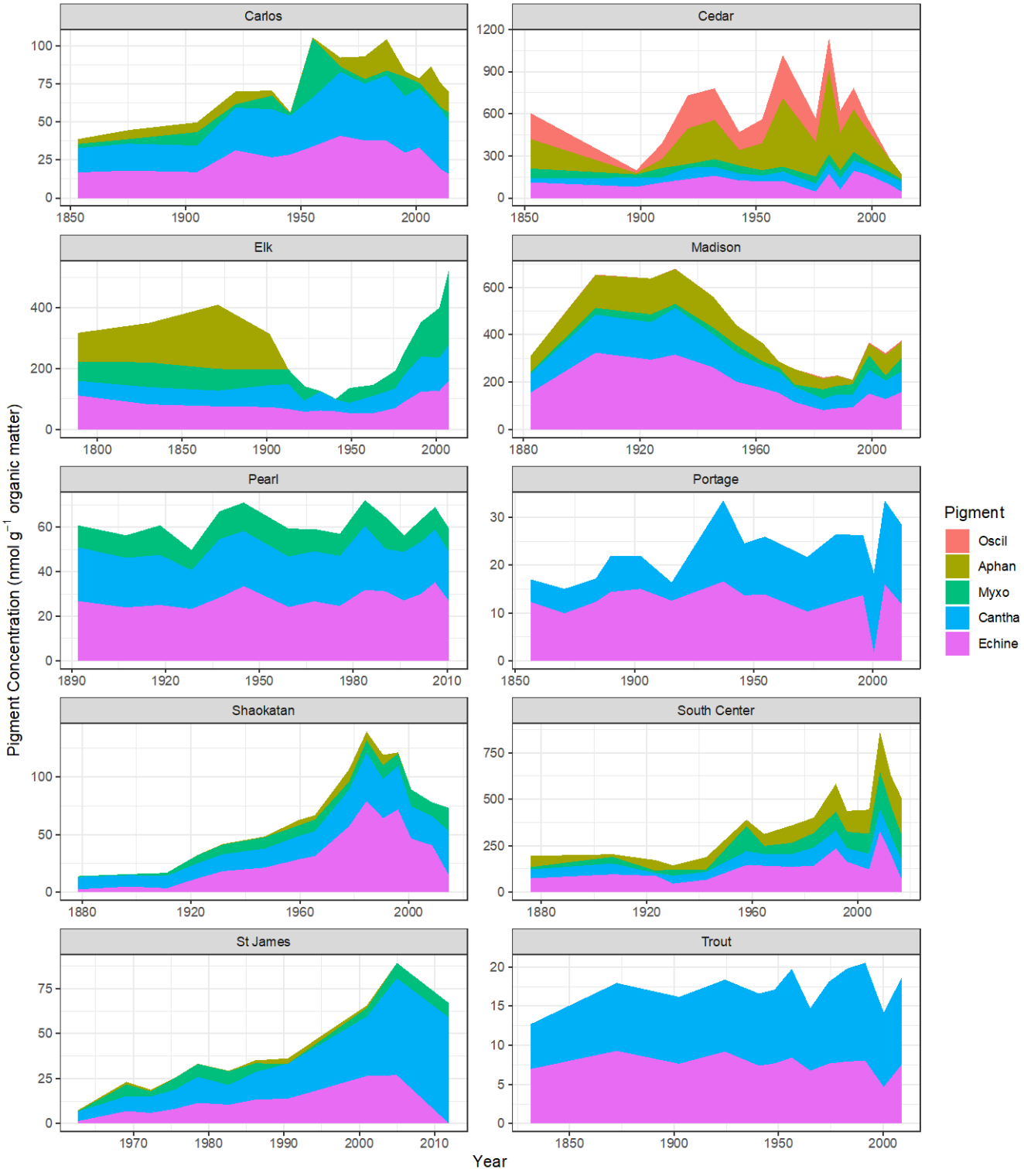


Figure 22. Fossil Cyanobacterial pigments from sediment cores in ten lakes of this study.

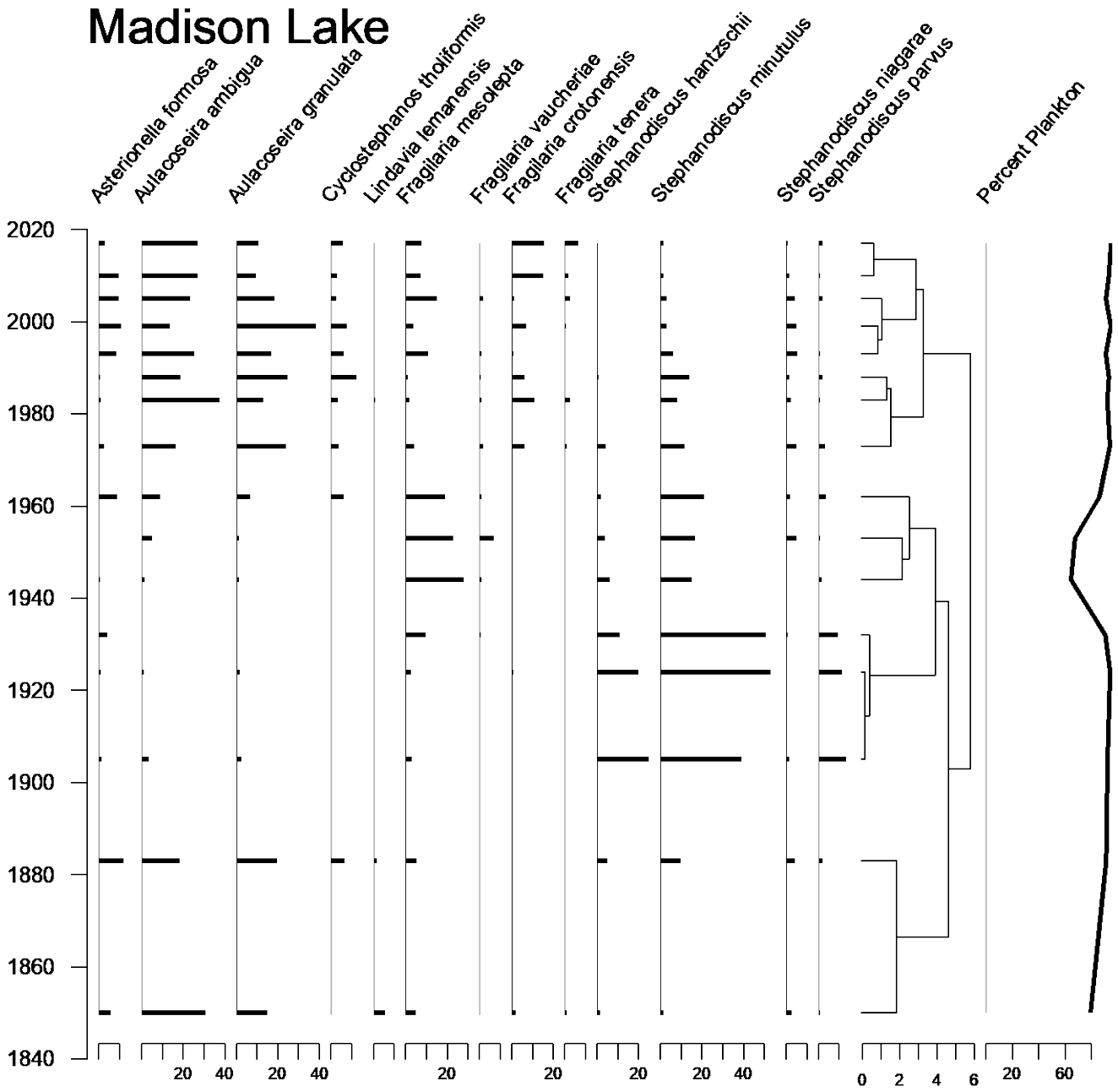


Figure 23. Stratigraphic plot of diatom abundances in Madison Lake. Dendrogram represents a constrained hierarchical clustering (CONISS) which separates distinct time periods based on the diatom flora.

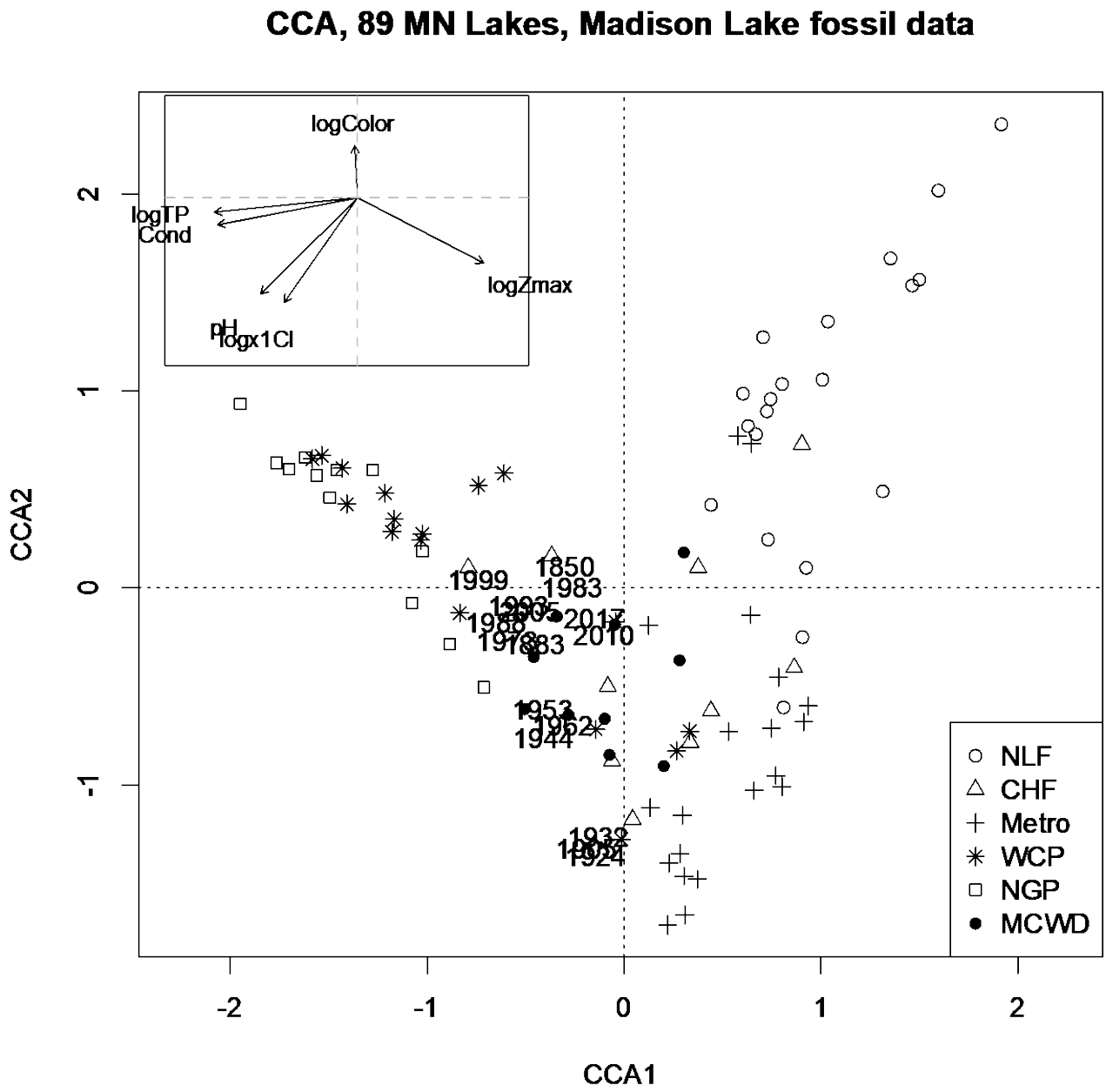


Figure 24. Constrained Coordinates Analysis (CCA) ordination of the 89-lake surface training set in Minnesota. This training set represents diatom populations in lakes all across the state and is organized along water quality axes show in the top-left inset. Diatom samples from Madison Lake are passively plotted (represented by the date of the section) onto the ordination to show how they move along these major water quality axes through time.

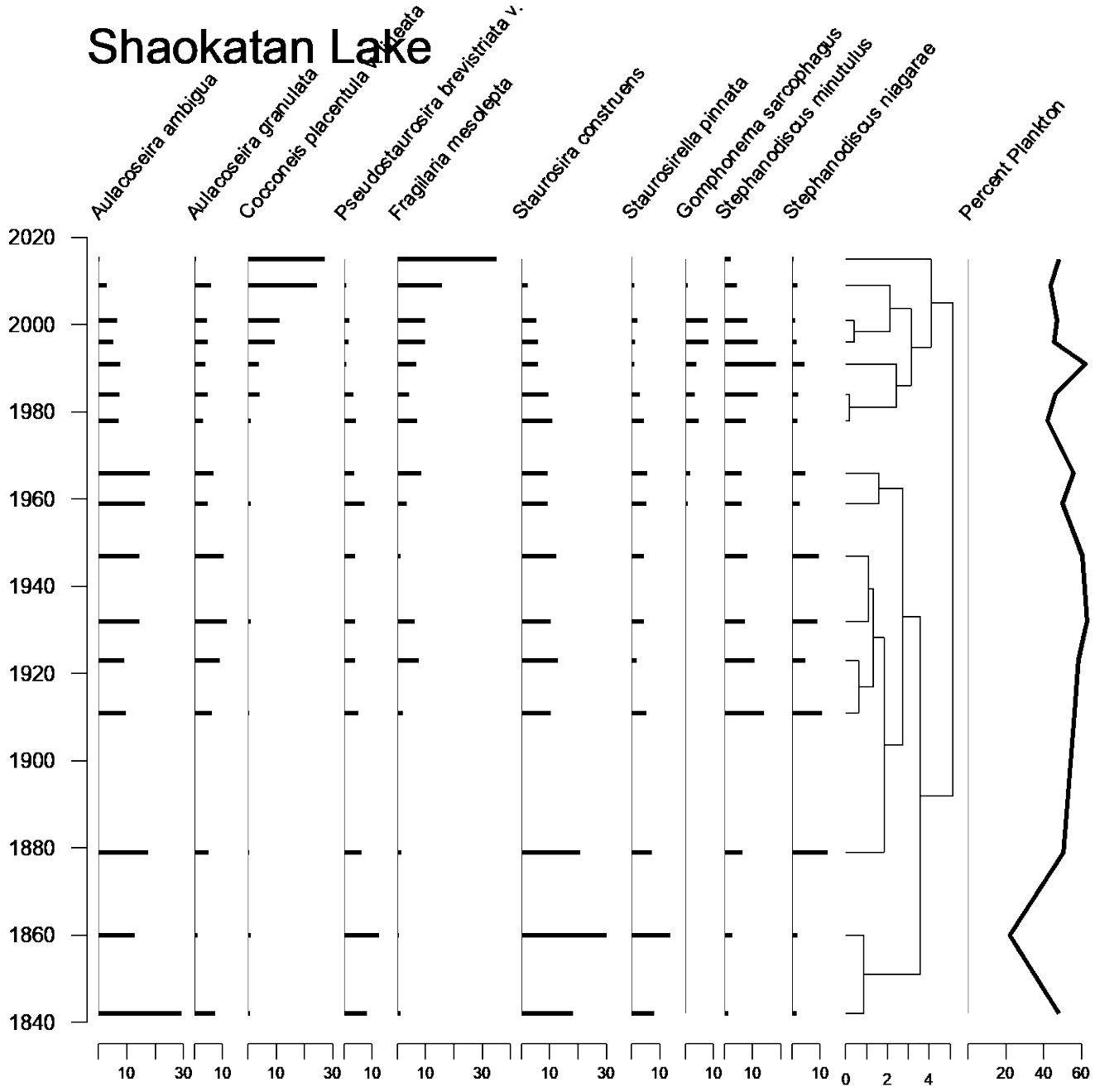


Figure 25. Stratigraphic plot of diatom **abundances** in Lake Shaokatan. Dendrogram represents a constrained hierarchical clustering (CONISS) which separates distinct time periods based on the diatom flora.

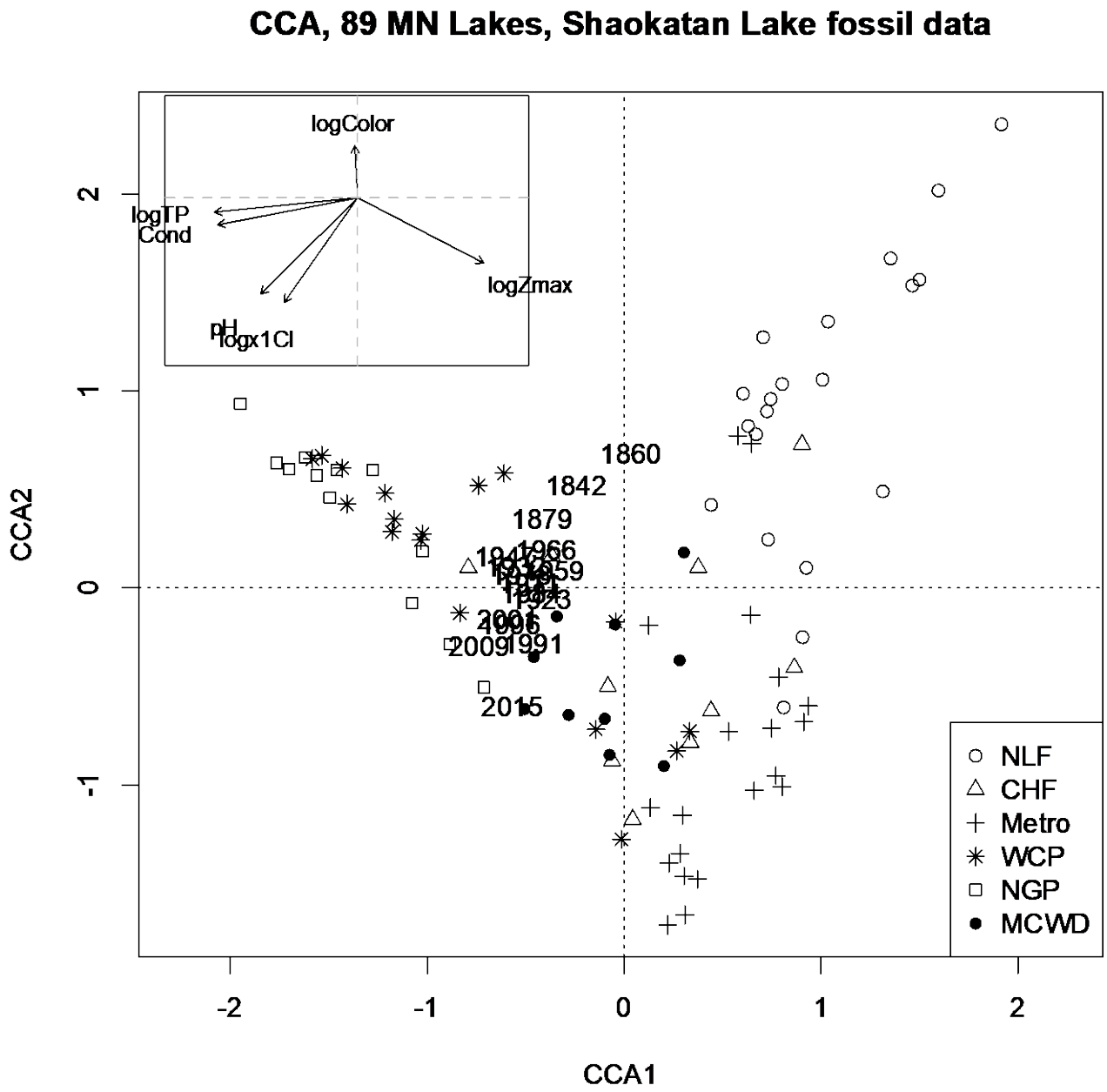


Figure 26. Constrained Coordinates Analysis (CCA) ordination of the 89-lake surface training set in Minnesota. This training set represents diatom populations in lakes all across the state and is organized **along** water quality axes show in the top-left inset. Diatom samples from Lake Shaokatan are passively plotted (represented by the date of the section) onto the ordination to show how they move along these major water quality axes through time.

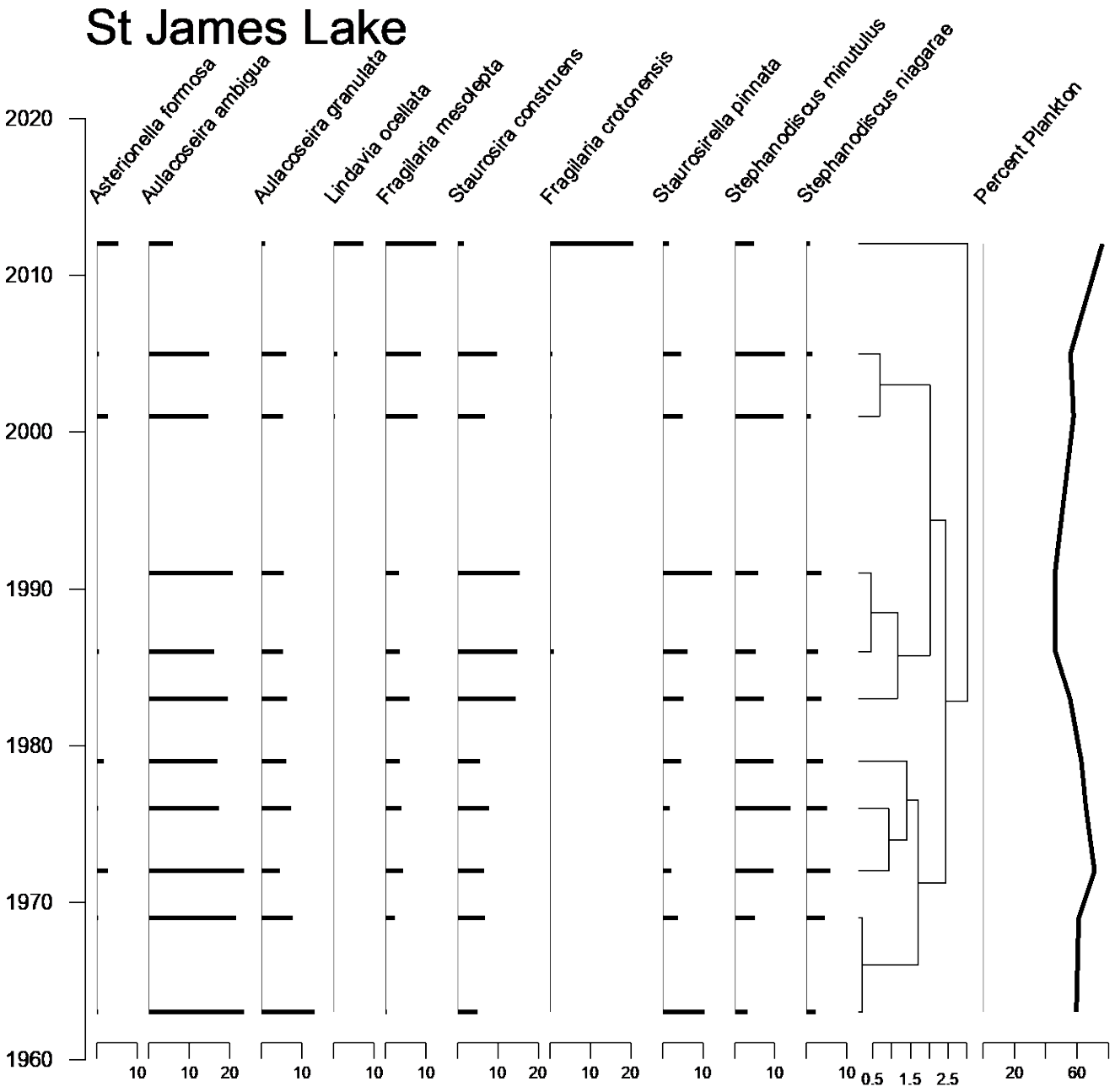


Figure 27. Stratigraphic plot of diatom abundances in Lake St. James. Dendrogram represents a constrained hierarchical clustering (CONISS) which separates distinct time periods based on the diatom flora.

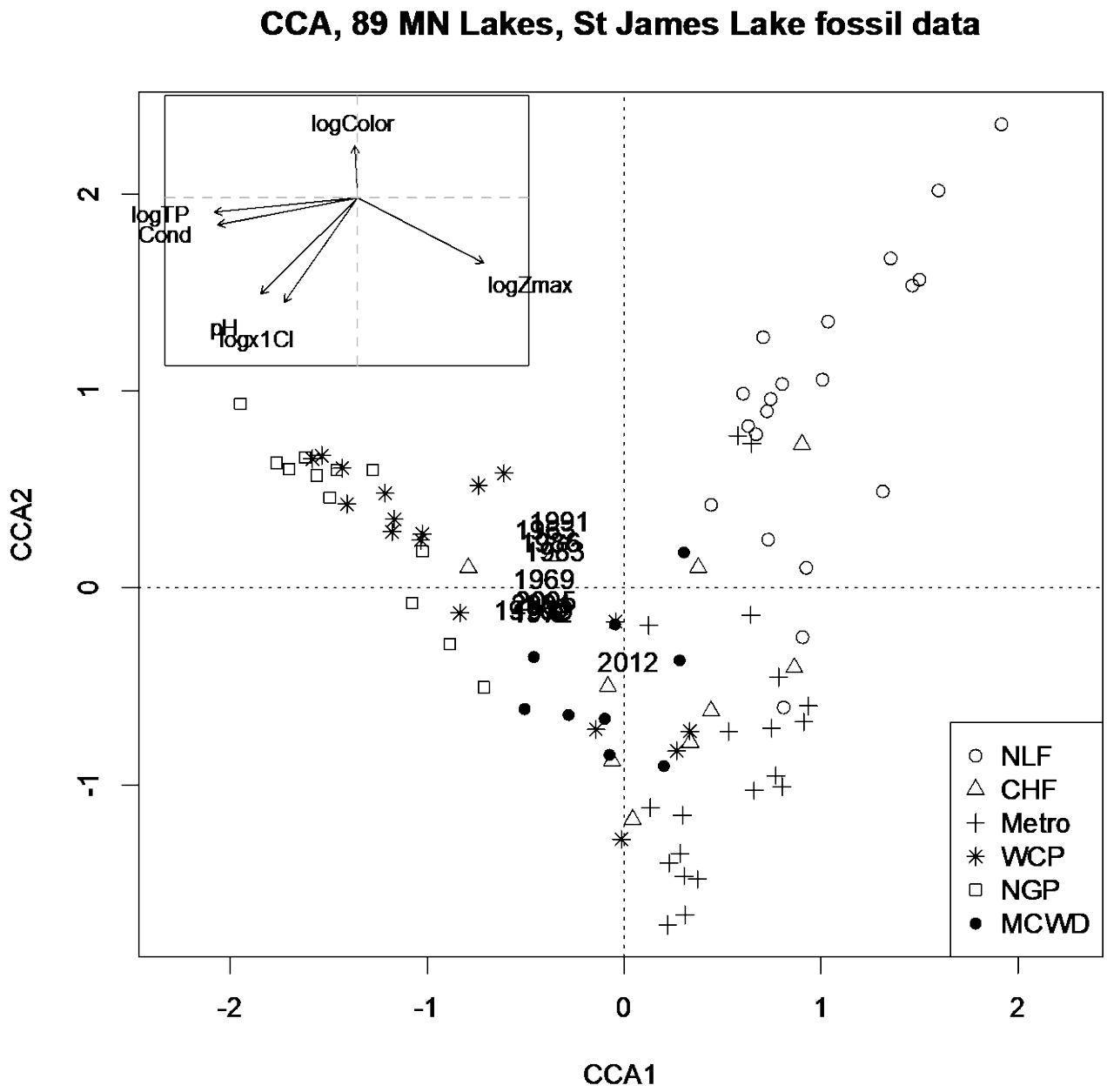


Figure 28. Constrained Coordinates Analysis (CCA) ordination of the 89-lake surface training set in Minnesota. This training set represents **diatom** populations in lakes all across the state and is organized along water quality axes show in the top-left inset. Diatom samples from Lake St. James are passively plotted (represented by the date of the section) onto the ordination to show how they move along these major water quality axes through time.

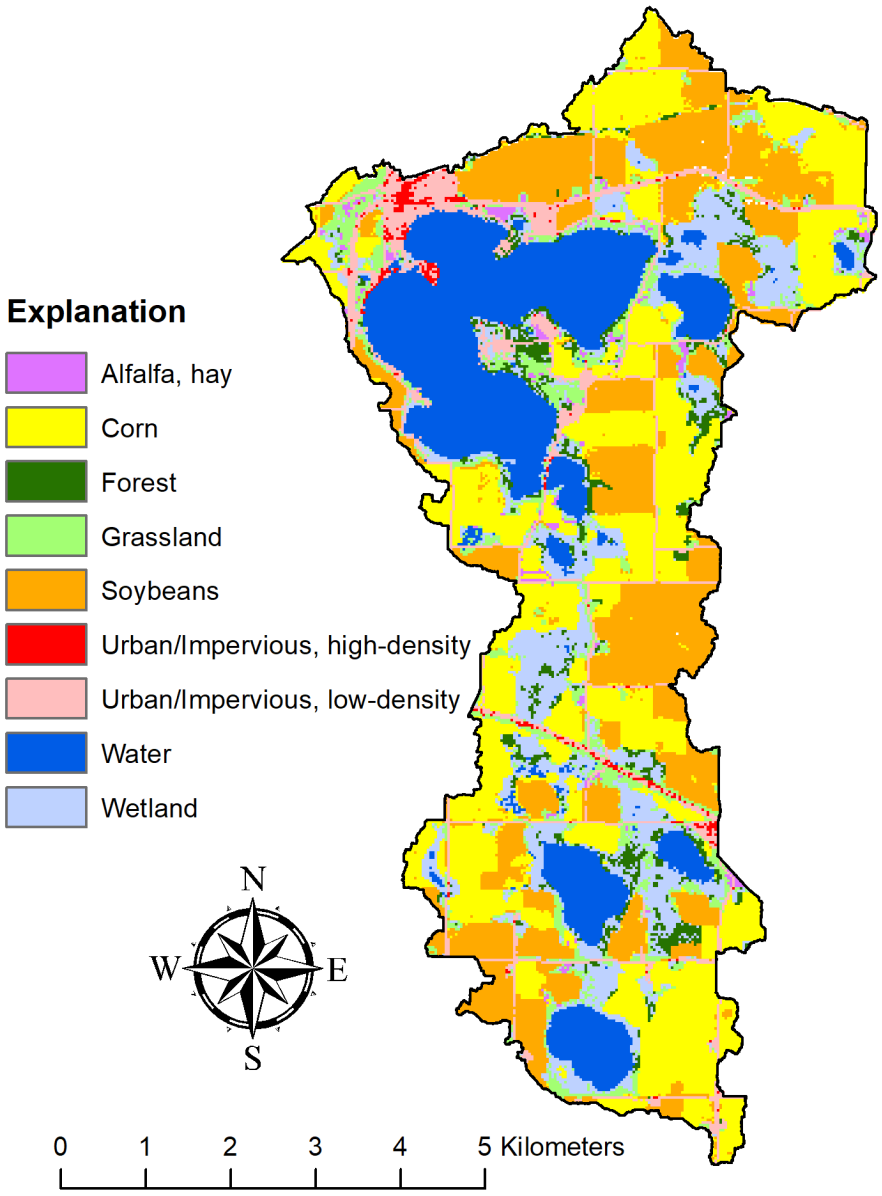


Figure 29. Land-use information for the Madison Lake watershed. Madison Lake is located in the NW corner of the map.

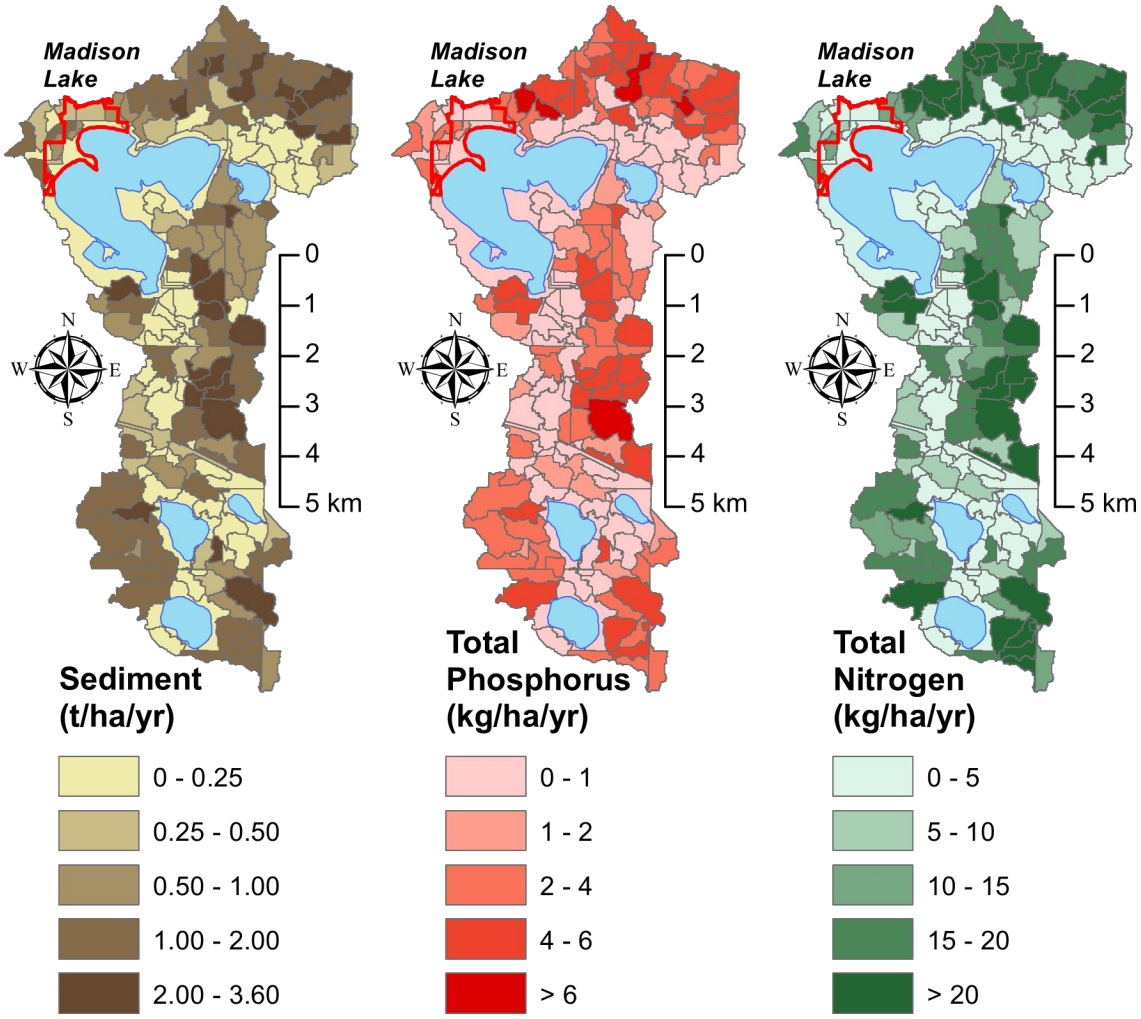


Figure 30. Soil and Water Assessment Tool (SWAT) output for annual sediment, phosphorus and nitrogen fluxes from each sub-basin within the Madison Lake watershed.

Environment and Natural Resources Trust Fund
Final Attachment A (Budget Sheet): Budget Detail for M.L. 2016 Environment and Natural Resources Trust Fund Projects

Project Title: Tracking and Preventing Harmful Algal Blooms
Legal Citation: M.L. 2016, Chp. 186, Sec. 2, Subd. 04a
Project Manager: Daniel R. Engstrom
Organization: St. Croix Watershed Research Station, Science Museum of Minnesota
M.L. 2016 ENRTF Appropriation: $ 593,000
Project Length and Completion Date: 3.5 Years, June 30, 2019
Date of Report: August 16, 2019

ENVIRONMENT AND NATURAL RESOURCES TRUST FUND BUDGET	Activity 1 Budget	Amount Spent	Activity 1 Balance	Activity 2 Budget	Amount Spent	Activity 2 Balance	Activity 3 budget	Amount Spent	Activity 3 Balance	Activity 4 Budget	Amount Spent	Activity 4 Balance	TOTAL BUDGET	TOTAL BALANCE
BUDGET ITEM	***M.L. 2015, Chp. 76, Sec. 2, Subd. 10Emerging Issues Account - Jump-start lake monitoring program***			***Identify species composition and timing of harmful algal blooms***				***Reconstruct frequency of algal blooms relative to natural conditions***				***Determine how nutrients and climate interact to favor harmful algae***		
Personnel (Wages and Benefits)	$42,900	$42,900	$0	$127,200	$127,200	$0	$107,200	$107,200	$0	$106,800	$106,800	$0	$384,100	$0
Engstrom, Research Director: Sediment dating; 8% FTE for 2 yr; Salary=77%, Benefits=23% ($21,600)														
Edlund, Senior Scientist (1 of 2); Diatom & BG algae analyses; 50% FTE for 2.5 yr; Salary=77%, Benefits=23% ($122,400)														
Almendinger, Senior Scientist (1 of 2); SWAT modeling; 35% FTE for 2 yr; Salary=77%, Benefits=23% ($70,100)														
Heathcote, Asst. Scientist; BG algae and toxins; data synthesis; 50% FTE for 3.5 yr; Salary=77%, Benefits=23% ($128,400)														
Field Technician; Lakea monitoring and sampling; 75% FTE for 2 yr; Salalry=77%, Benefits=23% ($41,600)														
Professional/Technical/Service Contracts														
U.S. Geological Survey (for CE-QUAL-W2 modeling of lake hydrodynamics and phosphorus cycling)										$50,000	$50,000	$0	$50,000	$0
University of Regina (for analysis of fossil plant pigments in sediment cores)							$18,800	$18,800	$0				$18,800	$0
Equipment/Tools/Supplies														
Field supplies (sediment traps, sample bottles & vials, reagents)	$1,000	$1,000	$0	$10,500	$10,500	$0							$11,500	$0
Dissolved oxygen and temperature recording probes	$18,500	$18,500	$0							$5,500	$5,500	$0	$24,000	$0
Capital Expenditures Over $5,000														
YSI multi-parameter sonde for water-column measurements	$20,000	$20,000	$0										$20,000	$0
ELISA micoplate reader and supplies for analysis of algal toxins				$10,000	$10,000	$0							$10,000	$0
Travel expenses in Minnesota	$6,500	$6,500	$0	$6,500	$6,500	$0	$1,100	$1,100	$0				$14,100	$0
Lake monitoring & coring (mileage and gas, ~70 trips) $11,500														
Lake monitoring & coring (meals) $6,200														
Lake monitoring & coring (lodging) $1,900														
Other														
Lab analysis of water samples (N, P, DOC, DIC) and sediment cores: radiometric dating (Lead-210, Cesium-137); biogenic silica; loss-on-ignition, sediment phosphorus and metals	$4,100	$4,100	$0	$11,600	$11,600	$0	$44,800	$44,800	$0				$60,500	$0
COLUMN TOTAL	**$93,000**	**$93,000**	**$0**	**$165,800**	**$165,800**	**$0**	**$171,900**	**$171,900**	**$0**	**$162,300**	**$162,300**	**$0**	**$593,000**	**$0**

Science Museum of Minnesota

St. Croix Watershed Research Station

SCWRS Fact Sheet 2019-01

Construction and Calibration of a Computer Model of the Madison Lake Watershed

Summary

• A computer model of the Madison Lake watershed can help identify sources and transport of nonpoint-source pollutants (sediment, phosphorus, and nitrogen), thus informing management decisions on how to clean up these pollutants and reduce noxious algal blooms in the lake.

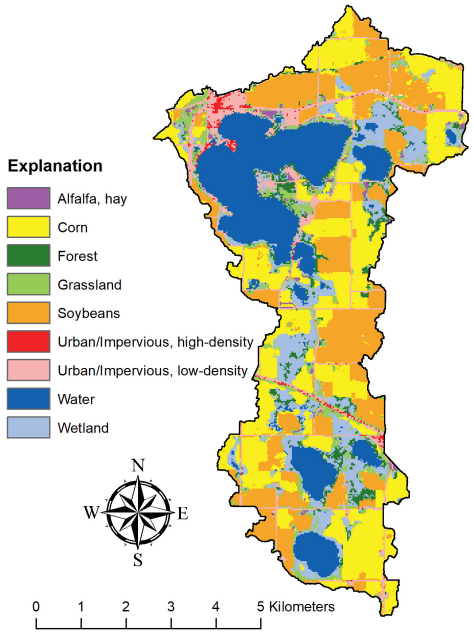


Figure 1. Land use in the Madison Lake watershed.

Issue: *Nonpoint-Source Pollution*

• Madison Lake is a relatively deep (18 m maximum) lake with high recreation value in an agricultural region where shallow lakes are more typical. The Minnesota Department of Natural Resources (MDNR) has deemed Madison Lake as one of their "Sentinel Lakes," a set of 25 representative lakes from across Minnesota selected for detailed studies on how landscapes and climate impact lake ecology.

• Economic policy has driven agriculture to become dominated by row crops (corn and soybeans), which occupy about 50% of the 45-km^2 Madison Lake watershed (Figure 1). Row cropping efficiently produces high yields of grain, but its monoculture nature reduces biodiversity and wildlife habitat.

• Because of fertilizer applications and tillage that leaves the fields without living cover for most of the year, row crops can be significant sources of sediment and nutrients that wash off the land and compromise our waterways. These pollutants are called "nonpoint-source" (NP-S) pollutants because they come from diffuse sources across the landscape. In particular, Madison Lake is impaired by eutrophication, i.e., noxious algal blooms, caused by excess NP-S phosphorus and nitrogen loads.

General Approach: *Monitoring and Modeling*

• To better characterize the problem and create solutions, the sources and quantities of nutrients entering Madison Lake need to be determined. The most direct way is to monitor (measure) the inputs where possible, such as the inlet streams to the lake. Our project partners at the U.S. Geological Survey (USGS) monitored lake inflows and outflows for the 2014-18 ice-free seasons. However, monitoring is relatively expensive and limited to just the few selected sites. What is going on in the rest of the watershed?

• A complementary approach is to construct a computer model that simulates the essential eco-hydrological processes within a watershed (Figure 2). Input to such a

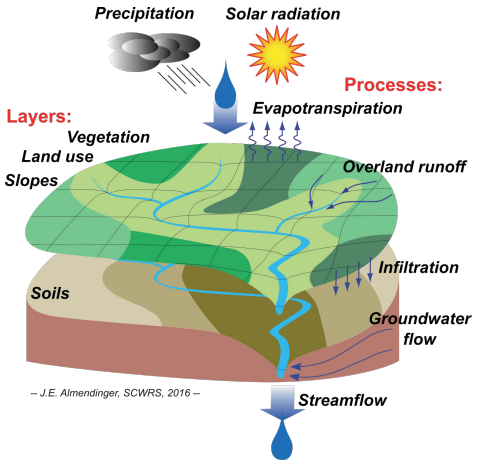


Figure 2. Components of a watershed model.

model includes topography, soil type, land cover, agricultural practices (crop rotations, tillage practices, and fertilizer applications), and daily precipitation and temperature. Output includes daily flows and export of sediment and nutrients from each land-use type as well as from the watershed as a whole.
• The best studies combine (a) monitoring data to measure observed flows and nutrient loads, with (b) modeling efforts to figure out how the watershed works -- that is, what are the landscape and weather processes that generated the observed data?
• The next steps would be to design remediation methods to clean up the sources of nutrients across the watershed. Innovative farming practices (i.e., best management practices, or BMPs) that introduce more diversity in the timing and spatial pattern of crop rotations could simultaneously increase habitat, improve soil fertility, and protect streams and lakes.

Specific Approach: *SWAT Model Construction and Calibration*

• The model applied to the Madison Lake watershed is called the Soil and Water Assessment Tool, or SWAT for short. SWAT was developed by the U.S. Department of Agriculture (USDA) Agricultural Research Service (ARS) to help understand and predict loads of NP-S pollutants (sediment and nutrients) from large river basins over long periods of time.
• Input to the SWAT model relies on readily-available data from government agency web sites. Topography was taken from LiDAR digital elevation models (DEMs) made available by the MDNR at a 3-m (meter) horizontal resolution. The DEM was hydro-modified to include drainage features (e.g., culverts) that correct for the false water impoundment by roads and other embankments. Soils data were taken from the SSURGO database made available by the USDA, which is the most spatially detailed soil data available.
• Land cover and crop types were taken from the USDA's crop data layer (CDL) datasets for 2014-18. This 5-year sequence of crops on the ground, at 30-m spatial resolution, provided an objective method for inferring typical crop areas, rotations, and locations in the watershed. Table 1 gives the areas of each crop, and Figure 1 shows their spatial distribution. Corn and soybeans were the most common crops by far, with minor amounts of alfalfa and even less of small grains. Representative amounts of inorganic fertilizer were added to all crops at the time of planting. Conservation tillage was assumed for all cropland, consisting of fall chisel plowing followed by spring disking or field cultivation.
• Weather data (daily precipitation and temperature) were taken from six weather stations (Amboy, LeSueur, Mankato, St. Peter, Waseca, and Faribault) and averaged for the watershed centroid by simple inverse-distance weighting.
• After watershed models are constructed, they need to be adjusted ("calibrated") so that their output matches known monitoring data from the watershed. Figure 3 shows the comparison between observed daily flows out of Madison Lake (thick gray lines) and the modeled values (thin black lines) for 2015-16. The Nash-Sutcliffe (NS) statistic shows the quality of the model fit (fraction of observed variance explained by the model). An NS value of 1.0 indicates a perfect fit, and a value of 0.5 or above indicates a good fit. The NS values were 0.65 and 0.76 for 2015 and 2016, respectively, indicating a very good fit. Unfortunately, observed data were not available for loads of sediment, phosphorus, and nitrogen entering the lake, and so the quality of these modeled quantities cannot be determined.

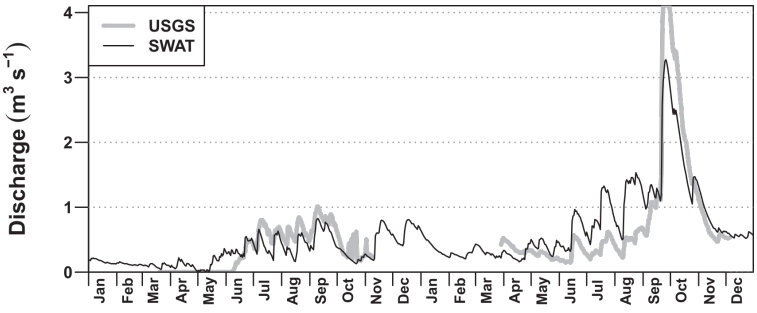


Figure 3. SWAT model calibration runs for daily flow out of Madison Lake, 2015-16
Observed = USGS, Modeled = SWAT

Table 1. SWAT-modeled loads and yields of sediment, total phosphorus, and total nitrogen from different land uses.

Land Cover	Area		Sediment			Total Phosphorus			Total Nitrogen		
	(km^2)	(%)	(t/yr)	(%)	(t/ha/yr)	(kg/yr)	(%)	(kg/ha/yr)	(kg/yr)	(%)	(kg/ha/yr)
Agricultural Lands	23.7	52.6%	5,546	98.6%	2.34	12,968	97.6%	5.47	63,045	97.3%	26.59
Corn	*12.9*	*28.6%*	*2,601*	*46.2%*	*2.02*	*6,193*	*46.6%*	*4.81*	*33,718*	*52.0%*	*26.20*
Soybeans	*10.6*	*23.6%*	*2,944*	*52.3%*	*2.77*	*6,772*	*51.0%*	*6.37*	*29,309*	*45.2%*	*27.55*
Alfalfa	*0.2*	*0.4%*	*1*	*0.0%*	*0.07*	*3*	*0.0%*	*0.16*	*18*	*0.0%*	*0.91*
Developed	3.2	7.1%	66	1.2%	0.21	234	1.8%	0.73	1,179	1.8%	3.71
Roads	*3.0*	*6.7%*	*38*	*0.7%*	*0.13*	*179*	*1.3%*	*0.60*	*1,047*	*1.6%*	*3.49*
Urban	*0.2*	*0.4%*	*29*	*0.5%*	*1.56*	*55*	*0.4%*	*3.00*	*132*	*0.2%*	*7.17*
Undeveloped	18.2	40.3%	14	0.3%	0.01	87	0.7%	0.05	560	0.9%	0.31
Grassland	*2.1*	*4.6%*	*4*	*0.1%*	*0.02*	*35*	*0.3%*	*0.17*	*363*	*0.6%*	*1.75*
Forest	*1.1*	*2.4%*	*2*	*0.0%*	*0.02*	*5*	*0.0%*	*0.05*	*24*	*0.0%*	*0.22*
Aquatic	*15.0*	*33.4%*	*8*	*0.1%*	*0.01*	*47*	*0.4%*	*0.03*	*173*	*0.3%*	*0.12*
Total	45.1	100%	5,626	100%		13,289	100%		64,785	100%	

Results: *Land-Use Sources of Sediment and Nutrients*

• A *load* is a mass of a constituent during a selected time period, e.g., metric tons per year (t/yr) or kilograms per year (kg/yr). A *yield* is a load per unit area of a selected land unit, e.g., tons per hectare per year (t/ha/yr) or kilograms per hectare per year (kg/ha/yr). We will use metric units in this report, even though in US agriculture, English units of short tons per acre, or pounds per acre, are far more commonly used.

• Table 1 shows average annual loads and yields of sediment, phosphorus, and nitrogen from different crops and other land covers for a 10-year model run from 2009-18. The values here represent the amounts of NP-S pollutants mobilized on the landscape. Not all of this mass makes it to Madison Lake; a significant portion gets trapped along the way in wetlands and ponds.

• Loads of all constituents were dominated by agriculture, both because it is the most prevalent land use in the watershed and because its yields tend to be higher than most other land uses. According to the model, corn and soybeans occupied a little more than half of the land area and generated about 98% of the sediment, phosphorus, and nitrogen loads in the watershed.

• Yields told a clearer story about which land uses were more "leaky" with regard to NP-S pollutants. Again, per unit area, row crops generated more sediment, phosphorus, and nitrogen than other land uses. Corn and soybeans had similarly large yields of sediment (over 2 t/ha/yr), phosphorus (about 5-6 kg/ha/yr), and nitrogen (26-27 kg/ha/yr). Urban areas likewise had significant yields of sediment and phosphorus, but their footprint was much smaller than agriculture. Highly permeable soils can limit yields of NP-S pollutants in surface runoff, but apparently these soil types are not significant in the Madison Lake watershed.

Results: *Spatial Distribution of Sediment and Nutrient Yields*

• Figure 4 shows yields of sediment, phosphorus, and nitrogen for each of the 197 modeled subbasins of the Madison Lake watershed. The darker colors represent "hot spots" of sediment and nutrient sources in the watershed. Yields here represent the amount of each constituent delivered to the stream reach via overland flow and groundwater, i.e., the initial mass mobilized in the uplands minus any losses to sediment and nutrient traps (e.g., wetlands) encountered between field and stream.

• In the Madison Lake watershed, sediment, phosphorus, and nitrogen yields are consistent with each other and are driven primarily by sources, namely, location of corn and soybean fields. Wetland, forest, and grasslands produce minimal yields of these NP-S pollutants. The cropland hot-spots of high yields are areas to target for remediation by alternative farming practices that reduce soil erosion and nutrient loss.

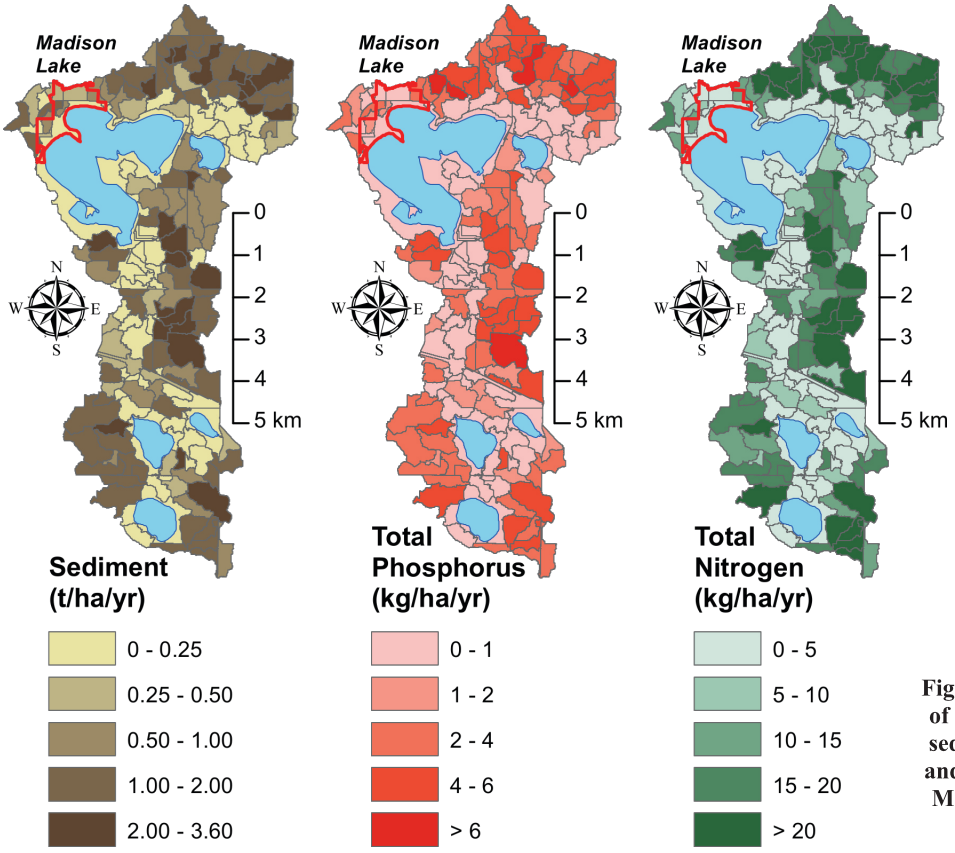


Figure 4. Spatial distribution of SWAT-modeled yields of sediment, total phosphorus, and total nitrogen across the Madison Lake watershed.

Summary and Conclusions

• The SWAT model for the Madison Lake watershed was able to simulate known stream flows in the watershed, and to identify the probable sources (land use and subbasin) of these constituents. The next steps will be to simulate possible remediation scenarios to see which ones will most efficiently reduce these pollutants while increasing landscape biodiversity and habitat, without undue burden on the farmers who are stewards of the land.

Acknowledgment

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Updates to the Madison Lake (Minnesota) CE-QUAL-W2 Water-Quality Model for Assessing Algal Community Dynamics

By Erik A. Smith and Richard L. Kiesling

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U.S. Geological Survey

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Conversion Factors

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
meter (m)	3.281	foot (ft)
meter (m)	39.37	inches (in.)
kilometer (km)	0.6215	mile (mi)
	Area	
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
cubic meter (m^3)	35.31	cubic foot (ft^3)
	Flow rate	
meter per year (m/yr)	3.281	foot per year (ft/yr)
	Energy	
watt per square meter (W/m^2)	0.3170	British thermal unit per hour per square foot ($Btu/hr/ft^2$)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as °F = (1.8 × °C) + 32.

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

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Elevation, as used in this report, refers to distance above the vertical datum.

Supplemental Information

Concentrations of chemical constituents in water are given in either milligrams per liter (mg/L) or micrograms per liter (µg/L).

Abbreviations

<	less than
DEM	digital elevation model
DHEL	Department of Health Environmental Laboratory
DO	dissolved oxygen
HAB	harmful algal bloom
MAE	mean absolute error
MNDNR	Minnesota Department of Natural Resources
NWIS	National Water Information System
R^2	coefficient of determination
RMSE	root mean square error
USACE	U.S. Army Corps of Engineers
USGS	U.S. Geological Survey
WSC	wind sheltering coefficient

Updates to the Madison Lake (Minnesota) CE-QUAL-W2 Water-Quality Model for Assessing Algal Community Dynamics

By Erik A. Smith and Richard L. Kiesling

Abstract

A previously developed CE-QUAL-W2 model for Madison Lake, Minnesota, simulated the algal community dynamics, water quality, and fish habitat suitability of Madison Lake under recent (2014) meteorological conditions. Additionally, this earlier model simulated the complex interplay between external nutrient loading, internal nutrient loading from sediment release of phosphorus, and the organic matter decomposition of the algal biomass. However, the partitioning of cyanobacteria within the modeling framework was simplified to one group and did not account for how different cyanobacteria populations are affected by light conditions, the usage of nitrogen, temperature growth ranges, and differences in settling rates. To get a better handle on the proliferation of cyanobacteria in Madison Lake, the model required updates to at least partition the cyanobacteria into a group that fixed nitrogen and a second, more buoyant cyanobacteria group, that did not independently fix nitrogen.

To address the shortcomings of simulating cyanobacteria in the earlier model, the U.S. Geological Survey (USGS), in cooperation with the St. Croix Watershed Research Station (Science Museum of Minnesota), updated the Madison Lake CE-QUAL-W2 model to better characterize cyanobacteria into two groups. In addition to updating the cyanobacteria group differentiation, the entire portion of the model that handles the simulation of algal community dynamics was updated while preserving the model's predictive capabilities for nutrients, water temperature, and dissolved oxygen. The calibration and validation of the model was done under recent meteorological conditions with large and persistent cyanobacteria blooms (2014 and 2016). Overall, the model simulations predicted the persistently large total phosphorus concentrations in Madison Lake's hypolimnion, key differences in nutrient concentrations between the two years, and cyanobacteria bloom persistence.

Introduction

Across the entire spectrum of freshwater lakes around the world, high anthropogenic nitrogen and phosphorus inputs into freshwater lakes have been implicated as one of the primary causes for the alarming rise in cyanobacteria blooms over the past several decades (Xu and others, 2010; Dolman and others, 2012; Paerl and Otten, 2013). These blooms can reduce the recreational and ecological value of lakes, including lakes across Minnesota. For Madison Lake, a fairly large and deep lake located in southern Minnesota, cyanobacteria have become an increasingly dominant component of the algal community. This creates a potential concern for the health of Madison Lake, as many cyanobacteria species can produce potent toxins and can lead to harmful algal blooms (HABs). When cyanobacteria form a toxic HAB, potential impairments include restricted recreational activities because of algal scums or algal mats and

the production of toxins (for example, microcystin) in amounts capable of threatening human health, domestic animals and wildlife (O'Neil and others, 2012; Graham and others, 2016). Exposure to environmental concentrations of cyanotoxins can cause hepatic, neurologic, respiratory, and dermatologic problems in humans (Merel and others, 2013; Loftin and others, 2016).

Although cyanobacteria (also known as cyanophyta) have been common components of the Madison Lake phytoplankton community for some time, recent Madison Lake data has shown a proliferation of cyanobacteria (Lindon and Heiskary, 2007; Lindon and Heiskary, 2009). From 2013 through 2018, routine field monitoring samples showed cyanobacteria as a fairly large percentage of the algal community, both by the overall number of individuals (counts) and the overall biovolume. Also, it was found that several cyanobacteria genera persisted throughout much of the summer and into the fall months (July through October).

The Minnesota Department of Natural Resources and other local resource managers are concerned that these cyanobacteria blooms could negatively impact Madison Lake. Madison Lake is a popular recreational lake for fishing, swimming, and boating and also has a dense community of year-round residents (Lindon and others, 2010). Persistent algal blooms, whether the blooms are cyanobacteria or other types of algae, can negatively impact the fishery indirectly by affecting dissolved oxygen. Madison Lake contains high-quality populations of fish species (Minnesota Department of Natural Resources, 2016) such as northern pike (*Esox lucius)*, smallmouth bass (*Micropterus dolomieu*), bigmouth buffalo (*Ictiobus cyprinellus*), and bluegill sunfish (*Lepomis macrochirus*). Continuous monitoring of epilimnetic and hypolimnetic dissolved oxygen in Madison Lake has documented prolonged periods of hypoxia, associated with periods of long water residence time and sustained high levels of algal biomass that last for

weeks. When blooms enter stationary phase growth or start to senesce, bacteria mineralize the sinking algal biomass, consuming large amounts of oxygen, thereby decreasing dissolved oxygen concentrations. Large blooms can result in hypoxia areas, which, in turn, can endanger the fishery by creating habitat bottlenecks. So aside from the obvious concerns related to toxins, large algal blooms can have multiple negative effects on the overall health of Madison Lake.

Previous summaries of Madison Lake water quality have documented large inputs of nitrogen and phosphorus into Madison Lake (Lindon and others, 2018). A U.S. Geological Survey report (Smith and others, 2017) showed high levels of total phosphorus in the hypolimnion, in that case from 2014. The same report documented the development of a hydrodynamic and water-quality model for Madison Lake which suggested that a large component of the total phosphorus was due to internal loading from sediments during hypoxic or anoxic conditions. However, to get a handle on how a diverse algal community responds to shifts in external and internal loading of nutrients, particularly nitrogen and phosphorus, a sophisticated model that can simulate algal community dynamics is necessary. This earlier model, developed with the CE-QUAL-W2 modeling framework (Cole and Wells, 2015), did simulate the algal community into four different groups, including a general group for cyanobacteria (termed blue-green algae in the earlier report). However, this model's differentiation did not account for how different cyanobacteria populations are affected by light conditions, the usage of nitrogen, temperature growth ranges, and differences in settling rates.

To address the shortcomings of simulating cyanobacteria in the earlier model, the U.S. Geological Survey (USGS), in cooperation with the St. Croix Watershed Research Station (Science Museum of Minnesota) with support from the Minnesota Environment and Natural Resources Trust Fund (ENRTF), updated the Madison Lake CE-QUAL-W2 model. In addition

to updating the cyanobacteria group differentiation, the entire portion of the model that handles the simulation of algal community dynamics was updated while preserving the model's predictive capabilities for nutrients, water temperature, and dissolved oxygen. The calibration and validation of the model was done under recent meteorological conditions with large and persistent cyanobacteria blooms (2014 and 2016). With the completed model, further scenarios can be run as new Soil and Water Assessment Tool (SWAT) simulations become available that can provide external nutrient loading information for different management scenarios or past environmental conditions.

Purpose and Scope

The purpose of this report is to document updates to a previously developed CE-QUAL-W2 hydrodynamic and water-quality model of Madison Lake, Minnesota (Smith and others, 2017). The previous version simulated phytoplankton into four general algal communities or groups: (1) bacillariophyta and crysophyta (diatoms); (2) chlorophyta (green algae); (3) cyanophyta (blue-green algae); and, (4) haptophyta and cryptophyta (flagellates). For the updated model, the blue-green algae group, referred to as cyanophyta in this report, has been divided into two groups: a nitrogen-fixing cyanophyta group, generally representative of *Anabaena*, *Dolichospermum*, and *Cylindrospermopsis*, and a non-fixing, buoyant cyanophyta group, generally representative of *Planktothrix*, *Microcystis*, and *Woronichinia*.

Study Area

Madison Lake (fig. 1) in Blue Earth County, Minnesota, is in the Le Sueur River Basin, part of the greater Minnesota River Basin (Lindon and others, 2010). Madison Lake is weakly dimictic, generally starting off as well-mixed before early summer, with a weak thermocline that

develops in the summer months; the lake mixes again in the late fall (Lindon and others, 2010). Dissolved oxygen is well-mixed in the early spring (April to May) and late fall (mid-October), with a substantial portion of the hypolimnion becoming anoxic by mid-summer; however, anoxia can develop earlier in some years and subsist late into the fall, especially when the lake's thermocline develops early (Lindon and others, 2010). The water balance of the drainage basin for Madison Lake is typically controlled by a spring snowmelt in late March or early April, followed by periodic large rain events in the summer. The mean precipitation in the region for 1981–2010 is 0.82 meter per year (m/yr) (National Centers for Environmental Information, 2016).

Figure 1. Map showing location of water-quality sampling sites for Madison Lake, Minnesota.

Table 1. Location of continuous pressure transducers, water-quality sondes, thermistors, and discrete water-quality measurements used for the development of model input or calibration/validation of water temperature, dissolved oxygen, and water-quality constituents.

Primary inflows to Madison Lake are in the northeast and southeast parts of the lake, both primary sampling locations for nutrient and major inorganic constituents, water temperature, and streamflow (table 1). The unnamed stream to Madison Lake at CR-48 near Madison Lake, Minn. (USGS station number 05320130 [U.S. Geological Survey, 2019a]; hereafter referred to as the "northeast inlet") flows into the relatively large and shallow northeast bay of Madison Lake. The unnamed stream between Schoolhouse and Goolsby Lakes southeast of Madison Lake, Minn. (USGS station number 05320140; hereafter referred to as the "southeast inlet") flows into the shallow part of the smallest bay (by area) along the southeast shoreline. The main primary outflow for Madison Lake is the site Madison Lake outlet to Mud Lake South of Madison Lake,

Minn. (USGS station number 05320170 [U.S. Geological Survey, 2019a]; hereafter referred to as the "Madison Lake outlet"), located along the southwest part of the lake.

The lake has three distinct bays, with two of the three bays containing deep areas. The deep area in the southwest bay, also the largest deep area by areal extent, was sampled at site Madison Lake southwest deep point near Madison Lake, Minn. (hereafter referred to as "southwest deep point") with a depth of approximately 18 m. This location was used for extensive in-lake water-quality sampling, periodic vertical profiles of water temperature and DO, and continuous monitoring of water temperature at various depths.

Methods and Data

The Madison Lake CE-QUAL-W2 model was previously developed for 2014 to simulate algal community dynamics, water-surface elevations, flow, water quality, and fish habitat suitability (Smith and others, 2017). This study updated the original model to re-distribute the algal community into five distinct algal groups or divisions rather than four groups, re-calibrating the updated model for 2014 and validating the model for 2016. Both the original and updated versions were developed with CE-QUAL-W2 (version 4.0, available at http://www.ce.pdx.edu/w2/), a two-dimensional, laterally averaged, hydrodynamic and water-quality model originally developed by the USACE and currently supported by Portland State University (Cole and Wells, 2015). The CE-QUAL-W2 model calculates the hydrodynamic properties of water-surface elevation, velocities, and temperature and can simulate water-quality variables in addition to temperature. An advantage of the CE-QUAL-W2 model is that the hydrodynamic and water-quality modules are coupled together through an equation of state for density, dependent on temperature, suspended solids, and dissolved solids. This enables the

water-quality model to feed back into the hydrodynamic part of the model; however, because of this coupling, changes to the model specifications for algal growth and senescence can affect the other parts of the model. Therefore, the changes to the algal dynamics and some other updates required a reassessment of the entire model fit.

The CE-QUAL-W2 computational grid, based on available bathymetric data (Minnesota Geospatial Information Office, 2016) and a digital elevation model (DEM) (U.S. Geological Survey, 2016, was left unaltered and is described in detail in Smith and others (2017). In summary, the CE-QUAL-W2 grid was separated into segments that laterally average across the lake, with individual segments grouped together into branches. Each branch is grouped together to represent the entire computational grid of the water body. For Madison Lake, the CE-QUAL-W2 water body (fig. 1) was grouped together from two separate branches: (1) branch 1 starts at segment 2 and continues through segment 9; (2) branch 2 separates out the southeast part of Madison Lake, where the southeast inlet flows into the lake, and connects to branch 1 at segment 9 via segment 15.

This project followed a similar calibration strategy as other CE-QUAL-W2 modeling projects completed by the Upper Midwest Water Science Center Integrated Ecosystems Systems team (Smith and others, 2014; Smith and others, 2017, Smith and others, 2018). Calibration targets included a water balance calibration based on water-surface elevation, chlorophyll *a*, algae, and nutrients (ammonia, nitrate plus nitrite, total Kjeldahl nitrogen, total phosphorus, orthophosphate). As vertical variations in temperature and dissolved oxygen are important for distinguishing temporal variations in the lake epilimnion, hypolimnion, and mixed layers, emphasis was considered for the synoptic depth profiles of temperature and dissolved oxygen from the southwest deep point.

The CE-QUAL-W2 model required time series inputs of hydrological, thermal, water quality, and meteorological data. A summary of the discrete and continuous data collected for Madison Lake, further split by sampling locations, is shown in table 1. All the input data used for calendar year 2014 was documented in Smith and others (2017). The same basic data and sources was used for 2016, with the exception that continuous streamflow and temperature was unavailable for 2016, so a surrogate dataset was required and discussed further in the "Hydraulic and Thermal Boundary Conditions".

Water Balance

The water balance of Madison Lake for May 15–November 1, 2014 was left unaltered (Smith and others, 2017), with a new water balance required for March 30–November 23, 2016. Similar to the 2014 water balance, the 2016 water balance was completed by comparing measured water levels to simulated water levels. However, unlike 2014, continuous water levels were unavailable for the Madison Lake outlet (USGS station number 05320170). Instead, the simulated water levels were compared to the daily water-surface elevations collected by the Lake Level Minnesota Monitoring Program (Minnesota Department of Natural Resources, 2019a) and available from the Minnesota Department of Natural Resources (MNDNR) Lake Finder website (Minnesota Department of Natural Resources, 2019b).

Hydraulic Boundary Conditions

Lake inflow used in the CE-QUAL-W2 model were obtained from two separate channels that flow into Madison Lake. The northeast inlet streamflow (fig. 1; table 1) was measured in the channel connecting several small lakes and wetlands to Madison Lake. The southeast inlet streamflow (fig. 1; table 1) was measured in the channel connecting Schoolhouse and Goolsby

Lake to Madison Lake. Submersible pressure transducers were installed for the northeast inlet, southeast inlet, and Madison Lake outflow from May–November 2014. These transducers collected continuous water-surface level (stage or gage height) measurements every 15 minutes. Three corresponding measurements of streamflow and water-surface level measurements were made at each inflow site in 2014 (U.S. Geological Survey, 2019) by the MNDNR to construct an elevation-streamflow rating table, as discussed in Smith and others (2017) and presented in appendix table 1–1 of the same report. In summary, the elevation-streamflow rating curves were developed using graphical plotting methods similar to those described in Rantz and others (1982a, 1982b), with linear extrapolations added to the upper and lower end of the rating curves to estimate streamflows outside of the range of measured streamflows. The Madison Lake outflow, located along the southwest part of the lake, was also estimated though an elevation-streamflow rating curve, based upon four direct measurements made in 2014.

For 2016, no continuous water level measurements were available for either of the two inflow sites or the lake outflow. However, the model still requires streamflow input into the model, ideally sub-daily measurements. Without such a record available, the 2014 elevation-streamflow rating table was applied to the daily water-surface elevations from the Lake Level Minnesota Monitoring Program (Minnesota Department of Natural Resources, 2019a; 2019b). By using this methodology, daily inflows and outflow were calculated and input into the model. The daily water-surface elevations used for 2016, available from the Lake Finder website, are also available as part of the full CE-QUAL-W2 model archive (Smith, 2019) in the el_obs.csv file (in meters) available on USGS ScienceBase.

For both 2014 and 2016, additional water inflows to Madison Lake were assumed from ungaged locations in the lake and from groundwater flow, known as distributed tributary flow.

This distributed tributary flow was input into the model in daily time steps and distributed evenly across all the model segments. To account for this additional flow, water was iteratively added to the distributed tributary flow (also known as QDT) through successive model runs until a satisfactory match was attained between simulated and measured water-surface elevations.

Thermal Boundary Conditions

Inflowing water temperature was collected in 2014 by the same submersible pressure transducers for water levels: the northeast inlet and the southeast inlet. The temperatures were then converted to the appropriate data format for CE-QUAL-W2 and applied as tin_br1, the inflowing water temperature via the northeast inlet into branch 1, and tin_br2, the inflowing water temperature via the southeast inlet into branch 2 (Smith and others, 2017; Smith, 2019). The distributed tributary flow also had associated temperature records within the model framework, applied as tdt_br1 and tdt_br2 for the two separate branches. In both cases, a continuous temperature record from a nearby observation well with a depth of 3.8 meters (Minnesota Unique Identification Number 792526) was assumed as the distributed tributary flow temperature. No conversion was done with this temperature record and is available as part of the CE-QUAL-W2 archive (Smith, 2019). As this continuous record was available from 2013 through 2018, both 2014 and 2016 had a full record available. The daily mean temperatures for the northeast inlet (USGS station number 05320130) and for the southeast inlet (USGS station number 05320140) are available online through NWIS (U.S. Geological Survey, 2019); additionally, the ScienceBase archive for the CE-QUAL-W2 model (Smith, 2019) includes the 2014 inflow water temperatures.

However, as with the flow data, no direct water temperatures were available for 2016 for either the northeast or southeast inlet. Instead, a surrogate water temperature dataset had to be

constructed. Using a relationship between water temperature and air temperature, similar to a technique applied to central United States streams by Preud'homme and Stefan (1992), a regression between the daily air temperature (available from the Mankato Regional Airport) and the daily water temperatures from the two inlet transducers was applied. For branch 1, the following mathematical relation between daily air temperature and daily water temperature (eqn. 1), based on 2014 data, was applied to 2016 daily air temperatures to create a surrogate branch 1 temperature record with a coefficient of determination (R^2) of 0.87:

$$\text{Temperature (Branch 1)} = 1.0208 * \text{Daily Air Temperature} - 1.4595,\ R^2 = 0.87 \quad (1)$$

For branch 2, the following mathematical relation between daily air temperature and daily water temperature (eqn. 2), also based on 2014 data, was applied to 2016 daily air temperatures to create a surrogate branch 2 temperature record with a R^2 of 0.90:

$$\text{Temperature (Branch 2)} = 0.993 * \text{Daily Air Temperature} + 0.4015,\ R^2 = 0.90 \quad (2)$$

As with the 2014 temperature data, the ScienceBase archive (Smith, 2019) includes the 2016 inflow and distributed tributary water temperatures.

Meteorological Data

Meteorological data are required as input to the CE-QUAL-W2 model because of the importance of surface boundary conditions to the overall behavior of the model, specifically surface heat exchange, solar radiation absorption, wind stress, and gas exchange. Required meteorological data include air temperature, dew point temperature, wind speed, wind direction, and cloud cover. All unit conversions from the meteorological data to the required units for the model were straightforward with the exception of cloud cover. The qualitative sky cover parameter (that is, clear, scattered, broken, and overcast) was converted to an integer value ranging from 0 to 10: clear is 1, scattered (1/8 to 1/2 cloud coverage) is 5, and overcast is 10. All

of the required data were available at hourly intervals for the Mankato Regional Airport (U.S. Air Force station identification number 726585) from the Climate Data Online portal (National Climatic Data Center, 2016; National Climatic Data Center, 2018), located <12.5 kilometers (km) west of Madison Lake. Based on the latitude and longitude of the lake and the required meteorological inputs, evapotranspiration was included in the water balance as an internal CE-QUAL-W2 calculation.

Water-Quality, Data Collection, Vertical Profiles, and Laboratory Analyses

Limnological characteristics, including properties that could affect trophic state, were examined at the southwest deep point. This site was sampled by MNDNR staff five times in 2014 (Smith and others, 2017) and between 5-11 times, depending on the constituent, for 2016. Samples were collected near the surface (between 0 and 2 meters) and at depth, averaging between 15.5 and 16.5 meters, using a Kemmerer sampler (Wildco 1200E; Wildlife Supply Co., Yulee, Florida) and were analyzed using the methods in table 2 to determine concentrations of nutrients, chlorophyll *a*, total dissolved solids, major ions (total silica and dissolved iron), and algal counts. Water samples were filtered (through a 0.45-micrometer filter for dissolved analysis or not filtered for total analysis) and preserved as required (U.S. Environmental Protection Agency, 1993a, 1993b, 1993c, 1994d). Alkalinity was determined by incremental titration at the field location (Wilde, 2006). Secchi-disk transparency (Wetzel, 2001) was measured at each vertical profile location to estimate photic depth. Vertical profiles (approximately 1-m intervals) of temperature, DO concentration, pH, and specific conductance were measured by MNDNR staff with a multiparameter Hydrolab sonde at each lake site in conjunction with the water samples.

Table 2. Water-quality methods for constituents analyzed in water samples from Madison Lake, 2014 and 2016.

Sampling also was done by the MNDNR at the inflows for both lakes (table 1). The same constituents and methodologies as the limnological sites were followed for these inflow sites. Sampling frequency for the inflow sites varied between the two inlets and two years, sampled 4-5 times in 2014 and 5-6 times in 2016. Water samples collected by the MNDNR at the lake, inflow, and outflow sites were analyzed by the Minnesota Department of Health Environmental Laboratory (DHEL) in St. Paul, Minn., except for the algae data. All the samples analyzed by the Minnesota DHEL have been previously reviewed and published and are available online (Minnesota Pollution Control Agency, 2018). The algae data were produced by a phytoplankton enumeration technique performed by PhycoTech, Inc. (PhycoTech, 2019); all of the raw algal data are presented in table 3, presented by relative count, and then converted to algal biomass by assuming an algal biomass (in milligrams per liter) to chlorophyll *a* (in micrograms per liter) ratio of 0.05 and multiplying by the chlorophyll *a* concentration collected on the same day. This ratio is different than the ratio applied for Smith and others (2017) for 2014 data, so table 3 supersedes the Smith and others (2017) unless applied with the earlier version of the model.

Table 3. Relative counts and converted algal biomass (in milligrams per liter) for Madison Lake southwest deep point near Madison Lake, Minnesota, 2014 and 2016.

A primary data-quality objective was to ensure that samples were representative of the water bodies under investigation. Quality assurance was assessed with specific procedures, such as instrument calibration, to ensure data reliability and assess the quality of the sample data. The quality-assurance plan for this study followed MNDNR guidelines (Anderson and Martin, 2015).

Additional quality assurance specific to Minnesota DHEL is available online (Minnesota Department of Health, 2016). Results from available quality-assurance data associated with water-quality data used for input to the model and for calibration and validation of the model were reviewed prior to the modeling efforts. Overall, the water-quality datasets (discrete samples collected at specific streamflow or lake elevations) for the calibration and validation periods were considered appropriate for the range of environmental conditions simulated for this study.

Initial Conditions

Water-quality modeling was incorporated into the lake hydrodynamic model. Each simulated constituent (including temperature) must have an initial, single concentration for the entire lake or a gridwide initial vertical profile of concentrations at the start of each model run. Initial constituent concentrations are presented in table 4 for the calibration (2014) run and the validation (2016) run; initial constituent concentrations were considered uniform throughout both lakes for every segment and layer, except in cases with a reported range of values in a vertical profile. It should be noted that differences exist between the starting initial constituent concentrations for the algal concentrations from the original Madison Lake CE-QUAL-W2 model (Smith and others, 2017) and the updated CE-QUAL-W2 model presented in this report (table 2). In addition to water quality constituents, an initial water-surface elevation and water temperature were also set to the measured value at the simulation start for both lakes.

Table 4. Initial constituent concentrations for the Madison Lake CE-QUAL-W2 model: 2014 calibration and 2016 validation runs.

Chemical Boundary Conditions

Each simulated water-quality constituent, including total dissolved solids, nutrients, silica, iron, organic matter, and inorganic carbon, must have a daily concentration value for all inflow tributaries (including distributed tributary flow). Because of the low frequency of discrete water-quality samples, a mean daily concentration value was linearly interpolated between the discrete samples for each inflow tributary or a single concentration was applied for the entire model run for each inflow tributary. The distributed tributary inflow constituents were based on the mean concentrations for the northeast inlet site for branch 1 and the southeast inlet for branch 2.

Organic matter concentrations were back-calculated from the total Kjeldahl nitrogen concentration minus the dissolved ammonia concentration, with an additional calculation based on a linear relation between streamflow and the particulate organic nitrogen to total organic nitrogen ratio (Smith and others, 2014). Organic matter concentrations were then further divided into four separate pools, as required by the CE-QUAL-W2 model (Cole and Wells, 2015): labile dissolved, refractory dissolved, labile particulate, and refractory particulate, with dissolved and particulate pools separated into labile and refractory at 30 and 70 percent, respectively.

Model Parameters

Numerous CE-QUAL-W2 models have shown that the default hydraulic parameters are robust across different hydrologic settings (Cole and Wells, 2015). Most of the default hydraulic parameters that control the hydrodynamics and heat exchange provided within CE-QUAL-W2 or the CE-QUAL-W2 manual (Cole and Wells, 2015). The density control for all inflows in the

model allowed for the water inflows to match up with the layers within the lake that corresponded to the inflow density.

For the water-quality algorithms, over 200 parameters control the constituent kinetics. An advantage of CE-QUAL-W2 is the modular design that allows for control of the water-quality constituents by adding specific subroutines. Many of these parameters were optional depending on the inclusion of groups such as epiphyton, zooplankton, macrophytes, and algae. As with the hydraulic and heat exchange parameters that control the hydrodynamics, all the parameters were time and space invariant. The option exists to vary some parameters, such as the extinction coefficient of water; however, not enough data were collected to justify dynamic control of any parameters. All the parameterization for the updated Madison Lake CE-QUAL-W2 is available through the CE-QUAL-W2 control file, available in the ScienceBase archive (Smith, 2019). Many of the parameters were left as the default values, whereas the remaining parameters were adjusted during the calibration process. Guidance for adjusting selected parameters also came from other USGS CE-QUAL-W2 model applications (Bales and Robbins, 1999; Flowers and others, 2001; Green and others, 2003; Sullivan and Rounds, 2004; Galloway and Green, 2006; Galloway and others, 2008; Sullivan and others, 2011; Smith and others, 2014; Cole and Wells, 2015).

Model Calibration and Validation

The degree of fit between the simulated results and measured lake values was considered during model calibration. The two values utilized to evaluate the degree of fit were the MAE and the RMSE. The MAE, computed by equation 3 (for example, see usage in Smith and others,

2017 and Smith and others, 2018), is a measure of the mean difference between the simulated (model) value and the measured value:

$$\text{MAE} = \frac{1}{n}\sum_{i=1}^{n}|simulated\ value - measured\ value| \quad (3)$$

where

n is the number of observations.

For example, an MAE of 1.0 milligram per liter (mg/L) for DO means that the simulated value is on average within 1.0 mg/L of the measured DO value. The RMSE is a slightly different metric in that it indicates the amount of deviation between the simulated value and the measured value. The RMSE, as computed by equation 4 (for example, see usage in Smith and others, 2014), gives the deviation between the simulated value and the measured value approximately 67 percent of the time:

$$\text{RMSE} = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(simulated\ value - measured\ value)^2} \quad (4)$$

where

n is the number of observations.

The degree of fit between the simulated and measured outlet water-surface elevation was only considered during the initial water balance calibration for each year. The early focus on the water balance made certain that the amount of flow in and out of the lake is properly considered before the subsequent water temperature, DO, algae, and nutrients followed, using the MAE and RMSE metrics.

Refined calibration focused on the vertical profiles of temperature and DO (fig. 1; table 1). Additionally, the refined calibration step included the water-quality parameters highlighted previously (ammonia, nitrate plus nitrite, total Kjeldahl nitrogen, total phosphorus,

orthophosphate, and chlorophyll *a*). Final refinement of model parameters was achieved with the realization of low MAE and RMSE values for most of the target constituents. Values of MAE and RMSE below 1 degree Celsius (°C) and <1 mg/L for DO were ideal but not possible for every profile. The MAE and RMSE values for other water-quality parameters were operationally defined by other USGS reports utilizing CE-QUAL-W2, such as Smith and others (2014), which included Lake Carlos, Elk Lake, and Trout Lake and Smith and others (2017), which included the original Madison Lake model and Pearl Lake, another Sentinel Lake. Most model runs included one adjustment with a subsequent model run to characterize the parameter sensitivity.

Water Balance

Before the water temperature and water-quality calibration could proceed, the differences between the simulated and measured water-surface elevations were rectified for 2016, as the 2014 water balance was completed during the initial model calibration (Smith and others, 2017). Similar to the calibration strategy for 2014 (Smith and others, 2017), the initial attempt to achieve a water balance for Madison Lake used the two gaged tributaries, the northeast inlet and southeast inlet (table 1), as the sole inflows for the calibration period of March 30–November 23, 2016; however, the simulated water-surface elevation was below the measured water-surface elevation, which indicated that additional water sources to the lake existed, such as ungaged tributaries and groundwater.

Two different distributed tributary flows were added iteratively for each of the two water bodies of Madison Lake to include unaccounted inflow. In addition to unaccounted inflows and groundwater flow, the 2016 water balance included a higher percentage of distributed tributary flow compared to the ratio of branch inflows to distributed tributary flow in 2014. For 2014, approximately 15 percent of the total flow during the calibration period (May 15–November 1,

2014) was from the distributed tributary flow. Alternatively, approximately 51 percent of the total flow during the validation period (March 30–November 23, 2016) was from the distributed tributary flow. A comparison between daily flows calculated directly from the transducer water levels to daily flows calculated from the 2014 Lake Level Minnesota Monitoring Program yielded an R^2 of 0.98 and 0.97, respectively, for branch 1 inflow and branch 2 inflow, demonstrating that using the lake water-surface elevation rather than transducer water levels was an appropriate technique for 2016. However, the lack of sub-daily resolution combined with the possibility of bias from using the lake water-surface elevation led to the higher percentage of distributed tributary flow for 2016. The water balance was still rectified for 2016, with MAE and RMSE values of <0.03 m for the simulated water-surface elevations.

Temperature

The simulated water temperature results from both the calibration (2014) and validation (2016) were compared to vertical profiles of lake water temperatures at the southwest deep point site, generally collected during MNDNR water-quality sampling trips. Comparisons to the same 2014 profiles were made for the original Madison Lake model calibration (Smith and others, 2017), but new comparisons were warranted to make sure that the updated model still adequately captured the lake's temperature dynamics.

A total of eight dates from 2014 are shown in figure 2, and nine dates from 2016 are shown in figure 3. For 2014, the model consistently attained MAE and RMSE values <1.0 °C for all eight dates, with several values <0.5 °C. The temperature calibration did not differ much from the original Madison Lake model. For the combined vertical profiles, the MAE and RMSE values were 0.55 and 0.70 °C, respectively (table 5), compared to the original Madison Lake model with 0.53 and 0.68 °C, respectively, so almost identical between the original calibration

and the updated model. As with the original model, the location and slope of the simulated thermocline matched the measured thermocline. For 2016, the model also consistently attained MAE and RMSE values <1.0 °C for eight of nine dates, with one exception early in the year slightly simulated too warm. For the combined 2016 vertical profiles, the MAE and RMSE values were 0.67 and 0.81 °C, respectively (table 5).

Figure 2. Simulated and measured water temperature for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minn. for eight dates in 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Figure 3. Simulated and measured water temperature for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minn. for nine dates in 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

Table 5. Summary of mean absolute error (MAE) and root mean square error (RMSE) values for calibration (2014) and validation (2016) runs for Madison Lake at Madison Lake southwest deep point near Madison Lake, Minnesota (also known as southwest deep point).

The influential boundary conditions that affect water temperature included sediment temperature, initial lake water temperature, and inflow water temperature. The temperature substitution for 2016, using air temperatures to simulate water temperatures, did not seem to have a large effect on the 2016 model validation given the low MAE and RMSE values. Meteorological effects include air temperature, wind velocity, wind direction, and solar radiation. Wind sheltering effects, as augmented through the wind sheltering coefficient (WSC) file, were still important for the 2016 validation. The WSC input file considers boundary effects on wind mixing, such as topography and shoreline tree cover, with a range from 50 to 72 percent

of the full wind value for 2016 and 50 to 64 percent for 2014, except for a value of 100 percent at the beginning of both years. Several hydraulic and thermal parameters also affect water temperature. Most of these parameters were left identical to the original Madison Lake model, with the exception of an increase from 1.5 to 2.0 for the CBHE coefficient that controls sediment heat exchange and a slight increase in the sediment temperature by 0.3 °C. One other set of critical parameters altered for the updated model were the short wave solar radiation extinction coefficients due to various algal groups (EXA1, EXA2, EXA3, EXA4, EXA5), which were all adjusted from 0.1 to 0.2 for each group, the recommended default CE-QUAL-W2 value (Cole and Wells, 2015; Smith, 2019).

Dissolved Oxygen

Accurately simulating DO is critical in determining the size of summer habitat refugia for important game fish species because their thermal requirements often confine them below the epilimnion where they are vulnerable to mass die offs because of a lack of DO. Even cool-water and warm-water fish species have upper thermal tolerances. If these fish subsist for long periods in warmer waters in combination with low DO levels, even noncold-water fish can be subject to die offs (Fang and others, 1999) based on oxythermal constraints.

Within the CE-QUAL-W2 model, many sources and sinks are available for DO, which makes DO likely the most complicated constituent to model. Sources include inflows, atmospheric exchange across the lake surface, and algal photosynthesis (Cole and Wells, 2015). Sinks include decay mechanisms such as bacterial respiration of dissolved and solid-phase organic matter (labile and refractory) in the water column and lake sediment. Other simulated sinks include algal respiration, macrophyte respiration, ammonia and nitrite nitrification, and

exchange back to the atmosphere and into sediments (Cole and Wells, 2015). The values used for these parameters are part of the CE-QUAL-W2 control file (Smith, 2019).

With varying success, the model captured the trajectories of DO concentrations at multiple depths over time, which indicated that the model was accurately simulating the underlying metabolic processes in each lake. For the DO calibration (2014) and validation (2016), the principal calibration targets were the lake profile data from the southwest deep point site, available from monthly vertical DO profiles collected by MNDNR personnel during water-quality sampling trips. Generally, DO measurements were recorded for each meter below water surface. Simulated and measured DO concentrations are shown for a total of eight dates for 2014 (fig. 4), and a total of nine dates for 2016 (fig. 5). Overall, the simulated DO concentrations tracked the measured concentrations from the southwest deep point site.

Figure 4. Simulated and measured dissolved oxygen concentration for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minn. for eight dates in 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Figure 5. Simulated and measured dissolved oxygen concentration for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minn. for nine dates in 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

Similar to water temperature, the same 2014 DO profiles were compared for the original Madison Lake model calibration (Smith and others, 2017), but new comparisons were warranted to make sure that the updated model still adequately captured the lake's DO dynamics. Generally, where the greatest change in DO occurred, the simulated concentrations matched the depth and slope of the measured concentrations. Also, compared to the earlier model

comparisons, the same observations and conclusions from Smith and others (2017) can be made for the updated model calibration (2014). For example, the maximum midwater DO maximum between 3 and 6 m on May 29, 2014, showed little difference between the simulated and measured values as reflected with the low MAE and RMSE values (<0.6 mg/L). Also similar to the original calibration, the minimal hypolimnion oxygen levels starting from around June 3, 2014, were maintained until sometime between the August 26 and September 17 DO profiles. By September 17, the lake began to overturn, as shown for DO (fig. 4) and lake water temperature (fig. 2). The simulated DO concentrations for September 17 were greater at depth, so the lake overturn started to occur 7 to 10 days earlier in the model than the measured lake values. For the combined vertical DO profiles, the MAE and RMSE values were 0.86 and 1.22 mg/L, respectively (table 5), compared to the original Madison Lake model with 0.68 and 1.15 mg/L, respectively,

For the 2016 validation, the simulated DO still captured the DO trajectories, but did not attain as low MAE and RMSE values; for the combined 2016 vertical profiles, the MAE and RMSE values were 0.91 and 1.46 mg/L, respectively. However, despite these higher values, the model simulated the general DO trajectories throughout the year from May to September with the nine profiles. The largest discrepancy between the measured and simulated results was the lack of simulated DO supersaturation in the shallower mixed layer. This discrepancy could be caused by the lack of simulated algal growth earlier in the year, such that an earlier algal bloom missed or not captured with the model would have caused larger simulated DO values in the shallow mixed layer. Alternatively, the model did an adequate simulation of the hypolimnetic oxygen minimums, both with depth and timing. Overall, the 2016 model validation did show that

the model could capture DO dynamics for two years with different algal community dynamics and a different total algal biomass.

As far as parameter changes with the updated model, significant changes were made to the algal community dynamics, as discussed in the "Algae" section. Since algal dynamics played a large part in controlling the DO dynamics, those changes that could have affected DO are discussed separately. However, other major parameters that can control limnological DO concentrations, such as the decay rates of different organic matter pools, were unaltered from the original Madison Lake model. Also, the sediment oxygen demand (parameter SOD) was unaltered from the original Madison Lake model (2.5 mg/L). The equation for calculating reaeration was changed from equation #9 to equation #3, upon a determination that equation #3 is more appropriate for water bodies with lower flow-through rates.

Algae

The previous model version simulated phytoplankton into four general algal communities or groups: (1) bacillariophyta and crysophyta (diatoms); (2) chlorophyta (green algae); (3) cyanophyta (blue-green algae); and, (4) haptophyta and cryptophyta (flagellates). For this previous version, the paradigm of four general algal communities or groups was pursued rather than a more diverse modeling regime. Algal group dynamics within CE-QUAL-W2 models are sensitive and the uncertainty in model parameterization beyond four different algal groups can be problematic (Cole and Wells, 2015).

For the updated model, the cyanophyta group has been divided into two groups: a nitrogen-fixing cyanophyta group, generally representative of *Anabaena*, *Dolichospermum*, and *Cylindrospermopsis*, and a non-fixing, buoyant cyanophyta group, generally representative of *Planktothrix*, *Microcystis*, and *Woronichinia*. This enhancement was added to improve the

model's predictive capacity of cyanophyta (also known as cyanobacteria) blooms and focus on populations known to exist in Madison Lake. As the original model and the measured 2014 algal and chlorophyll *a* data suggested, a mid- to late-summer dominance by cyanophyta existed in 2014 and again in 2016. Furthermore, an analysis of algal data going back to 2013 suggested cyanophyta dominance in the mid- to late-summer months (Minnesota Pollution Control Agency, 2018).

However, as noted above, simulating beyond three to four algal groups within CE-QUAL-W2 can be challenging. The model updates to the algal community sub-module required substantial adjustments to many of the parameters governing algal growth and senescence for all five groups. Also, the lack of measured Madison Lake algal data beyond algal counts and biomass did not adequately constrain model parameterization independently. Instead, the guidance for determining algal growth patterns was mainly provided by other CE-QUAL-W2 modeling efforts, such as the previous sentinel lake models (Smith and others, 2014; Smith and others, 2017) and the Lake St. Croix CE-QUAL-W2 model (Smith and others, 2018).

Similar to the original model, the zooplankton grazing dynamics were captured within algal specific constants such as the algal growth rate (parameter AG) and the algal mortality rate (parameter AM) (Smith, 2019). Algal growth temperature ranges (parameters AT1 through AT4) were different across all five algal groups, as were the algal growth rates (parameter AG) and the light saturation intensities at the maximum photosynthetic rate (parameter ASAT). One major change from the original Madison Lake model was the fraction of algal growth specific to temperature ranges (parameters AK1 through AK4). Rather than adjust these parameters to artificially attain a better model fit, these parameters were set to the CE-QUAL-W2 default rates, given the lack of further information on these parameters. Other algal growth parameters were

also set with more uniformity across the different groups, given the lack of specific information for the Madison Lake algal groups. With the updated model, the stoichiometric equivalences used for determining the nutrients in the algal biomass, such as phosphorus, nitrogen, and carbon, were also set with more uniformity across the different algal groups and generally closer to the CE-QUAL-W2 default rates. Most importantly, the ratio between algal biomass and chlorophyll *a* was adjusted to 0.05 across all groups rather than different ratios between the groups, and this same adjustment was made for the transformation of the measured algal data conversions to biomass. For the original model, this parameterization was out of sync between both the different groups and the measured data.

Overall, the simulated distribution of the five algal groups (fig. 6), instead of four algal groups, was improved for the updated Madison Lake model from the original model (Smith and others, 2017). The algae MAE and RMSE values were generally not as meaningful statistics for calibration and validation; however, the MAE and RMSE values dropped across all of the five algal groups compared to the algal group simulations for the original model. The largest change in the MAE/RMSE values was for cyanophyta. In the original report, cyanophyta (referred to as "blue-green algae") had MAE and RMSE values of 1.81 and 1.95 mg/L, respectively, whereas the updated model split this group into two groups with MAE and RMSE values <1.1 mg/L.

Figure 6. Simulated and measured algal group distributions (diatoms, green algae, fixing cyanophyta, non-fixing (buoyant) cyanophyta, and flagellates) for the 1-meter depth at Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014.

For the 2014 calibration year, diatoms were the first group to peak, as shown with the simulated and measured values. Diatoms commonly peak earlier in the year (Sigee, 2005). The simulated diatom values peaked by the middle of May and then approached 0 mg/L by late June.

For the measured values, a second peak occurred in late July and again in mid-September; however, the model did not capture these dynamics. Several factors controlled the lack of simulated diatom growth beyond early June. Simulated algal growth favored the other groups beyond the early part of the year. The temperature range for diatom growth was lower than the other algal groups, so once the lake warmed by early June, the diatoms were outcompeted by the other algal groups. The larger algal light saturation intensity for diatoms, which affected optimal algal growth, limited growth once the lake had greater concentrations of inorganic and organic suspended sediments, macrophytes, and algal biomass and thereby blocked the light. Combined with a larger settling rate, the diatoms would settle to a depth in the lake unfavorable for optimal light saturation set in the model.

The next group to succeed diatoms in 2014 were the fixing cyanophyta (fig. 6). As mentioned in the first Madison Lake modeling report, splitting up cyanophyta into at least two groups was warranted since the lake's nitrogen limitation at this time of year combined with warmer temperatures favored cyanophyta capable of fixing nitrogen. As less emphasis was placed on other factors compared to the original model, the wider temperature range for maximum algal growth (between 20 and 30 °C) combined with a lower algal light saturation intensity of 75 watts per square meter (W/m^2) compared to 120 W/m^2 for the diatoms allowed for more growth of fixing cyanophyta (*Anabaena*, *Dolichospermum*, and *Cylindrospermopsis*).

By mid-July, the light saturation and favorability for fixing cyanophyta growth began to wane in favor of the non-fixing cyanophyta group (*Planktothrix*, *Microcystis*, and *Woronichinia*). This algal group was also differentiated from the fixing cyanophyta group through a higher light saturation intensity (125 W/m^2) and for high buoyancy by adjusting the settling rate to 0. Throughout the remainder of the summer into September and October, the non-fixing

cyanophyta continued to grow, whereas the fixing cyanophyta group died off. The data supported this simulated growth, except the cyanophyta continued to have higher biomass in September for the measured data. This differentiation was difficult to simulate in the 2014 calibration, but for the 2016 validation the dynamics seemed to be closer to the measured data.

The other two algal community groups, green algae and flagellates, had similar growth rates and patterns for the simulated and measured values in 2014. The two groups were distinguished from each other in that the green algae showed a mid-August peak, whereas the flagellates showed a September peak (fig. 6). The maximum algal growth temperature range was similar for both groups, with 20 to 25 °C and 24 to 28 °C for the green algae and flagellates, respectively. Of the five groups, the flagellates had the lowest algal light saturation intensity of 20 W/m^2. Otherwise, as shown in the CE-QUAL-W2 control file (Smith, 2019), the parameterization of the two groups was similar for growth rate, algal mortality (parameter AM), and algal settling rate; and both groups had the same algal half-saturation constants for nitrogen- and phosphorus-limited growth (parameter AHSP).

Overall, the simulated algal biomass concentrations for the calibration were similar to measured algal biomass concentrations with the exception of the previously described deviation for diatoms later in the year. Also, the simulated cyanophyta concentrations did not match the large measured values in August and September, although the combined simulated growth of the fixing and non-fixing cyanophyta groups did match the measured values. Part of the likely discrepancy is that isolated sample points in time might not capture the general trend.

Figure 7. Simulated and measured chlorophyll *a* concentrations for the 1-meter depth at Madison Lake southwest deep point near Madison Lake, Minn. (segment 7) in Madison Lake, May 15 to November 1, 2014.

The chlorophyll *a* concentration data were used to help interpret if the overall magnitude of the algal group composition was in the correct range. Photosynthetic pigments, such as chlorophyll *a*, are accepted in the literature as surrogates for algal biomass given the large expense of measuring algal biomass directly (Lindenberg and others, 2008). Simulated and measured values of the chlorophyll *a* concentrations are shown for the Madison Lake southwest deep point site in figure 7 (segment 7, fig. 1). Measured chlorophyll *a* data were collected in the surface layer at approximately 1 m below the water surface as part of the monthly MNDNR water-quality sampling trips. Overall, the simulated values were a good approximation of the measured values, with the exception of the high simulated chlorophyll *a* value compared to the low measured chlorophyll *a* concentration in October.

Considerable differences occurred for the 2016 validation for both algal growth (fig. 8) and chlorophyll *a* (fig. 9). Unlike 2014, the simulated growth of green algae and flagellates occurred much earlier in the year than suggested by the measured data (fig. 8). Simulated diatom growth did occur, but at a much lower rate than 2014. The two cyanophyta groups also started to grow earlier in the year, but this growth was also supported by the measured data. The simulated fixing cyanophyta peaked in August at the same time as the measured fixing cyanophyta, whereas the non-fixing (buoyant) cyanophyta showed the same growth curve as 2014 except with an earlier start. The simulated non-fixing cyanophyta overall seemed to grow faster than the measured non-fixing cyanophyta, but this could also be due to a model limitation. The CE-QUAL-W2 model in general might have a difficult time distinguishing different nitrogen sources (organic versus non-organic sources), and in reality, the fixing cyanophyta group does not necessarily fix nitrogen at all times since nitrogen fixation is a highly energetic process (Maier, 2004). Within the CE-QUAL-W2 modeling framework, it is not possible to have the same group

set-up as using nitrogen fixation only part of the time, so this interplay of nitrogen fixation versus non-nitrogen fixation among different cyanophyta groups can be difficult to model. As with 2014, the chlorophyll *a* concentration data were used to help interpret if the overall magnitude of the algal group composition was in the correct range for 2016. Similar to 2014, the general trend was captured by the simulated chlorophyll a values but were lower than the measured data in the later summer mainly because the model did not capture the high measured non-fixing cyanophyta growth.

Figure 8. Simulated and measured algal group distributions (diatoms, green algae, fixing cyanophyta, non-fixing (buoyant) cyanophyta, and flagellates) for the 1-meter depth at Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016.

Figure 9. Simulated and measured chlorophyll *a* concentrations for the 1-meter depth at Madison Lake southwest deep point near Madison Lake, Minn. (segment 7) in Madison Lake, March 30 to November 23, 2016.

Macrophyte Growth

The macrophyte growth model was run for the 2014 calibration and 2016 validation, given the high amount of documented macrophyte growth for Madison Lake (Lindon and others, 2010). As noted in Smith and others (2017), most of the macrophyte growth parameters were kept at default rates except for the maximum macrophyte growth rate (MG), the light saturation intensity at maximum photosynthetic rate (MSAT), and the fraction of macrophyte biomass that is converted to particulate organic matter after macrophytes die (MPOM). Additionally, two adjustments were made to the macrophyte growth module from the original Madison Lake CE-QUAL-W2 model. For the original model (Smith and others, 2017), the fractions of phosphorus

(PSED) and nitrogen uptake (NSED) from sediments were set equal to 1, but for the updated model both parameters were adjusted to 0.5.

Nutrients

Nutrients are controlled by many processes, such as inflow loads, algal production, and organic matter decay rates (Cole and Wells, 2015). One of the most important controls is the amount of nutrients (loads, determined in the model as concentration multiplied by streamflow and a unit conversion factor) contributed by the inflows, which are different for both lakes. Madison Lake had a larger flux of nitrate earlier in 2014 season with a larger flux of ammonia later in the year, whereas in 2016 for the model validation the lake did not have the initial large flux of nitrate mid-summer or the large mid-summer flux of ammonia. It is known that loading into lakes such as Madison Lake would be expected to vary across ecoregions, with the soil fertility in the contributing drainage basin, and across different land uses (for example, row-crop agriculture compared to deciduous forest). However, the data suggested interannual variability that the model must be able to account for to reasonably simulate the nutrient conditions.

In-lake processing of the nutrients is the major factor controlling nutrient concentrations. An in-depth discussion of the sources and sinks for Madison Lake was given in Smith and others (2017). In summary, Madison Lake has fairly small flows from two different inflows and seems to have considerably large groundwater sources relative to surface inflows. Agricultural land use is the dominant land use at approximately 50 percent for the drainage areas for Madison Lake with only 2 percent forest cover (Lindon and others, 2010), and the drainage basin to lake area ratio for Madison Lake is 4:1. Generally, basins with a smaller percentage of forest or other undeveloped land cover combined with a larger ratio of agricultural land use will have higher nutrient loads (U.S. Geological Survey, 1999).

As with water temperature and dissolved oxygen, new comparisons were warranted to make sure that the updated model still adequately captured the lake's nutrient dynamics for 2014 despite earlier calibration efforts (Smith and others, 2017). The focus for evaluating the model calibration and validation was three constituents of nitrogen and two constituents of phosphorus: nitrate plus nitrite, ammonia, total Kjeldahl nitrogen, orthophosphate, and total phosphorus. For purposes of comparing simulated and measured concentrations, total Kjeldahl nitrogen was classified as the concentration of nitrogen present in ammonia, nitrate plus nitrite, and organically bound nitrogen (in living algal biomass and all organic matter pools). For purposes of comparing simulated and measured concentrations, total phosphorus was classified as the concentration of phosphorus present in orthophosphate and bound up in organic matter (in living algal biomass and all organic matter pools). The primary tools for evaluating the degree of fit for the nutrients were the MAE and RMSE values (table 5) and all comparisons were for samples taken either from 1 m below the water surface or in the hypolimnion from the southwest deep point in Madison Lake (segment 7, fig. 1). It is worth noting that these values could often be largely offset by only one or two measured samples because of the small number of total discrete samples (five samples for 2014; 4-10 samples for 2016, depending on the constituent).

Dissolved ammonia and dissolved nitrate plus nitrite distributions in Madison Lake were largely affected by the inflows and the lake hydrodynamics. Few differences between the simulated and measured dissolved ammonia concentrations were noted at the 1-m depth from July through September, both for the calibration year (2014 – fig. 11) and the validation year (2016– fig. 12). An exception occurred for the June and October 2014 samples, which were both much higher than all the other samples. Overall, algal uptake of available ammonia was fairly rapid in the simulation and actual lake for both years, with replenishment by organic matter

decay and inflows. This process of algal uptake accounted for the lower dissolved ammonia concentrations during the middle of the simulation period for the simulated and measured values. The 2014 MAE and RMSE values for dissolved ammonia were comparable between the old calibration (Smith and others, 2017) and the new calibration (this report), with MAE and RMSE values of 0.17 and 0.33 mg/L, respectively (fig. 10; table 5), for the new calibration. The 2016 validation closely matched the measured values, which were all at the sample detection limit; the MAE and RMSE values for the 2016 validation were 0.01 and 0.02 mg/L, respectively.

Figure 10. Simulated and measured dissolved ammonia concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Figure 11. Simulated and measured dissolved ammonia concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

Simulated and measured dissolved nitrate plus nitrite concentrations are shown in figures 12 (2014) and figure 13 (2016) for the Madison Lake southwest deep point site. For nitrate, the two years were very different: in 2014, nitrate started with a high initial value whereas 2016 started relatively low. The model simulated both years with very low MAE and RMSE values (table 5), including a drop in the MAE/RMSE values with the new 2014 calibration. The improvement was due in part to an important change to the NO3S parameter which controls the nitrate sediment diffusion rate. Additionally, changes were made across the five algal groups to

the algal stoichiometry and also updates to the algal growth rates; all of these changes would affect dissolved nitrate and nitrite concentrations.

Figure 12. Simulated and measured dissolved nitrate plus nitrite concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Figure 13. Simulated and measured dissolved nitrate plus nitrite concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

Dissolved orthophosphate concentrations in the Madison Lake measured data were relatively stable for both years (fig. 14; fig. 15), except a higher measured value in October 2014. The simulated orthophosphate concentrations were considerably more variable due to the algal dynamics incorporated into the model and the cycling of nutrients through the various organic pools, algal communities, and the lake's simulated macrophyte community. At the end of both the calibration (2014) and validations (2016) runs, a steady increase in simulated dissolved orthophosphate concentrations occurred primarily because of the lack of demand by the simulated algae and macrophytes. Overall, the MAE and RMSE values were 0.02 and 0.02 mg/L, respectively, for both years (fig. 14; fig. 15; table 5).

Figure 14. Simulated and measured dissolved orthophosphate concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake,

Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Figure 15. Simulated and measured dissolved orthophosphate concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

Simulated and measured concentrations are shown for total Kjeldahl nitrogen in figures 16 and 17. The 2014 MAE and RMSE values for total Kjeldahl nitrogen were 0.35 and 0.43 mg/L, respectively (fig. 16; table 5), slightly higher than the original calibration (Smith and others, 2017) of 0.29 and 0.33 mg/L, respectively, for MAE and RMSE. The measured data indicate a dynamic range, from approximately 1.4 to 2.2 mg/L. A peak in total Kjeldahl nitrogen for the simulated values occurred in late June because of the increase in ammonia and nitrate concentrations, with a steady increase from late July through mid-September due to an accumulation in organic matter from the deterioration of algal biomass, macrophytes, and inflows. The simulated results were generally the same pattern as the measured total Kjeldahl nitrogen concentrations, except for a steady decrease in total Kjeldahl nitrogen towards the end of the simulation period (fig. 16). For the 2016 validation, the model fit was improved over the 2014 calibration with MAE and RMSE values of 0.20 and 0.24 mg/L, respectively. The simulated total nitrogen tracks the measured values throughout the simulation, and includes the same late season decline. These decreases were likely because of the overall decay of the simulated organic matter pools and the decrease in simulated total algal biomass.

Figure 16. Simulated and measured total Kjeldahl nitrogen concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Figure 17. Simulated and measured total Kjeldahl nitrogen concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

Total phosphorus (fig. 18; fig. 19) is shown for the epilimnion and hypolimnion locations since measured hypolimnion values were available for total phosphorus. In the epilimnion, the measured total phosphorus concentrations were stable but the simulated concentrations for both the calibration (fig. 18) and validation (fig. 19) were too large. The model could have been fit to match the epilimnion concentrations better but would have sacrificed the hypolimnion phosphorus model fit with measured values and would have set phosphorus at unrealistically low stoichiometric equivalents for algal biomass and organic matter. In the hypolimnion, a steady and steep increase in the simulated total phosphorus occurred from late May until mid-September to greater than 1,200 µg/L, before crashing to the baseline of less than 150 µg/L. These high values were confirmed for both years with the measured data, with a high value of 1,130 µg/L in 2014 and 815 µg/L in 2016. The likely explanation for the large phosphorus concentrations in the simulated and measured values (fig. 18; fig. 19) was the large release rates in phosphorus from the lake sediments. The MAE values for the epilimnion (1-m depth) and hypolimnion (15.5-m depth or 16.5-m depth) were 82 and 54 μg/L, respectively; the RMSE

values for the epilimnion (1-m depth) and hypolimnion (16.5-m depth) were 86 and 69 μg/L, respectively (fig. 26; table 5). The large drop in total phosphorus coincides with the turnover of Madison Lake and the mixing of all the lake water, which redistributed the concentrated total phosphorus to the entire lake volume.

Figure 18. Simulated and measured total phosphorus concentrations at 1 meter and 16.5 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Figure 19. Simulated and measured total phosphorus concentrations at 1 meter and 15.5 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

Phosphorus Loads

Monthly total phosphorus budgets were calculated for the updated Madison Lake model for 2014 and 2016 (table 6), and included phosphorus subdivisions of external phosphorus load derived from organic matter, external phosphorus load derived from orthophosphate, and internal phosphorus load released by zero-order sediment release. Additionally, monthly phosphorus budgets were calculated for the original Madison Lake model (Smith and others, 2017) and two sensitivity analyses completed in the same report, both with a 20 percent variation in the incoming dissolved orthophosphate load (20 percent increase, 20 percent decrease). Phosphorus budgets included external sources from the two tributaries, the distributed tributary flow (unaccounted surface flow, groundwater flow), and internal phosphorus loading from sediment

release. Negative numbers in the table denote a loss term due to net export of phosphorus for the distributed tributary flow.

Table 6. Summary of phosphorus loading for updated Madison Lake model (2014, 2016), original Madison Lake model, and two phosphorus loading scenarios, according to load estimates and internal CE-QUAL-W2 calculations. Negative terms denote a loss term due to the net export of phosphorus (from the distributary tributary flow).

In comparisons between the 2014 and 2016 model runs (updated model), the 2016 model simulation had a higher overall total phosphorus budget. Even if only considering the periods of overlap (May through October), 2016 had approximately 25 percent more total phosphorus loading than 2014. More precipitation, and therefore higher flows, occurred for 2016 than 2014, and was the primary driver of the increased loads as the limited concentration data for the two different years were similar (Smith, 2019). When comparing the monthly data, the highest 2014 phosphorus loads occurred in June (2,750 kilograms) before dropping throughout the rest of the summer (fig. 20), whereas the 2016 monthly phosphorus loads stayed relatively high from June to October (range: 856 to 1,783 kilograms per month). Most of the load for the remainder of 2014 after August was internal loading, since there was little to no flow into Madison Lake after early September.

Figure 20. Total phosphorus concentrations monthly, in kilograms per month, for the 2014 and 2016 model years for the updated model.

For the other three model runs (scenarios 3-5; table 6), the original 2014 model was included to show that there was relatively little difference in the phosphorus budget between the original and updated model. For the sensitivity scenario increasing the external dissolved

orthophosphate load by 20 percent (scenarios 4; table 6), as compared to the original model (scenario 3; table 6), increasing this external load did increase the overall phosphorous load by approximately 7 percent (5,596 kilograms versus 5,244 kilograms). Alternatively, decreasing the external dissolved orthophosphate load by 20 percent (scenarios 5; table 6) decreased the total phosphorus load by approximately 7 percent (4,904 kilograms versus 5,244 kilograms).

As a percentage of the overall load, the internal sediment release of phosphorus accounted for between 39 to 48.1 percent of the model run load (table 6). On a month-by-month basis, the internal load covers a much wider range, ranging from almost no internal sediment release of phosphorus to dominating the overall monthly load. The high percent of internal load is particularly high in the summer months when hypoxic conditions dominated the lake's hypolimnion, with low release rates occurring before hypoxia dominates the lake or after the fall lake mixing. Little difference existed between the total internal load for 2014 and 2016 (scenarios 1-2; table 6), relative to the large differences from external loads.

Model Limitations

A full understanding of model limitations is necessary to better evaluate the performance of any water-quality model. The previous Madison Lake CE-QUAL-W2 model report elaborated further on these limitations, but it is important to reiterate the limitations due to the limited datasets available for Madison Lake. The fixed number of water-quality samples to which the model is calibrated may not have captured the full range of conditions in the dynamic systems. Also, all boundary conditions datasets had limitations. Water-quality data were linearly interpolated between sampling dates. The continuous streamflow for both tributaries and one outflow location, based on applying the 2014 elevation-streamflow ratings (Smith and others,

2017 – appendix table 1–1) to continuous water levels, were unavailable for 2016 so instead these elevation-streamflow ratings were applied to the daily water-surface elevation for the lake. Inherent errors in this approach would be captured by the constructed distributary tributary flows, but this still represents an important limitation. Finally, the continuous water temperatures were also unavailable for 2016 so a substituted dataset relating air temperature to water temperature had to be substituted for the tributary inflows.

Another source of limitations was the lack of specific information on algal growth rates, mortality rates, sinking rates, and algal light saturation coefficients. Also, the full stoichiometric equivalences for the individual algal groups was not known for the Madison Lake phytoplankton. Incubation experiments on some of these parameters could help constrain model parameterization, rather than depending on a manual parameter estimation process. Literature values for these constants do exist, but they tend to show a wide range that only help constrain the parameter estimation process rather than fix the parameters to a single value. Overall, the model did show the ability to simulate the different algal groups throughout the year, but better characterization of the algal community dynamics from either field or laboratory experimentation would improve the model further.

Summary

A previously developed CE-QUAL-W2 model for Madison Lake, Minnesota, simulated the algal community dynamics, water quality, and fish habitat suitability of Madison Lake under recent (2014) meteorological conditions. Additionally, this earlier model simulated the complex interplay between external nutrient loading, internal nutrient loading from sediment release of phosphorus, and the organic matter decomposition of the algal biomass. However, the

partitioning of cyanobacteria within the modeling framework was simplified to one group and did not account for how different cyanobacteria populations are affected by light conditions, the usage of nitrogen, temperature growth ranges, and differences in settling rates. To get a better handle on the proliferation of cyanobacteria in Madison Lake, the model required updates to at least partition the cyanobacteria into a group that fixed nitrogen and a second, more buoyant cyanobacteria group, that did not independently fix nitrogen.

To address the shortcomings of simulating cyanobacteria in the earlier model, the U.S. Geological Survey (USGS), in cooperation with the St. Croix Watershed Research Station (Science Museum of Minnesota), updated the Madison Lake CE-QUAL-W2 model to better characterize cyanobacteria into two groups. In addition to updating the cyanobacteria group differentiation, the entire portion of the model that handles the simulation of algal community dynamics was updated while preserving the model's predictive capabilities for nutrients, water temperature, and dissolved oxygen. The calibration and validation of the model was done under recent meteorological conditions with large and persistent cyanobacteria blooms (2014 and 2016). Overall, the model simulations predicted the persistently large total phosphorus concentrations in Madison Lake's hypolimnion, key differences in nutrient concentrations between the two years, and cyanobacteria bloom persistence.

For calibration targets, the CE-QUAL-W2 model successfully predicted water temperature on the basis of the two metrics of mean absolute error and root mean square error. One of the main calibration tools for CE-QUAL-W2 model development was the vertical profile temperature data. Altogether, simulated Madison Lake water temperatures tracked measured water temperatures throughout the water column. In addition to water temperature, the CE-QUAL-W2 model successfully predicted dissolved oxygen concentration based on the same two

metrics of mean absolute error and root mean square error. Along with temperature, dissolved oxygen is a key metric to illustrate the accuracy of the model's calibration. Simulated vertical profiles of dissolved oxygen concentration generally matched the largest change in measured dissolved oxygen concentration, including the approximate depth, slope, and timing of large shifts.

Monthly total phosphorus budgets calculated for the updated Madison Lake model for 2014 and 2016 found that 2016 had significantly more internal and external phosphorus loading. Most of the additional phosphorus loading was from external inputs into Madison Lake rather than internal phosphorus release from sediments. The additional loading was likely from increased precipitation, and therefore higher flows, for 2016 as little to no inflow occurred into Madison Lake after September 2014. As a percentage of the overall load, the internal sediment release of phosphorus accounted for between 39 to 48.1 percent of the total external and internal phosphorus loads.

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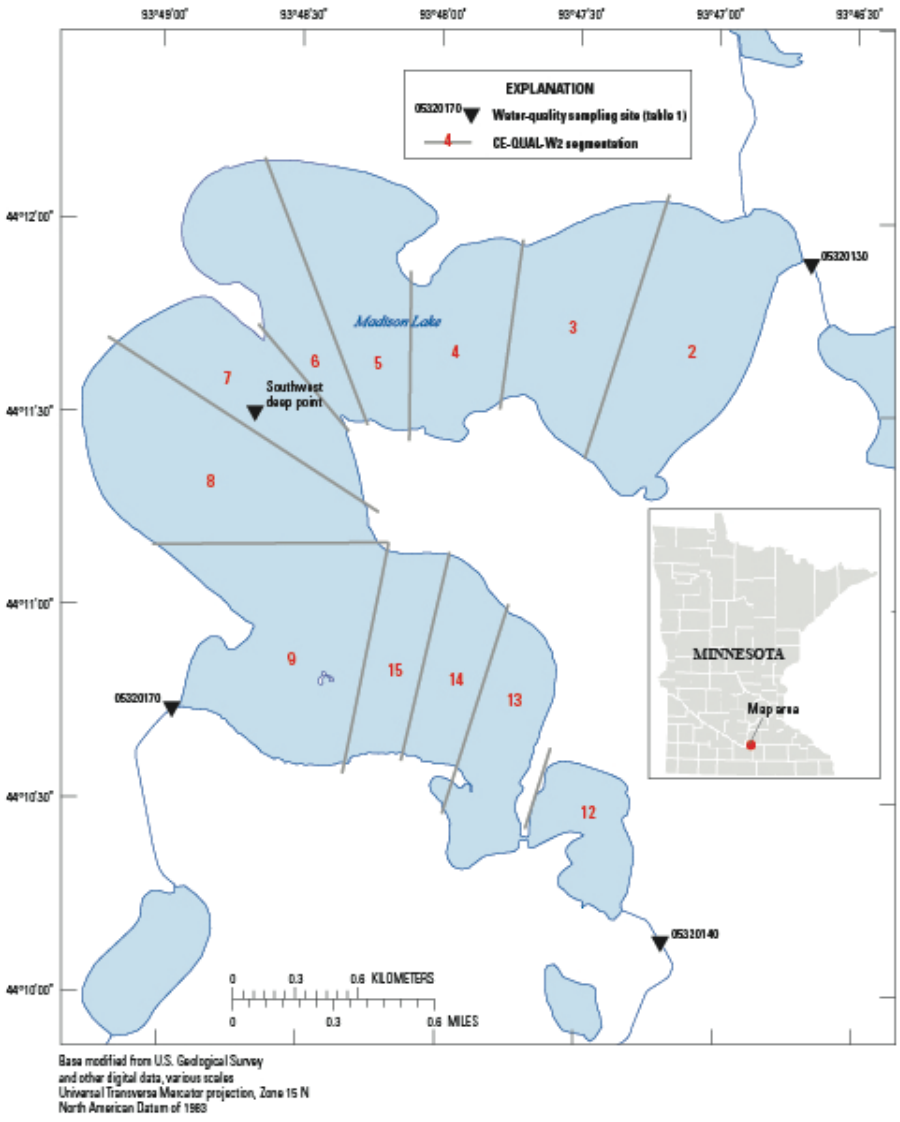

Figure 1: Map showing location of water-quality sampling sites for Madison Lake, Minnesota.

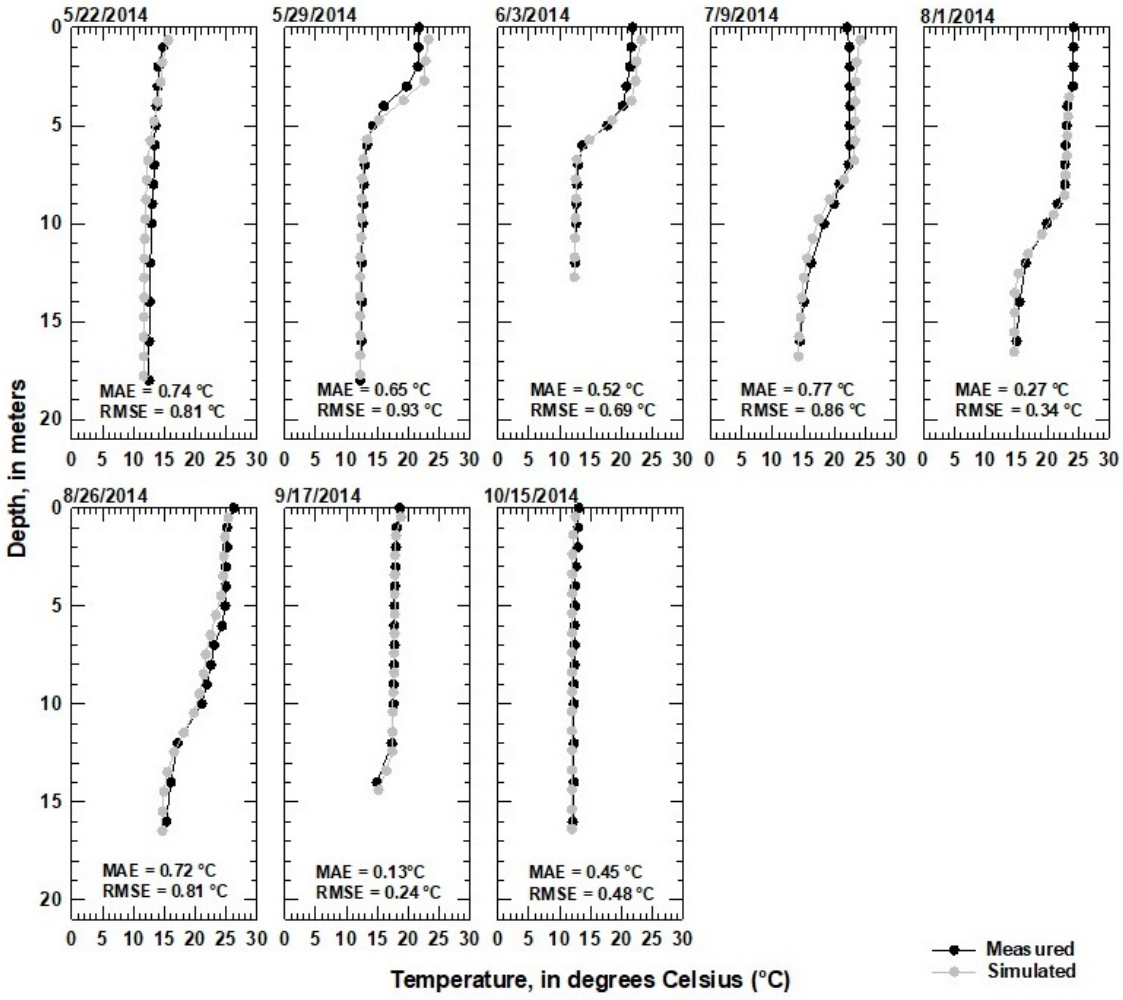

Figure 2: Simulated and measured water temperature for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minn. for eight dates in 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

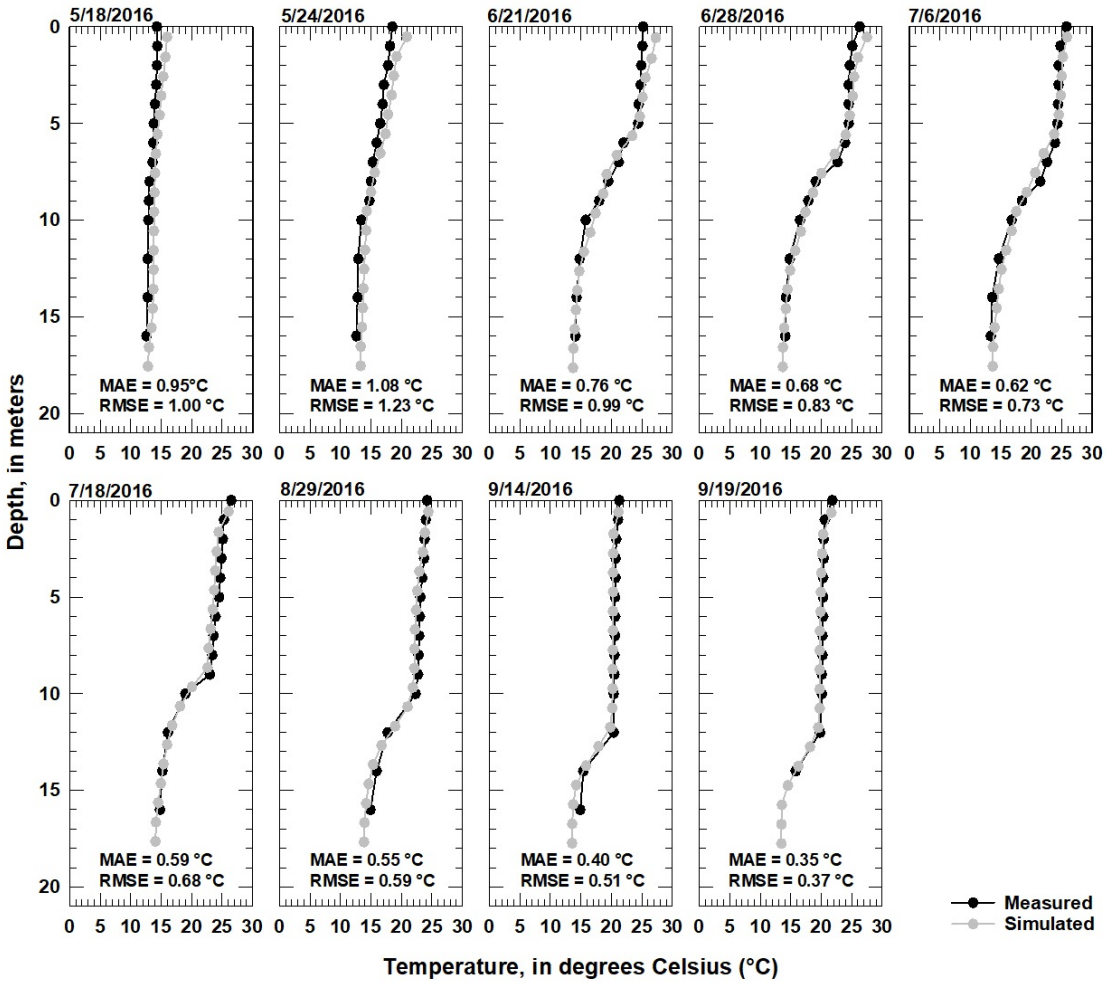

Figure 3: Simulated and measured water temperature for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minn. for nine dates in 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

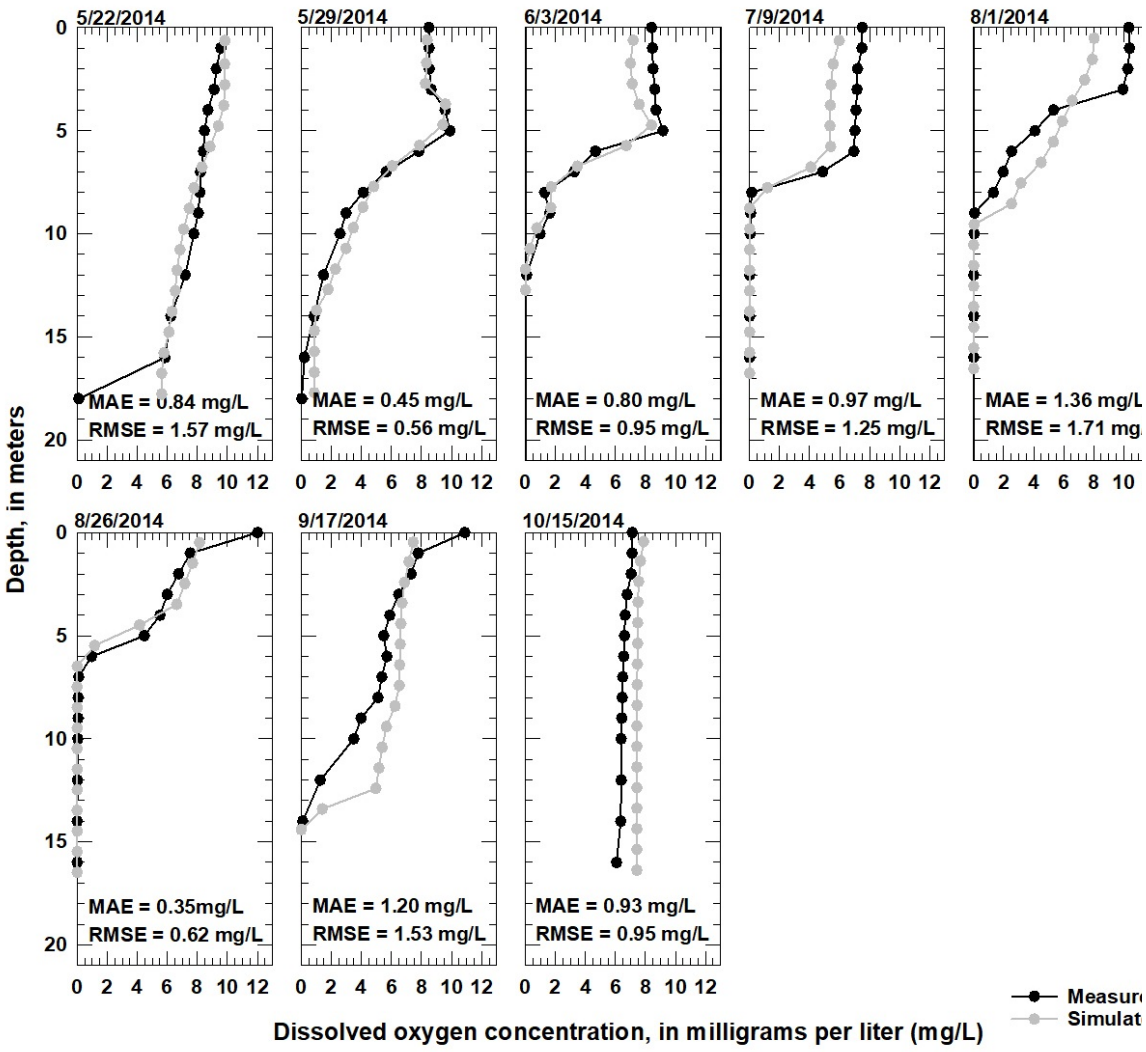

Figure 4: Simulated and measured dissolved oxygen concentration for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minn. for eight dates in 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

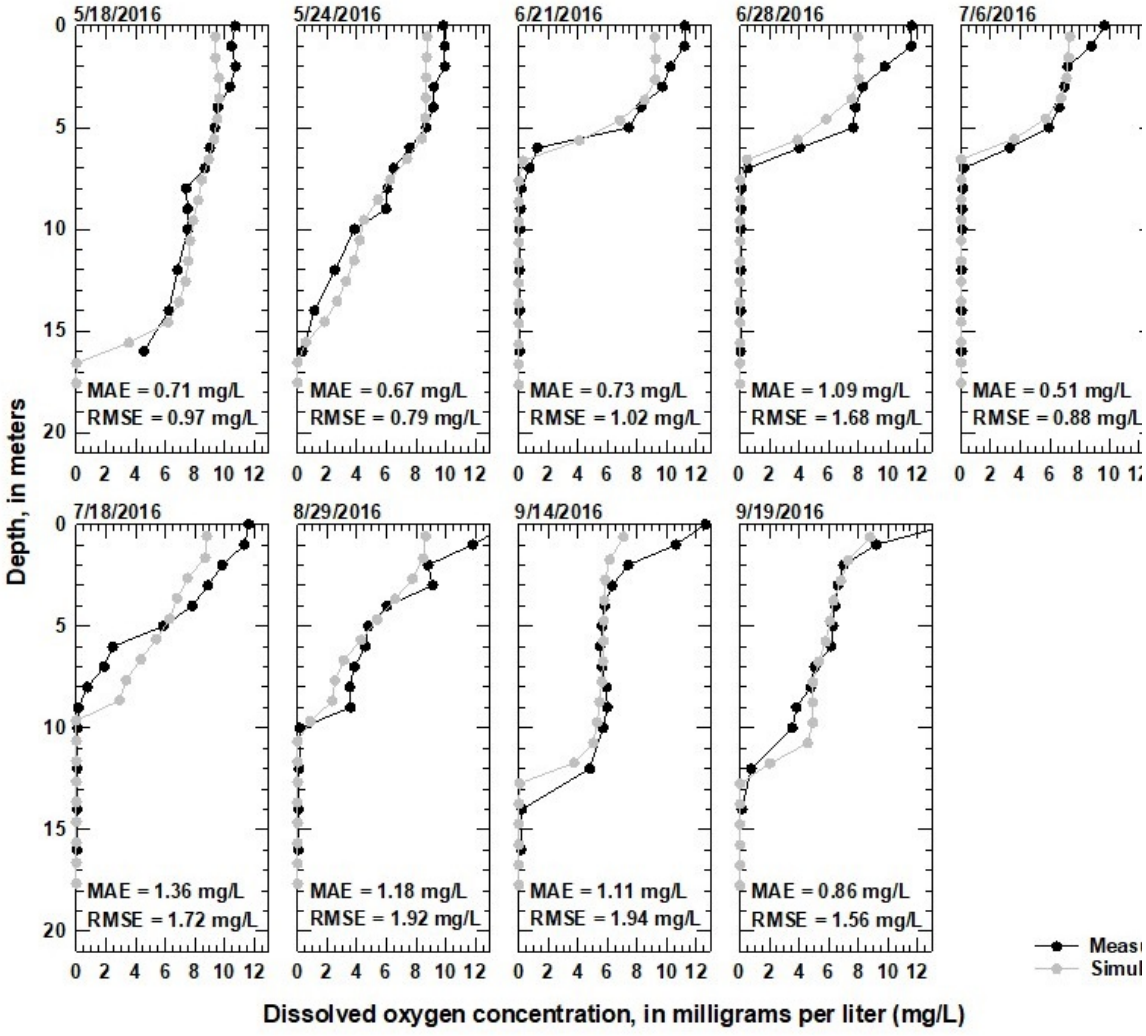

Figure 5: Simulated and measured dissolved oxygen concentration for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minn. for nine dates in 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

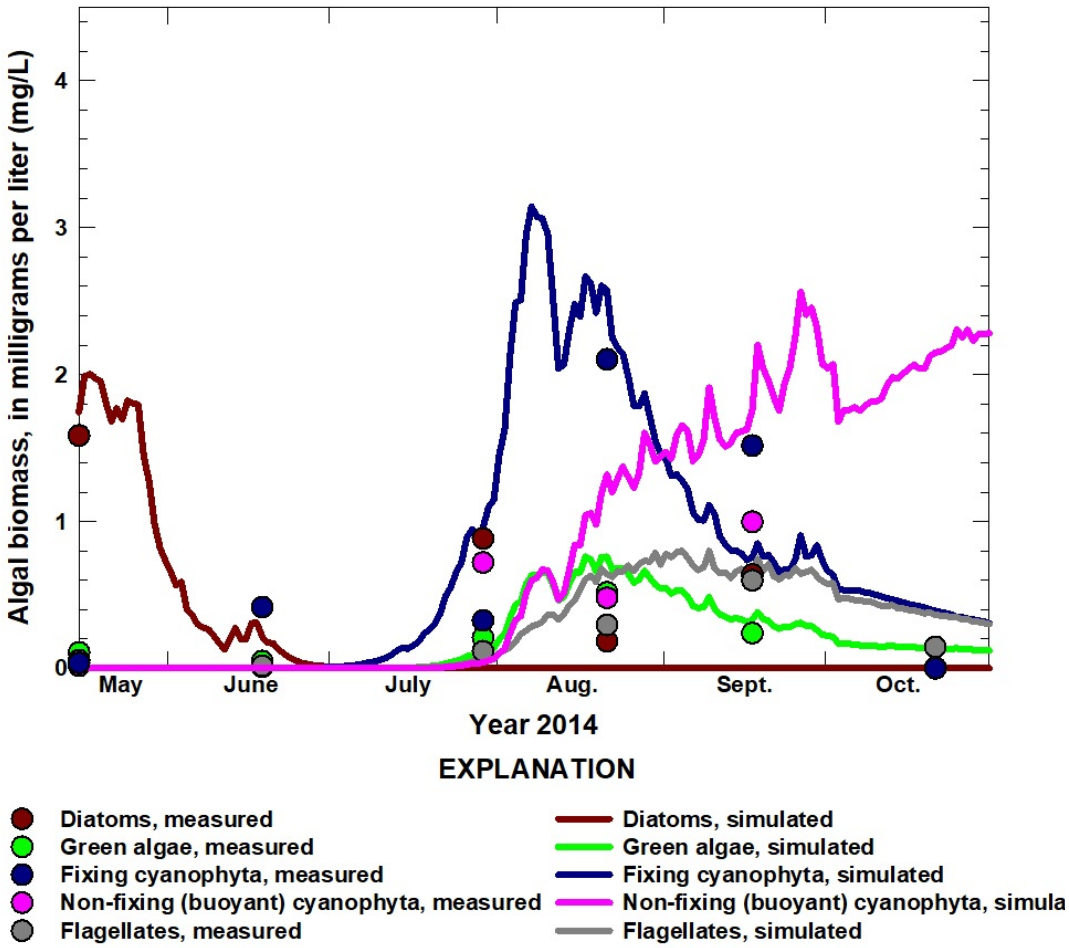

Figure 6: Simulated and measured algal group distributions (diatoms, green algae, fixing cyanophyta, non-fixing (buoyant) cyanophyta, and flagellates) for the 1-meter depth at Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014.

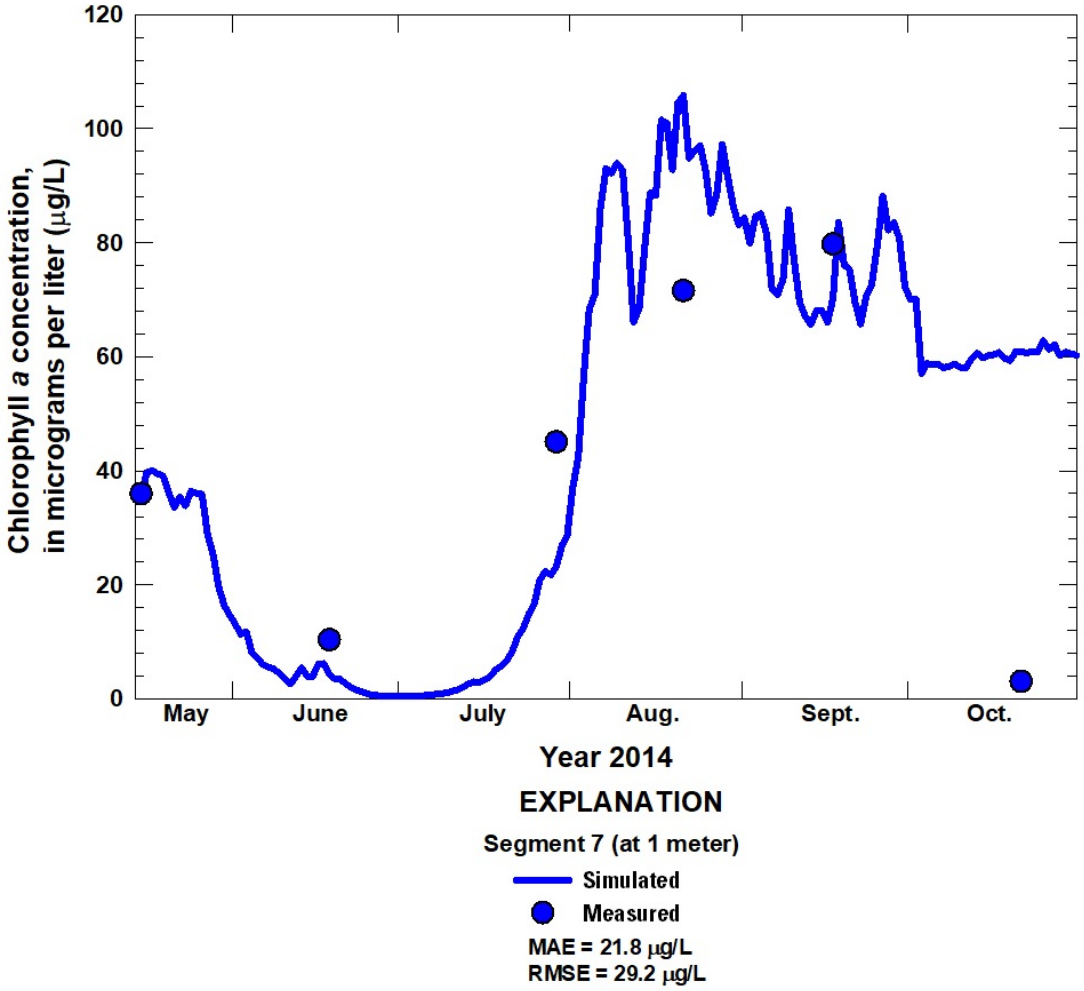

Figure 7: Simulated and measured chlorophyll *a* concentrations for the 1-meter depth at Madison Lake southwest deep point near Madison Lake, Minn. (segment 7) in Madison Lake, May 15 to November 1, 2014.

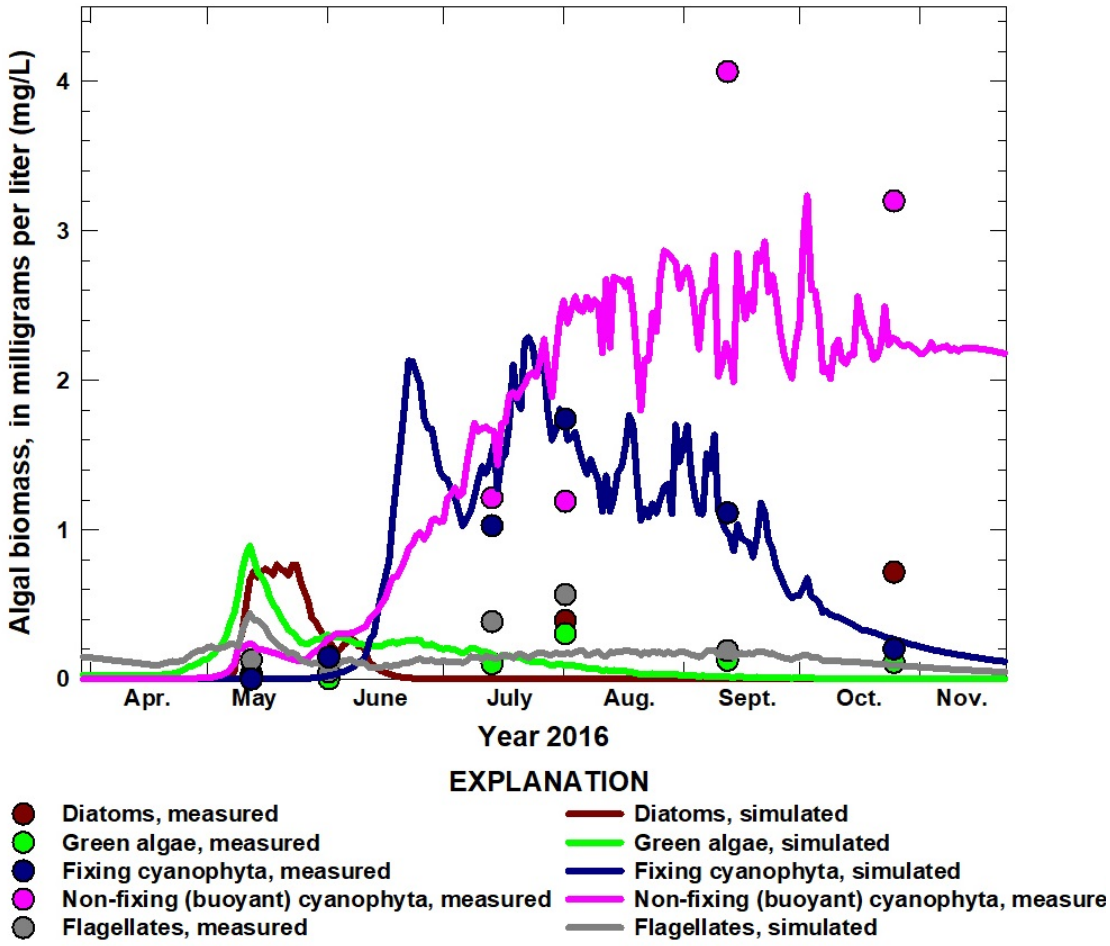

Figure 8: Simulated and measured algal group distributions (diatoms, green algae, fixing cyanophyta, non-fixing (buoyant) cyanophyta, and flagellates) for the 1-meter depth at Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016.

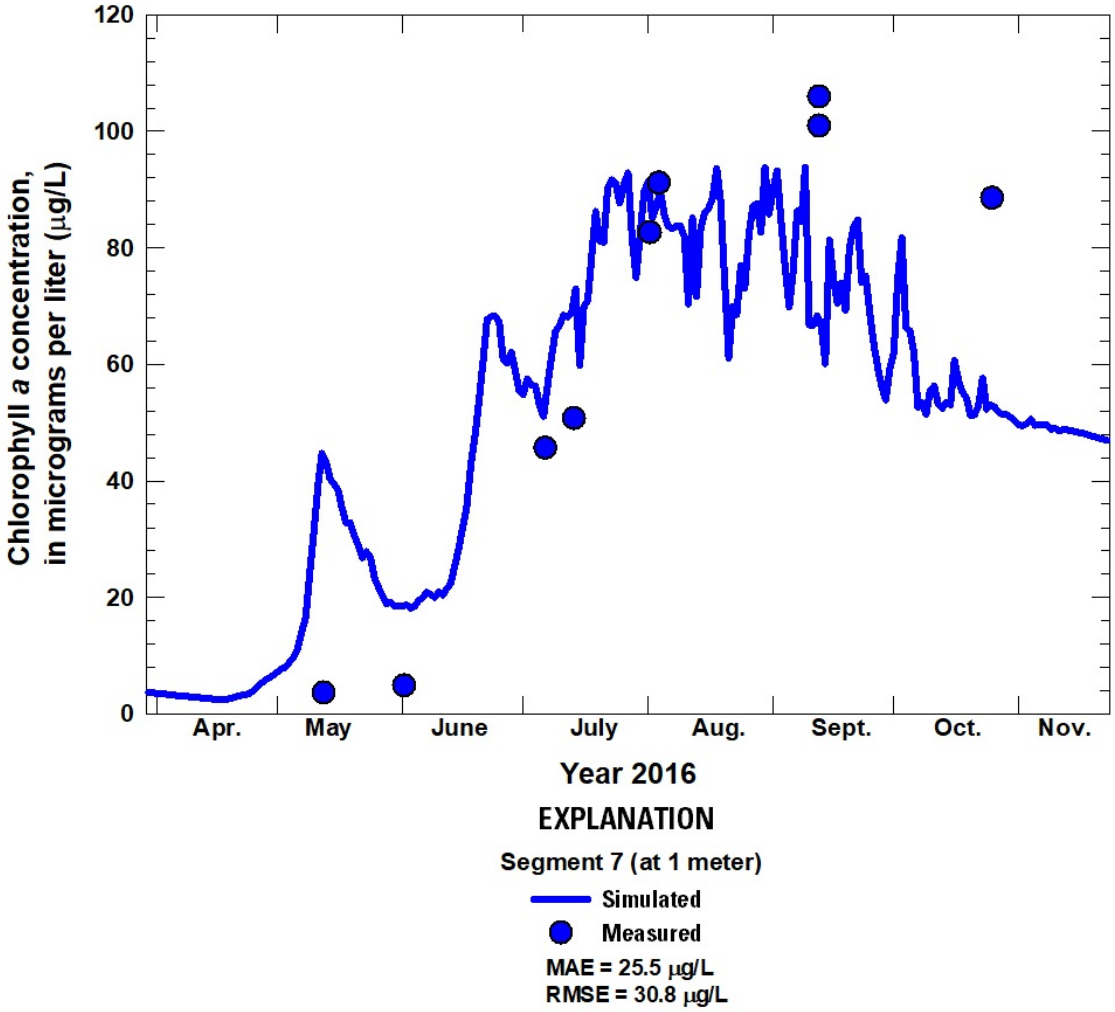

Figure 9: Simulated and measured chlorophyll *a* concentrations for the 1-meter depth at Madison Lake southwest deep point near Madison Lake, Minn. (segment 7) in Madison Lake, March 30 to November 23, 2016.

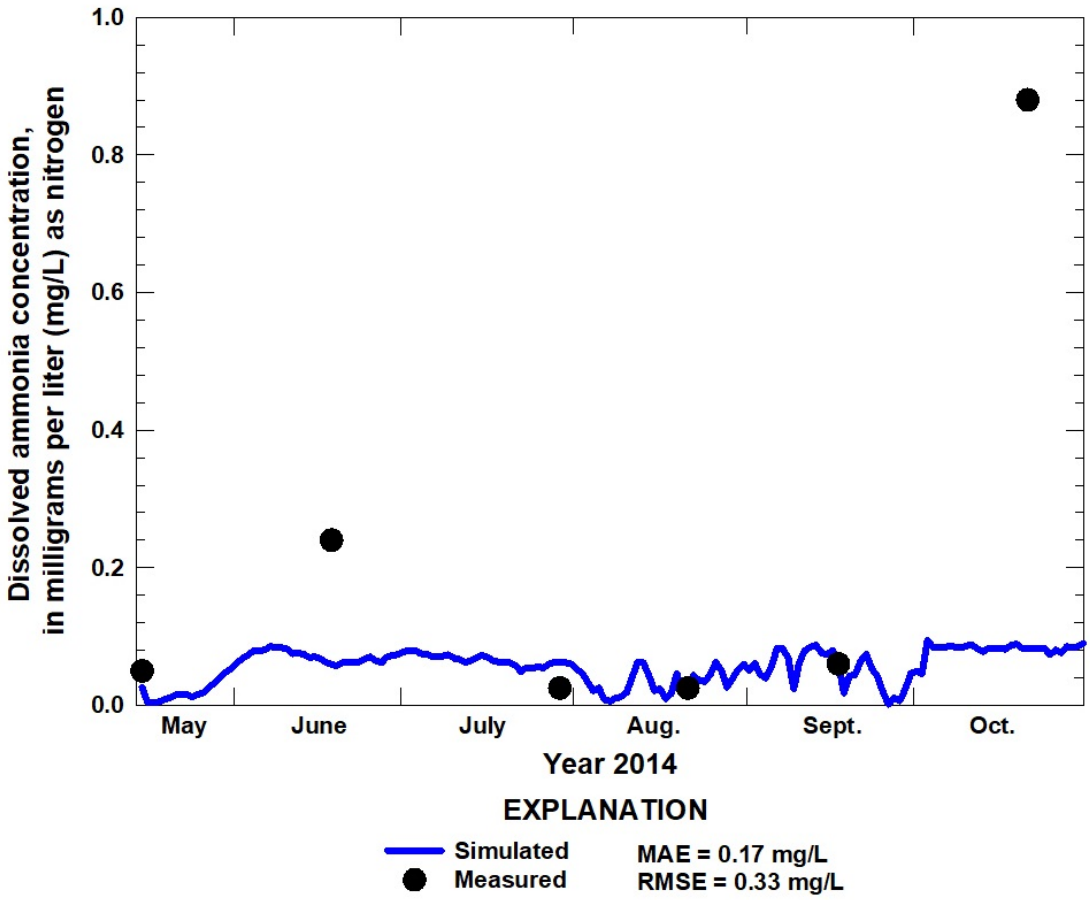

Figure 10: Simulated and measured dissolved ammonia concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

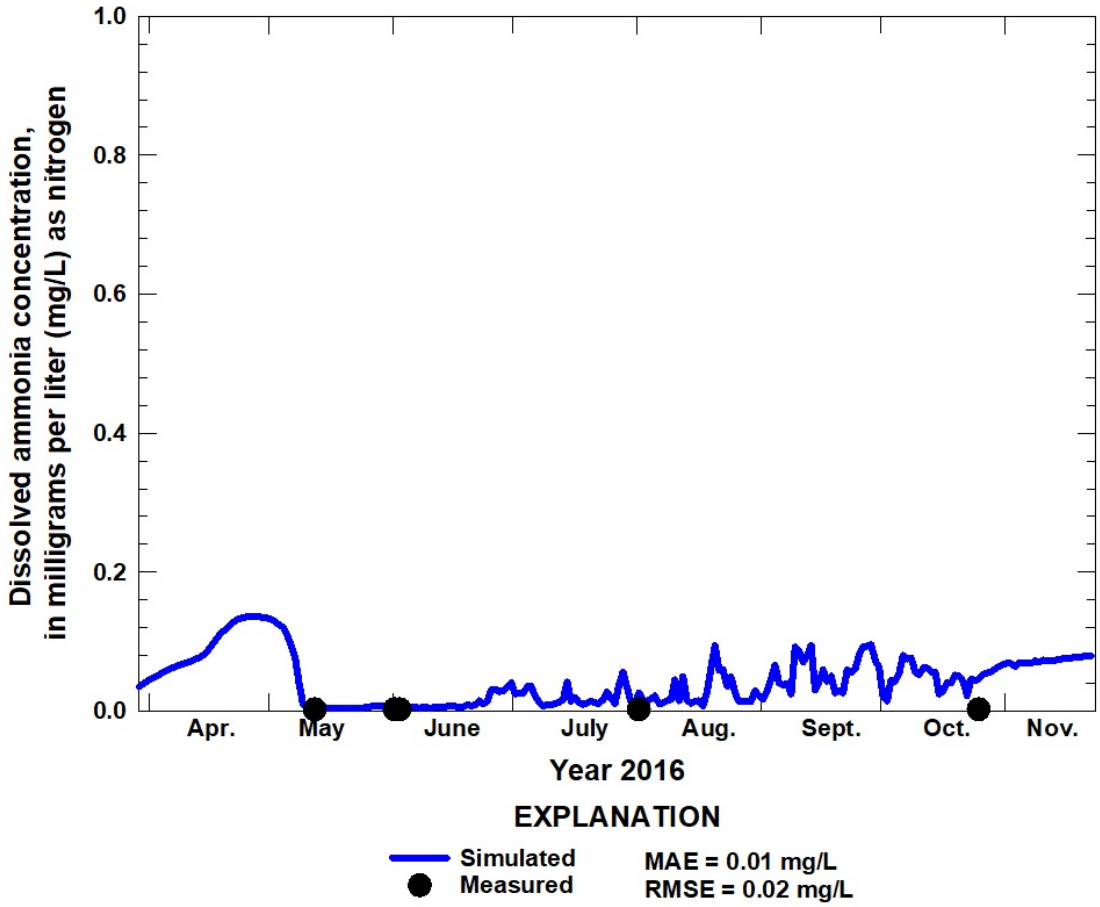

Figure 11: Simulated and measured dissolved ammonia concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

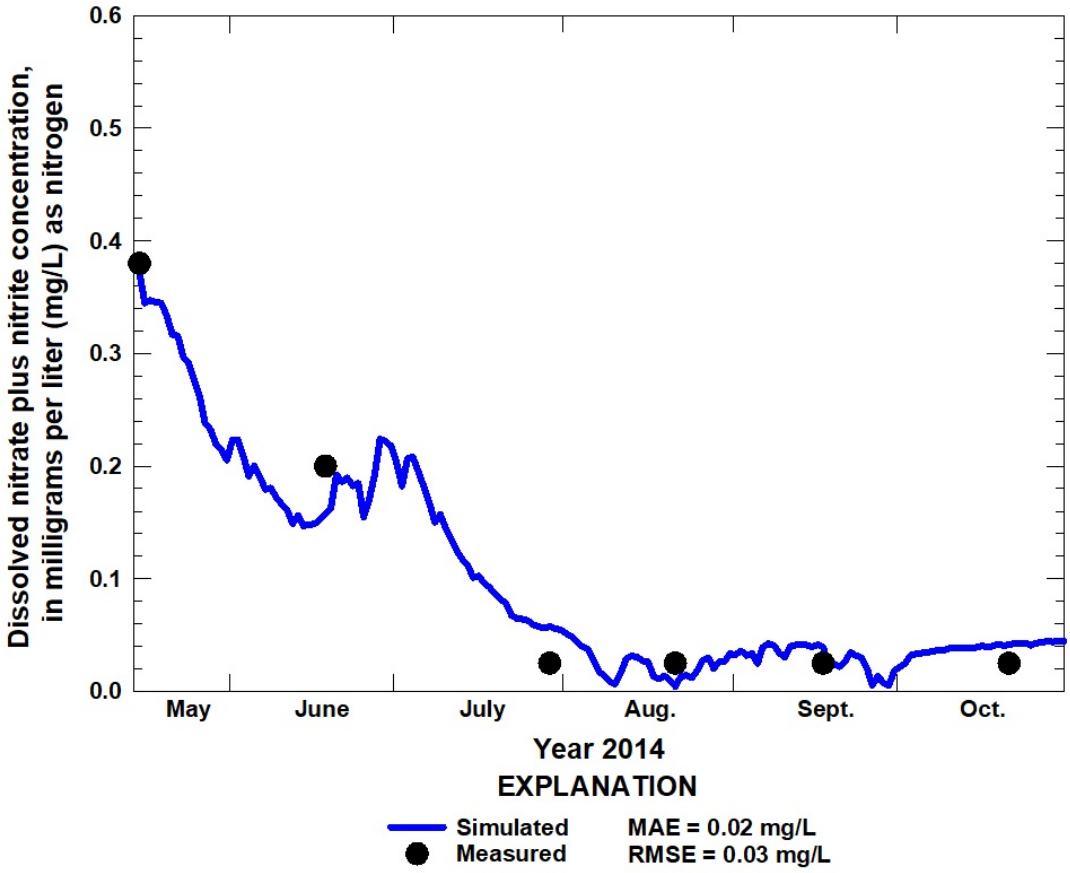

Figure 12: Simulated and measured dissolved nitrate plus nitrite concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

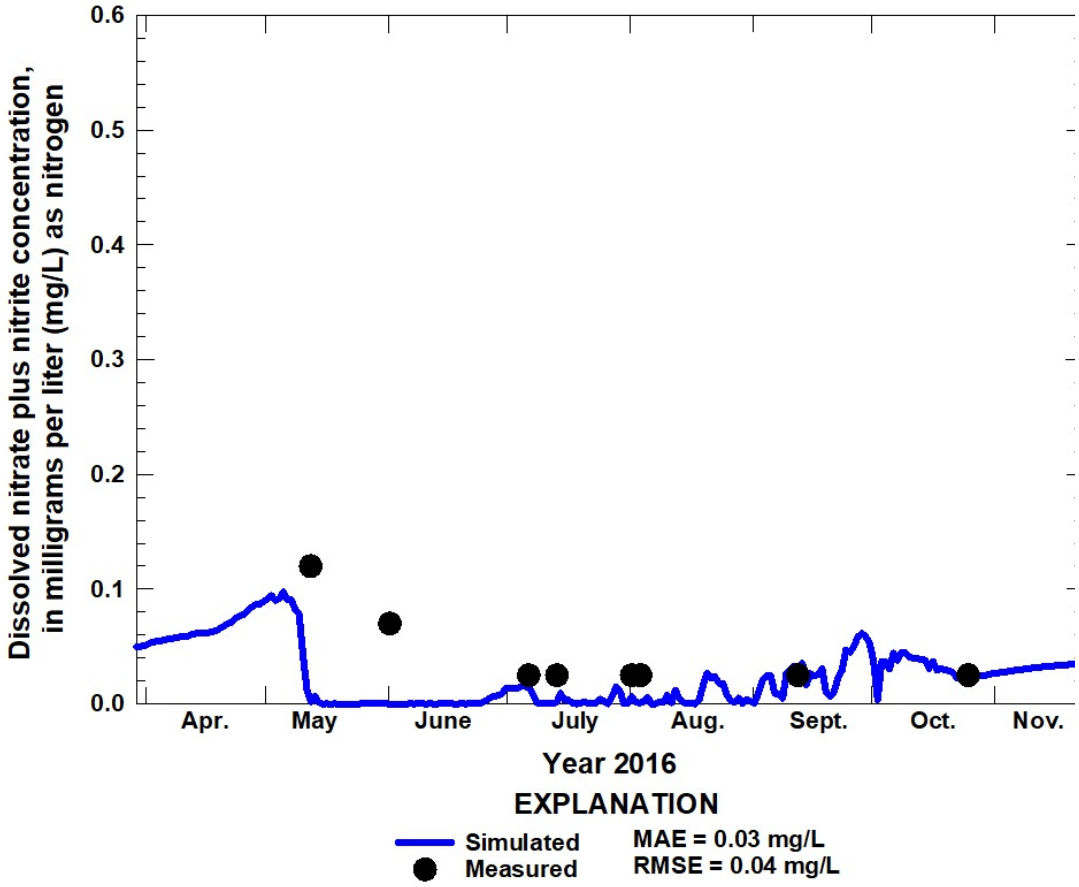

Figure 13: Simulated and measured dissolved nitrate plus nitrite concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

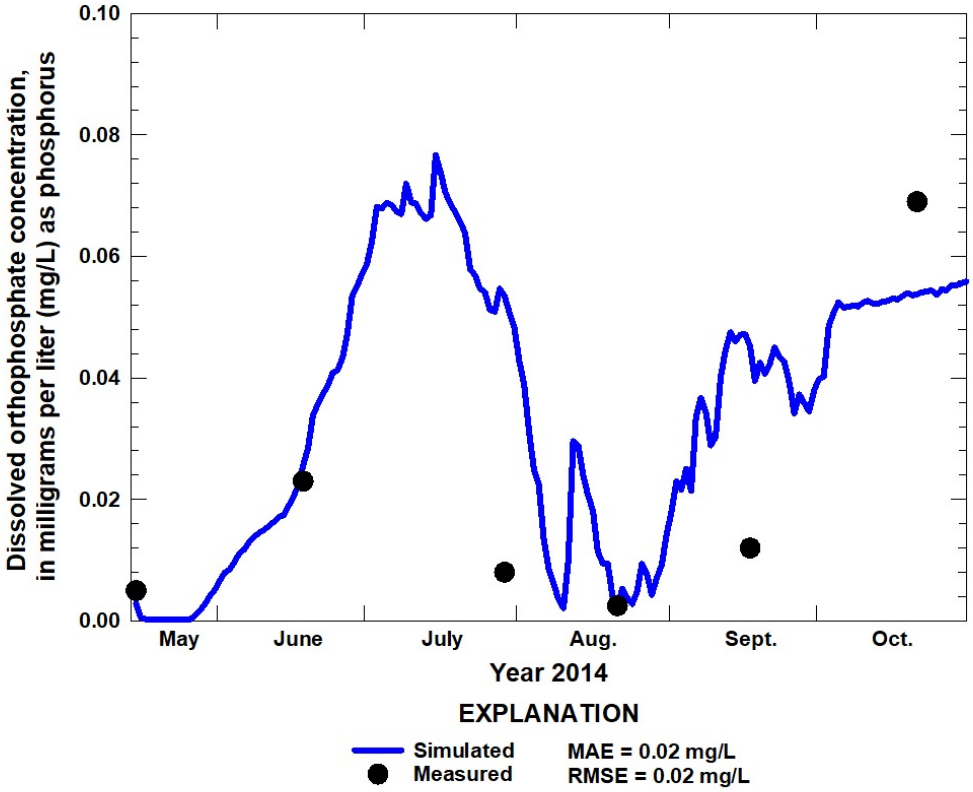

Figure 14: Simulated and measured dissolved orthophosphate concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

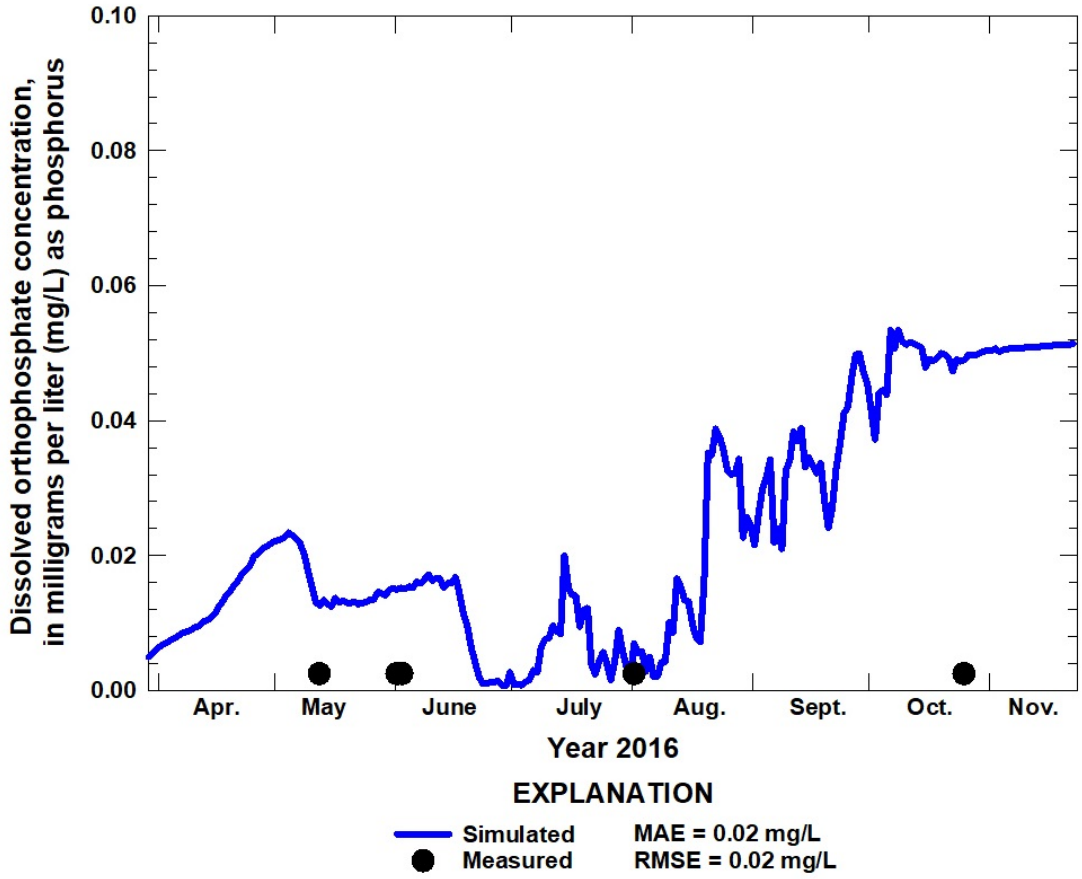

Figure 15: Simulated and measured dissolved orthophosphate concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

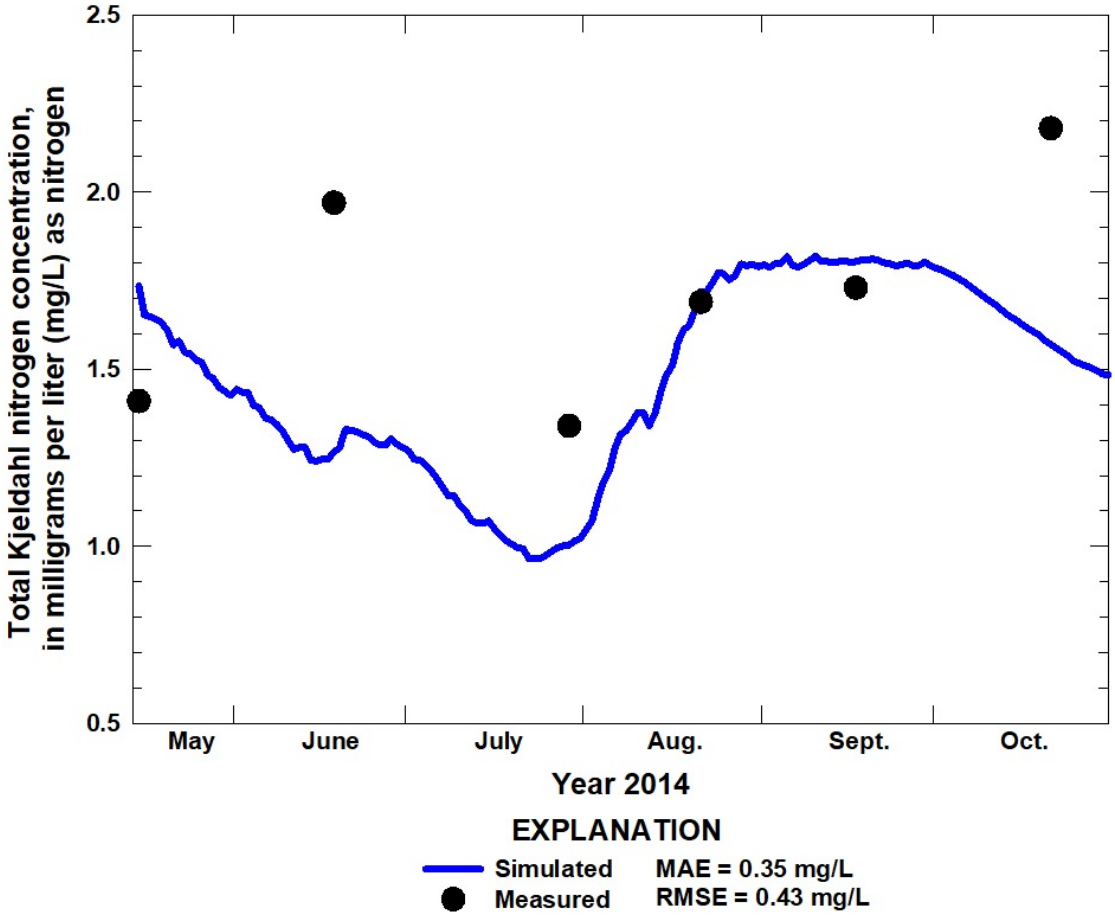

Figure 16: Simulated and measured total Kjeldahl nitrogen concentrations at 1 meter below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

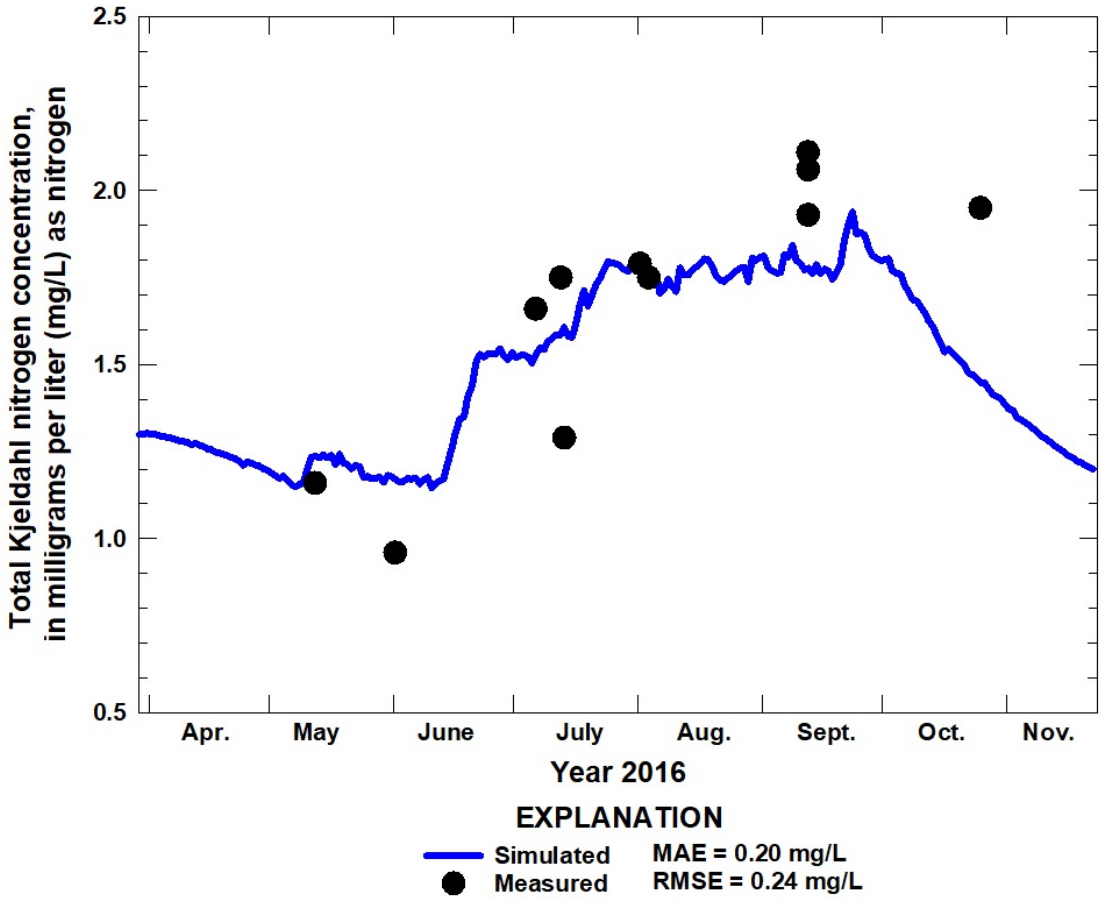

Figure 17: Simulated and measured total Kjeldahl nitrogen concentrations at 2 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

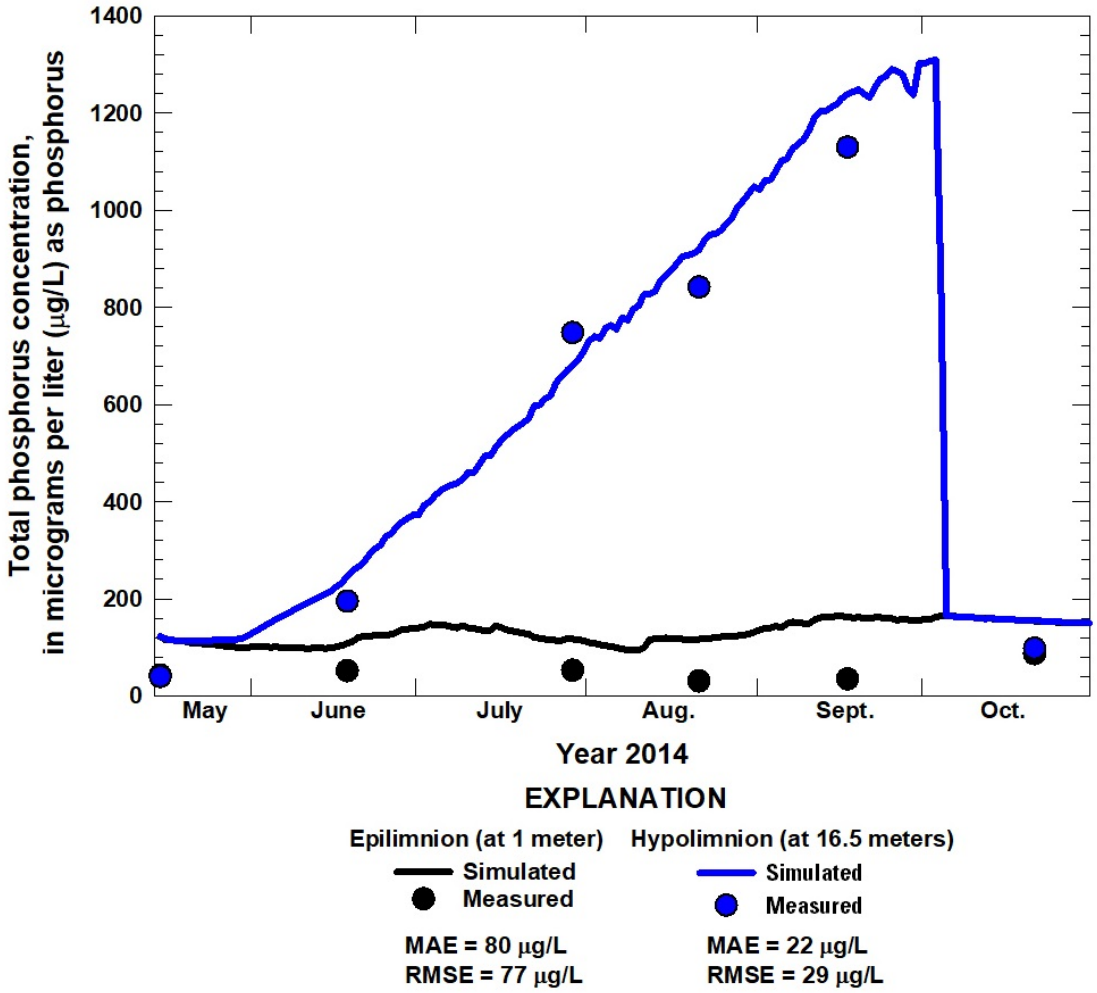

Figure 18: Simulated and measured total phosphorus concentrations at 1 meters and 16.5 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

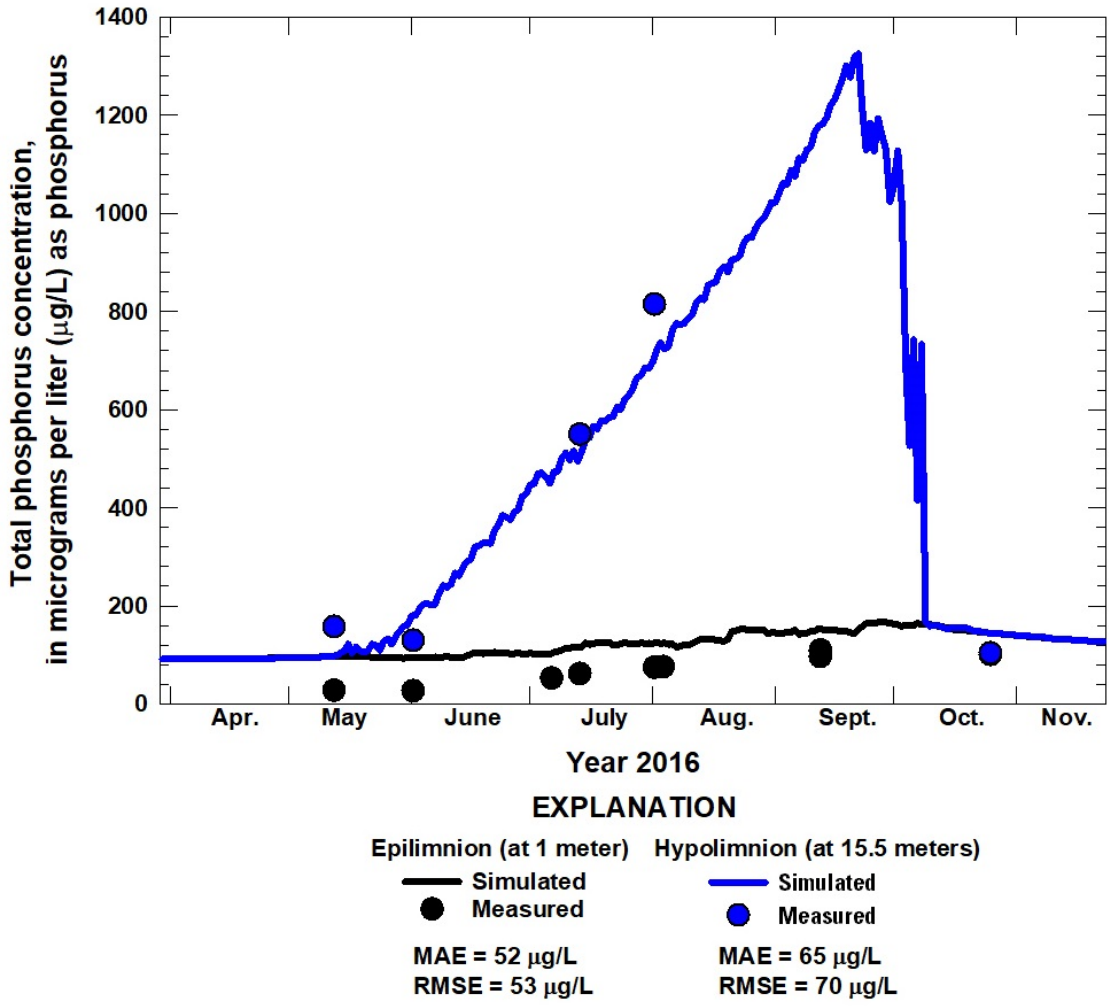

Figure 19: Simulated and measured total phosphorus concentrations at 1 meter and 16.5 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minn., March 30 to November 23, 2016, with values of mean absolute error (MAE) and root mean square error (RMSE).

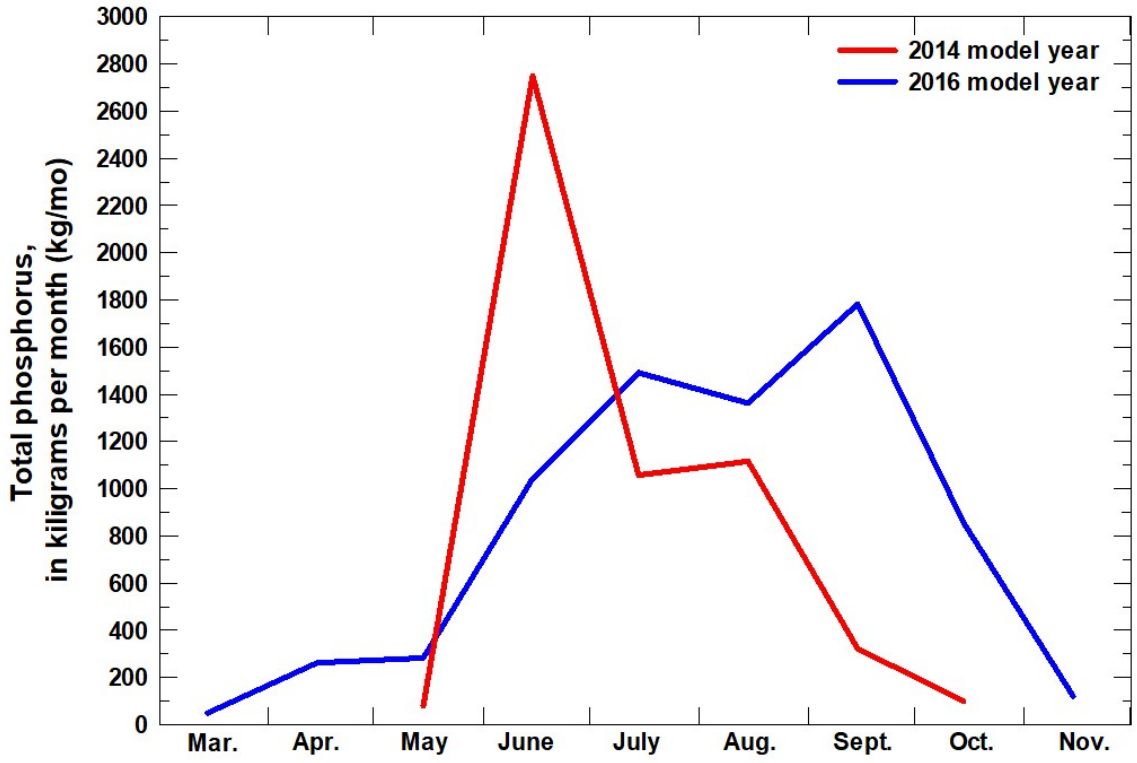

Figure 20: Total phosphorus concentrations monthly, in kilograms per month, for the 2014 and 2016 model years for the updated model.

Table 1. Location of continuous pressure transducers, water-quality sondes, thermistors, and discrete water-quality measurements used for the development of model input or calibration/validation of water temperature, dissolved oxygen, and water-quality constituents.

[All continuous measurements included regular monthly visits to download and calibrate continuous pressure transducers, water-quality sondes, and thermistors. USGS, U.S. Geological Survey; Minn., Minnesota; --, not applicable. Latitude/longitude given in degrees (°), minutes ('), and seconds (") referenced to the North American Datum of 1983. Continuous constituents: S, water stage level; S→Q, discharge/flow (derived from stage); DO, dissolved oxygen; SC, specific conductance; T, water temperature. Discrete constituents: MI, major inorganics; L-L N, low-level nutrients; TC/TN, total carbon/total nitrogen; Alk, alkalinity; Alg, algae. Use: C, calibration; I, input; WQ, water quality (including discrete constituents). Model segment: number identifies segment inflow/outflow attached to in model; I, inflow; O, outflow]

Site name	Common name in report	USGS station number[1]	Minnesota LakeFinder station number[2]	Latitude / longitude	Continuous constituents	Discrete constituents	Use	Model segment
Unnamed stream to Madison Lake at CR-48 near Madison Lake, Minn.	Northeast inlet	05320130	--	44° 11' 53.4"N -93° 46' 39.9"W	S, S→Q, T	DO, pH, SC, T, MI, L-L N, TC/TN, Alk	I	2 (I)
Unnamed stream between Schoolhouse and Goolsby Lakes southeast of Madison Lake, Minn.	Southeast inlet	05320140	--	44° 10' 7.9"N -93° 47' 11.2"W	S, S→Q, T	DO, pH, SC, T, MI, L-L N, TC/TN, Alk	I	12 (I)
Madison Lake outlet to Mud Lake south of Madison Lake, Minn.	Madison Lake outlet	05320170	--	44° 10' 43.7"N -93° 48' 57.1"W	S, S→Q, T	--	S (C), I (S→Q), T (C)	9 (O)
Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	--	07-0044-00-102; 07-0044-00-201	44° 11' 29.8"N -93° 48' 39.7"W	DO, T	DO, pH, SC, T, MI, L-L N, TC/TN, Alk, Alg	WQ (I), T (C), DO (C)	7

[1]U.S. Geological Survey, 2016

[2]Minnesota Department of Natural Resources, 2019b.

Table 2. Water-quality methods for constituents analyzed in water samples from Madison Lake, 2014 and 2016.

[EPA, U.S. Environmental Protection Agency; mg/L, milligrams per meter; SM, standard method; --, not analyzed; µg/L, micrograms per liter; SiO_2, silicon dioxide]

Constituent	Minnesota Department of Health Environmental Laboratory	
	Method	Method detection limit[1]
Dissolved nitrite, as nitrogen	EPA 353.2 (U.S. Environmental Protection Agency, 1993a)	0.01 mg/L
Dissolved nitrite plus nitrate nitrogen	EPA 353.2 (U.S. Environmental Protection Agency, 1993a)	0.05 mg/L
Dissolved ammonia, as nitrogen	EPA 350.1 (U.S. Environmental Protection Agency, 1993b)	0.05 mg/L
Total Kjeldahl nitrogen	EPA 351.2 (U.S. Environmental Protection Agency, 1993c)	0.20 mg/L
Total phosphorus	SM 4500-P (American Water Works Association and others, 1997a)	0.01 mg/L
Dissolved phosphorus	EPA 365.1 (U.S. Environmental Protection Agency, 1993d)	0.01 mg/L
Dissolved orthophosphate	EPA 365.1 (U.S. Environmental Protection Agency, 1993)	0.005 mg/L
Chlorophyll-*a*	SM 10200-H (American Water Works Association and others, 1997b)	0.001 mg/L
Total dissolved solids	SM 2540C (American Water Works Association and others, 1997c)	10 mg/L
Total silica, as silicon dioxide	SM 4500 SiO_2 (American Water Works Association and others, 1997d)	0.5 mg/L
Total alkalinity	Inflection point titration (Wilde, 2006)	1 mg/L
Algal counts	ASA (PhycoTech, 2017)	--
Dissolved iron	EPA 200.7 (U.S. Environmental Protection Agency, 2007)	0.001 mg/L

[1]The minimum detection limit is the minimum concentration of a substance that can be measured and reported with a 99-percent confidence that the analyte concentration is greater than 0 (U.S. Environmental Protection Agency, 2002).

Table 3. Relative counts and converted algal biomass for (in milligrams per liter) for Madison Lake southwest deep point near Madison Lake, Minnesota, 2014 and 2016.

[mg/L, milligrams per liter]

Algal group or genera	Date	Relative Count	Converted algal biomass (mg/L)
Diatoms (bacillariophyta/crysophyta)	2014-05-14	88	1.584
	2014-06-18	5	0.027
	2014-07-29	38	0.883
	2014-08-21	5	0.185
	2014-09-17	16	0.638
	2014-10-21	1	0.002
	2016-05-12	19	0.035
	2016-06-01	7	0.017
	2016-07-13	4	0.103
	2016-08-01	9	0.392
	2016-09-12	4	0.127
	2016-10-25	32	0.716
Green algae (chlorophyta)	2014-05-14	6	0.108
	2014-06-18	9	0.049
	2014-07-29	9	0.209
	2014-08-21	14	0.517
	2014-09-17	6	0.239
	2014-10-21	3	0.005
	2016-05-12	11	0.020
	2016-06-01	1	0.002
	2016-07-13	4	0.103
	2016-08-01	7	0.305
	2016-09-12	4	0.127
	2016-10-25	5	0.112
Fixing cyanophyta	2014-05-14	2	0.036
	2014-06-18	77	0.417
	2014-07-29	14	0.325
	2014-08-21	57	2.104
	2014-09-17	38	1.516
	2014-10-21	0	0.000
	2016-05-12	0	0.000
	2016-06-01	60	0.148
	2016-07-13	40	1.026
	2016-08-01	40	1.741
	2016-09-12	40	1.113
	2016-10-25	9	0.201

Algal group or genera	Date	Relative Count	Converted algal biomass (mg/L)
Non-fixing (buoyant) cyanophyta	2014-05-14	1	0.018
	2014-06-18	2	0.011
	2014-07-29	31	0.721
	2014-08-21	13	0.480
	2014-09-17	25	0.998
	2014-10-21	1	0.002
	2016-05-12	0	0.000
	2016-06-01	12	0.152
	2016-07-13	36	1.214
	2016-08-01	26	1.190
	2016-09-12	146	4.064
	2016-10-25	143	3.199
Flagellates (haptophyta/cryptophyta)	2014-05-14	3	0.054
	2014-06-18	3	0.016
	2014-07-29	5	0.116
	2014-08-21	8	0.295
	2014-09-17	15	0.599
	2014-10-21	94	0.145
	2016-05-12	70	0.129
	2016-06-01	20	0.049
	2016-07-13	15	0.385
	2016-08-01	13	0.566
	2016-09-12	6	0.191
	2016-10-25	9	0.201

Table 4. Initial constituent concentrations for the Madison Lake CE-QUAL-W2 model: 2014 calibration and 2016 validation runs.

[m NAVD 88; meters above North American Vertical Datum of 1988; mg/L, milligrams per liter; °C, degrees Celsius]

Constituent	Year	
	2014	2016
Initial water-surface elevation, m NAVD 88	310.57	340.38
Total dissolved solids (TDS), mg/L	177.7	272.0
Dissolved orthophosphate, mg/L	0.005	0.005
Dissolved ammonia, as nitrogen, mg/L	0.05	0.035
Dissolved nitrite plus nitrate nitrogen, mg/L	0.38	0.05
Dissolved silica, mg/L	3.95	3.00
Particulate silica, mg/L	1	1
Total iron, mg/L	0.014	0.014
Labile dissolved organic matter (DOM), mg/L	4.9510	3.6759
Refractory DOM, mg/L	11.5522	8.5772
Labile particulate organic matter (POM), mg/L	0.1490	0.2651
Refractory POM, mg/L	0.3478	0.6186
Diatoms/Crysophyta, mg/L	1.4	2.5×10^{-12}
Chlorophyta (Green algae), mg/L	5.0×10^{-9}	0.0300
Fixing cyanophyta, mg/L	8.0×10^{-6}	0.0035
Non-fixing (buoyant) cyanophyta, mg/L	7.5×10^{-13}	0.0010
Haptophyta/Cryptophyta, mg/L	1.0×10^{-7}	0.1500
Dissolved oxygen, mg/L	[1]0.75-10.25	10
Inorganic carbon, mg/L	170.4	182.2
Alkalinity, mg/L	140.0	149.7
Labile phosphorus partition	0.0065	0.0065
Labile nitrogen partition	0.0950	0.0950
Refractory phosphorus partition	0.0065	0.0065
Refractory nitrogen partition	0.0950	0.0950
Initial temperature, °C	9.9	6.0
Sediment temperature, °C	14.5	12.5
Macrophyte, mg/L	5.0	0.4

[1]Initial constituent concentrations were considered uniform throughout the lake for every segment and layer, except in cases with a reported range of values, which constitutes a vertical profile. The highest value is at the surface layer, with the lowest value at the bottom layer, with iterative values in between for each of the layers.

Table 5. Summary of mean absolute error (MAE) and root mean square error (RMSE) values for calibration (2014) and validation (2016) runs for Madison Lake at Madison Lake southwest deep point near Madison Lake, Minnesota (also known as southwest deep point).

[°C, degrees Celsius; Minn. Minnesota; mg/L, milligrams per liter; <, less than; µg/L, micrograms per liter; multiple, integrated vertical profile data]

Constituent	Depth (meters)	Number of compared data points	AME	RMSE	Number of compared data points	AME	RMSE
		Calibration Year (2014)			Validation Year (2016)		
Water temperature, °C	multiple	103	0.55	0.70	125	0.67	0.81
Dissolved oxygen, mg/L	multiple	103	0.86	1.22	125	0.91	1.46
Chlorophyll *a*, µg/L	2	6	22	29	10	26	31
Dissolved orthophosphate, mg/L	2	6	0.02	0.02	5	0.02	0.02
Dissolved ammonia, mg/L	2	6	0.17	0.33	5	0.01	0.02
Dissolved nitrite plus nitrate nitrogen, mg/L	2	6	0.02	0.03	10	0.03	0.04
Total Kjeldahl nitrogen, mg/L	2	6	0.35	0.43	11	0.20	0.24
Total phosphorus, µg/L	2	6	80	77	10	52	53
Total phosphorus, µg/L	16.5	6	22	29	5	65	70
Diatoms (bacillariophyta/crysophyta), mg/L	2	6	0.34	0.46	6	0.36	0.44
Green algae (chlorophyta), mg/L	2	6	0.12	0.13	6	0.28	0.39
Fixing cyanophyta, mg/L	2	6	0.45	0.50	6	0.13	0.19
Non-fixing (buoyant) cyanophyta, mg/L	2	6	0.74	1.03	6	0.81	1.02
Flagellates (haptophyta/cryptophyta), mg/L	2	6	0.13	0.17	6	0.19	0.23

Table 6. Summary of phosphorus loading for updated Madison Lake model (2014, 2016), original Madison Lake model, and two phosphorus loading scenarios, according to load estimates and internal CE-QUAL-W2 calculations. Negative terms denote a loss term due to the net export of phosphorus (from the distributary tributary flow).

Model Year/Scenario	Scenario Number	Component	March	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Total
			kilograms/month									kilograms/year
Model run, March 29-November 23, 2016	1	Organic Matter	28.82	141.2	123.1	180.1	285.0	274.6	746.4	412.2	64.27	2256
		Orthophosphate, external load	20.52	100.6	92.40	177.3	338.6	287.1	727.2	385.5	42.69	2172
		Orthophosphate, internal load	0.651	21.18	67.79	686.1	868.8	801.6	309.4	58.25	15.05	2829
		Total Phosphorus	49.99	262.9	283.3	1044	1492	1363	1783	855.9	122.0	7256
		Internal load, percentage of total phosphorus	1.3%	8.1%	23.9%	65.7%	58.2%	58.8%	17.4%	6.8%	12.3%	39.0%
Model run (updated model), May 15-November 1, 2014	2	Organic Matter	--	--	-46.81	909.3	187.0	98.36	18.30	23.80	--	1190
		Orthophosphate, external load	--	--	81.17	1252	258.6	75.54	13.17	16.32	--	1697
		Orthophosphate, internal load	--	--	45.71	588.0	612.6	942.6	291.2	58.92	--	2539
		Total Phosphorus	--	--	80.06	2750	1058	1116	322.7	99.04	--	5426
		Internal load, percentage of total phosphorus	--	--	57.1%	21.4%	57.9%	84.4%	90.2%	59.5%	--	46.8%
Model run (original model), May 15-November 1, 2014	3	Organic Matter	--	--	-46.81	909.3	187.0	98.36	18.30	23.80	--	1190
		Orthophosphate, external load	--	--	81.17	1252	258.6	75.54	13.17	16.32	--	1697
		Orthophosphate, internal load	--	--	41.01	479.8	628.0	951.0	204.0	53.77	--	2358
		Total Phosphorus	--	--	75.37	2641	1074	1125	235.5	93.89	--	5244
		Internal load, percentage of total phosphorus	--	--	54.4%	18.2%	58.5%	84.5%	86.6%	57.3%	--	45.0%

Model Year/Scenario	Scenario Number	Component	March	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Total
			kilograms/month									kilograms/year
Model run (original model), May 15-November 1, 2014, increasing external orthophosphorus load (20 percent)	4	Organic Matter	--	--	-46.81	909.3	187.0	98.36	18.30	23.80	--	1190
		Orthophosphate, external load	--	--	97.78	1503	310.3	90.52	15.76	19.56	--	2037
		Orthophosphate, internal load	--	--	40.73	480.9	628.8	956.5	208.7	53.73	--	2369
		Total Phosphorus	--	--	91.69	2893	1126	1145	242.8	97.08	--	5596
		Internal load, percentage of total phosphorus	--	--	44.4%	16.6%	55.8%	83.5%	86.0%	55.3%	--	42.3%
Model run (original model), May 15-November 1, 2014, decreasing external orthophosphorus load (20 percent)	5	Organic Matter	--	--	-46.81	909.3	187.0	98.36	18.30	23.80	--	1190
		Orthophosphate, external load	--	--	64.46	1001	206.8	60.38	10.48	12.96	--	1356
		Orthophosphate, internal load	--	--	41.31	480.5	630.7	949.3	202.0	54.05	--	2358
		Total Phosphorus	--	--	58.95	2391	1025	1108	230.7	90.81	--	4904
		Internal load, percentage of total phosphorus	--	--	70.1%	20.1%	61.6%	85.7%	87.5%	59.5%	--	48.1%